Residual Risk Assessment for the Surface Coating of Metal Coil Source Category in Support of the 2020 Risk and Technology Review Final Rule

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Table of Contents

Executive Summary 5
1 Introduction 7
2 Methods 8
2.1 Emissions and source data 9
2.2 Dispersion modeling for inhalation exposure assessment 10
2.3 Estimating chronic human inhalation exposure 13
2.4 Acute risk screening and refined assessments 14
2.5 Multipathway human health risk assessment 15
2.6 Environmental risk assessment 21
2.7 Dose-response assessment 23
2.7.1 Sources of chronic dose-response information 23
2.7.2 Sources of acute dose-response information 28
2.8 Risk characterization 30
2.8.1 General 30
2.8.2 Mixtures 32
3 Risk results for the Surface Coating of Metal Coil source category 33
3.1 Source category description and emissions 33
3.2 Baseline risk characterization 36
3.2.1 Risk assessment results based on actual emissions 37
3.2.2 Risk assessment results based on allowable emissions 39
3.3 Post-control risk characterization 40
4 General discussion of uncertainties in the risk assessment 40
4.1 Emissions inventory uncertainties 40
4.2 Exposure modeling uncertainties 41
4.2.1 Inhalation exposure modeling 41
4.2.2 Multipathway exposure modeling 43
4.2.3 Environmental risk screening assessment 45
4.3 Uncertainties in the dose-response relationships 46
5 References 55

Index of Tables

Table 2.2-1. AERMOD version 16216r Model Options for RTR Modeling 11
Table 2.5-1. Multipathway Scenarios and Ingestion Pathways 16
Table 3.1-1 Summary of Emissions from the Surface Coating of Metal Coil Source Category and Dose-Response Values Used in the Residual Risk Assessment 35
Table 3.2-1. Source Category Level Inhalation Risks for Surface Coating of Metal Coil Based on Actual Emissions 37
Table 3.2-2 Source Category Contribution to Facility-Wide Cancer Risks Based on Actual Emissions 39

Appendices

Appendix 1 Emissions Inventory Support Documents
Appendix 2 Technical Support Document for HEM-3 Modeling
Appendix 3 Meteorological Data for HEM-3 Modeling
Appendix 4 Dispersion Model Receptor Revisions and Additions
Appendix 5 Analysis of short-term emission rates relative to long-term emission rates
Appendix 6 Technical Support Document for TRIM-Based Multipathway Tiered Screening Methodology for RTR
Appendix 7 Protocol for Site-Specific Multipathway Risk Assessment
Appendix 8 Dose-Response Values Used in the RTR Risk Assessments
Appendix 9 Environmental Risk Screening Assessment
Appendix 10 Detailed Risk Modeling Results
Appendix 11 Site-Specific Human Health Multipathway Residual Risk Assessment Report

Index of Acronyms

AERMOD	American Meteorological Society/EPA Regulatory Model
AEGL	Acute exposure guideline level
ASTDR	US Agency for Toxic Substances and Disease Registry
CalEPA	California Environmental Agency
CTE	Central Tendency Estimate
ERPG	Emergency Response Planning Guideline
HAP	Hazardous Air Pollutant
HEM	Human Exposure Model
HI	Hazard index
HQ	Hazard quotient
IRIS	Integrated Risk Information System
MACT	Maximum Achievable Control Technology
MIR	Maximum Individual Risk
MOA	Mode of action
NAC	National Advisory Committee
NAAQS	National Ambient Air Quality Standards
NATA	National Air Toxics Assessment
NEI	National Emissions Inventory
NPRM	Notice of Proposed Rulemaking
PB-HAP	Persistent and Bioaccumulative – HAP
PAH	Polycyclic aromatic hydrocarbon
POM	Polycyclic organic matter
REL	Reference exposure level
RfC	Reference concentration
RfD	Reference dose
RTR	Risk and Technology Review

TOSHI	Target-organ-specific hazard index
TRIM	Total Risk Integrated Methodology
TRIM.FaTE	TRIM Environmental Fate, Transport, and Ecological Exposure
URE	Unit risk estimate

Executive Summary

This document describes the risk assessment that the U.S. Environmental Protection Agency (EPA) conducted to assess the human health and environmental risks posed by hazardous air pollutant (HAP) emissions from the Surface Coating of Metal Coil Industry. Section 112 of the Clean Air Act (CAA) establishes a two-stage regulatory process for addressing emissions of HAP from stationary sources. In the first stage, EPA must promulgate technology-based national emission standards for hazardous air pollutants (NESHAP) for categories of sources. EPA has completed this stage. For NESHAP that require maximum achievable control technology (MACT) standards, EPA is required to complete a second stage of the regulatory process – the residual risk review. In this second stage, EPA is required to assess the health and environmental risks that remain after implementation of the standards. EPA must also review each of the technology-based standards at least every eight years and revise them, as necessary, taking into account developments in practices, processes and control technologies. If appropriate based on the results of the risk and technology reviews, the Agency will revise the rule. For efficiency, the Agency includes the analyses in the same regulatory package and calls the rulemakings the Risk and Technology Review (RTR).

The specific source category results contained in this document are from the Surface Coating of Metal Coil Industry residual risk assessment, in support of EPA's 2018 proposed rule. The EPA is proposing amendments to the NESHAP for this source category, under 40 CFR part 63, subpart SSSS, to address the results of the RTR review of the MACT standards, required under Section 112. Metal coil surface coating is a process-specific rather than a product-specific operation. This source category includes facilities that contain a metal coil surface coating line, which is the process and the collection of equipment used to apply an organic coating to the surface of metal coil that is at least 0.15 millimeter (0.006 inch) thick. A coil coating line includes a web unwind or feed section, a series of one or more work stations, any associated curing oven, wet section, and quench station. The portions of the metal coil coating line to which the emission limitations apply are the coating application stations and associated curing ovens. We estimate that there are 48 major source Surface Coating of Metal Coil facilities operating in the U.S. The total emissions of HAP for the source category are approximately 291 tons per year. The HAP emitted in the largest quantities are xylenes, toluene, glycol ethers, naphthalene, isophorone, diethylene glycol monobutyl ether, ethyl benzene, methyl isobutyl ketone, ethylene glycol, methanol, dimethyl phthalate and formaldehyde. Emissions of these pollutants make up over 99 percent of the total HAP emissions by mass. Emissions of persistent and bioaccumulative HAP (PB-HAP) include lead compounds. Emissions of environmental HAP include lead compounds and hydrofluoric acid.

The below table summarizes the results of the risk assessment for the Surface Coating of Metal Coil source category. The results of the chronic inhalation cancer risk assessment based on actual emissions from the Surface Coating of Metal Coil source category indicate that the maximum lifetime individual cancer risk posed by the 48 facilities could be as high as 10-in-1 million, with naphthalene and formaldehyde from solvent storage as the major contributors to the risk. The total estimated cancer incidence from this source category is 0.005 excess cancer cases per year, or one excess case in every 200 years. Approximately 57,000,000 people live within 50 kilometers of the 48 modeled Metal Coil Surface Coating facilities, and 19,000 people are estimated to have cancer risks above 1-in-1 million from HAP emitted from the

facilities in this source category, with 40 of those people estimated to have cancer risks above 10-in-1 million.

Risk Summary for the Surface Coating of Metal Coil Source Category

	Inhalation Cancer Risk		Population Cancer Risk			Max Chronic Individual Noncancer Risk		Max Acute Noncancer Risk		Multipathway Assessment
	Maximum Individual Risk (in 1 million)	Risk Driver	Cancer Incidence (cases per year)	≥ 10 in 1 million	≥ 1 in 1 million	Hazard Index (TOSHI)	Risk Driver	Hazard Quotient	Risk Driver	Risk Driver and Health Endpoints
Baseline Actual Emissions										
Source Category	10	naphthalene, formaldehyde	0.005	40	19,000	0.1 (reproductive)	glycol ethers	3	diethylene glycol monobutyl ether	---
Whole Facility	40	naphthalene	0.03	1,700	270,000	5 (respiratory)	chlorine	---	---	---
Baseline Allowable Emissions										
Source Category	10	naphthalene, formaldehyde	0.006	40	24,000	0.1 (reproductive)	glycol ethers	---	---	---

Regarding the noncancer risk assessment, the maximum chronic noncancer hazard index value for the source category could be up to 0.1 (for reproductive effects) driven by emissions of glycol ethers from prime coating application and finish coating processes. Of the 57,000,000 people living within 50 kilometers of these facilities, no one is exposed to noncancer hazard index levels above 1, based on actual emissions from sources regulated under this source category.

Regarding acute health risks posed by actual baseline emissions, the highest refined acute hazard quotient is estimated to be 3 (based on an acute dose-response value for diethylene glycol monobutyl ether). Two facilities are expected to have an acute hazard quotient greater than 1 for the acute dose-response values examined.

Whole facility (or "facility-wide") emissions include those regulated under this source category plus all other emissions generated at each facility. The results of the chronic inhalation cancer risk assessment based on whole facility emissions are more uncertain and rely on the quality of the emissions data collected for source categories outside this regulatory review. These emissions sources may not undergo the same level of data quality review as those being assessed in this regulatory assessment. The maximum lifetime individual cancer risk posed by the 48 facilities, based on whole facility emissions, is 40-in-1 million with naphthalene from equipment cleanup of metal coil processes driving the risk. The total estimated cancer incidence based on whole facility emissions is 0.03 excess cancer cases per year, or one excess case in every 30 years. Approximately 270,000 people are estimated to have cancer risks above 1-in-1 million from HAP emitted from all sources at the facilities in this source category, with 1,700 of those people estimated to have cancer risks above 10-in-1 million. Regarding the noncancer risk assessment, the maximum chronic noncancer hazard index posed by whole facility emissions is estimated to be 5 (for respiratory effects) driven by emissions of chlorine from a secondary aluminum fluxing process. There are 120 people

exposed to a noncancer (respiratory) hazard index level above 1, based on whole facility emissions from 2 of the 48 facilities within this source category.

In evaluating the potential for multipathway effects from emissions of lead, modeled maximum annual lead concentrations were compared to the NAAQS for lead (0.15 $\mu g/m^3$). Results of this analysis confirm that the NAAQS for lead would not be exceeded by any facility.

A screening-level evaluation of the potential adverse environmental risk associated with emissions of hydrofluoric acid indicated that no ecological benchmarks are exceeded.

Finally, potential differences between actual emission levels and the maximum emissions allowable under EPA's standards (i.e., "allowable emissions") were also calculated for the Metal Coil Surface Coating facilities. Emission estimates based on the actual emissions were then scaled up using a factor of 1.1 to develop the allowable emissions. Risk results from the inhalation risk assessment indicate that the maximum lifetime individual cancer risk could be as high as 10-in-1 million for allowable emissions, with naphthalene and formaldehyde from solvent storage driving the cancer risks for both the actual and allowable emissions scenarios. The maximum chronic noncancer hazard index value would remain below one, with a hazard index as high as 0.1 (for the reproductive hazard index) at the allowable emissions level, with glycol ethers emissions from prime coating application and finish coating processes driving this maximum hazard index value. The total estimated cancer incidence from this source category, considering allowable emissions, is expected to be about 0.006 excess cancer cases per year, or one excess case in every 200 years. Based on allowable emission rates for this source category, 24,000 people are estimated to have cancer risks above 1-in-1 million, with 40 of those people estimated to have a cancer risk above 10-in-1 million. No people are estimated to have a noncancer hazard index above 1.

This document summarizes the methods used to conduct the risk assessment of this source category as well as the results. Section 1 discusses the relevant regulatory framework including background on the Clean Air Act sections which require the EPA to conduct these source category risk assessments. Methods described in Section 2 include those used by EPA to develop refined estimates of chronic inhalation exposures and human health risks for cancer and noncancer endpoints, as well as those used to screen for acute health risks, chronic non-inhalation (i.e., multipathway) health risks, and adverse environmental effects. The source category-specific results for the risks are presented in Section 3. Section 4 contains a discussion of the uncertainties of the risk assessment, including uncertainties in the exposure assessment and in the dose-response values. The appendices to this risk report contain detailed descriptions of the methods used and the results.

1 Introduction

Section 112 of the Clean Air Act (CAA) establishes a two-stage regulatory process for addressing emissions of hazardous air pollutants (HAP) from stationary sources. In the first stage, section 112(d) requires the Environmental Protection Agency (EPA, or the Agency) to

develop technology-based National Emission Standards for Hazardous Air Pollutants (NESHAP) for categories of sources (e.g., petroleum refineries, pulp and paper mills, etc.). EPA has completed this stage. For NESHAP that require maximum achievable control technology (MACT) standards, EPA is required to complete a second stage of the regulatory process – the residual risk review. In this second stage, EPA is required under section 112(f)(2) to assess the health and environmental risks that remain after implementation of the MACT standards. If additional risk reductions are necessary to protect public health with an ample margin of safety or to prevent an adverse environmental effect, EPA must develop standards to address these remaining risks. For each source category for which EPA issued MACT standards, the residual risk stage must be completed within eight years of promulgation of the initial technology-based standard.

Also, under section 112(d)(6), EPA must review each of the technology-based standards at least every eight years and revise it, as necessary, taking into account developments in practices, processes and control technologies. If appropriate based on the results of the risk and technology reviews, the Agency will revise the rule. For efficiency, the Agency includes the 112(f) and 112(d) analyses in the same regulatory package and calls the rulemakings the Risk and Technology Review (RTR).

In December 2006 we consulted with a panel from the EPA's Science Advisory Board (SAB) on the "Risk and Technology Review (RTR) Assessment Plan," and in June 2007 we received a letter with the results of that consultation. Subsequent to the consultation, in June 2009, EPA met with an SAB panel for a formal peer review of the "Risk and Technology Review (RTR) Assessment Methodologies" (USEPA, 2009a). We received the final SAB report on this review in May 2010 (USEPA, 2010a). Where appropriate, we responded to the SAB's key recommendations in developing our current risk assessments and continue our efforts to improve our assessments by incorporating updates that address the SAB's recommendations as they are developed and become available. Our responses to the key recommendations of the SAB are outlined in a memo entitled, "EPA's Actions in Response to Key Recommendations from the SAB Review of RTR risk Assessment Methodologies" (USEPA, 2010b). EPA has updated several aspects of the risk assessment methodologies contained in the 2009 document. In 2017, we submitted these updated methodologies to SAB for review. The updated methodologies are described in, Screening Methodologies to Support Risk and Technology Reviews (RTR): A Case Study Analysis. SAB's findings for this review was submitted to EPA September 2018.

This document contains the methods we use to conduct the risk assessment, the results of the residual risk assessment performed for the Surface Coating of Metal Coil source category, and a description of associated uncertainties.

2 Methods

A risk assessment consists of four steps: 1) hazard identification, 2) dose-response assessment, 3) exposure assessment, and 4) risk characterization. The first step, hazard identification, determines whether the pollutants of concern can be linked to the health effects in question (cancer and/or noncancer). Section 112 of the CAA identifies the HAP to be

considered in the risk assessment for this source category. The second step is the dose-response assessment, which quantifies the relationship between the dose of a pollutant and the resultant health effects. Dose-response assessments are performed by EPA through the Integrated Risk Information System (IRIS) process as well as by other agencies, such as the Agency for Toxic Substances and Disease Registry (ATSDR). See Section 2.7 of this document for more information on dose-response assessments. The third and fourth steps, the exposure assessment and the risk characterization, respectively, are specific to the source category and are described throughout this report. The exposure assessment includes characterization of HAP emissions, environmental fate and transport, and population exposure for both inhalation and non-inhalation pathways. The fourth and final step, risk characterization, integrates all the information from the previous steps and describes the outcome of the assessment. This four-step approach to risk assessment was endorsed by the National Academy of Sciences in its publication "Science and Judgment in Risk Assessment" (NAS, 1994) and subsequently was adopted in the EPA's "Residual Risk Report to Congress" (USEPA, 1999).

The EPA conducts a risk assessment that provides estimates of the maximum individual risk (MIR) posed by the HAP emissions from each source in the source category, the hazard index (HI) for chronic exposures to HAP with potential to cause chronic (or long-term) noncancer health effects and the hazard quotient (HQ) for acute exposures to HAP with the potential to cause acute (or short-term) noncancer health effects. The MIR is defined as the cancer risk associated with a lifetime of exposure at the highest concentration of HAP where people are likely to live. The HQ is the ratio of the potential exposure to the HAP to the level at or below which no adverse effects are expected; the HI is the sum of HQs for HAP that affect the same target organ or organ system. The risk assessment also provides estimates of the distribution of cancer risks within the exposed populations, cancer incidence and an evaluation of the potential for adverse environmental effects. The following sections describe how we estimate HAP emissions and conduct steps three and four of the risk assessment. The methods used to assess risks are consistent with those peer reviewed by a panel of the EPA's Science Advisory Board (SAB) in 2009 and described in their peer review report issued in 2010 (USEPA 2010a).

2.1 Emissions and source data

To conduct the exposure assessment, EPA gathers the best available data on emissions, emissions release parameters, and other relevant source category-specific parameters. EPA determines the HAP emissions levels from emission points in the source category and identifies the emissions release characteristics of these emission points (e.g., stack height). EPA often begins with the National Emissions Inventory (NEI) database as the starting point for emissions and emissions release characteristics for the source category. The NEI database contains information about sources that emit HAP, and it contains annual air pollutant emissions estimates. EPA's industry experts review the source category data for consistency and completeness. This includes an evaluation of facilities contained in the source category, the emissions units expected to be included for the processes in the source category, and the HAP compounds and emissions levels typically seen. If necessary, EPA will conduct a formal information collection request (CAA, Section 114) for emissions data and other data from the

industry associated with the source category under review. Following the creation of the initial data set, the EPA performs the technology review and the residual risk assessment. If appropriate, based on the results of these reviews, the EPA proposes regulatory action for the source category in a Notice of Proposed Rulemaking (NPRM) published in a *Federal Register* notice. The NPRM data sets are available for public review in the rulemaking docket. Industry, state and local agencies, as well as the public have an opportunity to provide comments on the data, analyses, and results used to support the proposed action. EPA incorporates the comments, as appropriate, conducts any re-assessment, and summarizes and responds to comments before finalizing the action. Through source category-specific engineering reviews, information collection efforts, and public comment, EPA ensures that the data used to conduct risk assessments in support of the RTR rulemakings are of high quality.

In order to put the source category risks in context, we also examine the risks from the entire "facility," where the facility includes all HAP-emitting operations within a contiguous area and under common control. In other words, we examine the HAP emissions not only from the source category emission points of interest, but also from all other emission sources at the facility for which we have data. Using the most current available NEI data at the time of the assessment, the EPA develops "facility-wide" emissions estimates. It is important to note that the NEI facility-wide inventory may not always reflect the level of detail or be representative of the same temporal period that is found in the source category-specific inventory. Further information on the NEI, which is developed from federal/state/local/tribal submitted data, can be found on the EPA's web site at: https://www.epa.gov/air-emissions-inventories/national-emissions-inventory.

Details on the development of the source data, emissions and associated uncertainties in the data for the Surface Coating of Metal Coil source category can be found in Appendix 1 (*Emissions Inventory Support Documents*). Section 3 provides a summary of the processes and emissions associated with this source category.

2.2 Dispersion modeling for inhalation exposure assessment

For the residual risk analyses, we estimate both long- and short-term inhalation exposure concentrations and associated health risks from each facility in the source category. To do this, we use the Human Exposure Model 3 (HEM-3 or HEM-AERMOD) modeling system – which combines the Human Exposure Model (HEM) with the American Meteorological Society/EPA Regulatory Model (AERMOD) dispersion modeling system. HEM-3 performs three main operations: atmospheric dispersion modeling, estimation of individual human exposures and health risks, and estimation of population risks. The approach used in applying this modeling system is outlined below. Further details are provided in Appendix 2 to this document (*Technical Support Document for HEM-3 Modeling*). This section focuses on the dispersion modeling component.

The dispersion model in the HEM-3 modeling system, AERMOD version 16216r, is a state-of-the-science Gaussian plume dispersion model that is preferred by EPA for modeling point, area, and volume sources of continuous air emissions from facility applications (USEPA,

2005a). Further details on AERMOD can be found in the AERMOD User's Guide (USEPA, 2016a) and the AERMOD Implementation Guide (USEPA, 2016b).[1] The model is used to develop annual average ambient concentrations through the simulation of hour-by-hour dispersion from the emission sources into the surrounding atmosphere. Unless data are available on the hours of operation for a source category, default hourly emission rates used for this simulation are generated by evenly dividing the total annual emission rate from the inventory into the 8,760 hours of the year.

The first step in the application of the HEM-3 modeling system is to predict ambient concentrations at locations of interest. The AERMOD model options employed are summarized in Table 2.2-1 and are discussed further below.

Table 2.2-1. AERMOD version 16216r Model Options for RTR Modeling

Modeling Option	*Selected Parameter for chronic exposure*
Type of calculations	Hourly Ambient Concentration
Source types	Point Volume Area Polygon Line Buoyant Line
Receptor orientation	Polar (13 rings and 16 radials) Discrete (census block centroids) and user-supplied receptors
Terrain characterization	Actual from USGS 1/3-arc-second DEM data
Building downwash	Not Included
Plume deposition/depletion	Not Included
Urban source option	Site Specific (See Appendix 2)
Meteorology	1-year representative NWS from nearest site (824 stations) for year 2016

In HEM-3, meteorological data are ordinarily selected from a list of more than 800 National Weather Service (NWS) surface observation stations across the continental United States, Alaska, Hawaii, and Puerto Rico, and HEM-3 defaults to the station closest to each modeled facility. We use data from other stations in special circumstances if we have reason to believe that other data are more representative for certain facilities. In this analysis, the average distance between a modeled facility and the respective meteorological station was 12 miles (19 km). The meteorological data in HEM-3's library are for a single year, and 2016 is the most recent full year of available data. EPA's Guideline on Air Quality Models addresses the regulatory application of air quality models for assessing criteria pollutants and requires five years of data to capture variability in weather patterns from year to year. We follow the guideline for air toxics modeling also; however, because dispersion model runtimes using five

[1] An explanation of the updates from the previous version of AERMOD can be at https://www.epa.gov/scram/air-quality-dispersion-modeling-preferred-and-recommended-models#aermod and corresponding updates to HEM can be found at https://www.epa.gov/fera/human-exposure-model-hem-3-users-guides.

years of meteorological data would be too long for RTR source categories with many sources, we model only a single year. While the selection of a single year may result in under-prediction of long-term ambient levels at some locations, it may result in over-prediction at others. The sensitivity of model results to the selection of the nearest weather station and the use of one year of meteorological data is discussed in "Risk and Technology Review (RTR) Risk Assessment Methodologies" (USEPA 2009a).

We use the AERMET meteorological data preprocessor and the Automated Surface Observing System (ASOS) surface data and Forecast Systems Laboratory (FSL) upper air data to generate nationwide surface and profile files for input into AERMOD. In 2016, the Agency released to the public on the EPA's Support Center for Regulatory Atmospheric Modeling (SCRAM) website both AERMET and AERMOD (version 16216r). Appendix 3 to this document (*Meteorological Data for HEM-3 Modeling*) provides a complete listing of meteorological stations and assumptions, along with further details used in processing the data through AERMET. EPA has posted the AERMET meteorological data (2016) used in this analysis on the EPA's Fate, Exposure, and Risk Analysis (FERA) website under the Human Exposure Model (HEM) page.

The HEM-3 modeling system estimates ambient concentrations at the geographic centroids of census blocks (using the 2010 Census) and at other receptor locations that can be specified by the user.[2] See Appendix 4 of this document (*Dispersion Model Receptor Revisions and Additions*) for a discussion of user receptors and centroid location changes specific to this source category. HEM-3 accounts for the effects of multiple facilities when estimating concentration impacts at each block centroid. We typically combine the impacts of all facilities within the same source category and assess chronic exposure and risk for all census blocks[3] with at least one resident (i.e., locations where people may reasonably be assumed to reside rather than receptor points at the fenceline of a facility). We then calculate ambient concentrations as the annual average of all estimated short-term (one-hour) concentrations at each block centroid. We do not consider possible future residential use of currently uninhabited areas.

To assess the potential impacts from short-term exposures, we estimate worst-case one-hour concentrations at the census block centroids and at points closer to the facility (using either the polar receptors or user-specified receptors) that represent locations where people may be present for short periods[4]. Note that this is in contrast to the development of ambient concentrations for evaluating long-term exposures, which we perform only for occupied census blocks. Since short-term emission rates are needed to screen for the potential for hazard via acute exposures, and since the emission data typically contain only annual emission totals, we generally apply the assumption to all source categories that the maximum

[2] We also estimate ambient concentrations for a grid of polar receptors that is specific to each facility, and these receptors are used to interpolate concentrations for census blocks in the outer part of the modeling domain, and for finding the maximum offsite concentrations.
[3] Census blocks, the finest resolution available in the census data, are typically comprised of approximately 50 people or about 20 households.
[4] Generally, we estimate these concentrations at locations no nearer than 100 meters from the center of the facility (note that for large facilities, this 100-meter ring could still contain locations inside the facility property).

one-hour emission rate from any source is ten times the average annual hourly emission rate for that source. However, source category-specific data may support the use of engineering judgement to determine a different acute emissions multiplier. Further information on the factor used to estimate short-term emissions for this source category is provided in Appendix 1, and further discussion of the acute risk assessment can be found in Section 2.4.

We determine census block elevations for HEM-3 nationally from the US Geological Survey 1/3 Arc Second National Elevation Dataset, which has a spatial resolution of about 10 meters. Each polar receptor is assigned the highest elevation of any census block in its neighborhood (all blocks closer to that polar receptor than any other polar receptor). If an elevation is not provided for an emission source, the model uses the average elevation of all polar receptors on the innermost polar ring. In addition to using receptor elevation to determine plume height, AERMOD adjusts the plume's flow if nearby elevated hills are expected to influence the wind patterns. For details on how hill heights are estimated and used in the AERMOD modeling, see Appendix 2 of this document.

2.3 Estimating chronic human inhalation exposure

We use the estimated annual average ambient air concentration of each HAP at each census block centroid or user-defined receptor as a surrogate for the lifetime inhalation exposure concentration of all the people who reside in the census block. The risk assessment does not consider either the short-term or long-term behavior (mobility) of the exposed populations and its potential influence on their exposure.

We do not address short-term human activity, including indoor air concentrations. Our experience with the National Air Toxics Assessment (NATA), which models daily human activity using EPA's HAPEM, suggests that given our current understanding of the ratio of exposure concentrations to ambient values, including short-term human activity in RTR analyses would, on average, reduce risk estimates by up to about 25 percent for particulate HAP and typically by much less for gaseous HAPs. To ensure the risk characterization is health protective, EPA risk assessors do not include this small potential reduction in exposure concentrations when calculating risks.

We do not address long-term migration or population growth or decrease over the 70-year modeling period. Instead, we assume that each person's predicted exposure is constant over the course of their lifetime, which is assumed to be 70 years. The assumption of not considering short- or long-term population mobility does not bias the estimate of the theoretical MIR (assumes a person stays in one location for 70 years) nor does it affect the estimate of cancer incidence since the total population number remains the same. It does, however, affect the shape of the distribution of individual risks across the affected population, shifting it toward higher estimated individual risks at the upper end and reducing the number of people estimated to be at lower risks, thereby increasing the estimated number of people at higher risk levels.

2.4 Acute risk screening and refined assessments

In establishing a scientifically defensible approach for the assessment of potential health risks due to acute exposures to HAP, we follow a similar approach to that for chronic health risk assessments under the residual risk program, in that we begin with a screening assessment and then, if appropriate, perform a refined assessment.

The approach for the acute health risk screening assessment is designed to eliminate from further consideration those facilities for which we have confidence that no acute adverse health effects of concern will occur. For this screening assessment, we use readily available data and conservative assumptions for emission rates, meteorology, and exposure location that, in combination, approximate a worst-case exposure.

The following are the steps we take and assumptions we make in the acute screening assessment:

- When available, we use peak 1-hour emission data obtained from data collection efforts or estimated based on the operating characteristics and engineering judgement of facility emission sources; otherwise, we use an emission adjustment factor. In some cases, a source category-specific emission adjustment factor is calculated using information from the source category and engineering judgement. In other cases, when category-specific information is not available, a default source category-specific emission factor of 10 is used. This default factor is based on an analysis using a short-term emissions data set from a number of sources located in Texas (originally reported on by Allen *et al.* 2004) (see Appendix 5 of this document, *Analysis of Data on Short-term Emission Rates Relative to Long-term Emission Rates*).
- We assume that the peak emissions occur at all emission points at the same time.
- For facilities with multiple emission points, 1-hour concentrations at each receptor are assumed to be the sum of the maximum concentrations due to each emission point, regardless of whether those maximum concentrations occurred during the same hour.
- Worst-case meteorology (from one year of local meteorology) is assumed to occur at the same time the peak emission rates occur. The recommended EPA local-scale dispersion model, AERMOD, is used for simulating atmospheric dispersion.
- A person is assumed to be located downwind at the point of maximum modeled impact during this same worst-case 1-hour period, but no nearer to the source than 100 meters.

As a result of this screening assessment, the maximum HAP concentration is compared to multiple acute dose-response values for the HAP being assessed to determine whether a possible acute health risk might exist. The acute dose-response values are described in section 2.6 of this report.

A facility will either be found to pose no potential acute health risks (i.e., it will "screen out") or will need to undergo a more refined assessment. When we identify levels of a HAP that exceed its acute health benchmarks, we perform a more refined assessment, if possible. For situations in which we have used engineering judgement to estimate emissions, a refinement may be to obtain facility-specific data on HAP emissions. Other refinements may include the

temporal pattern of emissions (number of working hours, batch vs continuous operation), the location of emission points, the boundaries of the facility, and/or the local meteorology. In some cases, all of these site-specific data are used to refine the assessment; in others, lesser amounts of site-specific data may be used to determine that acute exposures are not a concern, and significant additional data collection is not necessary. See Section 3 of this document for the approach used for this source category.

2.5 Multipathway human health risk assessment

Due to the potential for significant human health risks due to exposure via routes other than inhalation (e.g., ingestion), we determine whether any sources emit HAP known to be persistent and bioaccumulative in the environment (PB-HAP).[5] The set of PB-HAP compounds or compound classes initially identified for potential screening assessment (from EPA's Air Toxics Risk Assessment (ATRA) Library) included the following: cadmium compounds, chlordane, chlorinated dibenzodioxins and furans (dioxins), 1,1-dichloro-2,2-bis(p-chlorophenyl) ethylene (DDE), heptachlor, hexachlorobenzene, hexachlorocyclohexane, lead compounds, mercury compounds, methoxychlor, polychlorinated biphenyls (PCB), polycyclic organic matter (POM), toxaphene, and trifluralin. Of these, EPA identified cadmium compounds, dioxins, mercury compounds, lead, POM, as well as arsenic, as PB-HAP of primary concern, based on assessment of national emission totals, toxicity considerations, and bioaccumulation potential. We assess these six PB-HAP for human health risks due to non-inhalation exposure.

We use a tiered approach to evaluate emissions of these PB-HAP for potential non-inhalation risks. This approach is designed to eliminate from further consideration those facilities for which we have confidence that human health risks will not occur due to non-inhalation exposure to their PB-HAP emissions. The approach was developed for use with EPA's peer-reviewed Total Risk Integrated Methodology: Fate, Transport, and Ecological Exposure (TRIM.FaTE) model.

For each carcinogenic PB-HAP, we have derived a screening threshold emission rate at which the maximum excess lifetime cancer risk would be 1-in-1 million. For each PB-HAP that causes noncancer health effects, we have derived a screening threshold emission rate for which the maximum HQ would be 1. The ratio of facility emissions to the screening threshold emission rate is termed a "screening value;" facility emissions that exceed the screening threshold emission rate have a screening value greater than 1. A screening value greater than 1 in any of the tiered screening methods represents a high-end estimate of what the risk or hazard may be; it cannot be equated with a risk value or a HQ (or HI). For example, for a carcinogen, a screening value of 30 (i.e., facility emissions are 30 times above the screening threshold emission rate) means that we are confident that the cancer risk is lower than 30-in-1 million. Similarly, for a non-carcinogen, a screening value of 2 (i.e., facility emissions are 2 times above the screening threshold emission rate) can be interpreted to mean that we are

[5] Although the two-letter chemical symbol for lead is Pb, in this assessment PB-HAP refers to the many air pollutants known to be persistent and bioaccumulative in the environment. When this report is specifically referring to lead, the term is spelled out (i.e., the two-letter chemical symbol for lead is not used in this document).

confident that the noncancer HQ would be lower than 2.

For Tier 1, 2, and 3 assessments, we use hypothetical exposure scenarios to assess whether non-inhalation exposures pose a potential human health risk. Exposure scenarios were developed to simulate generic gardening and subsistence farming and subsistence fishing lifestyles. Each screening exposure scenario is designed to represent the upper end of the range of possible exposure levels, such that it is a conservative but not impossible scenario. The exposure scenarios were developed for use in conjunction with the TRIM.FaTE model. These hypothetical exposure scenarios and associated ingestion exposure pathways are shown in Table 2.5-1.

Table 2.5-2. Multipathway Scenarios and Ingestion Pathways

Hypothetical Exposure Scenario	**Fish**	**Breast Milk** [a]	**Beef/Pork /Chicken**	**Dairy Milk**	**Eggs**	**Soil**	**Fruits and Vegetables** [b]
Combined Fisher and Farmer (Tier 1)	x	x	x	x	x	x	x
Fisher (Tier 2)	x	x					
Gardener (urban or rural) (Tier 2)		x			x	x	x
Farmer [c] (Tier 2)		x	x	x	x	x	x
Pollutants of Concern [d]	Hg, Cd, As, dioxin, POM	dioxin	As, dioxin, POM	As, dioxin, POM	As, dioxin, POM	As, dioxin, POM	As, dioxin, POM

[a] Health risks from the breast milk pathway are only associated with exposure to dioxins.
[b] Both protected and unprotected fruits and vegetables are included.
[c] This scenario may be included in a Tier 2 assessment in cases where we have site-specific data indicating that farming operations are present.
[d] The health endpoint for exposure to Hg (as methylmercury) and Cd is noncancer and the health endpoint for exposure to As (as inorganic arsenic), dioxin, and POM is cancer.

For the Tier 1 screening assessment, we determine whether the facility-specific emission rates for each emitted PB–HAP are high enough to create the potential for significant non-inhalation human health risks under reasonable worst-case conditions. We do this by comparing the facility-specific emission rates to the screening threshold emission rates for each PB-HAP for a hypothetical upper-end screening exposure scenario – the combined fisher and farmer scenario. The subsistence fisher scenario assumes a high-end fish consumption rate of 373 g/day for adults, a 99th percentile ingestion rate (Burger, 2002); fish consumption

rates for other age groups are presented in Appendix 6. The farmer scenario involves an individual that lives for a 70-year lifetime on a farm near the source and consumes produce grown, and meat and animal products raised, on the farm. The ingestion rates used for these food groups, and for incidental soil ingestion, are set at the 90th percentile of EPA's Exposure Factors Handbook: 2011 Edition (USEPA, 2011) and are considered upper-bound levels. The fisher and farmer exposure scenarios are combined for the Tier 1 TRIM.FaTE model application. See Appendix 6 (*Technical Support Document for TRIM-Based Multipathway Tiered Screening Methodology for RTR*) for a complete discussion of the development and testing of the screening scenario and the screening threshold emission rates.

For those facilities with PB-HAP emissions that exceed the Tier 1 screening threshold emission rate, we conduct a Tier 2 multipathway screening assessment. For the Tier 2 screening assessment, we refine the assessment by using the facility locations and considering two separate exposure scenarios – the fisher scenario and the home gardener scenario (rural or urban, as appropriate). In some cases, if supported by site-specific information, the subsistence farmer scenario is also considered. For each facility, we use the Tier 1 PB-HAP screening threshold emission rate, but with adjustments based on the ingested media and based on an understanding of how exposure concentrations estimated for the screening scenario change with use of the local meteorology and environmental assumptions. The gardener and fisher scenarios replace the Tier 1 combined fisher and farmer scenario as more likely exposure scenarios. The gardener scenario is only evaluated for carcinogenic PB-HAP (i.e., arsenic, dioxin, and POM) because the evaluated non-carcinogens (i.e., mercury and cadmium) accumulate in soil and the farm food chain in much smaller amounts than in fish tissue. For the gardener scenario, the Tier 1 PB-HAP screening threshold emission rates are adjusted to reflect exposure only through soil and farm foods, based on the rural/urban classification of the facility site (with urban gardeners growing and ingesting less home-grown produce than rural gardeners). The gardener scenarios (rural and urban) involve an individual that maintains a garden and consumes produce from this garden for 70 years at his/her residence. The evaluated locations of the gardener correspond to the maximum impacted residential receptor according to the RTR inhalation cancer assessment for each of the 8 wind octants (N, NE, E, SE, …) for all carcinogenic HAPs combined. The screening threshold emission rate can be different at each of these gardener locations, based on distance from the facility and based on local meteorology conditions. The ingestion rates used for the food groups are set at the 90^{th} percentile and mean values for rural and urban, respectively, based on data from EPA's Exposure Factors Handbook: 2011 Edition (USEPA, 2011); both gardeners have incidental soil ingestion rates equal to those of the farmer. The largest of the gardener screening values is identified for each PB-HAP. The fisher scenario is conducted for all of the currently evaluated PB-HAP, whose Tier 1 PB-HAP screening threshold emission rates are adjusted to reflect exposure only through fish ingestion. For the Tier 2 assessment, to fulfill the adult ingestion rate for the fisher scenario, if needed, more than one lake may be included in the modeling in order to reach a cumulative total of 373 acres and achieve the 373-g/day fish ingestion rate. A complete discussion of the bioassay studies used to support the assumption that the biological productivity limitation of each lake is 1 gram of fish caught and consumed per acre of water per day is provided in Appendix 6 of this document. The screening threshold emission rate can be different at each lake location, based on distance from the facility and based on local meteorology conditions.

If we need to include more than one lake in the Tier 2 screening assessment to achieve the 373 g/day ingestion rate, we begin with the lake with the highest modeled chemical concentration of a given PB-HAP group and "fish" up to the lake's biological productivity. We then systematically proceed to other lakes based on concentration, until the 373 g/day target is met. A maximum travel radius of 50 km relative to the facility is used to maintain a realistic scenario for the fisher. The final Tier 2 screening result for the fisher can be expressed as the sum of the screening result from each lake that is fished (which is based on the amount of fish ingested from each lake multiplied by the chemical concentration in fish). If the highest-concentration lake is at least 373 acres in size, the adult fisher catches and consumes 373 g/day of fish from that lake. If the cumulative size of multiple visited lakes exceeds 373 acres, the model includes from the final lake only the amount of fish necessary to satisfy the ingestion rate (i.e., to reach 373 g/day). If the total acreage of lakes within 50 km is less than 373 acres, the screening result reflects a reduced ingestion rate based on the smaller lake acreage. The order of fished lakes for a facility follows the order of PB-HAP concentration in fish from highest to lowest based on the facility's emissions. However, the resulting screening value calculations described above also potentially consider chemical inputs from emissions from multiple facilities. If a fished lake for one facility ("Facility A") is also within 50 km of another facility ("Facility B") in the source category, then the lake receives chemical input from emissions from two facilities. The order of fished lakes for Facility A considers only Facility A's chemical inputs to the lake, but the final fisher screening values for Facility A include the summed chemical inputs of Facility A and Facility B. If that lake was also fished for the Facility B scenario, then the same process would be applied to Facility B.

The Tier 2 assessment yields a facility-specific screening value for each PB-HAP for the fisher scenario and for the gardener scenario. If information is available to identify subsistence farming operations, the Tier 2 assessment will also include a screening value for the farmer scenario. Tier 2 screening values are evaluated for the source category to determine whether further refined screening is necessary for those facilities that may pose a significant risk. A finding that a facility's emissions exceed the Tier 2 screening threshold emission rate does not necessarily mean that multipathway impacts are significant, only that we cannot rule out that possibility based on the results of the screening assessment. See Appendix 6 of this document for a complete discussion of the Tier 2 screening assessment.

For facilities for which the Tier 2 screening value(s) indicate a potential health risk to the public, we can conduct a Tier 3 multipathway screening assessment. The Tier 3 screening assessment has three individual stages; we progress through these stages until the facility's screening values indicate that the emissions are unlikely to pose health risks to the public, or until all three stages are complete.

The first stage of a Tier 3 screening assessment, the lake-assessment stage, is a refinement of the fisher scenario. We examine the fished lakes from Tier 2 and evaluate the existence, the potential purpose, the accessibility and fishability, and the suitability of the lakes for the models and methods used in the screening assessments. We do not reasonably expect a subsistence fisher to catch and consume fish from lakes or ponds that are for industrial or

wastewater disposal; are covered in thick plant growth (e.g., swamps or marshes); are clearly closed to public use; or no longer exist (i.e., filled or drained). TRIM.FaTE is not configured to model chemical processes and environmental fate and transport mechanisms in saltwater or brackish waters, nor is it configured to model the very large watersheds and water dynamics of rivers, bays or very large lakes (e.g., larger than 100,000 acres)[6]. We use aerial imagery and web inquires to evaluate whether any Tier 2 fished lakes meet these disqualifying criteria and, if so, remove those lakes from all future screening assessments. If we remove a lake from a facility's assessment, and the total acres of fished lakes drops below the target of 373 acres, we evaluate the previously unfished lake with the highest chemical concentration, and so on, until the sizes of the qualifying lakes collectively comprise at least 373 acres or all lakes have been evaluated. We then rerun the fisher screening scenario with the revised lake data set. If the PB-HAP emissions for a facility exceed the fisher screening threshold emission rate based on the revised lake data set, we can conduct the next stage of the Tier 3 screening assessment (i.e., the plume-rise screen); otherwise, the emissions are considered unlikely to pose significant health risks in the fisher scenario.

The second stage of a Tier 3 screening assessment, the plume-rise stage, is a refinement of the previously assessed scenarios (i.e., Tier 2 farmer, Tier 2 gardener, Tier 3 lake-assessment fisher) where emissions exceeded screening threshold emission rates. We use site-specific hourly meteorology and facility-specific emission-point characteristics to estimate the fraction of annual emissions that stay within TRIM.FaTE's mixing layer where exposure occurs (i.e., that do not exit the mixing layer). In Tiers 1 and 2, all chemicals are emitted inside the mixing layer and are available for ground-level exposure. In reality, meteorological conditions and emission-point characteristics can cause emissions occasionally to reach higher than the mixing layer. In TRIM.FaTE, any emissions exiting the mixing layer do not reenter the mixing layer, resulting in no ground-level exposure for those emissions. In this Tier 3 stage, we use thermodynamic equations with local hourly meteorology and facility stack parameters to calculate hourly plume-rise heights. The fraction of annual hours during which the plume-rise height is less than the mixing-layer height equals the fraction of annual emissions available for human exposure in the screening assessment. We calculate these fractions for the location of each fished lake and for each relevant garden because lakes and gardens can be in different directions from the facility; thus, these calculations are conditional on wind direction. The results of this stage of Tier 3 are revised fisher and/or gardener screening values for each relevant PB-HAP and facility, accounting for emissions deposited above the mixing layer. If the revised screening value still indicates potential health risks to the public, we can proceed to the final stage of the Tier 3 screening assessment (i.e., the time-series screen); otherwise, the PB-HAP emissions are considered unlikely to pose significant risks.

In the third and final stage of a Tier 3 screening assessment, the time-series assessment, we can conduct new runs of TRIM.FaTE for each relevant lake and/or garden location for a facility for every PB-HAP that represents a risk concern based upon the Tier 3 plume-rise assessment. For these model runs, we start with the screening configuration corresponding to

[6] Very large lakes and bays (i.e., those larger than 100,000 acres) are not included because their watersheds are too large and their lake dynamics are too complex to realistically model in the TRIM.FaTE system. Lakes and bays larger than 100,000 acres include the Great Lakes, the Great Salt Lake, Lake Okeechobee, Lake Pontchartrain, Lake Champlain, Green Bay, and Galveston Bay.

the lake and/or garden location, and we use site-specific hourly meteorology and the hourly plume-rise values calculated in the Tier 3 plume-rise assessment. Allowing TRIM.FaTE to model chemical fate and transport with hour-by-hour changes in meteorology and plume rise produces a more accurate estimate of chemical concentrations in media of interest, as compared to the static values used in Tier 2 and the post-processing adjustments made in the Tier 3 plume-rise assessment. If a facility's model-estimated PB-HAP screening-level cancer risk is below 1-in-1 million (or screening-level HQ is below 1 for non-carcinogens), the emissions are considered unlikely to pose significant risks.

If a facility's PB-HAP Tier 3 screening results still indicate a potential health risk to the public and data are available, we may elect to conduct a more refined multipathway assessment. A refined assessment replaces some of the assumptions made in the screening with site-specific data. The refined assessment also uses the TRIM.FaTE model and facility-specific emission rates for each PB-HAP. Many variables are available to consider in a refined multipathway assessment, and we have developed a protocol to maintain consistency across source categories. This protocol can be found in Appendix 7 of this document (*Protocol for Site-Specific Multipathway Risk Assessment*) and details of the site-specific multipathway assessment can be found in Appendix 11 of this document (*Site-Specific Human Health Multipathway Residual Risk Assessment Report*).

Lead

We take a different approach for assessing lead compounds. In evaluating the potential multipathway risks from emissions of lead compounds, rather than developing a screening emission rate for them, we compare maximum estimated chronic atmospheric concentrations with the current national ambient air quality standard (NAAQS) for lead. Values below the NAAQS are considered to have a low potential for multipathway risks.

The NAAQS value, a public health policy judgment, incorporates the Agency's most recent health evaluation of air effects of lead exposure for the purposes of setting a national ambient air quality standard. In setting this value, the Administrator promulgated a standard that was requisite to protect public health with an adequate margin of safety. We consider values below the level of the primary NAAQS to protect against multipathway risks because, as mentioned above, the primary NAAQS is set to protect public health with an adequate margin of safety. However, ambient air lead concentrations above the NAAQS are considered to pose the potential for increased risk to public health. We consider this NAAQS assessment to be a refined analysis given: 1) the numerous health studies, detailed risk and exposure analyses, and level of external peer and public review that went into the development of the primary NAAQS for lead, combined with 2) the site-specific dispersion modeling used in this assessment to estimated ambient lead concentrations due to the source category emissions. It should be noted, however, that this comparison does not account for possible population exposures to lead from sources other than the one being modeled; for example, via consumption of water from untreated local sources or ingestion of locally grown food. Nevertheless, the Administrator judged that such a standard would protect, with an adequate margin of safety, the health of children and other at-risk populations against an array of adverse health effects, most notably including neurological effects, particularly neurobehavioral and neurocognitive effects, in children (73 FR 67007). The Administrator,

in setting the standard, also recognized that no evidence or risk-based bright line indicated a single appropriate level. Instead, a collection of scientific evidence and other information was used to select the standard from a range of reasonable values (73 FR 67006).

We further note that comparing ambient lead concentrations to the NAAQS for lead, considering the level, averaging time, form and indicator, also informs whether there is the potential for adverse environmental effects. This is because the secondary lead NAAQS, set to protect against adverse welfare effects (including adverse environmental effects), has the same averaging time, form, and level as the primary standard. Thus, ambient lead concentrations above the NAAQS for lead also indicate the potential for adverse environmental effects.

2.6 Environmental risk assessment

The EPA has developed a screening approach to examine the potential for adverse environmental effects, as required under section 112(f)(2)(A) of the CAA. The environmental screening assessment focuses on the following eight environmental HAP:

- Six persistent bioaccumulative HAP (PB-HAP) – cadmium, dioxins, POM, mercury (both inorganic mercury and methylmercury), arsenic, and lead;
- Two acid gases – hydrochloric acid (HCl) and hydrofluoric acid (HF).

HAP that persist and bioaccumulate are of particular environmental concern because they accumulate in the soil, sediment, and water. The acid gases – HCl and HF – were included due to their well-documented potential to cause direct damage to terrestrial plants. See Appendix 9 of this document (*Environmental Risk Screening Assessment*) for a more detailed discussion of the environmental risk screening assessment.

For the environmental risk screening assessment, EPA first determines whether any facilities in the source category emit any of the eight environmental HAP. If one or more of the environmental HAP are emitted by at least one facility in the source category, we proceed to the second step of the environmental risk screening assessment.

For cadmium, mercury, POM, arsenic, and dioxins, the environmental screening assessment consists of the same three tiers used in the multipathway human health risk assessment (see Section 2.5). In the first tier, the same TRIM.FaTE modeling used in human health risk assessment is conducted, using reasonable worst-case environmental conditions to identify screening threshold emission rates corresponding to ecological benchmarks for soil, fish, surface water, and sediment. For each facility and PB-HAP, facility emissions are compared to these screening threshold emission rates to determine the potential for significant impacts on off-site ecological receptors. The ratio of facility emissions to the screening threshold emission rate is termed a "screening value." Facility emissions that exceed the screening threshold emission rate have a screening value greater than 1, and risks above levels of concern for ecological receptors are possible. Screening values below 1 indicate that risks to ecological receptors are likely below levels of concern.

For those facilities with PB-HAP emissions that exceed a Tier 1 screening threshold emission rate, we conduct a Tier 2 screening assessment. In Tier 2, the Tier 1 screening threshold emission rates are adjusted to account for local meteorology and environmental assumptions. For lake-related ecological receptors, actual locations of lakes within 50 km of the facility are identified, and the screening threshold emission rate can be different at each lake location based on distance from the facility and based on local meteorology conditions. After the screening value (i.e., ratio of facility emissions to screening threshold emission rate) is calculated at each lake, the largest screening value is identified. Screening threshold emission rates for soil receptors are evaluated at many locations surrounding the facility and are also impacted by distance from facility and local meteorology. For soil receptors in Tier 2, we are interested in the overall average screening value across all soil receptors (for a given facility and PB-HAP), and we are also interested in the total area in the vicinity of the facility where screening values are above 1 (for a given facility and PB-HAP). If a lake-related screening value is above 1, or the soil screening value is above 1 at any location, or the overall average soil screening value is above 1, it does not necessarily mean that the ecological effects are significant, but only that we cannot rule out that possibility. For facilities with Tier 2 screening values above 1, we can evaluate their emissions further in Tier 3.

Like in the multipathway human health risk assessment, in Tier 3 of the environmental screening assessment, we examine the suitability of the lakes around the facilities to support life and remove those that are not (e.g., lakes that have been filled in or are industrial ponds), adjust emissions for plume-rise, and conduct hour-by-hour time-series assessments. For the lake assessment, we remove from the screening any lakes that appear to be industrial, for wastewater disposal, or no longer exist. TRIM.FaTE is not configured to model chemical processes and environmental fate and transport mechanisms in saltwater or brackish waters, nor is it configured to model the very large watersheds and water dynamics of rivers or very large lakes (e.g., larger than 100,000 acres); these types of water bodies are also removed from the screening assessment. Unlike the multipathway human health risk assessment, we assume that if lakes that are swampy or are not publicly accessible, they still can support ecological life and some animals will still eat from them. After lakes are removed that meet these disqualifying criteria, lake-related receptors are rescreened. For the plume-rise assessment, as in the human health assessment, we adjust the facility's previously calculated screening value based on the fraction of facility emissions that remain in the mixing layer where exposure occurs, after accounting for plume rise (which is based on site-specific meteorology and facility-specific emission-point characteristics). If these Tier 3 adjustments still indicate that ecological risks could be above levels of concern (i.e., screening values are above 1), as in the human health assessment, we can conduct new TRIM.FaTE modeling using the screening configuration corresponding to the relevant lake and/or soil locations, site-specific hourly meteorology, and hourly plume-rise values. If such modeling results in screening-level media concentrations or doses above benchmark levels, we may elect to conduct a more refined assessment using more site-specific information. If, after additional refinement, the media concentrations or doses are above benchmark levels, the facility may have the potential to cause adverse environmental effects.

For acid gases, the environmental screening assessment evaluates the potential phytotoxicity and reduced productivity of plants due to chronic exposure to acid gases. The environmental risk screening methodology for acid gases is a single-tier screening assessment that compares the average off-site ambient air concentration over the modeling domain to ecological benchmarks for each of the acid gases. For purposes of an ecological risk screening assessment, EPA identifies a potential for adverse environmental effects to plant communities from exposure to acid gases when the average off-site ambient air concentration over the modeling domain for a facility exceeds the ecological benchmark for that acid gas. In such cases, we further investigate factors such as the magnitude of the exceedance and the characteristics of the area of exceedance (e.g., land use of exceedance area, size of exceedance area) to determine whether the facility's emissions have the potential to cause adverse environmental effects.

Lead

For lead compounds, we currently do not have the ability to calculate media concentrations using the TRIM.FaTE model. However, air concentrations of lead are already calculated as part of the human health exposure and risk assessment using HEM-3. To evaluate the potential for adverse environmental effects from lead, we compare the average modeled air concentrations of lead around each facility in the source category to the level of the secondary NAAQS for lead. The secondary lead NAAQS is a reasonable means of evaluating environmental risk because it is set to provide substantial protection against adverse welfare effects which can include "effects on soils, water, crops, vegetation, man-made materials, animals, wildlife, weather, visibility and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being." [7] We investigate any modeled exceedances of the lead NAAQS in a manner similar to that noted above for acid gases.

2.7 Dose-response assessment

2.7.1 Sources of chronic dose-response information

Dose-response assessments (carcinogenic and non-carcinogenic) for chronic exposure (either by inhalation or ingestion) for the HAP reported in the emissions inventory for this source category are based on the EPA Office of Air Quality Planning and Standards' (OAQPS) existing recommendations for HAP (USEPA, 2014a). This information has been obtained from various sources and prioritized according to (1) conceptual consistency with EPA risk assessment guidelines and (2) level of peer review received. The prioritization process was aimed at incorporating into our assessments the best available science with respect to dose-

[7] A secondary standard, as defined in Section 109(b)(2), must "specify a level of air quality the attainment and maintenance of which, in the judgment of the Administrator, based on criteria, is requisite to protect the public welfare from any known or anticipated adverse effects associated with the presence of [the] pollutant in the ambient air." Welfare effects as defined in section 302(h) (42 U.S.C. 7602(h)) include, but are not limited to, "effects on soils, water, crops, vegetation, man-made materials, animals, wildlife, weather, visibility and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being."

response information. The recommendations are based on the following sources, in order of priority:

1) **U.S. Environmental Protection Agency (EPA).** EPA has developed dose-response assessments for chronic exposure for many HAP. These assessments typically provide a qualitative statement regarding the strength of scientific data and specify a reference concentration (RfC, for inhalation) or reference dose (RfD, for ingestion) to protect against effects other than cancer and/or a unit risk estimate (URE, for inhalation) or slope factor (SF, for ingestion) to estimate the probability of developing cancer. The RfC is defined as an "estimate (with uncertainty spanning perhaps an order of magnitude) of a continuous inhalation exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime." The RfD is "an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime." The URE is defined as "the upper-bound excess cancer risk estimated to result from continuous lifetime exposure to an agent at a concentration of 1 µg/m^3 in air." The SF is "an upper bound, approximating a 95 percent confidence limit, on the increased cancer risk from a lifetime exposure to an agent. This estimate, [is] usually expressed in units of proportion (of a population) affected per mg/kg-day…"

   EPA disseminates dose-response assessment information in several forms, based on the level of review. The Integrated Risk Information System (IRIS) is an EPA database that contains scientific health assessment information, including dose-response information. All IRIS assessments since 1996 have also undergone independent external peer review. The current IRIS process includes review by EPA scientists, interagency reviewers from other federal agencies, and the public, as well as peer review by independent scientists external to EPA. New IRIS values are developed and old IRIS values are updated as new health effects data become available. Refer to the IRIS Agenda for detailed information on status and scheduling of current individual IRIS assessments and updates. EPA's science policy approach, under the current carcinogen guidelines, is to use linear low-dose extrapolation as a default option for carcinogens for which the mode of action (MOA) has not been identified. We expect future EPA dose-response assessments to identify nonlinear MOAs where appropriate, and we will use those analyses (once they are peer reviewed) in our risk assessments. At this time, however, there are no available carcinogen dose-response assessments for inhalation exposure that are based on a nonlinear MOA.

2) **U.S. Agency for Toxic Substances and Disease Registry (ATSDR).** ATSDR, which is part of the US Department of Health and Human Services, develops and publishes Minimal Risk Levels (MRLs) for inhalation and oral exposure to many toxic substances. As stated on the ATSDR web site: "Following discussions with scientists within the Department of Health and Human Services (HHS) and the EPA, ATSDR chose to adopt a practice similar to that of the EPA's Reference Dose (RfD) and Reference Concentration (RfC) for deriving substance specific health guidance levels

for non-neoplastic endpoints." The MRL is defined as "an estimate of daily human exposure to a substance that is likely to be without an appreciable risk of adverse effects (other than cancer) over a specified duration of exposure." ATSDR describes MRLs as substance-specific estimates to be used by health assessors to select environmental contaminants for further evaluation.

3) **California Environmental Protection Agency (CalEPA).** The CalEPA Office of Environmental Health Hazard Assessment has developed dose-response assessments for many substances, based both on carcinogenicity and health effects other than cancer. The process for developing these assessments is similar to that used by EPA to develop IRIS values and incorporates significant external scientific peer review. As stated in the CalEPA Technical Support Document for developing their chronic assessments, the guidelines for developing chronic inhalation exposure levels incorporate many recommendations of the U.S. EPA (USEPA, 1994a) and NAS (NAS, 1994). The noncancer information includes available inhalation health risk guidance values expressed as chronic inhalation reference exposure levels (RELs). CalEPA defines the REL as "the concentration level at or below which no health effects are anticipated in the general human population." CalEPA's quantitative dose-response information on carcinogenicity by inhalation exposure is expressed in terms of the URE, defined similarly to EPA's URE.

For certain HAP, the dose-response information, based on this prioritization, is limited. To address data gaps, increase accuracy, and avoid underestimating risk, we made additional changes to some of the chronic inhalation exposure values. These important changes, outlined below and reflected in Appendix 8 (*Dose-Response Values Used in the RTR Risk Assessments*) to this document, are as follows:

1) **Manganese**. The EPA considers the ATSDR MRL for manganese (Mn) the most appropriate chronic inhalation reference value to be used in RTR assessments. There is an existing IRIS RfC for Mn (USEPA, 1993a), and ATSDR published an assessment of Mn toxicity which includes a chronic inhalation reference value (i.e., an ATSDR Minimal Risk Level, MRL). (ATSDR, 2012). Both the 1993 IRIS RfC and the 2012 ATSDR MRL were based on the same study (Roels et al., 1992); however, ATSDR used updated dose-response modeling methodology (benchmark dose approach) and considered recent pharmacokinetic findings to support their MRL derivation. Because of the updated methods, EPA has determined that the ATSDR MRL is the appropriate health reference value to use in RTR risk assessments.

2) **Polycyclic Organic Matter.** EPA has identified appropriate UREs for many individual compounds of POM, published in the sources used for RTR risk assessments. When an individual POM compound is reported in the emission inventory for the source category, we use the appropriate URE for that compound. However, if in the emission inventory for the source category a POM compound is reported for which EPA has not identified a URE, or when POM are not speciated into individual compounds, then EPA applies simplifying assumptions so that cancer risk can be quantitatively evaluated without substantially under- or over-estimating risk

(which can occur if all reported POM emissions were assigned the same URE). To accomplish this, EPA places each POM compound into one of eight POM groups, generally defined by toxicity and the estimated emission profile of POM compounds. POM Groups 1 and 2 include unspeciated POM (emissions reported as "polycyclic organic matter") and individual POM compounds with no URE assigned from the sources used in RTR risk assessments. With two exceptions, both Groups 1 and 2 are assigned a URE equal to 5 percent of that for pure benzo[a]pyrene; the two exceptions are benzo[a]fluoranthene and generic "benzofluoranthenes", which received the URE of benzo[b]fluoranthene. POM Groups 3 through 7 comprise POM compounds for which UREs are available from the sources used for RTR risk assessments, except for benzo[b+k]fluoranthene and benzo[g,h,i]fluoranthene which receive the URE of benzo[b]fluoranthene. If reported emissions are for a specific compound in these groups, then EPA evaluates the cancer risk of the compound using its unique URE if one has been derived or its group URE if one has not been specifically derived. If the reported emissions are for a specific POM group rather than a compound within the group, then EPA evaluates the cancer risk of the POM group using a URE value that is close to the average of the UREs of the individual compounds within the group. POM Group 8 is composed of unspeciated polycyclic aromatic hydrocarbons (PAH) reported as 7-PAH and are assigned a URE equal to approximately 18 percent of that for pure benzo[a]pyrene. In addition, we have concluded that three PAHs—anthracene, phenanthrene and pyrene—are not carcinogenic and therefore no URE is assigned. Details of the analysis that led to this conclusion can be found in the document titled Development of a Relative Potency Factor (RPF) Approach for Polycyclic Aromatic Hydrocarbon (PAH) Mixtures: In Support of Summary Information of the Integrated Risk Information System (IRIS).

3) **Glycol Ethers.** Often in an emission inventory, the glycol ethers are reported only as the total mass for the entire group without distinguishing among individual glycol ether compounds. In other cases, emissions of individual glycol ether compounds that had not been assigned dose-response values were reported. To avoid underestimating the health hazard associated with glycol ethers, we protectively apply the RfC for ethylene glycol methyl ether (the most toxic glycol ether for which an assessment exists) to glycol ether emissions of unspecified composition.

4) **Lead.** We consider the primary NAAQS for lead, which incorporates an adequate margin of safety, to be protective of all potential health effects for the most susceptible populations. The NAAQS, developed using the EPA Integrated Exposure, Uptake, Biokinetic Model, was preferred over the RfC for noncancer adverse effects because the NAAQS for lead was developed using more recent toxicity and dose-response information on the noncancer adverse impacts of lead. The NAAQS for lead was set to protect the health of the most susceptible children and other potentially at-risk populations against an array of adverse health effects, most notably including neurological effects, particularly neurobehavioral and neurocognitive effects (which are the effects to which children are most sensitive). The lead NAAQS, a rolling 3-month average level of lead in total suspended particles, is used as a long-term value in the RTR risk assessment.

5) **Nickel compounds.** To provide a conservative estimate of the potential cancer risks, the EPA considers the IRIS URE value for nickel subsulfide (which is considered the most potent carcinogen among all nickel compounds) to be the most appropriate value to be used in RTR assessments. Based on consistent views of major scientific bodies, such as the National Toxicology Program (NTP) in their 14th Report of the Carcinogens (RoC) (NTP, 2016), the International Agency for Research on Cancer (IARC, 1990), and other international agencies (WHO, 1991) that consider all nickel compounds to be carcinogenic, we currently consider all nickel compounds to have the potential of being carcinogenic to humans. The 14th RoC states that "the combined results of epidemiological studies, mechanistic studies, and carcinogenic studies in rodents support the concept that nickel compounds generate nickel ions in target cells at sites critical for carcinogenesis, thus allowing consideration and evaluation of these compounds as a single group." Although the precise nickel compound (or compounds) responsible for carcinogenic effects in humans is not always clear, studies indicate that nickel sulfate and the combinations of nickel sulfides and oxides encountered in industrial emissions of nickel mixtures cause cancer in humans (these studies are summarized in a review by Grimsrud et al., 2010). The major scientific bodies mentioned above have also recognized that there may be differences in the toxicity and/or carcinogenic potential across the different nickel compounds. For this reason, and given that there are two additional URE values[8] derived for exposure to mixtures of nickel compounds (as a group) that are 2-3 fold lower than the IRIS URE for nickel subsulfide, the EPA considers it reasonable, in some instances (e.g., when high quality data are available on the composition of nickel emissions from a specific source category), to use a value that is 50 percent of the IRIS URE for nickel subsulfide for providing an estimate of the lower end of the plausible range of cancer potency values for different mixtures of nickel compounds.

6) **Carbonyl Sulfide.** Although the health effects data for carbonyl sulfide (COS) are very limited, a series of studies (Morgan et. al., 2004; Herr et. al., 2007; Sills et. al., 2004) conducted by the National Toxicology Program have shown that the major concern regarding exposure to COS is its potential for neurotoxicity. These studies have shown consistently and at the same range of COS concentrations that the brain is a target organ for COS toxicity. Since appropriate health effects benchmarks have not been derived by our preferred sources of dose-response data including IRIS, ATSDR, and Cal EPA, the EPA has used the data from the above referenced studies to derive a chronic screening benchmark level for COS. A chronic screening level of 163 $\mu g/m^3$ was developed for COS from a No Observed Adverse Effects Level (NOAEL) of 200 ppm based on brain lesions and neurophysiological alterations in rodents. Additional details on the derivation of the chronic screening level for COS can be found in Appendix 8.

[8] Two UREs (other than the current IRIS values) have been derived for nickel compounds as a group: one developed by the California Department of Health Services (http://www.arb.ca.gov/toxics/id/summary/nickel_tech_b.pdf) and the other by the Texas Commission on Environmental Quality (http://www.tceq.texas.gov/assets/public/implementation/tox/dsd/facts/nickel_&_compounds.pdf).

7) **Pollutant Groups.** In the case of HAP groups such as cyanide compounds, mercury compounds, antimony compounds and others, the most conservative dose-response value in the chemical group is used as a surrogate for other compounds in the group for which dose-response values are not available. This is done to examine, under conservative assumptions, whether those HAP that lack dose-response values may pose an unacceptable risk and require further examination.

8) **Mutagenic Mode of Action.** For carcinogenic chemicals acting via a mutagenic mode of action (i.e., chemicals that cause cancer by damaging genes), we estimate risks to reflect the increased carcinogenicity of such chemicals during childhood. This approach is explained in detail in the Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens. Where available data do not support a chemical-specific evaluation of differences between adults and children, the Supplemental Guidance recommends using the following default adjustment factors for early-life exposures: increase the carcinogenic potency by 10-fold for children up to 2 years old and by 3-fold for children 2 to 15 years old. These adjustments have the aggregate effects of increasing by about 60 percent the estimated risk (a 1.6-fold increase) for a lifetime of constant inhalation exposure. EPA uses these default adjustments only for carcinogens known to be mutagenic for which data to evaluate adult and juvenile differences in toxicity are not available. The UREs for several HAP (see Appendix 8) were adjusted upward, by multiplying by a factor of 1.6, to account for the increased risk during childhood exposures. Although trichloroethylene is carcinogenic by a mutagenic mode of action, the age-dependent adjustment factor for the URE only applies to the portion of the slope factor reflecting risk of kidney cancer. For full lifetime exposure to a constant level of trichloroethylene exposure, the URE is adjusted upward by a factor of 1.12 (rather than 1.6 as discussed above). For more information on applying age-dependent adjustment factors in cases where exposure varies over the lifetime, see Toxicological Review of Trichloroethylene. The URE for vinyl chloride includes exposure from birth, although the IRIS assessment contains UREs for both exposure from birth and exposure during adulthood. This value already accounts for childhood exposure; thus, no additional factor is applied.

2.7.2 Sources of acute dose-response information

Hazard identification and dose-response assessment information for preliminary acute inhalation exposure assessments is based on the existing recommendations of OAQPS for HAP (USEPA, 2014b). When the benchmarks are available, the results from acute screening assessments are compared to both “no effects” reference levels for the general public, such as the California Reference Exposure Levels (RELs), and to emergency response levels, such as Acute Exposure Guideline Levels (AEGLs) and Emergency Response Planning Guidelines (ERPGs), with the recognition that the ultimate interpretation of any potential risks associated with an estimated exceedance of a particular reference level depends on the definition of that level and any limitations expressed therein. Comparisons among different available inhalation health effect reference values (both acute and chronic) for selected HAP can be found in an EPA document of graphical arrays (USEPA, 2009b).

California Acute Reference Exposure Levels (RELs). The California Environmental Protection Agency (CalEPA) has developed acute dose-response reference values for many substances, expressing the results as acute inhalation RELs.

> The acute REL is defined by CalEPA as "the concentration level at or below which no adverse health effects are anticipated for a specified exposure duration (OEHHA, 2015). RELs are based on the most sensitive, relevant, adverse health effect reported in the medical and toxicological literature. RELs are designed to protect the most sensitive individuals in the population by the inclusion of margins of safety. Since margins of safety are incorporated to address data gaps and uncertainties, exceeding the REL does not automatically indicate an adverse health impact." Acute RELs are developed for 1-hour (and 8-hour) exposures. The values incorporate uncertainty factors similar to those used in deriving EPA's inhalation RfCs for chronic exposures.

Acute Exposure Guideline Levels (AEGLs). AEGLs are developed by the National Advisory Committee (NAC) on Acute Exposure Guideline Levels (NAC/AEGL) for Hazardous Substances and then reviewed and published by the National Research Council. As described in the Committee's Standing Operating Procedures, AEGLs "represent threshold exposure limits for the general public and are applicable to emergency exposures ranging from 10 min to 8 h." Their intended application is "for conducting risk assessments to aid in the development of emergency preparedness and prevention plans, as well as real time emergency response actions, for accidental chemical releases at fixed facilities and from transport carriers." The document states that "the primary purpose of the AEGL program and the NAC/AEGL Committee is to develop guideline levels for once-in-a-lifetime, short-term exposures to airborne concentrations of acutely toxic, high-priority chemicals." In detailing the intended application of AEGL values, the document states, "It is anticipated that the AEGL values will be used for regulatory and nonregulatory purposes by U.S. Federal and State agencies, and possibly the international community in conjunction with chemical emergency response, planning, and prevention programs. More specifically, the AEGL values will be used for conducting various risk assessments to aid in the development of emergency preparedness and prevention plans, as well as real-time emergency response actions, for accidental chemical releases at fixed facilities and from transport carriers."

The NAC/AEGL defines AEGL-1 and AEGL-2 as:

> "AEGL-1 is the airborne concentration (expressed as ppm or mg/m^3) of a substance above which it is predicted that the general population, including susceptible individuals, could experience notable discomfort, irritation, or certain asymptomatic nonsensory effects. However, the effects are not disabling and are transient and reversible upon cessation of exposure."

> "AEGL-2 is the airborne concentration (expressed as ppm or mg/m^3) of a substance above which it is predicted that the general population, including susceptible individuals, could experience irreversible or other serious, long-lasting adverse health effects or an impaired ability to escape."

> "Airborne concentrations above AEGL-1 represent exposure levels that can produce mild and progressively increasing but transient and nondisabling odor, taste, and sensory irritation or certain asymptomatic, nonsensory effects. With increasing airborne concentrations above each AEGL, there is a progressive increase in the likelihood of occurrence and the severity of effects described for each corresponding AEGL. Although the AEGL values represent threshold levels for the general public, including susceptible subpopulations, such as infants, children, the elderly, persons with asthma, and those with other illnesses, it is recognized that individuals, subject to unique or idiosyncratic responses, could experience the effects described at concentrations below the corresponding AEGL."

Emergency Response Planning Guidelines (ERPGs). The American Industrial Hygiene Association (AIHA) has developed ERPGs for acute exposures at three different levels of severity. These guidelines represent concentrations for exposure of the general population (but not particularly sensitive persons) for up to 1 hour associated with effects expected to be mild or transient (ERPG-1), irreversible or serious (ERPG-2), and potentially life-threatening (ERPG-3).

ERPG values are described in their supporting documentation as follows: "ERPGs are air concentration guidelines for single exposures to agents and are intended for use as tools to assess the adequacy of accident prevention and emergency response plans, including transportation emergency planning, community emergency response plans, and incident prevention and mitigation."

ERPG-1 and ERPG-2 values are defined by AIHA's Standard Operating Procedures as follows:

> "ERPG-1 is the maximum airborne concentration below which nearly all individuals could be exposed for up to 1 hour without experiencing more than mild, transient health effects or without perceiving a clearly defined objectionable odor."

> "ERPG-2 is the maximum airborne concentration below which nearly all individuals could be exposed for up to 1 hour without experiencing or developing irreversible or other serious adverse health effects or symptoms that could impair an individual's ability to take protective action."

2.8 Risk characterization

2.8.1 General

The final product of the risk assessment is the risk characterization, in which the information from the previous steps is integrated and an overall conclusion about risk is synthesized that is complete, informative, and useful for decision makers. In general, the nature of this risk characterization depends on the information available, the application of the risk information and the resources available. In all cases, major issues associated with determining the nature

and extent of the risk are identified and discussed. Further, it is EPA's policy that a risk characterization be prepared in a manner that is clear, transparent, reasonable, and consistent with other risk characterizations of similar scope prepared across programs in the Agency. These principles of transparency and consistency have been reinforced by the Agency's *Risk Characterization Handbook* (USEPA, 2000a), in the Agency's information quality guidelines (USEPA, 2002a), and in the Office of Management and Budget (OMB) Memorandum on Updated Principles for Risk Analysis (OMB, 2007), and they are incorporated in these assessments.

Estimates of health risk are presented in the context of uncertainties and limitations in the data and methodology. Through our tiered, iterative analytical approach, we have attempted to reduce both uncertainty and bias to the greatest degree possible in these assessments, within the limitations of available time and resources. We provide summaries of risk metrics (including maximum individual cancer risks and noncancer hazards, as well as cancer incidence estimates) along with a discussion of the major uncertainties associated with their derivation to provide decision makers with the fullest picture of the assessment and its limitations.

For each carcinogenic HAP included in an assessment for which a potency estimate is available, individual and population cancer risks are calculated by multiplying the corresponding lifetime average exposure estimate by the appropriate URE. This calculated cancer risk is defined as the upper-bound probability of developing cancer over a 70-year period (i.e., the assumed human lifespan) at that exposure. Because UREs for most HAP are upper-bound estimates, actual risks at a given exposure level may be lower than predicted.

Increased cancer incidence for the entire population within the area of analysis is estimated by multiplying the estimated lifetime cancer risk for each census block by the number of people residing in that block, then summing the results for the entire modeled domain. This lifetime population incidence estimate is divided by 70 years to obtain an estimate of the number of cancer cases per year.

Unlike linear dose-response assessments for cancer, noncancer health hazards generally are not expressed as a probability of an adverse occurrence. Instead, the estimated human health risk for noncancer effects is expressed by comparing an exposure to a reference level as a ratio. The hazard quotient (HQ) is the estimated exposure divided by a reference level (e.g., the RfC). For a given HAP, exposures at or below the reference level ($HQ \leq 1$) are not likely to cause adverse health effects. As exposures increase above the reference level (HQs increasingly greater than 1), the potential for adverse effects increases. For exposures predicted to be above the RfC, the risk characterization includes the degree of confidence ascribed to the RfC values for the compound(s) of concern (i.e., high, medium, or low confidence) and discusses the impact of this on possible health interpretations.

The risk characterization for chronic effects other than cancer is developed using the HQ for inhalation, calculated for each HAP at each census block centroid. As discussed above, RfCs incorporate generally conservative uncertainty factors in the face of uncertain extrapolations, such that an HQ greater than 1 does not necessarily suggest the onset of adverse effects. The

hazard index (HI) is the sum of hazard quotients for substances that affect the same target organ or organ system and is an approximation of the aggregate effect on a specific target organ (e.g., the lungs). The HQ and HI cannot be translated to a probability that adverse effects will occur, and it is unlikely to be proportional to adverse health effect outcomes in a population.

Screening for potentially significant acute inhalation exposures also follows the HQ approach. We divide the maximum estimated acute exposure by each available acute dose-response value to develop an array of HQs. In general, when none of these HQs is greater than one, there is no potential for acute risk. When one or more HQ is above 1, we evaluate additional information (e.g., proximity of the facility to potential exposure locations) to determine whether there is a potential for significant acute risks.

2.8.2 Mixtures

Since most or all receptors in these assessments receive exposures to multiple pollutants rather than a single pollutant, we estimate the aggregate health risks associated with exposure to all of the HAP from a particular source category.

To combine risks across multiple carcinogens, our assessments use the mixtures guidelines' default assumption of additivity of effects and combine risks by summing them using the independence formula in the mixtures guidelines (USEPA, 1986; USEPA, 2000b).

In assessing noncancer hazard from chronic exposures, in cases where different pollutants cause adverse health effects via completely different modes of action, it may be inappropriate to aggregate HQs. In consideration of these mode-of-action differences, the mixtures guidelines support aggregating effects of different substances in specific and limited ways. To conform to these guidelines, we aggregate noncancer HQs of HAP that act by similar toxic modes of action, or (where this information is absent) that affect the same target organ. This process creates, for each target organ, a target-organ-specific hazard index (TOSHI), defined as the sum of HQs for individual HAP that affect the same organ or organ system. For the RTRs, TOSHI calculations are based exclusively on effects occurring at the "critical dose" (i.e., the lowest dose that produces adverse health effects). Although HQs associated with some pollutants have been aggregated into more than one TOSHI, this has been done only in cases where the critical dose affects more than one target organ. Because impacts on organs or systems that occur above the critical dose have not been included in the TOSHI calculations, some TOSHIs may have been underestimated. As with the HQ, the TOSHI should not be interpreted as a probability of adverse effects or as strict delineation of "safe" and "unsafe" levels. Rather, the TOSHI is another measure of the potential for adverse health outcomes associated with pollutant exposure and needs to be interpreted carefully by health scientists and risk managers.

Because of the conservative nature of the acute inhalation screening assessment and the variable nature of emissions and potential exposures, acute impacts are screened on an individual pollutant basis, not using the TOSHI approach.

3 Risk results for the Surface Coating of Metal Coil source category

3.1 Source category description and emissions

Metal coil surface coating is a process-specific rather than a product-specific operation. This source category includes facilities that contain a metal coil surface coating line, which is the process and the collection of equipment used to apply an organic coating to the surface of metal coil that is at least 0.15 millimeter (0.006 inch) thick. A coil coating line includes a web unwind or feed section, a series of one or more work stations, any associated curing oven, wet section, and quench station. The portions of the metal coil coating line to which the emission limitations apply are the coating application stations and associated curing ovens. (A coil coating line does not include ancillary operations such as mixing/ thinning, cleaning, wastewater treatment, and storage of coating material.) The Surface Coating of Metal Coil source category is regulated under 40 CFR part 63, subpart SSSS. A complete description of this source category can be found in the text of the NPRM.

The emission estimates for this source category were obtained from the 2011 NEI v2, the 2014 NEI v1, and a default speciation profile for organic HAP. We did not perform a formal information collection request (ICR) under the authority of section 114 of the Clean Air Act. We estimate that there are 48 major source facilities with metal coil surface coating processes in the U.S. Emissions from the Surface Coating of Metal Coil source category are summarized in Table 3.1-1. The total HAP emissions from the source category are approximately 291 tons per year. The HAP emitted in the largest quantities are xylenes (mixed), toluene, glycol ethers, naphthalene, isophorone, diethylene glycol monobutyl ether, ethyl benzene, methyl isobutyl ketone, ethylene glycol, methanol, dimethyl phthalate and formaldehyde. Emissions of these 12 HAP make up 99 percent of the total emissions by mass. The PB-HAP reported as emissions from these facilities include small amounts of lead compounds. The following environmental HAP are emitted from the metal coil surface coating processes and are included in the environmental risk screening assessment: lead compounds and hydrofluoric acid.

The emissions for this source category are estimates of actual emissions on an annual basis. Data were not available on the hours of operation for this source category, therefore, default hourly emission rates were used by evenly dividing the total annual emission rate from the inventory into the 8,760 hours of the year. The risk results presented in the following sections are based on these actual emissions. MACT-allowable and facility-wide emissions were also estimated, and the risk results based on those emissions are presented below as well. Details on the development of the actual, allowable, and facility-wide emission estimates and the source of the data for this source category can be found in Appendix 1.

Although the development of the RTR emissions dataset involved quality assurance/quality control processes, the accuracy of emissions values will vary depending on the source of the data, the degree to which data are incomplete or missing, the degree to which assumptions made to complete the datasets are accurate, and errors in emission estimates and other factors.

The emission estimates considered in this analysis are annual totals for certain years, and they do not reflect short-term fluctuations during the course of a year or variations from year to year. The estimates of peak hourly emission rates for the acute effects screening assessment were based on an acute emission adjustment factor of 1.1 that was applied to the annual emission rate from the inventory. The adjusted annual emission rate was then and divided by 8,760 hours to determine the acute average annual hourly emission rate, which is intended to account for emission fluctuations due to normal facility operations.

Some facilities had VOC emissions in the NEI data but did not have organic HAP emissions associated with metal coil coating processes. We used the VOC and organic HAP emissions reported for other facilities to develop a default speciation profile using the weighted average ratio of each organic HAP to VOC. The speciation profile for organic HAP emissions was used to estimate HAP emissions for facilities that reported only VOC emissions. The use of the speciation profile to estimate HAP emissions introduces uncertainty in the emission estimates. Out of 48 facilities in the metal coil coating modelling file, the organic HAP speciation profile was used to estimate organic HAP emissions for twelve facilities.

For the chronic inhalation risk assessment, the emissions inventory for the Surface Coating of Metal Coil source category includes emissions of 24 HAP with available chronic inhalation dose-response values. Of these, 5 are classified as known, probable, or possible carcinogens, with quantitative cancer dose-response values available, and 24 HAP have quantitative noncancer dose-response values available. These HAP, their emissions and dose-response values are listed in Table 3.1-1, and the source of each dose-response value is listed in Appendix 8.

For the acute inhalation risk assessment, for the Surface Coating of Metal Coil source category, we used a multiplier of 1.1 times the annual average emissions to estimate maximum hourly emissions, rather than the default emissions multiplier of 10 as described in Section 2.4. See Appendix 1 to this document for a detailed description of how the maximum hourly emissions were developed for this source category.

The emissions inventory for the Surface Coating of Metal Coil source category includes emissions of 15 HAP with relevant and available quantitative acute dose-response values. These HAP, their emissions and acute and chronic dose-response values are listed in Table 3.1-1, and the source of each dose-response value is listed in Appendix 8.

As mentioned previously, when we identify acute impacts which exceed their relevant dose-response values, we refine our acute screening estimates to the extent possible. For the Surface Coating of Metal Coil source category, the highest refined acute hazard quotient, which is not located on facility property, is estimated to be 3 based on an acute dose-response value for diethylene glycol monobutyl ether. Two facilities are expected to have an acute hazard quotient greater than 1 for the acute dose-response values examined. The acute results for the source category are summarized in the following section, and detailed information is contained in Appendix 10 to this document (*Detailed Risk Modeling Results*).

For the multipathway risk assessment, PB-HAP identified in the emissions inventory for the Surface Coating of Metal Coil source category include a small amount of lead compounds. In evaluating the potential multipathway risks from emission of lead compounds, we compared maximum estimated chronic atmospheric concentrations with the current NAAQS for lead, as described in Section 2.5. The results of the multipathway evaluation of lead compounds for the source category are summarized in the following section.

For the environmental risk assessment, the PB-HAP identified above, lead compounds, as well as the acid gas hydrofluoric acid were screened for potential adverse environmental effects as described in Section 2.5. The benchmark values and a detailed discussion of the approach for this assessment can be found in Appendix 9. The results of the environmental assessment for the source category are summarized in the following section, and detailed information is contained in Appendix 10 to this document.

Table 3.1-1 Summary of Emissions from the Surface Coating of Metal Coil Source Category and Dose-Response Values Used in the Residual Risk Assessment

HAP	Emissions (tpy)	Number of Facilities Reporting HAP (48 facilities in data set)	Prioritized Inhalation Dose-Response Value Identified by OAQPS			PB-HAP Oral Benchmark Values for Cancer (1/(mg/kg/d)) and/or Noncancer (mg/kg/d)[a]
			Unit Risk Estimate for Cancer (1/(µg/m^3))	Reference Concentration for Noncancer (mg/m^3)	Health Benchmark Values for Acute Noncancer (mg/m^3)	
Xylenes (mixed)	77	41		0.1	22 (REL)	
Toluene	54	36		5	37 (REL)	
Glycol Ethers						
Glycol Ethers	45	22		0.02	0.093 (REL)	
Diethylene Glycol Monobutyl Ether	16	2		0.02	0.093 (REL)	
Butyl Carbitol Acetate	0.3	1		0.02	0.093 (REL)	
Naphthalene	27	35	0.000034	0.003		
Isophorone	24	25		2		
Ethyl Benzene	13	34	0.0000025	1	140 (AEGL-1 (1-hr))	
Methyl Isobutyl Ketone	13	31		3		
Ethylene Glycol	7	19		0.4		
Methanol	7	17		20	28 (REL)	
Dimethyl Phthalate	3	23				
Formaldehyde	3	26	0.000013	0.0098	0.055 (REL)	
Cumene	0.7	21		0.4	250 (AEGL-1 (1-hr))	
Triethylamine	0.6	2		0.007	2.8 (REL)	
Tetrachloroethene	0.1	2	0.00000026	0.04	20 (REL)	
Diethanolamine	0.1	1		0.003		
Antimony Compounds	0.1	3		0.0002		
Hydrofluoric Acid	0.08	2		0.014	0.24 (REL)	

Table 3.1-1 Summary of Emissions from the Surface Coating of Metal Coil Source Category and Dose-Response Values Used in the Residual Risk Assessment

HAP	Emissions (tpy)	Number of Facilities Reporting HAP (48 facilities in data set)	Prioritized Inhalation Dose-Response Value Identified by OAQPS			PB-HAP Oral Benchmark Values for Cancer (1/(mg/kg/d)) and/or Noncancer (mg/kg/d)[a]
			Unit Risk Estimate for Cancer (1/(µg/m^3))	Reference Concentration for Noncancer (mg/m^3)	Health Benchmark Values for Acute Noncancer (mg/m^3)	
Cobalt Compounds	0.04	1		0.0001		
Lead Compounds	0.009	14		0.00015[b]		
Dimethyl Formamide	0.007	2		0.03	6 (ERPG-1)	
Nickel Compounds	0.007	2	0.00048	0.00009		

Notes:
[a] Benchmark values are provided only for PB-HAPs for which multipathway risk is assessed (via TRIM). There may be other PB-HAPs in this table, even though no benchmark is presented.
[b] There is no reference concentration for lead. In considering noncancer hazards for lead in this assessment, we compared rolling three-month average exposure estimates to the national ambient air quality standard (NAAQS) for lead (0.15 µg/m3). The primary (health-based) standard is a maximum or not-to-be-exceeded, rolling three-month average, measured as total suspended particles (TSP). The secondary (welfare-based) standard is identical to the primary standard.

3.2 Baseline risk characterization

This section presents the results of the risk assessment for the Surface Coating of Metal Coil source category based on the modeling methods described in the previous sections. All baseline risk results are developed using the best estimates of actual HAP emissions summarized in the previous section. The basic chronic inhalation risk estimates presented here are the maximum individual lifetime cancer risk, the maximum chronic hazard index, and the cancer incidence. We also present results from our acute inhalation impact screening assessment in the form of maximum hazard quotients, as well as the results of our preliminary screening assessment for potential non-inhalation risks and environmental risk from PB-HAP. Also presented are the HAP "drivers," which are the HAP that collectively contribute 90 percent of the maximum cancer risk or maximum hazard at the highest exposure location. A detailed summary of the facility-specific inhalation and multipathway risk assessment results is available in Appendix 10 of this document.

Although the development of the RTR emissions dataset involved quality assurance/quality control processes, the accuracy of emissions values will vary depending on the source of the data, the degree to which data are incomplete or missing, the degree to which assumptions made to complete the datasets are accurate, and errors in emission estimates and other factors. The emission estimates considered in this analysis are annual totals for certain years, and they do not reflect short-term fluctuations during the course of a year or variations from year to year. The estimates of peak hourly emission rates for the acute effects screening assessment were based on an emission adjustment factor applied to the average annual hourly emission rates, which are intended to account for emission fluctuations due to normal facility operations.

Some facilities had VOC emissions in the NEI data but did not have organic HAP emissions associated with metal coil coating processes. We used the VOC and organic HAP emissions reported for other facilities to develop a default speciation profile using the weighted average ratio of each organic HAP to VOC. The speciation profile for organic HAP emissions was used to estimate HAP emissions for facilities that reported only VOC emissions. The use of the speciation profile to estimate HAP emissions introduces uncertainty in the emission estimates. Out of 48 facilities in the metal coil coating modelling file, the organic HAP speciation profile was used to estimate organic HAP emissions for twelve facilities.

3.2.1 Risk assessment results based on actual emissions

Inhalation

Table 3.2-1 summarizes the chronic and acute inhalation risk results for this source category based upon baseline actual emissions. The results of the chronic inhalation cancer risk assessment are that the maximum lifetime individual cancer risk posed by the 48 facilities could be as high as 10-in-1 million, with naphthalene and formaldehyde from solvent storage as the major contributors to the risk. The total estimated cancer incidence from this source category is 0.005 excess cancer cases per year, or one excess case in every 200 years. Approximately 19,000 people were estimated to have cancer risks above 1–in-1 million from HAP emitted from 9 of the 48 facilities in this source category, with 40 of those people estimated to have a cancer risk above 10-in-1 million. The maximum chronic noncancer hazard index value for the source category could be up to 0.1 (reproductive) driven by emissions of glycol ethers from prime coating application and finish coating application processes, and no one is exposed to TOSHI levels above 1.

Worst-case acute HQs were calculated for every HAP that has an acute dose-response value, as shown in Table 3.1-1. For cases where the screening HQ was greater than 1, we further refined the estimates by determining the highest HQ value that might occur outside facility boundaries. Based on actual baseline emissions, the highest refined screening acute HQ value of 3 (based on the acute REL for diethylene glycol monobutyl ether) is shown in Table 3.2-1. It is also important to note that the emissions were modeled with an hourly emissions multiplier of 1.1 times the annual emissions rate. No facilities are estimated to have an acute HQ, based on an AEGL or an EPRG, that is greater than 1. Acute estimates for each facility and pollutant are provided in Appendix 10 of this document.

Table 3.2-1. Source Category Level Inhalation Risks for Surface Coating of Metal Coil Based on Actual Emissions

Result		HAP "Drivers"
Facilities in Source Category		
Number of Facilities Estimated to be in Source Category	48	n/a

Table 3.2-1. Source Category Level Inhalation Risks for Surface Coating of Metal Coil Based on Actual Emissions

Result		HAP "Drivers"
Number of Facilities Modeled in Risk Assessment	48	n/a
Cancer Risks		
Maximum Individual Lifetime Cancer Risk (in 1 million)	10	naphthalene, formaldehyde
Number of Facilities with Maximum Individual Lifetime Cancer Risk:		
Greater than or equal to 100 in 1 million	0	n/a
Greater than or equal to 10 in 1 million	2	naphthalene, formaldehyde
Greater than or equal to 1 in 1 million	9	naphthalene, formaldehyde, ethyl benzene
Chronic Noncancer Risks		
Maximum Reproductive Hazard Index	0.1	glycol ethers
Number of Facilities with Maximum Reproductive Hazard Index:		
Greater than 1	0	n/a
Acute Noncancer Screening Results		
Maximum Acute Hazard Quotient	3	diethylene glycol monobutyl ether (REL)
Number of Facilities With Potential for Acute Effects	2	n/a
Population Exposure		
Number of People Living Within 50 Kilometers of Facilities Modeled	57,000,000	n/a
Number of People Exposed to Cancer Risk:		
Greater than or equal to 100 in 1 million	0	n/a
Greater than or equal to 10 in 1 million	40	n/a
Greater than or equal to 1 in 1 million	19,000	n/a
Number of People Exposed to Noncancer Reproductive Hazard Index:		
Greater than 1	0	n/a
Estimated Cancer Incidence (excess cancer cases per year)	0.005	n/a
Contribution of HAP to Cancer Incidence		
naphthalene	92%	n/a

Facility-wide Inhalation

The facility-wide chronic MIR and TOSHI, available in Appendix 10, are based on emissions from all sources at the identified facilities (both MACT and non-MACT sources). The results of the facility-wide assessment for cancer risks, as compared to the source category assessment, are summarized in Table 3.2-2. The results indicate that 16 facilities have a facility-wide cancer MIR greater than or equal to 1-in-1 million, and 7 of those facilities have a facility-wide cancer MIR greater than or equal to 10-in-1-million. The maximum facility-

wide cancer MIR is 40-in-1 million, mainly driven by naphthalene from equipment cleanup of metal coil coating processes. The total estimated cancer incidence from the whole facility is 0.03 excess cancer cases per year, or one excess case in every 30 years. Approximately 270,000 people were estimated to have cancer risks above 1-in-1 million from exposure to HAP emitted from both MACT and non-MACT sources at the 48 facilities in this source category. The maximum facility-wide TOSHI for the source category is estimated to be 5, mainly driven by emissions of chlorine from a secondary aluminum fluxing process.

Table 3.2-2 Source Category Contribution to Facility-Wide Cancer Risks Based on Actual Emissions

Surface Coating of Metal Coil	Number of Facilities Binned by Facility-Wide MIR (in 1 million)				
Source Category MIR Contribution to Facility-Wide MIR	<1	1≤ MIR<10	10≤ MIR<100	≥ 100	**Total**
> 90%	23*	17*	4	2	0
50-90%	6	5	1	0	0
10-50%	6	3	1	2	0
< 10%	13**	7**	3**	3	0
Total	48	32	9	7	0

* Four facilities have no (0) MACT level or facility-wide risk.

** Four facilities have no (0) MACT level risk, but non-zero facility-wide risk.

Multipathway

In evaluating the potential for multipathway effects from emissions of lead, modeled maximum annual lead concentrations were compared to the NAAQS for lead (0.15 $\mu g/m^3$). Small amounts of lead emissions are reported from 14 facilities. Results of this analysis confirmed that the NAAQS for lead would not be exceeded by any facility.

Environmental

We conducted a screening-level evaluation of the potential adverse environmental risks associated with emissions of the environmental HAP hydrofluoric acid and lead from the Surface Coating of Metal Coil source category.

For hydrofluoric acid, each individual concentration (i.e., each off-site data point in the modeling domain) is below the ecological benchmarks for all facilities. For lead, we did not estimate any exceedances of the secondary lead NAAQS.

3.2.2 Risk assessment results based on allowable emissions

Inhalation

Potential differences between actual emissions levels and the maximum emissions allowable under the MACT standards (i.e., MACT-allowable emissions) were also calculated for the facilities. Emission estimates based on the actual emissions were then scaled up using a factor of 1.1 to develop the allowable emissions. See Appendix 1 of this document for a discussion of the development of the allowable factors. Risk results from the inhalation risk assessment using the MACT-allowable emissions indicate that the maximum lifetime individual cancer risk could be as high as 10-in-1 million with naphthalene and formaldehyde emissions from solvent storage driving the risks, and that the maximum chronic noncancer TOSHI value could be as high as 0.1 at the MACT-allowable emissions level with glycol ethers emissions from prime coating application and finish coating processes driving the TOSHI. The total estimated cancer incidence from this source category considering allowable emissions is expected to be about 0.006 excess cancer cases per year or one excess case in every 200 years. Based on allowable emission rates approximately 24,000 people were estimated to have cancer risks above 1-in-1 million, with 40 of those people estimated to have cancer risk above 10-in-1 million. No people are estimated to have a noncancer hazard index above 1.

3.3 Post-control risk characterization

A post-control risk assessment is not warranted for this category because we did not identify any add-on control technologies, other equipment or work practices and procedures that had not previously been considered during development of the Surface Coating of Metal Coil NESHAP, nor did we identify any developments in the same since the promulgation of the NESHAP.

4 General discussion of uncertainties in the risk assessment

The uncertainties in virtually all of the RTR risk assessments can be divided into three areas: 1) uncertainties in the emission data sets, 2) exposure modeling uncertainties, and 3) uncertainties in the dose-response relationships. Uncertainties in the emission estimates and in the air quality models lead to uncertainty in air concentrations. Uncertainty in exposure modeling can arise due to uncertain activity patterns, the locations of individuals within a census tract, and the microenvironmental concentrations as reflected in the exposure model. Finally, uncertainty in the shape of the relationship between exposure and effects, the URE and the RfC, also contributes to uncertainties in the risk assessment. These three areas of uncertainty are discussed below.

4.1 Emissions inventory uncertainties

Although the development of the RTR emissions data set involves an extensive quality assurance/quality control process, the accuracy of emission values will vary depending on certain factors, for example, the source of the data, the degree to which data are incomplete or missing, the degree to which assumptions made to complete the data sets are accurate, and the extent to which there are errors in these emission estimates. The emission estimates used in the risk assessment generally are annual totals for certain years, and they do not reflect short-term fluctuations during the course of a year or variations from year to year.

For the acute effects screening assessment, therefore, in the absence of available specific estimates or measurements, we use estimates of peak hourly emission rates. These estimates typically are calculated by applying an acute emission adjustment factor to the annual emission rate from the inventory. The adjusted annual emission rate is then divided by 8,760 hours to determine the acute average annual hourly emission rate, which is intended to account for emission fluctuations due to normal facility operations. The adjustment factor can be based on actual fluctuations seen in the available emission data for sources in a category or on engineering judgment; in the absence of such information, a default factor is applied.

To prepare the emissions data set, EPA gathers the best available data on emissions, emission release parameters, and other relevant source category-specific parameters. EPA often begins with its National Emissions Inventory (NEI) database as the starting point for emission rates, emissions release characteristics, and locations of the emission release points for each facility in the source category. The NEI is a composite of emission measurements and estimates produced by state and local regulatory agencies, industry, and EPA. EPA's industry experts then review the data for consistency and completeness and conduct extensive quality assurance/quality control checks. Available information, which may include compliance data, information from project files, permits, and other sources regarding facilities and emission sources, are also incorporated into the data set. This additional information may be incorporated in addition to the NEI data or in place of the NEI data, depending on EPA's evaluation of the quality of the various sources of data. In order to fill data gaps, EPA may conduct a formal information collection request (ICR) under the authority of section 114 of the Clean Air Act to obtain current, complete emissions data and other data from the facility owners and operators associated with the source category under review.

Uncertainty in the emissions data set stems from data gaps, default assumptions, and the emission models used to develop emissions inventory estimates. A variety of methods, such as emission factors, material balances, engineering judgement, air permit information and source testing, are used to develop emission estimates. Other parameters that are part of the emissions data set, including facility location and emission point parameters, may also be a source of uncertainty. Some release point locations use an average facility location instead of the location of each specific unit within the facility. In some instances, default release point parameters may be in the inventory. Where fugitive release parameters are not available, default values are included. Another potential source of emission estimate uncertainty may be low or poor quality data (e.g., out-of-date parameter values). For more information on the uncertainties in the emission estimates for this source category see Appendix 1 *(Emissions Inventory Support Documents*) of this document.

4.2 Exposure modeling uncertainties

4.2.1 Inhalation exposure modeling

Although every effort is made to identify all of the relevant facilities and emission points, as well as to develop accurate estimates of the annual emission rates for all relevant HAP, the uncertainties in our emission inventory likely dominate the uncertainties in the exposure assessment. The ambient air modeling uncertainties are considered relatively small in

comparison, since we are using EPA's refined local dispersion model with site-specific parameters and reasonably representative meteorology. If anything, the population exposure estimates are biased high by not accounting for short- or long-term population mobility and by not addressing processes like deposition, plume depletion, and atmospheric degradation. Additionally, estimates of maximum individual risk (MIR) contain uncertainty because they are derived at census block centroid locations rather than actual residences. This uncertainty is known to create potential underestimates and overestimates of the actual MIR values for individual facilities; however, overall, it is not thought to have a significant impact on the estimated MIR for a source category. We also do not factor in the possibility of a source closure occurring during the 70-year chronic exposure period, leading to a potential upward bias in both the MIR and population risk estimates. Nor do we factor in the possibility of population growth during the 70-year chronic exposure period, which could lead to a potential downward bias in both the MIR and population risk estimates. Finally, we do not factor in time an individual spends indoors.

We did not include the effects of human mobility on exposures in the assessment. Specifically, short-term mobility and long-term mobility between census blocks in the modeling domain were not considered. (Short-term mobility is movement from one micro-environment to another over the course of hours or days. Long-term mobility is movement from one residence to another over the course of a lifetime.) The approach of not considering short or long-term population mobility does not bias the estimate of the theoretical MIR (by definition), nor does it affect the estimate of cancer incidence because the total population number remains the same. It does, however, affect the shape of the distribution of individual risks across the affected population, shifting it toward higher estimated individual risks at the upper end and reducing the number of people estimated to be at lower risks, thereby increasing the estimated number of people at specific high risk levels (e.g., 1-in-10 thousand or 1-in-1 million).

In addition, the assessment predicted the chronic exposures at the centroid of each populated census block as surrogates for the exposure concentrations for all people living in that block. Using the census block centroid to predict chronic exposures tends to over-predict exposures for people in the census block who live farther from the facility and under-predict exposures for people in the census block who live closer to the facility. Thus, using the census block centroid to predict chronic exposures may lead to a potential understatement or overstatement of the true maximum impact, but is an unbiased estimate of average risk and incidence. We reduce this uncertainty by analyzing large census blocks near facilities using aerial imagery and adjusting the location of the block centroid to better represent the population in the block, as well as adding additional receptor locations where the block population is not well represented by a single location.

The assessment evaluates the cancer inhalation risks associated with pollutant exposures over a 70-year period, which is the assumed lifetime of an individual. In reality, both the length of time that modeled emission sources at facilities actually operate (i.e., more or less than 70 years) and the domestic growth or decline of the modeled industry (i.e., the increase or decrease in the number or size of domestic facilities) will influence the future risks posed by a given source or source category. Depending on the characteristics of the industry, these

factors will, in most cases, result in an overestimate both in individual risk levels and in the total estimated number of cancer cases. However, in the unlikely scenario where a facility maintains, or even increases, its emissions levels over a period of more than 70 years, residents live beyond 70 years at the same location, and the residents spend more of their days at that location, then the cancer inhalation risks could potentially be underestimated. However, annual cancer incidence estimates from exposures to emissions from these sources would not be affected by the length of time an emissions source operates.

The exposure estimates used in these analyses assume chronic exposures to ambient (outdoor) levels of pollutants. Because most people spend the majority of their time indoors, actual exposures may not be as high, depending on the characteristics of the pollutants modeled. For many of the HAP, indoor levels are roughly equivalent to ambient levels, but for very reactive pollutants or larger particles, indoor levels are typically lower. This factor has the potential to result in an overestimate of 25 to 30 percent of exposures (USEPA, 2001).

A sensitivity analysis, discussed in "Risk and Technology Review (RTR) Risk Assessment Methodologies" (USEPA 2009a), found that the selection of the meteorology data set location could have an impact on the risk estimates. The analysis found that cancer MIR derived using different meteorological stations varied by as much as 63 percent below to 51 percent above the value derived using the nearest meteorological station. Cancer incidence estimated using different meteorological stations varied by as much as 68 percent below to 120 percent above the value estimated using the nearest meteorological station. Similarly, air concentrations estimated using different meteorological stations varied by as much as 49 percent below to 21 percent above the value estimated using the nearest meteorological station. Since this analysis was performed EPA has increased the number of meteorological stations used in our risk assessments; thus, we expect variability to be reduced.

For the acute screening assessment, the results are intentionally biased high, and thus health-protective, by assuming the co-occurrence of independent factors, such as hourly emission rates, meteorology and human activity patterns. Furthermore, in cases where multiple acute dose-response values for a pollutant are considered scientifically acceptable, we choose the most conservative of these dose-response values, erring on the side of overestimating potential health risks from acute exposures. In cases where these results indicate the potential for exceeding acute HQs, we refine our assessment by developing a better understanding of the geography of the facility relative to potential exposure locations.

4.2.2 Multipathway exposure modeling

In modeling the fate and transport of pollutants through the environment and the non-inhalation exposure (i.e., ingestion) to these pollutants, TRIM.FaTE uses simplified representations of many complex real-world processes. This simplified representation introduces uncertainty. Uncertainties arise from model assumptions and structure, as reflected in the algorithms that describe the environmental movement of pollutants, and in the input values for numerous environmental parameters.

Uncertainty in the algorithms is inherent to any model attempting to represent complex processes in the real world. How persistent, bioaccumulative chemicals such as mercury, cadmium, arsenic, PAHs, and dioxins behave in the environment is highly complex, and many natural processes are represented in a simplified manner by TRIM.FaTE, including, for example:

- gaseous and particulate deposition from air;
- biogeochemical cycling in the aquatic environment, particularly mercury transformations through methylation and demethylation at the sediment-surface interface;
- mixing processes in air, water, and sediment;
- suspended and benthic sediment dynamics in lakes; and
- biotic processes such as growth, reproduction, and predation.

Even though some processes, such as diffusion, are known to follow second-order dynamics, the TRIM.FaTE model represents all fate and transport processes in terms of first-order differential equations. TRIM.FaTE also does not explicitly deal with lateral or vertical dispersion in the air compartments. Some algorithms, such as those addressing methylation and sediment transport, for example, do not consider all of the factors known to affect the process. Biotic processes including chemical absorption, chemical elimination, growth, reproduction, predation, and death have been represented relatively simplistically in the model. Although the model's algorithms have been validated and are based on professional judgment, some level of uncertainty results from such simplifications.

The input values for parameters are also associated with uncertainty. Algorithms that describe the environmental movement of pollutants depend on numerous environmental parameters for which the values might be naturally variable and for which available data are often limited. Examples of parameters for which input values are variable and uncertain include aquatic food web structure (e.g., diet of each fish species), biokinetic parameters that influence bioaccumulation (e.g., assimilation efficiencies and elimination rates), topographic characteristics (e.g., lake depth, runoff rates, and erosion rates), meteorological parameters (e.g., evaporation and precipitation rates), chemical transformation rates (e.g., methylation and demethylation rates, in the case of mercury), and human exposure parameters (especially fish consumption rates).

For TRIM.FaTE modeling, we use central tendency values and combinations of values that would lead to estimates of reasonable maximum exposures to bound risk estimates. We have conducted analyses of the sensitivity of risk estimates to parameter input values. For those parameters to which the model is particularly sensitive, we have continued to collect additional data to better quantify the variability and distribution of input values.
A more comprehensive explanation of the uncertainties related to fate, transport, and exposure modeling using TRIM.FaTE is provided in Appendix 6 (*Technical Support Document for TRIM-Based Multipathway Tiered Screening Methodology for RTR)* of this report for the tiered assessments and Appendix 11 (*Site-Specific Human Health Multipathway Residual Risk Assessment Report)* of this report for a site-specific assessment if one was conducted.

4.2.3 Environmental risk screening assessment

For each source category, we generally rely on site-specific levels of environmental HAP emissions to perform an environmental screening assessment. The environmental screening assessment is based on the outputs from models that estimate environmental HAP concentrations. The same models, specifically the TRIM.FaTE multipathway model and the AERMOD air dispersion model, are used to estimate environmental HAP concentrations for both the human multipathway screening analysis and for the environmental screening analysis. Therefore, both screening assessments have similar modeling uncertainties. Two important types of uncertainty associated with the use of these models in RTR environmental screening assessments (and inherent to any assessment that relies on environmental modeling) are model uncertainty and input uncertainty.

Model uncertainty concerns whether the selected models are appropriate for the assessment being conducted and whether they adequately represent the movement and accumulation of environmental HAP emissions in the environment. For example, does the model adequately describe the movement of the pollutant through the soil? This type of uncertainty is difficult to quantify. However, based on feedback received from previous EPA SAB reviews and other reviews, we are confident that the models used in the screening assessments are appropriate and state-of-the-art for the environmental risk assessments conducted in support of our RTR analyses.

Input uncertainty is concerned with how accurately the models have been configured and parameterized for the assessment at hand. For Tier 1 of the environmental screening assessment for PB-HAP, we configured the models to avoid underestimating exposure and risk to reduce the likelihood that the results indicate the risks are lower than they actually are. This was accomplished by selecting upper-end values from nationally-representative datasets for the more influential parameters in the environmental model, including selection and spatial configuration of the area of interest, the location and size of any bodies of water, meteorology, surface water and soil characteristics, and structure of the aquatic food web. In Tier 1, we use the maximum facility-specific emissions for the PB-HAP (other than lead compounds, which were evaluated by comparison to the Secondary Lead NAAQS) that are included in the environmental screening assessment and each of the media when comparing to ecological benchmarks. This is consistent with the conservative design of the Tier 1 screening assessment. In Tier 2 of the environmental screening assessment for PB-HAP, we refine the model inputs to account for meteorological patterns in the vicinity of the facility versus using upper-end national values, and we identify the locations of water bodies near the facility location. By refining the screening approach in Tier 2 to account for local geographical and meteorological data, we decrease the likelihood that concentrations in environmental media are overestimated, thereby increasing the usefulness of the screening assessment. To better represent widespread impacts, the modeled soil concentrations are averaged in Tier 2 to obtain one average soil concentration value for each facility and for each PB-HAP. For PB-HAP concentrations in water, sediment, and fish tissue, the highest value for each facility for each pollutant is used.

For the environmental screening assessment for acid gases, we employ a single-tiered approach. We use the modeled air concentrations and compare those with ecological benchmarks.

For both Tiers 1 and 2 of the environmental screening assessment, our approach to addressing model input uncertainty is generally cautious. We choose model inputs from the upper end of the range of possible values for the influential parameters used in the models, and we assume that the exposed individual exhibits ingestion behavior that would lead to a high total exposure. This approach reduces the likelihood of not identifying potential risks for adverse environmental impacts.

4.3 Uncertainties in the dose-response relationships

In the sections that follow, separate discussions are provided on uncertainty associated with cancer potency factors and for noncancer reference values. Cancer potency values are derived for chronic (lifetime) exposures. Noncancer dose-response values are generally derived for chronic exposures (up to a lifetime) but may also be derived for acute (less than 24 hours), short-term (from 24 hours up to 30 days), and subchronic (30 days up to 10 percent of lifetime) exposure durations, all of which are derived based on an assumption of continuous exposure throughout the duration specified. For the purposes of assessing all potential health risks associated with the emissions included in an assessment, we rely on both chronic (cancer and noncancer) and acute (noncancer) dose-response values, which are described in more detail below.

Although every effort is made to identify peer-reviewed dose-response values for all HAP emitted by the source category included in an assessment, some HAP have no peer-reviewed values. Since exposures to these pollutants cannot be included in a quantitative risk estimate, an understatement of risk for these pollutants at estimated exposure levels is possible. To help alleviate this potential underestimate, where we conclude similarity with a HAP for which a dose-response assessment value is available, we use that value as a surrogate for the assessment of the HAP for which no value is available. To the extent use of surrogates indicates appreciable risk, we may identify a need to increase priority for a new IRIS assessment of that substance. We additionally note that, generally speaking, HAP of greatest concern due to environmental exposures and hazards are those for which dose-response assessments have been performed, reducing the likelihood of understating risk. Further, HAP not included in the quantitative assessment are assessed qualitatively and considered in the risk characterization that informs the risk management decisions, including with regard to consideration of HAP reductions achieved by various control options.

Additionally, chronic dose-response values for certain compounds included in the assessment may be under EPA IRIS review. In those cases, revised assessments may determine in the future that these pollutants are more or less potent than currently thought.

For a group of compounds that are unspeciated (e.g., glycol ethers), we conservatively use the most protective reference value of an individual compound in that group to estimate risk. Similarly, for an individual compound in a group (e.g., ethylene glycol diethyl ether) that does

not have a specified reference value, we apply the most protective reference value from the other compounds in the group to estimate risk.

Cancer assessment

The discussion of dose-response uncertainties in the estimation of cancer risk below focuses on the uncertainties associated with the specific approach currently used by the EPA to develop cancer potency factors. In general, these same uncertainties attend the development of cancer potency factors by CalEPA, the source of peer-reviewed cancer potency factors used where EPA-developed values are not yet available. To place this discussion in context, we provide a quote from the EPA's *Guidelines for Carcinogen Risk Assessment* (herein referred to as *Cancer Guidelines*). (USEPA, 2005d) "The primary goal of EPA actions is protection of human health; accordingly, as an Agency policy, risk assessment procedures, including default options that are used in the absence of scientific data to the contrary, should be health protective." The approach adopted in this document is consistent with this approach as described in the *Cancer Guidelines*.

For cancer endpoints EPA usually derives an oral slope factor for ingestion and a unit risk value for inhalation exposures. These values allow estimation of a lifetime probability of developing cancer given long-term exposures to the pollutant. Depending on the pollutant being evaluated, EPA relies on both animal bioassay and epidemiological studies to characterize cancer risk. As a science policy approach, consistent with the *Cancer Guidelines*, EPA uses animal cancer bioassays as indicators of potential human health risk when other human cancer risk data are unavailable.

Extrapolation of study data to estimate potential risks to human populations is based upon EPA's assessment of the scientific database for a pollutant using EPA's guidance documents and other peer-reviewed methodologies. The EPA *Cancer Guidelines* describe the Agency's recommendations for methodologies for cancer risk assessment. EPA believes that cancer risk estimates developed following the procedures described in the *Cancer Guidelines* and outlined below generally provide an upper bound estimate of risk. That is, EPA's upper bound estimates represent a plausible upper limit to the true value of a quantity (although this is usually not a true statistical confidence limit). In some circumstances, the true risk could be as low as zero; however, in other circumstances the risk could also be greater.[9] When developing an upper bound estimate of risk and to provide risk values that do not underestimate risk, EPA generally relies on conservative default approaches.[10] EPA also

[9] The exception to this is the URE for benzene, which is considered to cover a range of values, each end of which is considered to be equally plausible, and which is based on maximum likelihood estimates.

[10] According to the NRC report Science and Judgment in Risk Assessment (NRC, 1994) "[Default] options are generic approaches, based on general scientific knowledge and policy judgment, that are applied to various elements of the risk-assessment process when the correct scientific model is unknown or uncertain." The 1983 NRC report Risk Assessment in the Federal Government: Managing the Process defined default option as "the option chosen on the basis of risk assessment policy that appears to be the best choice in the absence of data to the contrary" (NRC, 1983a, p. 63). Therefore, default options are not rules that bind the Agency; rather, the Agency may depart from them in evaluating the risks posed by a specific substance when it believes this to be appropriate. In keeping with EPA's goal of protecting public health and the environment, default assumptions are used to ensure that risk to chemicals is not underestimated (although defaults are not intended to overtly

uses the upper bound (rather than lower bound or central tendency) estimates in its assessments, although it is noted that this approach can have limitations for some uses (e.g. priority setting, expected benefits analysis).

Such health risk assessments have associated uncertainties, some which may be considered quantitatively, and others which generally are expressed qualitatively. Uncertainties may vary substantially among cancer risk assessments associated with exposures to different pollutants, since the assessments employ different databases with different strengths and limitations and the procedures employed may differ in how well they represent actual biological processes for the assessed substance. Some of the major sources of uncertainty and variability in deriving cancer risk values are described more fully below.

(1) The qualitative similarities or differences between tumor responses observed in experimental animal bioassays and those which would occur in humans are a source of uncertainty in cancer risk assessments. In general, EPA does not assume that tumor sites observed in an experimental animal bioassay are necessarily predictive of the sites at which tumors would occur in humans.[11] However, unless scientific support is available to show otherwise, EPA assumes that tumors in animals are relevant in humans, regardless of target organ concordance. For a specific pollutant, qualitative differences in species responses can lead to either under-estimation or over-estimation of human cancer risks.

(2) Uncertainties regarding the most appropriate dose metric for an assessment can also lead to differences in risk predictions. For example, the measure of dose is commonly expressed in units of mg/kg/d ingested or the inhaled concentration of the pollutant. However, data may support development of a pharmacokinetic model for the absorption, distribution, metabolism and excretion of an agent, which may result in improved dose metrics (e.g., average blood concentration of the pollutant or the quantity of agent metabolized in the body). Quantitative uncertainties result when the appropriate choice of a dose metric is uncertain or when dose metric estimates are themselves uncertain (e.g., as can occur when alternative pharmacokinetic models are available for a compound). Uncertainty in dose estimates may lead to either over or underestimation of risk.

(3) For the quantitative extrapolation of cancer risk estimates from experimental animals to humans, EPA uses scaling methodologies (relating expected response to differences in physical size of the species), which introduce another source of uncertainty. These methodologies are based on both biological data on differences in rates of process according to species size and empirical comparisons of toxicity between experimental animals and humans. For a particular pollutant, the quantitative difference in cancer potency between experimental animals and humans may be either greater than or less than that estimated by baseline scientific scaling predictions due to uncertainties associated with limitations in the test data and the correctness of scaled estimates.

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overestimate risk). See EPA 2004 An Examination of EPA Risk Assessment Principles and Practices, EPA/100/B-04/001.

[11] Per the EPA Cancer Guidelines: "The default option is that positive effects in animal cancer studies indicate that the agent under study can have carcinogenic potential in humans." and "Target organ concordance is not a prerequisite for evaluating the implications of animal study results for humans."

(4) EPA cancer risk estimates, whether based on epidemiological or experimental animal data, are generally developed using a benchmark dose (BMD) analysis to estimate a dose at which there is a specified excess risk of cancer, which is used as the point of departure (or POD) for the remainder of the calculation. Statistical uncertainty in developing a POD using a benchmark dose (BMD) approach is generally addressed though use of the 95 percent lower confidence limit on the dose at which the specified excess risk occurs (the BMDL), decreasing the likelihood of understating risk. EPA has generally utilized the multistage model for estimation of the BMDL using cancer bioassay data (see further discussion below).

(5) Extrapolation from high to low doses is an important source of uncertainty in cancer risk assessment. EPA uses different approaches to low dose risk assessment (i.e., developing estimates of risk for exposures to environmental doses of an agent from observations in experimental or epidemiological studies at higher dose) depending on the available data and understanding of a pollutant's mode of action (i.e., the manner in which a pollutant causes cancer). EPA's *Cancer Guidelines* express a preference for the use of reliable, compound-specific, biologically-based risk models when feasible; however, such models are rarely available. The mode of action for a pollutant (i.e., the manner in which a pollutant causes cancer) is a key consideration in determining how risks should be estimated for low-dose exposure. A reference value is calculated when the available mode of action data show the response to be nonlinear (e.g., as in a threshold response). A linear low-dose (straight line from POD) approach is used when available mode of action data support a linear (e.g., nonthreshold) response or as the most common default approach when a compound's mode of action is unknown. Linear extrapolation can be supported by both pollutant-specific data and broader scientific considerations. For example, EPA's *Cancer Guidelines* generally consider a linear dose-response to be appropriate for pollutants that interact with DNA and induce mutations. Pollutants whose effects are additive to background biological processes in cancer development can also be predicted to have low-dose linear responses, although the slope of this relationship may not be the same as the slope estimated by the straight line approach.

EPA most frequently utilizes a linear low-dose extrapolation approach as a baseline science-policy choice (a "default") when available data do not allow a compound-specific determination. This approach is designed to not underestimate risk in the face of uncertainty and variability. EPA believes that linear dose-response models, when appropriately applied as part of EPA's cancer risk assessment process, provide an upper bound estimate of risk and generally provide a health protective approach. Note that another source of uncertainty is the characterization of low-dose nonlinear, non-threshold relationships. The National Academy of Sciences (NAS, 1994) has encouraged the exploration of sigmoidal type functions (e.g., log-probit models) in representing dose-response relationships due to the variability in response within human populations. Another National Research Council report (NRC, 2006) suggests that models based on distributions of individual thresholds are likely to lead to sigmoidal-shaped dose-response functions for a population. This report notes sources of variability in the human population: "One might expect these individual tolerances to vary extensively in humans depending on genetics, coincident exposures, nutritional status, and various other susceptibility factors..." Thus, if a distribution of thresholds approach is considered for a carcinogen risk assessment, application would depend on ability of modeling to reflect the

degree of variability in response in human populations (as opposed to responses in bioassays with genetically more uniform rodents). Note also that low dose linearity in risk can arise for reasons separate from population variability: due to the nature of a mode of action and additivity of a chemical's effect on top of background chemical exposures and biological processes.

As noted above, EPA's current approach to cancer risk assessment typically utilizes a straight line approach from the BMDL. This is equivalent to using an upper confidence limit on the slope of the straight line extrapolation. The impact of the choice of the BMDL on bottom line risk estimates can be quantified by comparing risk estimates using the BMDL value to central estimate BMD values, although these differences are generally not a large contributor to uncertainty in risk assessment (Subramaniam et. al., 2006). It is important to note that earlier EPA assessments, including the majority of those for which risk values exist today, were generally developed using the multistage model to extrapolate down to environmental dose levels and did not involve the use of a POD. Subramaniam et. al. (2006) also provide comparisons indicating that slopes based on straight line extrapolation from a POD do not show large differences from those based on the upper confidence limit of the multistage model.

(6) Cancer risk estimates do not generally make specific adjustments to reflect the variability in response within the human population — resulting in another source of uncertainty in assessments. In the diverse human population, some individuals are likely to be more sensitive to the action of a carcinogen than the typical individual, although compound-specific data to evaluate this variability are generally not available. There may also be important life stage differences in the quantitative potency of carcinogens and, with the exception of the recommendations in EPA's Supplemental Cancer Guidance for carcinogens with a mutagenic mode of action, risk assessments do not generally quantitatively address life stage differences. However, one approach used commonly in EPA assessments that may help address variability in response is to extrapolate human response from results observed in the most sensitive species and sex tested, resulting typically in the highest URE which can be supported by reliable data, thus supporting estimates that are designed not to underestimate risk in the face of uncertainty and variability.

Chronic noncancer assessment

Chronic noncancer reference values represent chronic exposure levels that are intended to be health-protective. That is, EPA and other organizations, such as the Agency for Toxic substances and disease Registry (ATSDR), which develop noncancer dose-response values use an approach that is intended not to underestimate risk in the face of uncertainty and variability. When there are gaps in the available information, uncertainty factors (UFs) are applied to derive reference values that are intended to be protective against appreciable risk of deleterious effects. Uncertainty factors are commonly default values[12] (e.g., factors of 10 or 3)

[12] According to the NRC report *Science and Judgment in Risk Assessment* (NRC, 1994) "[Default] options are generic approaches, based on general scientific knowledge and policy judgment, that are applied to various elements of the risk-assessment process when the correct scientific model is unknown or uncertain." The 1983 NRC report *Risk Assessment in the Federal Government: Managing the Process* defined *default option* as "the

used in the absence of compound-specific data; where data are available, uncertainty factors may also be developed using compound-specific information. When data are limited, more assumptions are needed and more default factors are used. Thus, there may be a greater tendency to overestimate risk—in the sense that further study might support development of reference values that are higher (i.e., less potent) because fewer default assumptions are needed. However, for some pollutants it is possible that risks may be underestimated.

For noncancer endpoints related to chronic exposures, EPA derives a reference dose (RfD) for exposures via ingestion, and a reference concentration (RfC) for inhalation exposures. As stated in the IRIS Glossary, these values provide an estimate (with uncertainty spanning perhaps an order of magnitude) of daily oral exposure (RfD) or of a continuous inhalation exposure (RfC) to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. To derive values that are intended to be "without appreciable risk," EPA's methodology relies upon an uncertainty factor (UF) approach (USEPA, 1993b; USEPA, 1994b) which includes consideration of both uncertainty and variability.

EPA begins by evaluating all of the available peer-reviewed literature to determine noncancer endpoints of concern, evaluating the quality, strengths and limitations of the available studies. EPA typically chooses the relevant endpoint that occurs at the lowest dose, often using statistical modeling of the available data, and then determines the appropriate POD for derivation of the reference value. A POD is determined by (in order of preference): (1) a statistical estimation using the BMD approach; (2) use of the dose or concentration at which the toxic response was not significantly elevated (no observed adverse effect level—NOAEL); or (3) use of the lowest observed adverse effect level (LOAEL).

A series of downward adjustments using default UFs is then applied to the POD to estimate the reference value (USEPA, 2002b). While collectively termed "UFs", these factors account for a number of different quantitative considerations when utilizing observed animal (usually rodent) or human toxicity data in a risk assessment. The UFs are intended to account for: (1) variation in susceptibility among the members of the human population (i.e., inter-individual variability); (2) uncertainty in extrapolating from experimental animal data to humans (i.e., interspecies differences); (3) uncertainty in extrapolating from data obtained in a study with less-than-lifetime exposure (i.e., extrapolating from subchronic to chronic exposure); (4) uncertainty in extrapolating from a LOAEL in the absence of a NOAEL; and (5) uncertainty when the database is incomplete or there are problems with applicability of available studies. When scientifically sound, peer-reviewed assessment-specific data are not available, default adjustment values are selected for the individual UFs. For each type of

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option chosen on the basis of risk assessment policy that appears to be the best choice in the absence of data to the contrary" (NRC, 1983a, p. 63). Therefore, default options are not rules that bind the Agency; rather, the Agency may depart from them in evaluating the risks posed by a specific substance when it believes this to be appropriate. In keeping with EPA's goal of protecting public health and the environment, default assumptions are used to ensure that risk to chemicals is not underestimated (although defaults are not intended to overtly overestimate risk). See EPA 2004 *An examination of EPA Risk Assessment Principles and Practices*, EPA/100/B-04/001.

uncertainty (when relevant to the assessment), EPA typically applies an UF value of 10 or 3 with the cumulative UF value leading to a downward adjustment of 10-3000 fold from the selected POD. An UF of 3 is used when the data do not support the use of a 10-fold factor. If an extrapolation step or adjustment is not relevant to an assessment (e.g., if applying human toxicity data and an interspecies extrapolation is not required) the associated UF is not used. The major adjustment steps are described more fully below.

1) Heterogeneity among humans is a key source of variability as well as uncertainty. Uncertainty related to human variation is considered in extrapolating doses from a subset or smaller-sized population, often of one sex or of a narrow range of life stages (typical of occupational epidemiologic studies), to a larger, more diverse population. In the absence of pollutant-specific data on human variation, a 10-fold UF is used to account for uncertainty associated with human variation. Human variation may be larger or smaller; however, data to examine the potential magnitude of human variability are often unavailable. In some situations, a smaller UF of 3 may be applied to reflect a known lack of significant variability among humans.

2) Extrapolation from results of studies in experimental animals to humans is a necessary step for the majority of chemical risk assessments. When interpreting animal data, the concentration at the POD (e.g. NOAEL, BMDL) in an animal model (e.g. rodents) is extrapolated to estimate the human response. While there is long-standing scientific support for the use of animal studies as indicators of potential toxicity to humans, there are uncertainties in such extrapolations. In the absence of data to the contrary, the typical approach is to use the most relevant endpoint from the most sensitive species and the most sensitive sex in assessing risks to the average human. Typically, compound specific data to evaluate relative sensitivity in humans versus rodents are lacking, thus leading to uncertainty in this extrapolation. Size-related differences (allometric relationships) indicate that typically humans are more sensitive than rodents when compared on a mg/kg/day basis. The default choice of 10 for the interspecies UF is consistent with these differences. For a specific chemical, differences in species responses may be greater or less than this value.

Pharmacokinetic models are useful to examine species differences in pharmacokinetic processing and associated uncertainties; however, such dosimetric adjustments are not always possible. Information may not be available to quantitatively assess toxicokinetic or toxicodynamic differences between animals and humans, and in many cases a 10-fold UF (with separate factors of 3 for toxicokinetic and toxicodynamic components) is used to account for expected species differences and associated uncertainty in extrapolating from laboratory animals to humans in the derivation of a reference value. If information on one or the other of these components is available and accounted for in the cross-species extrapolation, a UF of 3 may be used for the remaining component.

3) In the case of reference values for chronic exposures where only data from shorter durations are available (e.g., 90-day subchronic studies in rodents) or when such data are judged more appropriate for development of an RfC, an additional UF of 3 or 10-fold is typically applied unless the available scientific information supports use of a different value.

4) Toxicity data are typically limited as to the dose or exposure levels that have been tested in individual studies; in an animal study, for example, treatment groups may differ in exposure by up to an order of magnitude. The preferred approach to arrive at a POD is to use BMD analysis; however, this approach requires adequate quantitative results for a meaningful analysis, which is not always possible. Use of a NOAEL is the next preferred approach after BMD analysis in determining a POD for deriving a health effect reference value. However, many studies lack a dose or exposure level at which an adverse effect is not observed (i.e., a NOAEL is not identified). When using data limited to a LOAEL, a UF of 10 or 3-fold is often applied.

5) The database UF is intended to account for the potential for deriving an underprotective RfD/RfC due to a data gap preventing complete characterization of the chemical's toxicity. In the absence of studies for a known or suspected endpoint of concern, a UF of 10 or 3-fold is typically applied.

Acute noncancer assessment

Many of the UFs used to account for variability and uncertainty in the development of acute reference values are quite similar to those developed for chronic durations. For acute reference values, though, individual UF values may be less than 10. UFs are applied based on chemical- or health effect-specific information or based on the purpose of the reference value. The UFs applied in acute reference value derivation include: 1) heterogeneity among humans; 2) uncertainty in extrapolating from animals to humans; 3) uncertainty in LOAEL to NOAEL adjustments; and 4) uncertainty in accounting for an incomplete database on toxic effects of potential concern. Additional adjustments are often applied to account for uncertainty in extrapolation from observations at one exposure duration (e.g., 4 hours) to arrive at a POD for derivation of an acute reference value at another exposure duration (e.g., 1 hour).

Not all acute dose-response values are developed for the same purpose and care must be taken when interpreting the results of an acute assessment of human health effects relative to the reference value or values being exceeded. Where relevant to the estimated exposures, the lack of dose-response values at different levels of severity should be factored into the risk characterization as potential uncertainties.

Environmental Risk Screening Assessment

Uncertainty also exists in the ecological benchmarks for the environmental risk screening assessment. We established a hierarchy of preferred benchmark sources to allow selection of benchmarks for each environmental HAP at each ecological assessment endpoint. In general, EPA benchmarks used at a programmatic level (e.g., Office of Water, Superfund Program) were used if available. If not, we used EPA benchmarks used in regional programs (e.g., Superfund Program). If benchmarks were not available at a programmatic or regional level, we used benchmarks developed by other agencies (e.g., NOAA) or by state agencies.

In all cases (except for lead compounds, which were evaluated through a comparison to the NAAQS), we searched for benchmarks at the following three effect levels, as described in Section 2.6 of this report and in Appendix 9 (*Environmental Risk Screening Assessment*) of

this report: a no-effect level (i.e., NOAEL), threshold-effect level (i.e., LOAEL), and probable-effect level (i.e., PEL).

For some ecological assessment endpoint/environmental HAP combinations, we could identify benchmarks for all three effect levels, but for most we could not. In one case, where different agencies derived significantly different numbers to represent a threshold for effect, we included both. In several cases, only a single benchmark was available. In cases where multiple effect levels were available for a particular PB-HAP and assessment endpoint, we used all of the available effect levels to help us determine whether risk exists if risks could be considered significant and widespread.

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MEMORANDUM

TO: Eric Goehl, Paula Hirtz, Kaye Whitfield, and Kim Teal, U.S. EPA/OAQPS/SPPD

FROM: Eastern Research Group, Inc.

DATE: April 22, 2019

SUBJECT: Documentation for Surface Coating of Metal Coil Risk and Technology Review (RTR) Modeling File

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1.0 INTRODUCTION

The purpose of this memorandum is to provide information on the Environmental Protection Agency (EPA) Risk and Technology Review (RTR) database used in estimating residual risk for the National Emission Standards for Hazardous Air Pollutants (NESHAP) for Surface Coating of Metal Coil (40 CFR part 63, subpart SSSS). This NESHAP covers only Clean Air Act (CAA) section 112 major sources.

Section 112 of the CAA establishes a two-stage regulatory process to address emissions of hazardous air pollutants (HAPs) from stationary sources. In the first stage, section 112(d) requires the EPA to develop maximum achievable control technology (MACT)-based standards for categories of industrial sources. In the second stage, known as the residual risk stage, section 112(f)(2) requires EPA to assess the health and environmental risks that remain after sources come into compliance with the NESHAP based on the MACT. If additional risk reductions are necessary to protect public health with an ample margin of safety or to prevent adverse environmental effects, EPA must develop standards to address these remaining risks. As part of this second stage, data were gathered to assess the residual risks from the surface coating of metal coil source category.

2.0 FACILITY LIST DEVELOPMENT

The list of facilities potentially subject to the NESHAP for Surface Coating of Metal Coil was initially developed using the Office of Enforcement and Compliance Assurance's (OECA) Enforcement and Compliance History Online (ECHO) database tool in January of 2016 and again in 2017. The ECHO facility list was compared to a list of facilities provided with the original rulemaking. We then reviewed air permits (those available online) to help determine whether each facility remaining on the list is subject to 40 CFR part 63, subpart SSSS. The ECHO searches and permit reviews resulted in a list of 48 facilities subject to subpart SSSS that were included in the modeling file. The list of facilities is provided in Appendix A.

3.0 RECORDS ASSIGNED TO CATEGORY

Emissions data were pulled from the 2011 NEI v2 for the original ECHO facility list and from the 2014 NEI v1 for the additional facilities added from the second ECHO search. The

2014 NEI v1 data were not available at the time the data were compiled for the original ECHO facility list; the 2011 NEI v2 were the most recent data available at that time. The Source Classification Codes (SCCs) and unit descriptions were reviewed and compared to air permits, where available, in order to assign records to the metal coil coating category. The SCC and unit descriptions were also used to assign Emission Process Groups in the modeling file. The Emission Process Groups are used to understand the relative contribution of each type of process to the overall risk results after modeling is completed.

4.0 SPECIATION PROFILES

After records were assigned to the category, we performed an overall review of the emissions to determine if the data for each facility were both complete and representative. We found that some facilities reported organic and metal HAP for metal coil coating operations, while others reported only criteria pollutants such as particulate matter (PM) or criteria pollutant precursors such as volatile organic compounds (VOC).

4.1 ORGANIC SPECIATION PROFILE

Twenty-two of the 48 facilities in the final modeling file reported both VOC and organic HAP emissions for metal coil coating processes. The emissions for these facilities were used to develop speciation profiles using the weighted-average ratio of each organic HAP to VOC. The HAP-to-VOC ratio was calculated for each organic HAP as the sum of the reported HAP emissions divided by the sum of the reported VOC emissions for those processes with both VOC and HAP.

The NEI data included some organic HAP that were only reported once in the dataset. These HAP were excluded from the profile, because they were not considered to be typical or expected for all coil coating facilities. These HAP included N,N-dimethylformamide, tetrachloroethylene, and triethylamine. These HAP appear in the modeling file only for those facilities that reported them to the NEI.

A small amount of benzene (0.01 lb/yr) was reported for several process IDs related to coil coating at one facility. Because benzene was reported by only one facility, and because this HAP is not expected to be used as a coating solvent or a component of solvent blends, the benzene emissions were excluded from the speciation profile and the modeling file.

Some glycol ether emissions in NEI appear as a generic glycol ether pollutant code, while others reported emissions as 2-butoxyethyl acetate, diethylene glycol monobutyl ether, or butyl carbitol acetate. These pollutants and the generic glycol ether are assigned the same dose response values in risk modeling; thus emissions were combined to calculate a single glycol ether ratio value for the emissions profile.

The speciation profiles were applied to 12 facilities that reported VOC but no organic HAP associated with metal coil coating. The remaining 14 facilities in the category reported organic HAP for metal coil coating but did not report VOC for the same NEI process IDs or the total reported organic HAP exceeded the reported VOC for the process and was considered to be an outlier. The data for these 14 facilities were not used in developing the organic speciation profile.

4.2 METALS SPECIATION PROFILE

Two facilities reported emissions of lead and PM10-Primary from metal coil surface coating processes and eight facilities reported emissions of PM10-Primary from metal coil surface coating processes without reporting metal HAP. The facilities with both lead and PM10-Primary were used to develop a speciation profile based on the weighted-average ratio of lead to PM10-Primary. The lead to PM10-Primary ratio was calculated as the sum of the reported lead emissions divided by the sum of the reported PM10-Primary emissions for the processes with both PM10-Primary and lead. Six facilities reported metal HAPs from metal coil surface coating processes but did not report PM10-Primary for the same processes. Therefore, the metal HAP emissions from these six facilities were included in the modeling file but were not used in developing the speciation profile.

5.0 INDUSTRY REVIEW FILE

The EPA prepared an industry review file and shared it with the National Coil Coating Association for review and comment. We did not receive any revisions in response so no changes were made to the file.

6.0 QA OF MODELING FILE DATA

In addition to reviewing the modeling file for completeness of pollutants and emissions data, the stack and fugitive vent parameters and locational coordinates were also reviewed and revised as needed. This review is discussed in detail in sections 6.1 through 6.3.

6.1 STACK PARAMETERS

Stack parameter fields include stack (or fugitive) height, exit gas temperature, stack diameter, exit gas velocity, and exit gas flow rate. We used the stack parameter data reported in NEI where possible. If non-fugitive stack parameters were not populated in NEI, then we first attempted to calculate missing values. For example, missing flow rate coil be calculated using the reported diameter and velocity. If there was not enough information to calculate missing values or if reported values were outside of typical QA range check minimum and maximum values, we used averages based on other facilities in the category and approximate stack heights based on satellite imagery or images from Google Street View©.

Fugitive vents are required to have the same five stack parameter fields populated in the modeling file. The stack diameter, exit gas velocity, and exit gas flow rate were populated using national default values of 0.003 ft, 0.0003 ft/sec, and 0 actual cubic ft/sec, respectively. If fugitive height or temperature values were not provided in NEI, we assigned national default values of 10 ft and 72 degrees Fahrenheit, respectively.

6.2 LOCATIONAL COORDINATES

All latitudes and longitudes were mapped and reviewed using ArcGIS.

6.3 EMISSIONS VALUES

The 2011 NEI included emissions of hexavalent chromium for four Precoat Metals (Precoat) facilities. The EPA contacted Precoat and the National Coil Coating Association (NCCA), the industry trade association, to ask specifically about these emissions. No other

facilities in the source category reported chromium emissions. The EPA also conducted a site visit to the Precoat facility located in Columbia, South Carolina to review the coil coating operations and potential sources of these emissions. The Precoat representative stated that the hexavalent emissions were not reported by Precoat, but that the Precoat facilities report total chromium and the states and EPA apply different speciation factors to speciate these emissions into trivalent chromium and hexavalent chromium fractions in the inventories. He stated that the chromium releases were primarily offsite disposal releases, not air releases, and that the air releases of chromium were based on the historical assumption that 0.5 percent of the chromium in the applied prime coat was emitted to the air. He stated this assumption was made by the previous owners of the facilities and that Precoat had continued to use this factor. The factor was not based on any measured emissions, no testing was ever conducted, and no analytical data exist to support the factor. The NCCA confirmed that the hexavalent chromium emissions were the result of state-applied and EPA-applied factors to the total chromium reported by the facilities but that the coil coating processes and the oven temperatures did not support the emissions of hexavalent chromium reflected in the dataset.

During the Precoat site visit, EPA observed the coil coating application and curing processes. EPA observed that once the metal coil is cleaned, it goes through a pretreatment process consisting of a wet process followed by a dry process. The wet process consists of a series of inorganic chemical baths (can be either Ni/Zn or Ni/Zn/phosphate), followed by a dry chromic acid process that is roll applied to prepare the surface for coating. The following coating application process includes primers, some of which contain strontium chromium, that are also roll applied. The oven curing temperature for the prime coat at Precoat facilities vary depending on line speed but were reported to range well below temperatures that would cause any emissions of hexavalent chromium.

Because no data were available to support the use of this factor, and because there is no identifiable mechanism to cause these emissions (e.g., the material is roll applied and is not spray applied and oven temperatures are sufficiently low), the EPA did not apply chromium emissions to other facilities in the source category and removed these emissions from the modeling file.

7.0 ALLOWABLE EMISSIONS MULTIPLIER

Potential differences between actual emission levels and the maximum emissions allowable under the MACT standard (i.e., “allowable emissions”) were calculated for the metal coil coating source category. The MACT allowable emissions are the actual emissions multiplied by a factor that represents the difference between the actual emissions (which are based on actual production rates, the HAP content of the coatings and other materials used by the facility, the coating application method, and the add-on control device destruction efficiency or outlet concentration) and the MACT emission limits.

The portions of the metal coil coating line to which the NESHAP emission limits apply are the coating application stations and associated curing ovens. Compliance with the NESHAP emission limits can be achieved using several different options including: a compliant material as-purchased or as-applied (averaging) option, an emission rate with add-on controls option or a combination option of compliant coatings and an add-on control device. The standard also establishes operating limits for capture systems and add-on control devices. The compliance

options apply to an individual coil coating line, to multiple lines as a group, or to the entire affected source.

To use the as-purchased compliant material option facilities must demonstrate that the organic HAP content of each individual coating meets the emission limit (0.046 kg of organic HAP per liter solids). To use the as-applied compliant material option, facilities must demonstrate that the average of each coating or the average of all coating materials applied meets the emission limit (0.046 kg of organic HAP per liter solids). To use the emission rate with add-on controls option, facilities must demonstrate 1) an overall control efficiency of 98 percent or 2) an outlet concentration of 20 ppmv as carbon and 100 percent capture efficiency. (If facilities use a solvent recovery system, they may determine the overall control efficiency using a liquid-liquid material balance instead of conducting an initial performance test.) To use any compliance option involving an add-on control, except for a solvent recovery system and using a material balance, facilities are required to install control device parameter monitoring system equipment to demonstrate compliance with operating limits established for the emission capture and control systems.

During development of the 2002 MACT, the EPA conducted a survey and results were received from 89 major source coil coating facilities operating a total of 105 coil coating lines. According to the survey results, 75 percent of the coil coating lines employed the use of thermal oxidizers to control organic emissions, 23 percent of the coil coating lines used catalytic oxidizers and 2 percent used condenser/scrubber systems.

As part of the RTR rulemaking process, operating permits for 39 major source coil coating facilities were reviewed and 37 were found to employ the use of add-on controls. Three of the operating permits showed that not all coating lines had add-on controls which indicates the facilities are complying with the NESHAP emission limits using the compliant material option or the emission averaging option.

The primary add-on controls for these 37 facilities are thermal oxidizers, regenerative thermal oxidizers, and catalytic oxidizers. Based on review of the facility operating permits, when a facility specific compliance option is listed, the control efficiency option (98 percent) is the primary option for these control devices. Further review of the facility operating permits show that emission reductions for several thermal oxidizers are performing slightly above the 98 percent control efficiency level required by the NESHAP (in the range of 98.5 to 99.9 percent) which suggests that the actual emissions are slightly lower than the NESHAP allowable level for these control devices. For the majority of the facilities the control efficiency from the initial performance test, or most recent performance test, if periodic testing is required, was not available. All facility operating permits reflected the NESHAP requirements for add-on control devices. For each add-on control, we concluded that the organic HAP emissions were either equal to or slightly lower than the MACT allowable levels.

Three of the operating permits showed that not all coating lines had add-on controls which indicates that the facility was complying with the NESHAP emission limits using the compliant material option or the emission averaging option. Facilities using these options are required to use compliant materials that meet the NESHAP emission rate limit (0.046 kg of organic HAP per

liter solids), either individually or collectively, during each monthly compliance period. For these processes we believe the actual emission levels for these metal coil coating operations are a reasonable estimation of levels allowed by the NESHAP.

To account for the slightly lower level of actual organic HAP emissions (due to the higher control efficiency of the add-on control devices) we chose to multiply the actual emissions by factor of 1.1 to represent the allowable emissions from the metal coil source category.

8.0 ACUTE EMISSIONS MULTIPLIER

As part of the risk assessment analysis, the EPA develops an acute risk estimate that is based on variations in the coating process and the expected highest annual hourly emissions for the source category. We based our estimate on the knowledge we gained about the source category during the RTR review process, site visits, information from coil coating facility contacts and the coil coating association, and review of facility operating permits.

The coil coating industry is mostly a toll industry, where a facility produces painted coil for many clients. The coil coating category is process specific, meaning the same basic process is used from facility to facility. The hours of production for each facility are based on customer demand, so some facilities run 12-hour shifts, while others run a full 8,760 hours per year. As described above in section 7.0, most coil coating operations roll-apply prime coat and finish coat to a moving metal strip (coil) in a continuous coating process (long batches) inside enclosures that are routed to add-on control devices. The coated coil is then cured in an oven that is routed to an add-on control device. Coil coating operations for three facilities roll apply compliant materials (as-purchased or as-applied) and do not require add-on controls.

The coils are seamed together end to end for continuous runs, resulting in consistent emissions. We expect there may be slight variations in emissions due to variations in the organic HAP content of the coatings from run to run. To account for this variability, we chose an acute multiplier of 1.1 for the coil coating category. Since coil coating is a continuous coating process, we don't expect significant changes in hour-to-hour emissions, which is typical of industries with batch operations. We believe that a default acute multiplier of 10 is not reasonable and does not appropriately represent the variation in hourly emissions for the coil coating category. Based on our knowledge of coil coating operations we believe that a conservative acute multiplier of 1.1 is appropriate to account for the slight variation in coating composition for this category.

Appendix A – Metal Coil Coating Modeling File Facility List

EIS Facility Identifier	Facility Name	Location Address	City	State	County
10642311	Precoat Metals	5888 East County Road 180	Blytheville	AR	Mississippi County
12577111	Precoat Metals Division Sequa Corporation	3399 Davey Allison Blvd	Hueytown	AL	Jefferson County
13431011	MSC Walbridge Coatings Inc (0387000046)	30610 East Broadway	Walbridge	OH	Wood County
16808311	Metal Coaters (1409000037)	2400 Yankee Road	Middletown	OH	Butler County
1866611	Wismarq Corp	11440 Addison St	Franklin Park	IL	Cook County
2551411	Signode Corp	7701 W 71st St	Bridgeview	IL	Cook County
2994211	Nichols Aluminum - Davenport - Rockingham Rd	1725 Rockingham Rd	Davenport	IA	Scott County
4019511	Precoat Metals Houston	16402 Jacinto Port	Houston	TX	Harris County
4785611	NCI Group Inc., Dba, Metal Coaters of CA	9133 Center Ave	Rancho Cucamonga	CA	San Bernardino County
4800211	Fuji Film Inc P Plant	211 Puckett Ferry Rd	Greenwood	SC	Greenwood County
5276311	Wolverine Gasket Division - Eagle-Picher Ind	3175 State Street	Blacksburg	VA	Montgomery County
5344111	Aleris Rolled Products Inc	1372 KY 1957	Lewisport	KY	Hancock County
5344211	Precoat Metals	2604 River Rd	Hawesville	KY	Hancock County
5694311	Wolverine Advanced Materials – Blacksburg	201 Industrial Park Rd	Blacksburg	VA	Montgomery County
6068311	Wismarq Corporation	930 Armour Rd	Oconomowoc	WI	Waukesha County
6234611	Wheeling Corrugating Company	West Virginia Route 2	Beech Bottom	WV	Brooke County
6235311	Precoat Metals	4502 Freedom Way	Weirton	WV	Brooke County
6263011	Edco Products Inc	8700 Excelsior Blvd	Hopkins	MN	Hennepin County
641911	Wolverine Advanced Materials	10825 County Road 44	Leesburg	FL	Lake County
6559911	Alumax Mill Prod Inc/Mill Prod	1480 Manheim Pike	Lancaster	PA	Lancaster County
6654511	Consolidated Systems, Inc	650 Rosewood Dr	Columbia	SC	Richland County
7212211	Constellium	3509 E 2nd St	Muscle Shoals	AL	Colbert County
7249611	Precoat Metals A Division Of Sequa Corp	1950 E Main St	Greenfield	IN	Hancock County
7304011	Precoat Metals-Precoat Metals	4301 S Spring	St. Louis	MO	St. Louis City
7316911	Material Sciences Corp	2300 E Pratt Blvd	Elk Grove Village	IL	Cook County
7330811	Novelis Corporation (0278080136)	390 Griswold Street, N.E.	Warren	OH	Trumbull County
7376711	Precoat Metals Division Sequa Coatings	6300 Us Hwy 12	Portage	IN	Porter County
7409411	US Steel Corp/Irvin Plt	Camp Hollow Rd	Dravosburg	PA	Allegheny County

EIS Facility Identifier	Facility Name	Location Address	City	State	County
7745111	Edcoat Limited Partnership	30350 Edison Road	New Carlisle	IN	St. Joseph County
7746911	USG Interiors, Inc., Americoil Metals Corp., Westlake (1318617346)	1000 Crocker Rd.	Westlake	OH	Cuyahoga County
7822411	Vulcraft Of New York Inc	5362 Railroad St	Chemung	NY	Chemung County
7874411	Brightsmith Llc/Falls Twp	120 Enterprise Ave	Morrisville	PA	Bucks County
7882611	Precoat Metals	25 Northgate Industrial Dr	Granite City	IL	Madison County
7891111	Alcoa Inc	4879 State St	Riverdale	IA	Scott County
7905611	Agfa Corporation	50 Meister Ave	Branchburg	NJ	Somerset County
7972711	Precoat Metals Corp.	2nd Rd & Hupp Rd	La Porte	IN	La Porte County
8109311	Logan Aluminum Inc	6920 Lewisburg Rd	Russellville	KY	Logan County
8115811	Ak Steel - Zanesville Works (0660010006)	1724 Linden Avenue	Zanesville	OH	Muskingum County
8163211	Aleris Rolled Products, Inc. (0165000045)	1 Reynolds Road	Ashville	OH	Pickaway County
8181811	Alcoa Inc. - Warrick Operations	State Road 66 & State Road 61	Newburgh	IN	Warrick County
8230511	Precoat Metals	1095 Mendell Davis Drive	Jackson	MS	Hinds County
8235311	Jupiter Aluminum Corporation Jupiter	205 E Carey St	Fairland	IN	Shelby County
8267211	Crown Cork & Seal (0448010466)	5201 Enterprise Blvd	Toledo	OH	Lucas County
8373611	Centria (0630010006)	530 North Second Street	Cambridge	OH	Guernsey County
8405611	Springs Window Fashions/Montgomery Plt	8601 State Route 405	Montgomery	PA	Lycoming County
8463911	Ply Gem (0575010103)	2615 Campbell Road	Sidney	OH	Shelby County
8499411	Metal Coaters	951 Prisock Road	Byram	MS	Hinds County
948011	Southwire Company –Carrollton	One Southwire Drive	Carrollton	GA	Carroll County

Appendix 2
Technical Support Document for HEM-AERMOD Modeling

Modeling for the Residual Risk and Technology Review Using the Human Exposure Model 3 – AERMOD Version

Updated 7/31/2017

Technical Support Document

Prepared for:

U.S. Environmental Protection Agency
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Office of Air Quality Planning and Standards
Health and Environmental Impacts Division
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Disclaimer

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Contents

1. Introduction 1

2. Overview of the HEM-3 – AERMOD System 2
2.1 Preparation of Dispersion Modeling Inputs 3
2.1.1 Compiling Emission Source Data 3
2.1.2 Defining the Modeling Domain and Receptors 4
Treatment of Nearby Census Blocks and Screening for Overlapping Blocks 5
Polar receptor network 5
Elevations and hill heights for model receptors 6
2.1.3 Selection of Meteorological Data 6
2.2 Running of AERMOD 6
2.2.1 AERMOD Dispersion Options Used by HEM-3 6
2.2.2 Use of Dilution Factors 7
2.3 Postprocessing of AERMOD Results in HEM-3 8
2.3.1 Calculation of Impacts at Modeled Receptors 9
2.3.2 Interpolation of Impacts at Outer Census Blocks 10
2.3.3 Calculation of Population Exposures and Incidence 11
2.3.4 Model Outputs 12
2.4 Data Libraries Used in HEM-3 13
2.4.1 Chemical Health Effects Information 13
2.4.2 Census Block Locations and Elevation Data 14
2.4.3 Meteorological Data 15
2.4.4 Gaseous Deposition Parameters 15

3. Modeling for the Residual Risk Technology Review 15
3.1 Emission Source Inputs 15
3.2 Pollutant Cross-Referencing 16
3.3 Meteorological Data 17
3.4 Model Options Selected 19
3.4.1 Urban or Rural Dispersion Characteristics 19

3.4.2 Deposition and Plume Depletion 19
3.4.3 Cutoff Distance for Modeling of Individual Blocks 20
3.4.4 Facility Boundary Assumptions 20
3.5 Modeling of Multiple Facilities 22

4. Quality Assurance 23
4.1 Engineering Review 23
4.2 Geographic Pre-Modeling Checks 23
4.3 Geographic Post-Modeling Checks 24

5. Uncertainties 26

6. References 32

Figures

Figure 3-1. HEM-3 Meteorological Stations 18

Tables

Table 2-1. Parameters Used to Delineate the Modeling Domain in HEM-3 4

Table 3-1. HEM-3 Domain and Set-Up Options As Used in the Residual Risk and Technology Review Assessments 21

Table 5-1. Summary of General Uncertainties Associated with Risk and Technology Review Risk Assessments 28

1. Introduction

This document describes the general modeling approach used to estimate the risks to human populations in support of the Residual Risk and Technology Review (RTR) currently being carried out by the U.S. Environmental Protection Agency (EPA). It is important to note that risk characterizations of individual source categories under the RTR program may not follow every item/approach noted in this document. The reader is referred to the main body of the risk assessment document for more details on source category specific approaches that may have been included in the analysis.

The model used in these risk assessments is the Human Exposure Model, Version 3 (HEM-3). HEM-3 incorporates AERMOD, a state of the art air dispersion model developed under the direction of the American Meteorological Society / Environmental Protection Agency Regulatory Model Improvement Committee (AERMIC).

Section 2 of this report provides an overview of the HEM-3-AERMOD system; and Section 3 describes inputs and choices made in implementing the model for the RTR program. Quality assurance efforts undertaken in the modeling effort are discussed in Section 4, and uncertainties associated with the modeling effort are discussed in Section 5.

2. Overview of the HEM-3 – AERMOD System

HEM-3 performs three main operations: dispersion modeling, estimation of population exposure, and estimation of human health risks. The state-of-the-art American Meteorological Society (AMS) / EPA Regulatory Model (AERMOD)[1,2] is used for dispersion modeling. AERMOD can handle a wide range of different source types which may be associated with an industrial source complex, including stack (point) sources, area and polygon sources, volume sources, line and buoyant line sources.

To prepare dispersion modeling inputs and carry out risk calculations, HEM-3 draws primarily on three data libraries, which are provided with the model. The first is a library of meteorological data for over 800 stations, which are used for dispersion calculations. A second library of Census block ("centroid") internal point locations and populations provides the basis of human exposure calculations. The Census library also includes the elevations of every Census block, which are used in the dispersion calculations for the RTR assessments. A third library of pollutant unit risk estimates and reference concentrations is used to calculate population risks. These unit risk estimates (URE) and reference concentrations (RfCs) are based on the latest dose response values recommended by EPA for hazardous air pollutants (HAPs) and other toxic air pollutants. A fourth data library, which contains deposition parameters for gaseous pollutants, is also provided with HEM-3 but used only when the user opts to compute gaseous deposition with or without plume depletion. (Note: Deposition has not been computed for the RTR assessments to date).

HEM-3 has been implemented in two versions: a single facility version ("Single HEM-3"), and a multiple facility version ("Multi HEM-3"). Multi HEM-3 is used in the RTR risk assessment modeling. Both versions operate under the same general principles. In essence, Multi HEM-3 provides a platform for running the single facility version multiple times. In both versions, source location and emissions data are input through a set of Excel™ spreadsheets. The main difference is in the user interface for other model inputs. Single HEM-3 includes a graphical user interface (GUI) for the selection of various dispersion modeling options. In Multi HEM-3, a control file replaces many of these GUI inputs.

The model estimates cancer risks and non-cancer adverse health effects due to inhalation exposure at Census block internal point locations (or "centroids"), at concentric rings surrounding the facility center, and at other receptor locations that can be specified by the user. Cancer risks are computed using EPA's recommended unit risk estimates for HAPs and other toxic air pollutants. The resulting estimates reflect the excess cancer risk for an individual breathing the ambient air at a given receptor site 24-hours per day over a 70-year lifetime. The model estimates the numbers of people exposed to various cancer risk levels. In addition, HEM-3 estimates the total incremental cancer risks for people living within different distances of the modeled emission sources.

Potential non-cancer health effects due to chronic exposures are quantified using hazard quotients and hazard indices for various target organs. The "hazard quotient" (HQ) for a given chemical and receptor site is the ratio of the ambient concentration of the chemical to the reference concentration. The "hazard index" (HI) for a given organ is the sum of hazard quotients for substances that affect that organ. HEM-3 computes target-organ-specific hazard

indices (TOSHIs) for HAPs and other toxic air pollutants, and estimates the numbers of people exposed to different hazard index levels. In addition, maximum short term ("acute") concentrations are computed for all pollutants, and concentrations are compared with various threshold levels for acute health effects.

The following sections outline the methodologies used in the HEM-3–AERMOD system. Section 2.1 describes the preparation of dispersion modeling inputs, Section 2.2 describes the running of AERMOD, Section 2.3 describes calculations performed by HEM-3 to calculate risks and exposures, and Section 2.4 details the sources and methods used to produce HEM-3's data libraries. The HEM-3 User's Manuals – for Single HEM-3 and Multi HEM-3 – provide additional details on the input data and algorithms used in the model.[3] Specific model options used in the RTR assessments are discussed in Chapter 3.

2.1 Preparation of Dispersion Modeling Inputs

HEM-3 compiles data that will be needed for dispersion modeling, and prepares an input file suitable for running AERMOD. The dispersion modeling inputs can be divided into three main components: emission source data, information on the modeling domain and receptors for which impacts will be computed, and meteorological data.

2.1.1 Compiling Emission Source Data

A series of Excel™ spreadsheet files are used to specify the emissions and configuration of the facility to be modeled. At a minimum, two files are needed: a HAP emissions file, and an emissions location file. The HAP emissions file includes an emission source identification code for each emission source at the facility, the names of pollutants emitted by each source, and the emission rate for each pollutant. In addition, if the model run is to incorporate deposition or plume depletion, the HAP emissions file must also specify the percentage of each pollutant that is in the form of particulate matter. The balance is assumed to be in gaseous/vapor form.

The emissions location file includes the coordinates of each source, as well as information on the configuration and other characteristics of the source. HEM-3 can analyze point sources, area and polygon sources, volume sources, and line and buoyant line sources - configurations that are described in AERMOD's documentation.[1, 2] For stack (point) sources, such as a vertical non-capped, capped or horizontal stacks the emissions location file must provide the stack height, stack diameter, exit velocity, and emission release temperature. The file must also provide dimensions for each area, polygon, volume or line source, as well as the height of the source above the ground. For area sources, the angle of rotation from north can also be specified. The user can also provide the terrain elevation at the base of each source. (The controlling hill height is also used in AERMOD's flow calculations. Calculation of the controlling hill height by HEM-3 is discussed in Section 2.4.2.) If the terrain elevations are not provided by the user, HEM-3 will calculate elevations and controlling hill heights based on elevations and hill heights provided by the Census database for the Census blocks nearest to the facility.

If buoyant line source types are to be considered, particularly when computing building downwash effects, then HEM-3 requires an additional input file to specify the source type's

parameters. For buoyant line sources, the average buoyancy parameter, the average building dimensions (i.e., average building length, height, and width), the average line source width, and the average separation distance between buoyant lines are required inputs for an associated buoyant line parameters input file.

If particulate deposition and plume depletion are to be considered, then HEM-3 requires an input file to specify the particle size distribution. This input file must include the average particle diameter, the mass fraction percentage, and the average particle density for each size range emitted. Another optional file can be used to specify building dimensions if building wake effects are to be modeled.

2.1.2 Defining the Modeling Domain and Receptors

HEM-3 defines a modeling domain for each facility that is analyzed based on parameters specified by the model user or calculated by the model. These parameters are summarized in Table 2-1. The modeling domain is circular, and is centered on the facility, with a radius specified by the user. For the RTR analysis, the radius of the modeling domain is 50 kilometers (km). HEM-3 identifies all of the Census block locations in the modeling domain from its Census database, and divides the blocks into two groups based on their distance from the facility. For the inner group of Census blocks (closest to the facility), each block location is modeled as a separate receptor in AERMOD. The cutoff distance for modeling individual Census blocks is generally set to 3,000 m (3 km) for the RTR assessments, although it can be set differently by the model user. The model user can also provide an Excel™ spreadsheet specifying additional locations to be included as model receptors in AERMOD. These additional discrete "user receptors" may include facility boundary locations, monitoring sites, individual residences, schools, or other locations of interest.

Table 2-1. Parameters Used to Delineate the Modeling Domain in HEM-3

Parameter	Typical value
Modeling domain size – maximum radial distance to be modeled from facility center	50 km
Cutoff distance for modeling of individual blocks[a]	3,000 m
Overlap distance – where receptors are considered on facility property[a]	30 m
Polar receptor network:	
Distance to the innermost ring[b]	≥100 m
Number of concentric rings	13
Number of radial directions	16

[a] Measured from each stack at the facility, and from the edges of each area or volume source.

[b] Generally model-calculated to encompass all emission sources but not less than 100 meters from the facility center.

For Census blocks in the outer group, beyond this modeling cutoff distance, emission impacts are interpolated based on modeling results for a polar receptor network. The user also specifies an

"overlap" distance, within which Census block coordinates will be considered to be on facility property. The following paragraphs provide more details on the treatment of blocks near the facility, on the polar receptor network, and on the determination of receptor elevations and controlling hill heights to be used in AERMOD.

Treatment of Nearby Census Blocks and Screening for Overlapping Blocks

Census block locations near the facility are modeled as separate receptors within AERMOD. The cutoff distance for modeling of individual Census blocks may be chosen by the user, but is typically 3,000 meters for the RTR assessments. This distance is not measured from the center of the facility, but is the minimum distance from any source at the facility. Therefore, any Census block location that is within the cutoff distance from any emission source is treated as a discrete AERMOD receptor.

HEM-3 checks Census blocks that are very close to the facility in order to assess whether they overlap any point, area, volume, line or buoyant line emission sources. In addition, the user can specify an overlap distance, within which receptors will be considered to be on facility property. The default value for the overlap distance is 30 meters, or approximately equal to the width of a narrow buffer and a roadway. HEM-3 tests each nearby receptor to determine whether it is within this distance from any stack or from the perimeter of any area, volume, line or buoyant line source. If a receptor falls within this distance, HEM-3 will not calculate risks based on the location of that receptor, but will instead assume that the risks associated with the receptor are the same as the highest predicted value for any receptor that is not overlapping. The location for calculating the default impact may be either another Census block, one of the polar grid receptors, or one of the additional discrete user-specified receptor locations. [Note: An exception to this occurs when modeling polygon sources. Unlike other sources, when modeling polygons, the overlap function is disabled. This allows the impacts for a census tract modeled as a polygon source (e.g. mobile source emissions modeled uniformly across a census tract) to be calculated within the census tract being modeled.]

Polar receptor network

The polar receptor network used in HEM-3 serves three functions. First, it is used to estimate default impacts if one or more Census locations are inside the overlap cutoff distance used to represent the facility boundary. Second, it is used to evaluate potential acute effects that may occur due to short-term exposures in locations outside the facility boundary. Third, the polar receptor network is used to interpolate long-term and short-term impacts at Census block locations that are outside the cutoff distance for modeling of individual blocks.

Generally, the model calculates the inner radius (or first ring distance) for the polar receptor network to be just outside the emission source locations, but not less than 100 meters from the facility center. However, the user can override the default distance calculated by the model to fit the size and shape of the facility properties to be modeled. Likewise, the model will also use default values for the number of concentric rings to be analyzed (13 rings by default), and the number of radial directions (16 radials by default), although these default values can also be changed by the user to meet the needs of a specific modeling study. The inner radius of the

polar network should be the minimum distance from the facility center that is generally outside of facility property. (For complex facility shapes, it is sometimes useful to specify an inner ring that encroaches on facility property in some directions.) HEM-3 will distribute the radial directions evenly around the facility. For the concentric rings, the model will generate a logarithmic progression of distances starting at the inner ring radium and ending at the outer radium of the modeling domain.

Elevations and hill heights for model receptors

HEM-3 includes terrain elevations by default for the RTR assessments, but the user can choose to exclude terrain effects when running AERMOD. If the default terrain option is used, HEM-3 obtains elevations and controlling hill heights for Census block receptors from its internal Census location library. Section 2.4.2 describes the derivation of these elevations and hill heights.

Elevations and controlling hill heights for the polar grid receptors are also estimated based on values from the Census library. HEM-3 divides the modeling domain into sectors based on the polar receptor network, with each Census block assigned to the sector corresponding to the closest polar grid receptor. Each polar grid receptor is then assigned an elevation based on the highest elevation for any Census block in its sector. The controlling hill height is also set to the maximum hill height within the sector. If a sector does not contain any blocks, the model defaults to the elevation and controlling hill height of the nearest block outside the sector.

2.1.3 Selection of Meteorological Data

In addition to source and receptor information, AERMOD requires surface and upper air meteorological observations in a prescribed format. The model user can select a meteorological station from the HEM-3 meteorological data library, or add new files to the library if site-specific data are available. If the user does not specify a meteorological station, HEM-3 will select the closest station to the center of the modeling domain, as is generally done for the RTR assessments.

2.2 Running of AERMOD

Based on the user input data and other data described in the previous section, HEM-3 produces an input file suitable for AERMOD. HEM-3 then runs AERMOD as a compiled executable program. No changes have been made from the version of AERMOD released to the public by EPA. The following sections give additional information on how AERMOD is used within HEM-3.

2.2.1 AERMOD Dispersion Options Used by HEM-3

AERMOD provides a wide array of options for controlling dispersion modeling calculations. In general, HEM-3 uses the regulatory default options when running AERMOD.[1] These options include the following:

- Use stack-tip downwash (except for Schulman-Scire downwash);
- Use buoyancy-induced dispersion (except for Schulman-Scire downwash);
- Do not use gradual 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 profile exponents;
- Use low wind speed threshold;
- Use default vertical potential temperature gradients;
- Use of missing-data processing routines; and
- Consider terrain effects.

The following additional AERMOD options are available to the HEM-3 user:

- Calculation of wet and dry deposition rates for gaseous and particulate pollutants;
- Consideration of plume depletion (due to deposition) when calculating air concentrations;
- Consideration of building wake effects;
- Calculation of short term (acute) impacts;
- Use of the FASTALL option, which conserves model runtime by simplifying the AERMOD algorithms used to represent meander of the pollutant plume; and
- Use of the buoyant line plume option.

As noted in Section 2.1, the calculation of deposition or depletion and the consideration of building wake effects require additional user inputs.

The user can opt to analyze short term impacts on a number of different time scales (i.e., 1 hour, 2 hours, 3 hours, 4 hours, 6 hours, 8 hours, 12 hours, or 24 hours) however only one short term time scale can be selected per run. If the user chooses to analyze short term (acute) impacts, a multiplier must be specified to reflect the ratio between the maximum short term emission rate and the long term average emission rate. If available, acute multipliers specific to source classification codes (SCCs) are used in RTR assessments. If SCC-specific acute multipliers are not available, the default multiplier for short term emissions is a factor of 10. This means that in the default case the maximum short term emission rate is assumed to be 10 times the long term average emission rate. The multiplier can be set to one (1) if emissions from the facility are known to be constant. For RTR assessments, acute impacts are generally included in the modeling and the default multiplier of 10 is used, unless more source-specific information is available upon which to base the acute factor for the source category being modeled.

2.2.2 Use of Dilution Factors

To save computer run time when analyzing the impacts of multiple pollutants, HEM-3 does not model each pollutant separately. Instead, AERMOD is used to compute a series of dilution factors, specific to each emission source and receptor. The dilution factor for a particular

emission source and receptor is defined as the predicted ambient impact from the given source and at the given receptor, divided by the emission rate from the given source.

If the user chooses not to analyze deposition (with or without plume depletion), the dilution factor does not vary from pollutant to pollutant. If deposition and/or depletion is chosen as a model option, separate dilution coefficients must be computed for each gaseous pollutant. In addition, separate dilution factors must be computed for different components of particulate matter if the components do not have the same particle size distribution. In the current version of HEM-3, this can be done by creating a separate emission record for each pollutant emitted by from each source. (Common location data and source configurations can be used for different pollutant records representing the same emission source.)

2.3 Postprocessing of AERMOD Results in HEM-3

HEM-3 estimates total excess cancer risks and potential chronic non-cancer health effects for all Census block locations in the modeling domain, all user-defined receptors, and all points in the polar receptor network. Potential chronic non-cancer health effects are expressed in terms of TOSHI. Based on the results for Census blocks and other receptors, HEM-3 estimates the maximum individual risk (MIR) and maximum TOSHI for populated receptors, and determines the locations of these maximum impacts. The model also determines the concentrations of different pollutants at the site(s) of maximum risk and maximum TOSHI, and the contributions of different emission sources to these locations of maximum impact. It should be noted that the locations of maximum impact may differ for the maximum individual cancer risk and for the hazard indices of different target organs.

For acute impacts, HEM-3 calculates the maximum short term concentrations for all pollutants emitted by the facility. These maximum short term concentrations are compared with various threshold levels for acute health effects (e.g., the California EPA reference exposure level [REL] for no adverse effects).

At the option of the model user, HEM-3 will also compute the long term and short term predicted ambient concentrations of all pollutants emitted by the facility at all of the receptors in the modeling domain. In addition, pollutant contributions from each emission source at the facility are computed under this option. In RTR assessments, this option is standard and concentrations are computed for all receptors.

Section 2.3.1 describes methods used to calculate cancer risks and hazard indices for receptors that are explicitly modeled using AERMOD. Section 2.3.2 describes the interpolation approach used to estimate cancer risks and hazard indices at Census blocks that are not explicitly modeled.

2.3.1 Calculation of Impacts at Modeled Receptors

As noted in Section 2.2.2, HEM-3 does not model each pollutant separately unless deposition or depletion is being analyzed. Instead, AERMOD is used to compute a series of dilution factors, specific to each emission source and receptor. The following algorithms are used to compute cancer risks and TOSHI for chronic non-cancer health effects.

For cancer risk:

$$CR_T = \sum_{i,j} CR_{i,j}$$

$$CR_{i,j} = DF_{i,j} \times CF \times \sum_k [E_{i,k} \times URE_k]$$

For TOSHI:

$$TOSHI_T = \sum_{i,j} TOSHI_{i,j}$$

$$TOSHI_{i,j} = DF_{i,j} \times CF \times \sum_k [E_{i,k} / RfC_k]$$

where:

CR_T =	total cancer risk at a given receptor (probability for one person)
$\sum_{i,j}$ =	the sum over all sources i and pollutant types j (particulate or gas)
$CR_{i,j}$ =	cancer risk at the given receptor for source i and pollutant type j
$DF_{i,j}$ =	dilution factor [(μg/m^3) / (g/sec)] at the given receptor for source i and pollutant type j
CF =	conversion factor, 0.02877 [(g/sec) / (ton/year)]
$\sum_k$ =	sum over all pollutants k within pollutant type j (particulate or gas)
$E_{i,k}$ =	emissions of pollutant k from source i and in pollutant type j
URE_k =	cancer unit risk factor for pollutant k
$TOSHI_T$ =	total target-organ-specific hazard index at a given receptor
$TOSHI_{i,j}$ =	target-organ-specific hazard index at the given receptor for source i and pollutant type j
RfC_k =	non-cancer health effect reference concentration for pollutant k

The above equations are equivalent to the following simpler equations:

$$CR_T = \sum_{i,k} AC_{i,k} \times URE_k$$

$$TOSHI_T = \sum_{i,k} AC_{i,k} / RC_k$$

where:

$AC_{i,k}$ =	ambient concentration (μg/m^3) for pollutant k at the given receptor. This is the same as [$E_{i,k} \times DF_{i.j} \times CF$]

However, use of these simpler equations would require modeling all pollutants individually in AERMOD, and performing separate risk calculations for each pollutant.

If the cancer unit risk estimate is not available for a given chemical, then that chemical is not included in the calculation of cancer risk. Likewise, if the non-cancer reference concentration is not available for a given chemical, that chemical is not included in the calculation of hazard indices. Note also that separate reference concentrations are used for acute and chronic hazard indices.

HEM-3 computes short term concentrations and records the highest short term concentration for each pollutant. In addition, the user can opt to compute and record the short term and long concentrations at each receptor. Concentrations are computed as follows.

Long term concentrations:

$$AC_{T,k} = \sum_i AC_{i,k}$$

$$AC_{i,k} = E_{i,k\,x}\, DF_{i,j} \times CF$$

Short term concentrations:

$$AC_T = \sum_i AC_{i,k}$$

$$AC_{i,k} = E_{i,k\,x}\, DF_{i,j} \times CF \times M$$

where:

$AC_{T,k}$ = total estimated ambient concentration for pollutant k at a given receptor

$\sum_i$ = the sum over all sources i ($\mu g/m^3$)

$AC_{i,k}$ = estimated ambient concentration of pollutant k at the given receptor as a result of emissions from source i ($\mu g/m^3$)

M = ratio between the estimated maximum short term emission rate and the long term average emission rate (dimensionless)

2.3.2 Interpolation of Impacts at Outer Census Blocks

For Census blocks outside of the cutoff distance for individual block modeling, HEM-3 estimates cancer risks and hazard indices by interpolation from the polar receptor network. HEM-3 estimates impacts at the polar grid receptors using AERMOD modeling results and the algorithms described in Section 2.3.1. If terrain elevation is part of the modeling, then an elevation is estimated for each polar receptor. HEM-3 estimates elevations and controlling hill heights for the polar grid receptors based on values from the census library. HEM-3 divides the modeling domain into sectors based on the polar grid receptor network, with each census block assigned to the sector corresponding to the closest polar grid receptor.

HEM-3 then assigns each polar grid receptor an elevation based on the highest elevation for any census block in its sector. The controlling hill height is also set to the maximum hill height within the sector. If a sector does not contain any blocks, the model defaults to the elevation and controlling hill height of the nearest block outside the sector.

HEM-3 interpolates the impacts at each outer Census block from the four nearest polar grid receptors. The interpolation is linear in the angular direction, and logarithmic in the radial direction, as summarized in the following equations:

$$I_{a,r} = I_{A1,r} + (I_{A2,r} - I_{A1,r}) \times (a - A1) / (A2 - A1)$$

$$I_{A1,r} = \exp\{(\ln (I_{A1,R1}) + [(\ln (I_{A1,R2}) - \ln (I_{A1,R1})] \times [(\ln r) - \ln(R1)] / [\ln(R2) - \ln(R1)]\}$$
$$I_{A2,r} = \exp\{(\ln (I_{A2,R1}) + [(\ln (I_{A2,R2}) - \ln (I_{A2,R1})] \times [(\ln r) - \ln(R1)] / [\ln(R2) - \ln(R1)]\}$$

where:

$I_{a,r}$ = the impact (cancer risk, hazard index, or concentration) at an angle, a, from north, and radius, r, from the center of the modeling domain

a = the angle of the target receptor, from north

r = the radius of the target receptor, from the center of the modeling domain

A1 = the angle of the polar network receptors immediately counterclockwise from the target receptor

A2 = the angle of the polar network receptors immediately clockwise from the target receptor

R1 = the radius of the polar network receptors immediately inside the target receptor

R2 = the radius of the polar network receptors immediately outside the target receptor

2.3.3 Calculation of Population Exposures and Incidence

Using the predicted impacts for Census blocks, HEM-3 estimates the numbers of people exposed to various cancer risk levels and TOSHI levels. This is done by adding up the populations for receptors that have predicted cancer risks or TOSHI above the given threshold.

The model also estimates the annual excess cancer risk (incidence) for the entire modeling region. The following equation is used:

$$TCR = \sum_m [CR_m \times P_m] / LT$$

where:

TCR = the estimated annual cancer incidence (excess cancers/year) to the population living within the modeling domain

$\sum_m$ = the sum over all Census blocks m within distance the modeling domain

CR_m = the total lifetime cancer risk (from all modeled pollutants and emission sources) at Census block m

P_m = the population at Census block m

LT = the average lifetime used to develop the cancer unit risk factor, 70 years

HEM-3 also estimates the contributions of different chemicals and emission sources to total annual cancer incidence for the overall modeling domain using the following equations:

$$TCR_{i,j} = \sum_m [(\sum_k E_{i,k} \times URE_k) \times DF_{i,j,m} \times CF / LT]$$

$$TCR_{i,k} = TCR_{i\,j} \times E_{i,k} \times URE_k / (\sum_k E_{i,k} \times URE_k)$$

where:

$TCR_{i,j}$ = the estimated total annual cancer incidence (cancers/year) to the population in the modeling domain due to emissions from pollutant type j (1 = particulate, 2 = gas) and emission source i

$\sum_m$ = the sum over all Census blocks m within distance the modeling domain
$\sum_k$ = the sum over all pollutant k, within pollutant type j
$E_{i,k}$ = emissions of pollutant k from source i (tons/year)
URE_k = unit risk factor for pollutant k
$DF_{i,j,m}$ = dilution factor at receptor m, for emissions of pollutant type j (which includes pollutant k), from source i
CF = conversion factor, 0.02877 [(g/sec) / (tons/year)]
$TCR_{i,k}$ = the estimated annual cancer incidence (cancers/year) of the population in the modeling domain due to emissions of pollutant k (in pollutant type j) from emission source i

2.3.4 Model Outputs

The following is a summary of the outputs produced by HEM-3. These are written to a collection of files in Excel™ and dBase™ format (dbf).

- Maximum long term impacts at populated locations
    - maximum long term ambient concentration for each chemical
    - maximum lifetime individual cancer risk (MIR)
    - maximum TOSHI for the following health effects
      - respiratory system effects
      - liver effects
      - neurological system effects
      - developmental effects
      - reproductive system effects
      - kidney effects
      - ocular system effects
      - endocrine system effects
      - hematological system effects
      - immunological system effects
      - skeletal system effects
      - spleen effects
      - thyroid effects
      - whole body effects
    - locations of the maximum cancer risk and maximum TOSHIs
    - Census block identification codes for the maximum concentration, maximum cancer risk and maximum TOSHIs, and number of people in the Census block
    - contributions of different chemicals and emission sources to the maximum risk and TOSHI

- Maximum acute impacts
    - maximum short term ambient concentration for each chemical
    - threshold levels for acute health effects of each chemical (compared with the maximum short term concentrations)
    - locations of the maximum impacts for different chemicals (often polar receptors)
    - Census block identification codes at the locations of maximum concentration, and number of people in the block

    - contribution of each emission source at the facility to the maximum short term concentration of each chemical

- Outputs for all receptors
    - maximum individual cancer risk and TOSHI (all target organs) for each Census block and each user-specified discrete receptor (monitoring sites, etc.)
    - maximum individual cancer risk and TOSHI (all target organs) for each polar grid receptor
    - estimated deposition flux (optional)
    - predicted ambient concentration resulting from each emission source at each Census block and polar grid receptor (optional)

- Population exposures and total cancer risk, or incidence
    - estimated numbers of people exposed to different levels of lifetime individual cancer risk (1 in a million, 1 in 100,000, etc.)
    - estimated numbers of people exposed to different levels of TOSHI (1, 2, 10, etc.)
    - total cancer risk, or incidence, in estimated cancer deaths per year, over the entire modeling domain, and for each pollutant and source combination

2.4 Data Libraries Used in HEM-3

2.4.1 Chemical Health Effects Information

HEM-3 includes a library of available health effects data for HAPs. For each pollutant, the library includes the following parameters, where available:

- unit risk estimate (URE) for cancer;
- reference concentration (RfC) for chronic non-cancer health effects;
- reference benchmark concentrations for acute health effects; and
- target organs affected by the chemical for chronic non-cancer health effects.

Unit risk estimates and reference concentrations included in the HEM-3 chemical library have been taken from EPA's database of recommended dose-response factors for HAPs, which is updated periodically, consistent with continued research on these parameters.[4] The URE represents the upper-bound excess lifetime cancer risk estimated to result from continuous exposure to an agent (HAP) at a concentration of 1 microgram per cubic meter ($\mu g/m^3$) in air (e.g., if the URE = 1.5×10^{-6} per $\mu g/m^3$, then 1.5 excess tumors are expected to develop per 1 million people if all 1 million people were exposed daily for a lifetime to 1 microgram of the chemical in 1 cubic meter of air). UREs are considered plausible upper limits to the true value; the true risk is likely to be less but could be greater.[5]

The RfC is a concentration estimate of a continuous inhalation exposure to the human population that is likely to be without an appreciable "risk" of deleterious non-cancer health effects during a lifetime (including to sensitive subgroups such as children, asthmatics and the elderly). No adverse effects are expected as a result of exposure if the ratio of the potential

exposure concentration to the RfC, defined as the hazard quotient (HQ), is less than one. Note that the uncertainty of the RfC estimates can span an order of magnitude.[5] Target organs are those organs (e.g., kidney) or organ systems (e.g., respiratory) which may be impacted with chronic non-cancer health effects by exposure to the chemical in question. The hazard index (HI) is the sum of HQs for substances that affect the same target organ or organ system, also known as the target organ specific hazard index (TOSHI).

The reference benchmark concentration for acute health effects, similar to the chronic RfC, is the concentration below which no adverse health effects are anticipated when an individual is exposed to the benchmark concentration for 1 hour (or 8 hours, depending on the specific acute benchmark used and the formulation of that benchmark). A more in-depth discussion of the development and use of these parameters for estimating cancer risk and non-cancer hazard may be found in the EPA's Air Toxics Risk Assessment Library.[6]

The model user can add pollutants and associated health effects to HEM-3's chemical health effects (dose-response and target organ endpoints) library, as needed.

2.4.2 Census Block Locations and Elevation Data

The HEM-3 Census library includes Census block identification codes, locations, populations, elevations, and controlling hill heights for all of the over 6 million Census blocks identified in the 2010 Census and the over 5 million Census blocks identified in the 2000 Census. The model user may choose to use either Census database according to their modeling needs. The location coordinates reflect the internal "centroid" of the block, which is a point selected by the Census to be roughly in the center of the block. For complex shapes, the internal point may not be in the geographic center of the block. Locations and population data for Census blocks in the 50 states and Puerto Rico were extracted from the LandView® database For the 2000 Census[7] and from the U.S. Census Bureau website for the 2010 Census.[8] Locations and populations for blocks in the Virgin Islands were obtained from the U.S. Census Bureau website.

U.S. Geological Survey data was used to estimate the elevation of each census block in the continental U.S. and Hawaii. The data used for the 2000 Census elevations have a resolution of 3 arc-seconds, or about 90 meters.[9] The data used for the 2010 Census elevations have a resolution of 1/3 of an arc second, or about 10 meters.[10] Using analysis tools (ArcGIS® 9.1 software application for the 2000 Census, and ArcGIS® 10 for the 2010 Census), elevation was estimated for each census block in Alaska and the U.S. Virgin Islands. The point locations of the census blocks in Alaska and the U.S. Virgin Islands were overlaid with a raster layer of North American Digital Elevation Model (DEM) elevations (in meters).[9] An elevation value was assigned to each census block point based on the closest point in the ArcGIS elevation raster file.

An algorithm used in AERMAP, the AERMOD terrain processor, is used to determine controlling hill heights.[11,12] These values are used for flow calculations within AERMOD. To save run time and resources, the HEM-3 census block elevation database is substituted for the DEM data generally used in AERMAP. As noted above, the census block elevations were originally derived from the DEM database. To determine the controlling hill height for each census block, a cone is projected away from the block centroid location, representing a 10%

elevation grade. The controlling hill height is selected based on the highest elevation above that 10% grade (in accordance with the AERMAP methodology). The distance cutoff for this calculation is 100 km. (This corresponds to an elevation difference at a 10% grade of 10,000 m, which considerably exceeds the maximum elevation difference in North America.)

2.4.3 Meteorological Data

HEM-3 includes an extensive library of meteorological data to support the AERMOD dispersion model. Currently over 800 meteorological stations have been preprocessed for AERMOD as part of the RTR effort. Section 3.3 includes a depiction of these meteorological stations and Appendix 3 discusses the preparation of meteorological data for the RTR in more detail.

2.4.4 Gaseous Deposition Parameters

HEM-3 provides options to compute the deposition of air pollutants, and to take into account the impacts of plume depletion due to deposition of gaseous and particulate pollutants. If the deposition and depletion option is selected by the model user for gaseous pollutants, a number of pollutant properties are required by AERMOD. (These include the diffusivity of the pollutant in air, the diffusivity of the pollutant in water, the Henry's Law constant, and a parameter reflecting the cuticular resistance to uptake of the pollutant by leaves r_{cl}).[13] HEM-3 includes a library of these parameters for approximately 130 gaseous HAPs. This library is based on a compendium of gaseous deposition parameters developed by Argonne National Laboratories.[14] The HEM-3 user can edit these values, if appropriate, including adding additional pollutant values available in the literature or calculated based on recommended methodology, as discussed in the Single HEM-3 User's Guide.[3] It should be noted, however, that the deposition and depletion option of HEM-3 and AERMOD have not been used to date for the RTR assessments.

3. Modeling for the Residual Risk Technology Review

This section discusses the general approach used to implement the HEM-3 AERMOD system for the RTR modeling analyses. Separate reports have been prepared for each of the emission source categories analyzed to date. These reports provide information on the emissions inputs and results for specific emission categories.

3.1 Emission Source Inputs

HEM-3 and AERMOD require detailed data on emissions from each emission source included in the modeling analysis. These data include, for example:

- pollutants emitted;
- emission rate for each pollutant;
- emission source coordinates;
- stack height (or emission height for fugitive and other area sources);

- stack diameter (or configuration of fugitive and other area sources);
- emission velocity; and
- emission temperature.

Emissions data for the RTR assessments are compiled from a variety of data sources (e.g., the National Emissions Inventory (NEI)[15], information collection requests). Each source category evaluated under the RTR program utilizes the best available data. These data include HAP emission rates, emission source coordinates, stack heights, stack diameters, flow rates, exit temperatures, and other emission parameters depending on the emission source types modeled. EPA performs an engineering review of the NEI data. In cases where new or better data are known to exist for a particular source category, that information is integrated into the data used in modeling that category. For each source category, the emissions are summarized in the source category specific report. Detailed computer files containing all emission and release characteristics are available in the docket prepared for the specific RTR source category under proposed or final rulemaking.

As noted in the previous section, industrial emission sources can be characterized in AERMOD as point (vertical, capped and horizontal), area, polygon, volume, line or buoyant line sources. Fugitive emissions are generally characterized as low point sources with minimal exit velocities. For some categories, additional information is available on the configuration of fugitive emission sources. This information is incorporated into the emissions database as part of the engineering review. For example, fugitive emission sources are characterized as area or volume sources when sufficient configuration information is available.

3.2 Pollutant Cross-Referencing

Because the NEI is developed from a number of different data sources, a single chemical may be listed in the inventory under different names (i.e., a "common name" and one or more structure-based names). In addition, pollutant groupings such as polycyclic organic matter (POM), can be listed in the NEI under the names of individual member compounds, and under different synonyms (e.g. polynuclear aromatic hydrocarbons). HEM-3 requires an exact match with the chemical name in order to link emissions to the appropriate dose-response factors. The model will not process any pollutant that is not specifically listed in the chemical library. Therefore, all of the HAP names used in the NEI are linked to the appropriate chemical names in the HEM-3 reference file.

Pollutant-specific dose response values are used in the HEM-3 modeling whenever available, including when modeling POM pollutants and metal compounds. Pollutant groupings, such as POM groupings, are used for POMs without a chemical-specific unit URE's. These POMs are assigned a URE associated with various POM compounds having similar characteristics. The "Technical Support Document – EPA's 2011 National-scale Air Toxics Assessment" 2015 document[16] provides more details regarding POM modeling, including (p. 121):

> [S]ome emissions of POM were reported in [the] NEI as "7-PAH" or "16-PAH," representing subsets of certain POM, or simply as "total PAH" or "polycyclic organic

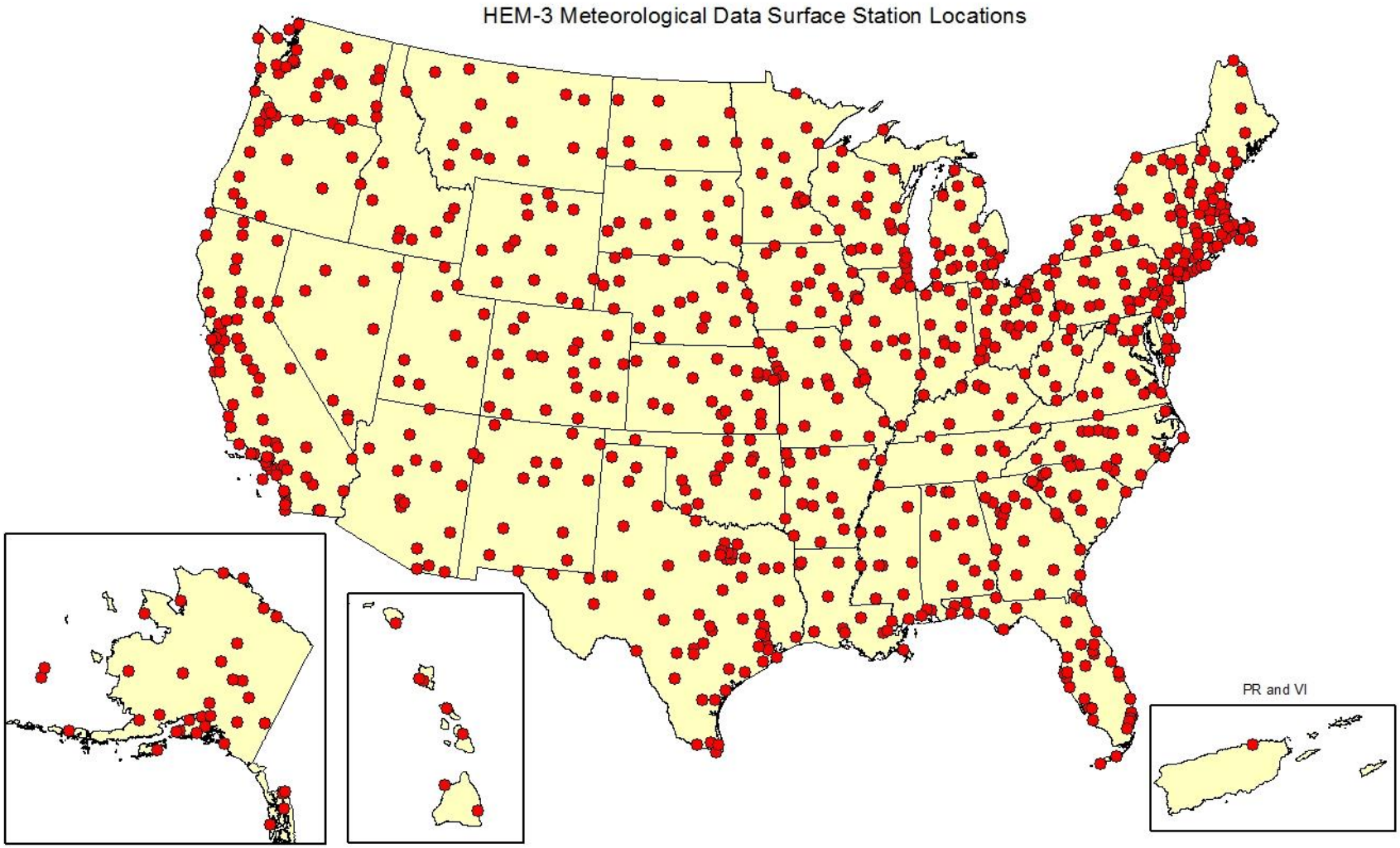
matter." In other cases, individual POM compounds are reported for which no quantitative cancer dose-response value has been published in the sources used for NATA. As a result, simplifying assumptions that characterize emissions reported as POM are applied so that cancer risk can be quantitatively evaluated for these chemicals without substantially under- or overestimating risk (which can occur if all reported emissions of POM are assigned the same URE). To accomplish this, POM emissions as reported in NEI were grouped into categories. EPA assigns dose-response values based on the known or estimated toxicity for POM within each group and on information for the POM speciation of emission sources, such as wood fires and industrial processes involving combustion.

Toxicity values used for metal compounds are also discussed in EPA's 2011 National Air Toxics Assessment Technical Support Document, including the treatment of chromium (VI) compounds, lead and nickel compounds.[16]

3.3 Meteorological Data

Nationwide meteorological data files are accessed by HEM-3 and used for the RTR modeling. The current HEM-3 AERMOD Meteorological Library includes over 800 nationwide locations, depicted in Figure 3-1. This library contains surface and upper air 2016 meteorological data from National Weather Service (NWS) observation stations, which span the entire U.S. as well as Puerto Rico and the U.S. Virgin Islands. AERMOD requires surface and upper air meteorological data that meet specific format requirements.[17, 18] Appendix 3 discusses the preprocessing performed on the meteorological data used by AERMOD and includes a detailed listing of the 824 meteorological surface and upper air station pairs, including coordinates, ground elevation and anemometer height for each station.

Figure 3-1.


HEM-3 Meteorological Data Surface Station Locations
PR and VI

3.4 Model Options Selected

HEM-3 presents a number of options for characterizing the modeling domain and data sources. As many sources are generally modeled in RTR assessments, established defaults and common practices are relied on to make these choices. The choices available to a HEM-3 user and the selections that are made in most RTR assessments are presented in Table 3-1. Some of the key selections are discussed in more detail in the paragraphs below.

It should be noted that although routine emissions are not expected to vary significantly with time, nonroutine (upset) emissions can be significant relative to routine emissions. Upset emissions occur during periods of startup, shutdown, and malfunction. Upset emissions are not likely for equipment or storage tanks, but do result from malfunctioning control devices and leaks in cooling tower heat exchangers. There is some limited data on upset emissions available,[19] but no facility-specific analyses of these data were performed to characterize short-term emissions from these emission sources, and upset emissions are generally not modeled for the RTR risk assessments.

3.4.1 Urban or Rural Dispersion Characteristics

Current RTR source category assessments which use the 2010 Census are based on either urban or rural dispersion characteristics, depending on the land characteristics surrounding each modeled facility. The EPA provides guidance on whether to select urban or rural dispersion coefficients in its Guideline on Air Quality Models.[20] In general, the urban option is used if (1) the land use is classified as urban for more than 50% of the land within a 3-kilometer radius of the emission source, or (2) the population density within a 3-kilometer radius is greater than 750 people per square kilometer. Of these two criteria, the land use criterion is more definitive.

Using the 2010 Census, the HEM-3 model determines, by default, whether to use rural or urban dispersion characteristics. HEM-3 will find the nearest census block to the facility center and determine whether that census block is in an urban area, as designated by the 2010 Census.[21] The population of the designated urban area will be used to specify the population input for AERMOD's urban mode. (Alternatively, a user may select the rural or urban option to override determination by the model. If a user selects an urban dispersion environment, then the user must provide the urban population as well.)

For the 2008 and prior screening-level RTR assessments of 51 source categories, the rural option was chosen to be most conservative (i.e., more likely to overestimate risk results). The rural option is also chosen by default by the HEM-3 model whenever the 2000 Census is selected by the user.

3.4.2 Deposition and Plume Depletion

The RTR modeling analysis to date has not taken into account the depletion of pollutant concentrations in the plume due to wet or dry deposition, although HEM-3 can model deposition with or without depletion using AERMOD. In addition, reactivity and decay have not been considered. It is possible that this approach may overestimate air concentrations and therefore

risk. However, one of the main metrics used by EPA in the residual risk program is the risk to the individual most exposed (the maximum individual risk, or MIR). Because the maximum risk usually occurs at a receptor very close to the emission source, it is unlikely to be influenced by altered plume dispersion characteristics of this type. For more refined, multipathway assessments, EPA may consider deposition and depletion.

3.4.3 Cutoff Distance for Modeling of Individual Blocks

The cutoff distance for modeling individual Census blocks is initially set to 3 km by default. This distance generally ensures that the maximum individual cancer risk and the maximum TOSHI are modeled explicitly and not interpolated. Following a modeling run, the results for each facility are checked to determine whether the maximum impacts are located inside the modeling cutoff distance. If the maximum impacts are outside the cutoff distance, and if any of the impacts are significant, then HEM-3 is rerun for the facility with a cutoff distance greater than 3 km. In general, this is done if the cancer risk exceeds 1 in 1 million or any TOSHI exceeds one. However, the risks for such facilities are generally very low, since the maximum impacts are in most cases only interpolated when the nearest Census block is more than 3 km from the facility (i.e., in sparsely populated areas).

3.4.4 Facility Boundary Assumptions

The main input mechanisms for incorporating facility boundary information in HEM-3 are the overlap distance, the distance to the innermost polar receptor ring, and user-specified receptor locations. The NEI does not provide information on facility boundaries. However, satellite/aerial images are used to locate residential populations that are closer to a facility than the Census block centroid. User-specified receptor locations are used in such assessments to avoid underestimating risk. Conservative default assumptions are used for the overlap distance and the innermost polar receptor ring. However, these are adjusted for some categories where facility sites are known to be large. In addition, satellite imagery is used to check the facility boundary assumptions for facilities with large projected impacts. These checks are discussed further in the section on Quality Assurance (Section 4).

Table 3-1. HEM-3 Domain and Set-Up Options As Used in the Residual Risk and Technology Review Assessments

Option	Selection
Dispersion model	AERMOD
Census database: 2010 or 2000	2010, unless retrospective analysis
Type of analysis: chronic, acute, or both	Both
Averaging time for short term impacts	1-hour
Multiplier for short term emissions	Source type-specific factors are used if available; a factor of 10 used otherwise
Dispersion characteristics: urban or rural, as determined by model, based on closest 2010 Census block to each facility (when using 2010 Census). Rural by default, when using the 2000 Census.	Urban or Rural based on facility location;
Include terrain impacts	Yes
Include building wake effects	No
Calculate deposition (wet, dry, or both) & include impacts of plume depletion	No[d]
User-specified receptor locations (for residential population locations, facility boundary sites, or other sites of interest)	Yes, for some facilities
Modeling domain size – maximum distance to be modeled	50 km
Cutoff distance for modeling of individual blocks	3 km[a]
Overlap distance where receptors are considered to be on facility property – measured from each source measured from each source	30 m[b]
Polar receptor network specifications:	
Distance from the facility center to the innermost ring	≥ 100 m[c]
Number of rings	13
Number of directions	16
Meteorology data	Closest site

[a] The individual block modeling cutoff is increased for categories and for some facilities to ensure that the maximum individual risk values are not interpolated.

[b] The overlap distance is adjusted for some facilities to avoid modeling locations that are on facility property (see section 4.2).

[c] HEM-3 sets the innermost ring distance to be just outside the emission sources but not < 100 m.

[d] RTR assessments typically do not calculate deposition and/or depletion, although the option to use AERMOD to model deposition with or without depletion is available in HEM-3.

3.5 Modeling of Multiple Facilities

HEM-3 models one facility at a time. However, clusters of nearby facilities may impact the same people, resulting in higher risk to those people. To account for this situation, risks are summed at each Census block for all facilities affecting the Census block.

As described earlier (Section 2.3.4), HEM-3 produces detailed output tables containing the risk and population for every Census block in the modeling domain. These detailed tables are combined for all facilities in a source category and the risk for each Census block is summed, using the RTR Summary Program add-on module to the Multi HEM-3 model, as described in the Multi HEM-3 User's Guide.[3] Thus, the effect of multiple facilities in the same source category on the same receptor are estimated. The resulting "combined facility" or "cluster-effect" census block risks are used to calculate population exposure to different cancer risk levels, non-cancer hazard indices, and source category incidence.

4. Quality Assurance

The National Emissions Inventory (NEI) is subject to an extensive program of quality assurance (QA) and quality control (QC). The QA/QC program for the point source component of the NEI is documented in a separate report, available from the NEI website.[22] This section describes QA activities carried out under the RTR modeling analysis.

4.1 Engineering Review

In addition to the standardized QA steps taken for the entire NEI, EPA performs an engineering review of NEI data for the emission source categories included in the RTR analysis. This engineering review includes two main components. The first component addresses the list of facilities included in each source category. EPA engineers review independent sources of information to identify all sources in the category that are included in the NEI. In addition, EPA reviews the list of sources represented as part of each category in the NEI to ensure that the facilities actually manufacture products characteristic of the source category.

The second component of the engineering review focuses on the appropriateness of facility emissions. EPA reviews the list of HAPs reportedly emitted by each facility to ensure that the pollutants are appropriate to the source category. In addition, EPA engineers review the magnitude of those HAP emissions. In cases where new or better data are known to exist for a particular source category, that information is integrated into the data used in the HEM-3/AERMOD modeling for that category. In these cases, the source category specific documents provide additional details on the emissions inputs used.

4.2 Geographic Pre-Modeling Checks

The NEI QA process includes some basic checks on location data for point sources. The coordinates for each source are checked to ensure that they are in the county that has been specified for the source. If this is not the case, or if no geographic coordinates are available for the emission source, then the coordinates are set to a default location based on the nature of the emission source category.[22] In addition, coordinates for all emission sources at a given facility are checked to ensure that they are within 3 km of one another. These QA checks happen prior to HEM-3 modeling and the results of such checks are reflected in the HEM-3 input files.

Another pre-modeling geographic QA check regards the location of the census block receptors. As noted above, to estimate ambient concentrations for evaluating long-term exposures, the HEM-3 model uses the census block centroids as dispersion model receptors. The census block centroids are often good surrogates for where people live within a census block. A census block generally encompasses about 40 people or 10-15 households. However, in cases where a block centroid is located on industrial facility property, or where a census block is large and the centroid less likely to be representative of the block's residential locations, the block centroid may not be an appropriate surrogate.

Census block centroids that are on facility property can sometimes be identified by their proximity to emission sources. In cases where a census block centroid is within 300 meters of any emission source, aerial images of the facility are reviewed to determine whether the block centroid is likely located on facility property. The selection of the 300-meter distance reflects a compromise between too few and too many blocks identified as being potentially on facility property. Distances smaller than 300 meters would identify only block centroids very near the emission sources and could exclude some block centroids that are still within facility boundaries, particularly for large facilities. Distances significantly larger than 300 meters would identify many block centroids that are outside facility boundaries, particularly for small facilities. Block centroids confirmed to be located on facility property are moved to a location that best represents the residential locations in the block.

In addition, census block centroids for blocks with large areas may not be representative of residential locations. Risk estimates based on such centroids can be understated if there are residences nearer to a facility than the centroid, and overstated if the residences are farther from the facility than the centroid. To avoid understating the maximum individual risk associated with a facility, block centroids are relocated in some cases, or additional user-specified receptors are added to a block. Aerial images of all large census blocks within one kilometer of any emission source are examined. Experience from previous risks characterizations show that in most cases the MIR is generally located within 1 km of the facility boundary. If the block centroid does not represent the residential locations, it is relocated in the HEM-3 input files to better represent them. If residential locations cannot be represented by a single receptor (that is, the residences are spread out over the block), additional user-specified receptors are included in the HEM-3 input files to represent residences nearer to the facility than the centroid.

4.3 Geographic Post-Modeling Checks

As part of the RTR modeling analysis, additional geographical QA checks are made for some facilities, after initial HEM-3 modeling results are reviewed. Facilities subjected to these additional checks include:

- cases where the initial estimates of maximum risks are particularly high
    - maximum individual cancer risk of over 1 in 10,000
    - any maximum TOSHI above 10
- cases where no Census blocks are identified by the model within 3 km of the facility

HEM-3 produces a detailed Google Earth™ map of the modeled point, area, polygon, volume, line and buoyant line emission sources and surrounding receptors (including Census block centroids, polar receptors and user-specified receptors) overlaying Google Earth™'s satellite imagery. This map allows a QA check of the specific source locations, as well as an approximate check of the facility boundaries. The emission source coordinates are reviewed for each of these facilities and compared with the address reported for the facility. If the address and the coordinates represent the same location, then the coordinates are taken to be correct. For more recent modeling of source categories, the emission coordinates initially modeled by HEM-3 tend to be correct, as they undergo pre-modeling scrutiny and QA checks (as discussed in Section 4.2).

More rarely, the modeled emission coordinates will be determined post initial modeling not to be located on facility property. If the facility and emission coordinate locations are different, then the satellite imagery for the address and the coordinate location are reviewed to determine whether either photograph includes an industrial facility. If emission source coordinates are found to be incorrect, HEM-3 is rerun using corrected coordinates. These changes are described in the source category documents.

For the high-risk facilities, the coordinates used to represent the most impacted Census blocks are also reviewed. This review draws on detailed Census block boundary maps and satellite imagery. Large industrial facilities will frequently occupy one or more entire Census blocks. However, these blocks may also include one or more residences on the periphery of the industrial land. Generally, the centroid coordinates listed for a Census block are near the center of the block. In these cases of mixed industrial and residential blocks, the coordinates may be on facility property.

In general, block coordinates are considered to be on facility property if they are located between the different emission source locations listed for the facility. In these situations, HEM-3 is rerun with an expanded overlap distance, in order to exclude the Census block coordinates that appear to be located on facility property. The distance to the innermost polar receptor ring is also adjusted to ensure that this ring is not on facility property, but as close to the apparent facility boundaries as possible.

5. Uncertainties

The RTR risk assessments using HEM-3 and AERMOD are subject to a number of uncertainties. For instance, model verification studies for AERMOD show predicted maximum annual concentrations ranging from 0.3 to 1.6 times measured values, with an average of 0.9. Predicted maximum short term (1 to 24 hours) concentrations were 0.25 to 2.5 times measured values, with an average of one.[23]

In addition, a number of simplifying assumptions are made in these modeling analyses. First, the coordinates reported by the U.S. Census Bureau for Census block internal points ("centroids") have been used as a surrogate for long-term population exposures. Locations of actual residences have not been modeled. In addition, the current version of HEM-3 does not take into account the movement of people from one Census block to another during the course of their lives, or commuting patterns during a given day. Nor does the model take into account the attenuation of pollutant from outside emission sources in indoor air. Ideally, risks to individuals would be modeled as they move through their communities and undertake different activities. However, such modeling is time-and resource-intensive and can only capture a portion of the uncertainty associated with the full range of human activities. In general, it is expected that long-term exposures will be overstated for high-end estimates (as most individuals will not spend all their time at their highly affected residences), but may understate the total population exposed (as some individuals living outside the modeled area may regularly commute into the area for work or school).

When considering long-term or lifetime exposures, it should be noted that relatively few people in the United States reside in one place for their entire lives. For the purposes of this assessment, cancer risk estimates are based on a lifetime exposure at the Census-identified place of residence. While it is impossible to know how this assumption affects the risk experiences by a particular individual (as people can move into higher- or lower-risk areas), it is likely that this assumption will overstate the exposure to those most exposed (i.e., people already living in high exposure areas are unlikely to move to yet higher exposure areas). However, this assumption will also tend to underestimate the total number of people exposed and population risk (i.e., incidence) because population levels are generally increasing.

In the current analyses, only direct inhalation is modeled. Other pathways such as the deposition of pollutants to drinking water, and to bioaccumulation of deposited pollutants in the food supply may be a significant source of exposure for persistent and bioaccumulative hazardous air pollutants (PB HAP). Screening level evaluations of the potential human health risks associated with emissions of PB HAP from the modeled facilities are used to determine if additional analyses are needed, but these analyses are outside the scope of this document. Because the HEM-3 AERMOD analyses are restricted to the inhalation pathway and depleting the plume would not be a conservative approach to modeling air concentrations, the impacts of plume depletion due to deposition are not taken into account. Thus, inhalation impacts may be overestimated for some pollutants, but exposures through other pathways would be underestimated.

A number of other simplifications are made in the dispersion modeling analyses, as noted in Table 3-1. For instance, building wake effects are not considered. In addition, meteorological observations are based on the closest station in the HEM-3 meteorological library (see Figure 3-1). Alternative meteorological stations may be more appropriate for some facilities. Ideally, facility-specific meteorological observations would be used. A single year of meteorological data (2016) is currently used for AERMOD's dispersion modeling. (The 2008 and prior screening-level RTR assessments of 51 source categories used meteorological data based on the year 1991.) When considering off-site meteorological data most site specific dispersion modeling efforts will employ up to five years of data to capture variability in weather patterns from year to year. However, because of the large number of facilities in the analyses and the extent of the dispersion modeling analysis (national scale), it is not practical to model five years of data. Other national studies such as NATA also consider only a single year of meteorological data. A sensitivity analyses performed by the NATA assessment found that variability attributable to the selection of the meteorology location/time (both temporal and spatial) resulted in a 17-84% variation in predicted concentrations at a given station.[24]

Finally, risk and exposure factors are also subject to uncertainty. Not all individuals experience the same degree of exposure or internal dose of a given pollutant due to individual-specific parameters such as weight, age, and gender. While the health benchmarks used in the analyses crudely account for sensitive populations, a prototypical human (e.g., body weight, ventilation rate) is used to define the benchmark. Because of the variability of these parameters in the population, this factor will result in a degree of uncertainty in the resulting risk estimate.

Table 5-1 summarizes the general sources of uncertainty for the RTR modeling analyses. The table also gives a qualitative indication of the potential direction of bias on risk estimates. The sources of uncertainty in Table 5-1 are divided into four categories, based on the major components of the analyses:

- emissions inventory;
- fate and transport modeling;
- exposure assessment; and
- toxicity assessment.

It must also be noted that individual source categories may be subject to additional uncertainties. These are discussed in separate reports which are prepared for each emission source category included in the RTR assessments.

Table 5-1. Summary of General Uncertainties Associated with Risk and Technology Review Risk Assessments

Parameter	Assumption	Uncertainty/Variability Discussion	Potential Direction of Bias on Risk Estimates
Emissions Inventory			
Individual HAP emissions rates and facility characteristics (stack parameters, property boundaries)	Emissions and facility characteristics from the NEI provide an accurate characterization of actual source emissions.	Our current emissions inventory is based on source category specific ICR and/or the latest NEI, our internal review, and public comments received. The degree to which the data in our inventory represents actual emissions is likely to vary across sources. For the 2008 screening level assessments, nearly half of the sources in a given source category submitted a review of their emissions and facility characteristics data. Some detailed data, such as property boundary information is not available for most facilities. This is an important consideration in determining acute impacts.	Unbiased overall, magnitude variable
Multiplier for short-term emission rates	Generally, maximum short term emission rates are estimated by applying a simple multiplier (a factor of 10) to average annual emissions.	The ratio between short-term and long-term average emission rates may vary among the different emission sources at a facility. In addition, the use of a simple multiplier means that impacts of maximum short term emissions are modeled for all meteorological conditions, including the worst-case conditions for population exposure.	Potential overestimate due to the fact that worst-case emissions are assumed to occasionally coincide with worst-case meteorology. Overestimate due to lack of actual information on short-term emission rates.
Fate and Transport Modeling			
Atmospheric dispersion model choice	AERMOD is EPA's recommended dispersion model for assessing pollutant concentrations from industrial facilities	Field testing of dispersion models, including AERMOD, have shown results to generally be within a factor of 2 of measured concentrations.	Unbiased overall
Building downwash	Not included in assessments	Use of this algorithm in AERMOD could improve the dispersion calculations at individual facilities. However, data are not readily available to utilize this option.	Potential underestimate of maximum risks near facility. No effect on risks further out.
Plume deposition and depletion	Not included in assessments	Ignoring these impacts for pollutants that deposit minimally, and whose risks derive predominantly from inhalation, should have minimal effect on risk estimates.	Unbiased or minimal overestimate.

Table 5-1. Summary of General Uncertainties Associated with Risk and Technology Review Risk Assessments (continued)

Parameter	Assumption	Uncertainty/Variability Discussion	Potential Direction of Bias on Risk Estimates
Meteorology	One year of meteorological data from the nearest weather station (selected from 824 nationwide) is representative of long-term weather conditions at the facility.	The use of one year of data rather than the five or more adds uncertainty based on whether that year is representative of each location's climatology. Use of weather station data rather than on-site data can add to uncertainty. Additionally, the use of default surface parameters in the generation of the meteorological datasets imparts uncertainty to the results from any individual facility.	Minimal underestimate or overestimate.
Reactivity	Not included in the assessments	Chemical reactions and transformations of individual HAP into other compounds due to solar radiation and reactions with other chemicals happens in the atmosphere. However, in general, the HAP in this assessment do not react quickly enough for these transformations to be important near the sources, where the highest individual risks are estimated. Further, most of the HAP do not react quickly enough for these transformations to be important to risk estimates in the entire modeled domain (i.e., within 50 km of the source).	No impact on maximum risk estimates. Minimal impact on population risks and incidence.
Maximum modeling distance	50 kilometers from center of facility	This distance is considered to be the maximum downwind distance for a Gaussian plume model such as AERMOD. This is because, in general, winds cannot be considered to follow straight line trajectories beyond this distance.	No effect on maximum individual risks. Minimal underestimation of incidence.
Exposure Assessment			
Locations and short-term movements of individuals	Ambient concentration at centroid of each off-site census block is equal to the exposure concentration for all people living in that census block. Effect of human activity patterns on exposures is not included in the assessment.	People live at different areas within block that may have higher or lower exposures than at the centroid. Individuals also move from outdoors to indoors and from home to school/work to recreation, etc., and this can affect their total exposure from these sources.	Unbiased across population for most pollutants and individuals, likely overestimate for most exposed and underestimate for least exposed persons.

Table 5-1. Summary of General Uncertainties Associated with Risk and Technology Review Risk Assessments (continued)

Parameter	Assumption	Uncertainty/Variability Discussion	Potential Direction of Bias on Risk Estimates
Long-term movements of individuals	MIR individual is exposed continuously to the highest exposure concentration for a 70-year lifetime.	Maximum individual risk (MIR) is defined in this way to be a maximum theoretical risk at a point where a person can actually reside.	Unbiased for most individuals, likely overestimate for the actual individual most exposed and likely underestimate for the least exposed. Incidence remains unbiased unless population around facilities increases or decreases over 70 years.
Toxicity Assessment			
Reference concentrations (RfC)	Consistent with EPA guidance, RfCs are developed including uncertainty factors to be protective of sensitive subpopulations. Additionally, RfCs are developed based on the level producing an effect in the most sensitive target organ or system.	While other organ systems may be impacted at concentrations above the RfC, these are not included in the calculation of target organ-specific hazard indices.	In general, EPA derives RfCs using procedures whose goal is to avoid underestimating risks in light of uncertainty and variability. The greater the uncertainties, the greater the potential for overestimating risks.
Unit Risk Estimate (URE)	Use of unit risk estimates developed from dose-response models such as linear low-dose extrapolation.	Uncertainty in extrapolating the impacts from short-duration, high-dose animal or work-related exposures to longer duration, lower-dose environmental impacts.	Overestimate of risks for nonlinear carcinogens and for linear carcinogens with sparse health effects data. In general, EPA derives URE values using procedures aimed at overestimating risks in light of uncertainty and variability.
Toxicity of mixtures	Cancer risks and non-cancer hazard quotients were calculated for each HAP individually and then summed into a total risk or hazard index (assumption of additivity).	Concurrent exposures to multiple chemicals may result in either increased or decreased toxicity due to chemical interactions but the data needed to quantify these effects are generally not available.	Unbiased overall. Some mixtures may have underestimated risks, some overestimated, and some correctly estimated.

Table 5-1. Summary of General Uncertainties Associated with Risk and Technology Review Risk Assessments (continued)

Parameter	Assumption	Uncertainty/Variability Discussion	Potential Direction of Bias on Risk Estimates
Surrogate dose-response values for HAPs without values	In the case of groups of HAPs such as glycol ethers, the most conservative dose-response value of the chemical group was used as a surrogate for missing dose-response values in the group. For others, such as unspeciated metals, we have applied speciation profiles appropriate to the source category to develop a composite dose-response value for the group. For HAP which are not in a group and for which no URE's or RfC's are available from credible sources, no assessment of risk is made.	Rather than neglecting the assessment of risks from some HAPs lacking dose response values, conservative assumptions allow the examination of whether these HAPs may pose an unacceptable risk and require further examination, or whether the conservative level examination with surrogates screens out the HAPs from further assessment.	Overestimate where most conservative values used. Unbiased where category-specific profiles applied. There is the potential to underestimate risks for pollutants which are not included in the assessment.

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Appendix 3
Meteorological Data for HEM3 Modeling

METEOROLOGICAL DATA PROCESSING USING AERMET FOR HEM3

US EPA
Air Toxic Assessment Group
RTP, NC 27711

April, 2017

BACKGROUND

The AERMOD meteorological processor, AERMET, and its supporting modeling system (AERSURFACE and AERMINUTE) were used to process one year of meteorological data for over 800 observation stations across the continental United States, Alaska, Hawaii, and Puerto Rico.

METEOROLOGICAL DATA

To estimate the boundary layer parameters required by AERMOD, AERMET requires hourly surface weather observations (which may include hourly values calculated from 1-minute data) and the full (i.e., meteorological variables reported at all levels) twice-daily upper air soundings. The surface and upper air stations are paired to produce the required input data for AERMOD.

USEPA meteorologists obtained calendar year 2016 Integrated Surface Hourly Data (ISHD) for over 800 Automated Surface Observation System (ASOS) stations spanning the entire US, as well as Puerto Rico and the US Virgin Islands, from the National Climatic Data Center (NCDC). To support AERMET, ASOS 1-minute data for each surface station were also obtained from NCDC in a DSI 6405 format.

Further, upper air sounding data for the same time period for over 80 observation sites were obtained from the "NOAA/ESRL" online Radiosonde Database. These datasets were produced by ESRL in Forecast Systems Laboratory (FSL) format. Attachment 1 lists the surface and upper air stations, as well as the coordinates, ground elevation, and anemometer height for each station.

AERMET PROCESSING

Utilizing the AERMET meteorological data preprocessor, and the ASOS surface and FSL upper air stations, surface and profile files for input into AERMOD were generated nationwide. The surface stations were paired with representative upper air stations by taking the upper air station closest to each surface station. The AERSURFACE tool was used to estimate the surface characteristics for input into AERMET utilizing land cover data surrounding the surface station. In addition, the AERMINUTE preprocessor was used to process 1-minute ASOS wind data for input into AERMET. Table 1 and Attachment 1 outline the approach and site specific inputs each of the data preprocessors and tools used to generate the AERMOD meteorological data.

Table 1. AERMET Processing Options

AERMET Options	Version	16216
	ASOS Site	Yes
	Surface Data Format	NCDC TD-3505 (ISHD)
	Upper Air Data Format	FSL, all levels, tenths m/s
	Wind Speed Threshold	0.5 m/s
	Beta Option (U*)	Yes
AERMINUTE Options	Version	15272
	Include 1 minute ASOS Data	Yes, where available TD-6405 format
AERSURFACE Options	Version	Nonpublic release version[1]
	Landcover data	USGS NLC for 2011
	Radius for Surface Roughness Calculations	1 km
	Seasons	Winter – Dec, Jan, Feb Spring – Mar, Apr, May Summer – June, July, Aug Fall – Sept, Oct, Nov
	Temporal resolution	Monthly, 12 sectors
	Site Surface Moisture	See Attachment 1
	Snow Cover	See Attachment 1

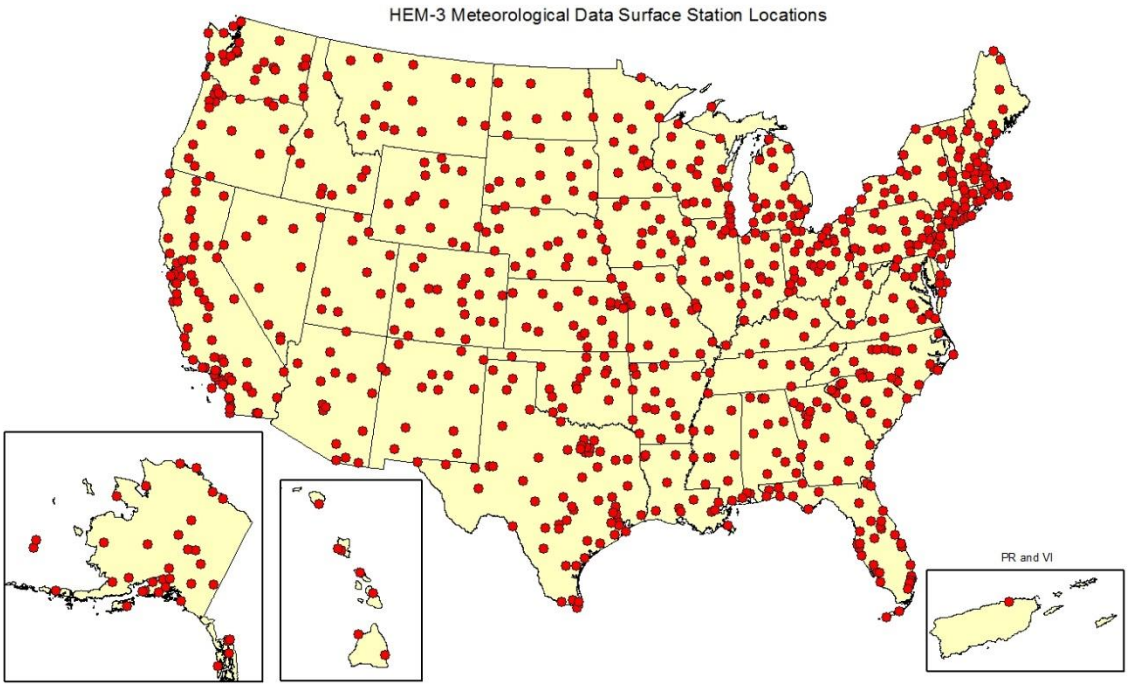
1 : generated with the pre-public release version of AERSURFACE date.

RESULTS

To assure that the data would support an AERMOD run, USEPA meteorologists ran AERMOD using a model plant with each AERMET SFC and PLF pairs. Further, the surface files were examined for completeness. If more than 10 percent of specific data including the Monin-Obukhov length, wind speed, and cloud cover were missing, the station was not considered suitable for the meteorological database and therefore excluded.

In all, 824 met station pairs ran successfully in AERMOD and passed completeness criteria. Figure 1 is a map that shows the locations of the 824 surface stations. The processed 2016 meteorological data generated by the above approach is posted on the EPA's FERA (Fate, Exposure, and Risk Analysis) website under the HEM model page.

Figure 1. 824 HEM-3 Meteorological Stations.



Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
13871	KANB	ANB	11409	ANNISTON METRO AP	33.5872	-85.8556	181	-6	8
13876	KBHM	BHM	30509	BIRMINGHAM AP	33.56556	-86.745	187	-6	10
53852	KDCU	DCU	51007	DECATUR PRYOR FLD	34.6525	-86.9453	179	-6	10
13839	KDHN	DHN	53007	DOTHAN RGNL AP	31.3167	-85.45	114	-6	10
53820	KGZH	GZH	32007	EVERGREEN MIDDLETON FLD	31.41556	-87.0442	77	-6	10
3856	KHSV	HSV	50807	HUNTSVILLE INTL AP	34.64389	-86.7861	190	-6	10
13838	KBFM	BFM	41307	MOBILE DWTN AP	30.62639	-88.0681	6	-6	10
13894	KMOB	MOB	30507	MOBILE RGNL AP	30.68833	-88.2456	66	-6	10
13895	KMGM	MGM	22009	MONTGOMERY AP	32.2997	-86.4075	62	-6	10
13896	KMSL	MSL	52407	MUSCLE SHOALS RGNL AP	34.7441	-87.5997	165	-6	10
3878	KTOI	TOI	12109	TROY MUNI AP	31.86056	-86.0122	120	-6	10
93806	KTCL	TCL	11209	TUSCALOOSA MUNI AP	33.2119	-87.6161	46	-6	10
63872	KEUF	NA		WEEDON FLD AP	31.95139	-85.1289	87	-6	10
26451	PANC	ANC	91806	ANCHORAGE INTL AP	61.169	-150.028	37	-9	8
26409	PAMR	MRI	41707	ANCHORAGE MERRILL FLD	61.21694	-149.855	42	-9	5
25308	PANT	ANN	52606	ANNETTE WSO AP	55.0389	-131.579	33	-9	10
27502	PABR	BRW	61703	BARROW POST ROGERS AP	71.2834	-156.782	9	-9	8
26615	PABE	BET	91306	BETHEL AP	60.785	-161.829	31	-9	8
26533	PABT	BTT	51806	BETTLES AP	66.9167	-151.519	196	-9	10
26415	PABI	BIG	100705	BIG DELTA AP	63.9944	-145.721	389	-9	10
25624	PACD	CDB	91906	COLD BAY AP	55.22083	-162.733	24	-9	10
26410	PACV	CDV	110705	CORDOVA M K SMITH AP	60.4888	-145.451	9	-9	8
27406	PASC	SCC	82406	DEADHORSE AP	70.1917	-148.477	19	-9	8
26411	PAFA	FAI	82306	FAIRBANKS INTL AP	64.8039	-147.876	132	-9	10
26425	PAGK	GKN	111705	GULKANA AP	62.1591	-145.459	476	-9	10
25323	PAHN	HNS	102705	HAINES AP	59.2433	-135.509	5	-9	10
25507	PAHO	HOM	92306	HOMER AP	59.642	-151.491	20	-9	8
25506	PAIL	ILI	11407	ILIAMNA AP	59.7494	-154.909	44	-9	10
25309	PAJN	JNU	82206	JUNEAU INTL AP	58.3566	-134.564	5	-9	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
26523	PAEN	ENA	92106	KENAI MUNI AP	60.5797	-151.239	28	-9	8
25325	PAKT	KTN	11003	KETCHIKAN INTL AP	55.35667	-131.712	23	-9	8
25503	PAKN	AKN	32107	KING SALMON AP	58.6829	-156.656	20	-9	10
25501	PADQ	ADQ	41007	KODIAK AP	57.75111	-152.486	24	-9	10
26616	PAOT	OTZ	83006	KOTZEBUE RALPH WEIN AP	66.86667	-162.633	9	-9	8
26510	PAMC	MCG	60106	MCGRATH AP	62.9574	-155.61	101	-9	10
26435	PANN	ENN	102705	NENANA MUNI AP	64.55	-149.072	110	-9	10
26617	PAOM	OME	90206	NOME MUNI AP	64.5111	-165.44	4	-9	8
26412	PAOR	ORT	82406	NORTHWAY AP	62.9617	-141.938	522	-9	10
27515	PAQT	AQT	82206	NUIQSUT AP	70.21167	-151.002	18	-9	8
25331	PAAQ	PAQ	91906	PALMER MUNI AP	61.5961	-149.092	70	-9	10
26492	PATO	POR	41707	PORTAGE GLACIER V C	60.785	-148.839	31	-9	10
25516	PASO	SOV	103005	SELDOVIA AP	59.44333	-151.702	19	-9	10
26438	PAWD	SWD	102705	SEWARD AP	60.12833	-149.417	15	-9	8
25333	PASI	SIT	12403	SITKA AP	57.0481	-135.365	4	-9	8
25335	PAGY	SGY	110705	SKAGWAY AP	59.4556	-135.324	6	-9	8
25628	PAPB	PBV	70906	ST GEORGE ISLAND AP	56.6	-169.565	27	-9	10
25713	PASN	SNP	100106	ST PAUL ISLAND AP	57.15528	-170.222	11	-9	10
26528	PATK	TKA	70203	TALKEETNA AP	62.32	-150.095	107	-9	8
26529	PATA	TAL	111705	TANANA CALHOUN MEM AP	65.175	-152.107	68	-9	10
27503	PAWI	AWI	81606	WAINWRIGHT AP	70.63917	-159.995	8	-9	8
53869	KHKA	HKA	92508	BLYTHEVILLE MUNI AP	35.94028	-89.8308	77	-6	10
53925	KDEQ	DEQ	42607	DE QUEEN SEVIER CO AP	34.05	-94.4008	108	-6	8
93992	KELD	ELD	71107	EL DORADO S AR RGNL AP	33.22083	-92.8142	77	-6	10
93993	KFYV	FYV	42409	FAYETTEVILLE DRAKE FLD	36.0097	-94.1694	381	-6	10
53922	KXNA	XNA	103008	FAYETTEVILLE NW AR AP	36.28333	-94.3	392	-6	10
13964	KFSM	FSM	21909	FT SMITH RGNL AP	35.333	-94.3625	137	-6	10
13971	KHRO	HRO	52709	HARRISON BOONE CO AP	36.2668	-93.1566	419	-6	8
3962	KHOT	HOT	72407	HOT SPRINGS ASOS	34.479	-93.096	163	-6	8

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
3953	KJBR	JBR	100108	JONESBORO MUNI AP	35.83111	-90.6464	78	-6	10
13963	KLIT	LIT	52109	LITTLE ROCK AP ADAMS FLD	34.7273	-92.2389	79	-6	10
53919	KLLQ	LLQ	71707	MONTICELLO MUNI AP	33.6361	-91.7556	88	-6	10
53921	KMWT	MWT	60607	MOUNT IDA ASOS	34.5467	-93.5781	214	-6	10
53918	KBPK	BPK	71107	MOUNTAIN HOME BAXTER AP	36.36889	-92.4703	281	-6	8
93988	KPBF	PBF	61907	PINE BLUFF GRIDER FLD	34.175	-91.9347	61	-6	8
53920	KRUE	RUE	120606	RUSSELLVILLE MUNI AP	35.25778	-93.0947	116	-6	10
13977	KTXK	TXK	81607	TEXARKANA WEBB FLD	33.4536	-94.0074	110	-6	8
93026	KDUG	DUG	22706	DOUGLAS BISBEE INL AP	31.4583	-109.606	1251	-7	10
3103	KFLG	FLG	20807	FLAGSTAFF PULLIAM AP	35.1441	-111.666	2135	-7	10
3124	KFHU	NA		FORT HUACHUCA	31.58833	-110.344	1415	-7	10
3195	KGCN	GCN	21207	GRAND CANYON NP AP	35.94611	-112.155	2014	-7	10
93167	KIGM	IGM	22007	KINGMAN MOHAVE CO AP	35.2577	-113.933	1042	-7	10
3196	KOLS	OLS	92606	NOGALES INTL AP	31.42083	-110.846	1194	-7	10
3162	KPGA	PGA	21507	PAGE MUNI AP	36.92611	-111.448	1314	-7	8
3184	KDVT	DVT	22107	PHOENIX DEER VALLEY MUNI AP	33.68833	-112.082	443	-7	10
23183	KPHX	PHX	40307	PHOENIX SKY HARBOR INTL AP	33.4277	-112.004	337	-7	10
23184	KPRC	PRC	20907	PRESCOTT LOVE FLD	34.65167	-112.421	1524	-7	10
93084	KSAD	SAD	30706	SAFFORD MUNI AP	32.85472	-109.635	968	-7	10
3192	KSDL	SDL	32007	SCOTTSDALE MUNI AP	33.62278	-111.911	449	-7	10
93027	KSJN	SJN	13006	ST JOHNS INDUSTRIAL AP	34.51833	-109.379	1746	-7	10
23160	KTUS	TUS	41007	TUCSON INTL AP	32.1313	-110.955	777	-7	10
3029	KRQE	RQE	13006	WINDOW ROCK AP	35.6575	-109.061	2054	-7	8
23194	KINW	INW	13006	WINSLOW MUNI AP	35.0281	-110.721	1489	-7	10
94299	KAAT	AAT	42406	ALTURAS MUNI AP	41.49139	-120.564	1334	-8	10
24283	KACV	ACV	13107	ARCATA EUREKA AP	40.97806	-124.109	61	-8	10
23191	KAVX	AVX	90507	AVALON CATALINA AP	33.405	-118.416	472	-8	10
23155	KBFL	BFL	31407	BAKERSFIELD AP	35.4344	-119.054	149	-8	10
23161	KDAG	DAG	13006	BARSTOW DAGGETT AP	34.8536	-116.786	584	-8	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
23157	KBIH	BIH	102705	BISHOP AP	37.3711	-118.358	1250	-8	10
23225	KBLU	BLU	103102	BLUE CANYON AP	39.2774	-120.71	1608	-8	8
23158	KBLH	BLH	102705	BLYTHE AP	33.6186	-114.714	120	-8	10
23152	KBUR	BUR	20707	BURBANK GLENDALE PASADENA AP	34.20056	-118.358	222	-8	8
23136	KCMA	CMA	12507	CAMARILLO AP	34.21667	-119.083	25	-8	10
3177	KCRQ	CRQ	41107	CARLSBAD PALOMAR AP	33.12806	-117.279	93	-8	8
3179	KCNO	CNO	32807	CHINO AP	33.97528	-117.636	192	-8	8
23254	KCCR	CCR	82605	CONCORD BUCHANAN FLD	37.9917	-122.055	5	-8	10
24286	KCEC	CEC	90706	CRESCENT CITY MCNAMARA AP	41.78028	-124.237	18	-8	8
3104	KTRM	TRM	92806	DESERT RESORTS RGNL AP	33.62667	-116.159	-39	-8	10
23199	KNJK	NA		EL CENTRO NAF	32.81667	-115.683	-14	-8	10
93193	KFAT	FAT	40307	FRESNO YOSEMITE INTL AP	36.78	-119.719	101	-8	8
3166	KFUL	FUL	40307	FULLERTON MUNI AP	33.87194	-117.979	29	-8	10
53119	KHJO	HJO	110705	HANFORD MUNI AP	36.31889	-119.629	77	-8	10
3167	KHHR	HHR	12507	HAWTHORNE MUNI AP	33.92278	-118.334	19	-8	8
93228	KHWD	HWD	20807	HAYWARD AIR TERMINAL	37.6542	-122.115	13	-8	10
93115	KNRS	NA		IMPERIAL BEACH REAM FLD NAS	32.56667	-117.117	7	-8	10
3144	KIPL	IPL	101706	IMPERIAL CO AP	32.83417	-115.579	-18	-8	10
3159	KWJF	WJF	40507	LANCASTER WM J FOX FLD	34.7411	-118.212	713	-8	10
23285	KLVK	LVK	32307	LIVERMORE MUNI AP	37.6927	-121.814	120	-8	8
23129	KLGB	LGB	40407	LONG BEACH DAUGHERTY FLD	33.8116	-118.146	9	-8	8
23174	KLAX	LAX	102706	LOS ANGELES INTL AP	33.938	-118.389	30	-8	10
93242	KMAE	MAE	21306	MADERA MUNI AP	36.98778	-120.111	77	-8	10
93205	KMYV	MYV	21306	MARYSVILLE YUBA CO AP	39.1019	-121.568	19	-8	10
23257	KMCE	MCE	22706	MERCED MUNI AP	37.28472	-120.513	46	-8	8
23258	KMOD	MOD	40207	MODESTO CITY CO AP	37.6241	-120.951	22	-8	8
23244	KNUQ	NA		MOFFETT FEDERAL AIRFIELD	37.40583	-122.048	12	-8	10
24259	KSIY	SIY	102705	MONTAGUE SISKIYOU AP	41.78139	-122.468	807	-8	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
23259	KMRY	MRY	20607	MONTEREY PENINSUL AP	36.58806	-121.845	50	-8	8
24215	KMHS	MHS	22706	MT SHASTA	41.3325	-122.333	1077	-8	10
93227	KAPC	APC	31808	NAPA CO AP	38.2102	-122.285	4	-8	10
23179	KEED	EED	20507	NEEDLES AP	34.7675	-114.619	271	-8	10
23230	KOAK	OAK	21507	OAKLAND METRO INTL AP	37.72139	-122.221	2	-8	10
53121	KOKB	OKB	20306	OCEANSIDE MUNI AP	33.21944	-117.349	10	-8	10
3102	KONT	ONT	92407	ONTARIO INTL AP	34.05611	-117.6	289	-8	8
93210	KOVE	OVE	21306	OROVILLE MUNI AP	39.49	-121.618	57	-8	10
93110	KOXR	OXR	11007	OXNARD VENTURA CO AP	34.20083	-119.207	11	-8	8
93138	KPSP	PSP	92607	PALM SPRINGS RGNL AP	33.8222	-116.504	125	-8	10
23182	KPMD	PMD	20807	PALMDALE AP	34.62944	-118.084	769	-8	10
93209	KPRB	PRB	22706	PASO ROBLES MUNI AP	35.6697	-120.628	247	-8	10
24216	KRBL	RBL	21306	RED BLUFF MUNI AP	40.1519	-122.254	108	-8	10
24257	KRDD	RDD	80907	REDDING MUNI AP	40.5175	-122.299	151	-8	10
3171	KRAL	RAL	71008	RIVERSIDE MUNI AP	33.95194	-117.439	245	-8	10
23232	KSAC	SAC	81507	SACRAMENTO EXECUTIVE AP	38.5069	-121.495	5	-8	10
23233	KSNS	SNS	20607	SALINAS MUNICIPAL AP	36.6636	-121.608	23	-8	10
23188	KSAN	SAN	82307	SAN DIEGO LINDBERGH FLD	32.7336	-117.183	5	-8	10
3131	KMYF	MYF	30707	SAN DIEGO MONTGOMERY FLD	32.81583	-117.139	127	-8	8
23234	KSFO	SFO	73103	SAN FRANCISCO INTL AP	37.6197	-122.365	2	-8	10
23293	KSJC	SJC	30807	SAN JOSE	37.3591	-121.924	16	-8	8
93206	KSBP	SBP	41707	SAN LUIS OBISPO AP	35.23722	-120.641	61	-8	10
23187	KSDB	SDB	21306	SANDBERG	34.7436	-118.724	1375	-8	10
93184	KSNA	SNA	81507	SANTA ANA JOHN WAYNE AP	33.68	-117.866	13	-8	6
23190	KSBA	SBA	62207	SANTA BARBARA MUNI AP	34.4258	-119.843	3	-8	8
23273	KSMX	SMX	60607	SANTA MARIA PUBLIC AP	34.8994	-120.449	74	-8	10
93197	KSMO	SMO	41807	SANTA MONICA MUNI AP	34.01583	-118.451	53	-8	10
23213	KSTS	STS	31407	SANTA ROSA SONOMA CO AP	38.5038	-122.81	35	-8	10
93230	KTVL	TVL	51707	SOUTH LAKE TAHOE AP	38.8983	-119.995	1925	-8	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
23237	KSCK	SCK	81607	STOCKTON METRO AP	37.8891	-121.226	8	-8	10
23275	KUKI	UKI	101206	UKIAH MUNI AP	39.12583	-123.201	183	-8	6
93241	KVCB	VCB	80707	VACAVILLE NUT TREE AP	38.3775	-121.958	33	-8	10
23130	KVNY	VNY	60707	VAN NUYS AP	34.20972	-118.489	235	-8	8
23277	KWVI	WVI	20607	WATSONVILLE MUNI AP	36.93583	-121.789	47	-8	10
24015	KAKO	AKO	12507	AKRON WASHINGTON CO AP	40.16667	-103.217	1421	-7	10
23061	KALS	ALS	52407	ALAMOSA SAN LUIS AP	37.4389	-105.861	2296	-7	10
93073	KASE	ASE	31507	ASPEN PITKIN CO AP	39.23	-106.871	2353	-7	8
3026	KITR	ITR	110705	BURLINGTON CARSON AP	39.24472	-102.284	1278	-7	10
93037	KCOS	COS	92308	COLORADO SPRINGS MUNI AP	38.81	-104.688	1884	-7	10
93069	KCEZ	CEZ	40407	CORTEZ MONTEZUMA COUNTY AP	37.30694	-108.626	1801	-7	10
24046	KCAG	CAG	32607	CRAIG MOFFAT COUNTY AP	40.49278	-107.524	1887	-7	10
93067	KAPA	APA	20607	DENVER CENTENNIAL AP	39.57028	-104.849	1793	-7	10
3017	KDEN	DEN	91205	DENVER INTL AP	39.8328	-104.658	1650	-7	10
93005	KDRO	DRO	40307	DURANGO LA PLATA CO AP	37.14306	-107.76	2033	-7	10
23066	KGJT	GJT	31307	GRAND JUNCTION WALKER FLD	39.1342	-108.54	1481	-7	10
23067	KLHX	LHX	121605	LA JUNTA MUNI AP	38.0494	-103.512	1278	-7	10
3013	KLAA	LAA	82608	LAMAR MUNI AP	38.07	-102.688	1124	-7	10
93009	KLXV	LXV	82808	LEADVILLE LAKE CO AP	39.22917	-106.317	3011	-7	10
93010	KLIC	LIC	100405	LIMON WSMO	39.18944	-103.716	1635	-7	10
94050	KEEO	EEO	31307	MEEKER AIRPORT	40.04417	-107.889	1940	-7	10
93013	KMTJ	MTJ	102705	MONTROSE REGIONAL AP	38.50583	-107.899	1743	-7	10
93058	KPUB	PUB	40407	PUEBLO MEM AP	38.2901	-104.498	1439	-7	10
23070	KTAD	TAD	102808	TRINIDAD PERRY STOKES AP	37.26222	-104.338	1750	-7	10
94702	KBDR	BDR	61709	BRIDGEPORT SIKORSKY MEM AP	41.15833	-73.1289	2	-5	8
54734	KDXR	DXR	71509	DANBURY MUNI AP	41.37139	-73.4828	138	-5	8
14707	KGON	GON	111708	GROTON NEW LONDON AP	41.3275	-72.0494	3	-5	10
14740	KBDL	BDL	40907	HARTFORD BRADLEY INTL AP	41.9381	-72.6825	58	-5	10
14752	KHFD	HFD	102308	HARTFORD BRAINARD FLD	41.73611	-72.6506	6	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
54788	KMMK	MMK	100708	MERIDEN MARKHAM MUNI AP	41.50972	-72.8278	30	-5	10
14758	KHVN	HVN	33109	NEW HAVEN TWEED AP	41.26389	-72.8872	1	-5	8
54767	KIJD	IJD	102705	WILLIMANTIC WINDHAM AP	41.74194	-72.1836	75	-5	1
13764	KGED	GED	82608	GEORGETOWN SUSSEX CO AP	38.68917	-75.3592	15	-5	10
13781	KILG	ILG	91808	WILMINGTON NEW CASTLE CO AP	39.6728	-75.6008	24	-5	10
12832	KAAF	AAF	52207	APALACHICOLA AP	29.73333	-85.0333	6	-5	10
12818	KBKV	BKV	52407	BROOKSVILLE HERNANDO CO AP	28.47361	-82.4544	20	-5	10
13884	KCEW	CEW	32907	CRESTVIEW BOB SIKES AP	30.77972	-86.5225	58	-6	10
12834	KDAB	DAB	13107	DAYTONA BEACH INTL AP	29.1828	-81.0483	9	-5	10
53853	KDTS	DTS	31407	DESTIN FT WALTON AP	30.4	-86.4717	4	-6	8
12885	KFXE	FXE	73009	FT LAUDERDALE EXECUTIVE AP	26.19694	-80.1708	4	-5	10
12849	KFLL	FLL	80109	FT LAUDERDALE HOLLYWOOD AP	26.0719	-80.1536	3	-5	10
12835	KFMY	FMY	20609	FT MYERS PAGE FLD AP	26.585	-81.8614	5	-5	8
12894	KRSW	RSW	20909	FT MYERS SW FL RGNL AP	26.53611	-81.755	9	-5	10
12895	KFPR	FPR	41607	FT PIERCE ST LUCIE CO INTL AP	27.49806	-80.3767	7	-5	8
12816	KGNV	GNV	30907	GAINESVILLE RGNL AP	29.6919	-82.2755	37	-5	10
92809	KHWO	HWO	80109	HOLLYWOOD NORTH PERRY AP	25.99889	-80.2411	2	-5	10
53860	KCRG	CRG	30507	JACKSONVILLE CRAIG MUNI AP	30.3361	-81.5147	12	-5	8
13889	KJAX	JAX	22707	JACKSONVILLE INTL AP	30.495	-81.6936	8	-5	10
12836	KEYW	EYW	102204	KEY WEST INTL AP	24.555	-81.7522	1	-5	10
12819	KLEE	LEE	60607	LEESBURG MUNI AP	28.82083	-81.8097	20	-5	10
12896	KMTH	MTH	22707	MARATHON AP	24.72583	-81.0517	2	-5	8
3818	KMAI	MAI	52507	MARIANNA MUNI AP	30.83556	-85.1839	33	-6	10
12838	KMLB	MLB	91506	MELBOURNE INTL AP	28.1011	-80.6439	8	-5	10
12839	KMIA	MIA	71409	MIAMI INTL AP	25.7905	-80.3163	9	-5	10
12888	KTMB	TMB	81309	MIAMI KENDALL TAMIAMI EXEC AP	25.6475	-80.4331	2	-5	10
12882	KOPF	OPF	70809	MIAMI OPA LOCKA AP	25.90694	-80.2803	3	-5	10
12897	KAPF	APF	72809	NAPLES MUNI AP	26.1522	-81.7752	3	-5	10
73805	KECP	NA		NW FLORIDA BEACHES INTL AP	30.349	-85.788	17	-6	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
12841	KORL	ORL	51007	ORLANDO EXECUTIVE AP	28.54528	-81.3331	33	-5	10
12815	KMCO	MCO	53107	ORLANDO INTL AP	28.4339	-81.325	27	-5	8
12854	KSFB	SFB	50307	ORLANDO SANFORD AP	28.77972	-81.2436	15	-5	10
13899	KPNS	PNS	32707	PENSACOLA RGNL AP	30.47806	-87.1869	34	-6	10
12812	KPGD	PGD	20409	PUNTA GORDA CHARLOTTE CO AP	26.91722	-81.9914	7	-5	10
12871	KSRQ	SRQ	11409	SARASOTA BRADENTON AP	27.40139	-82.5586	7	-5	10
92806	KSPG	SPG	10809	ST PETERSBURG ALBERT WHITTED	27.76472	-82.6275	2	-5	8
12873	KPIE	PIE	41607	ST PETERSBURG INTL AP	27.91056	-82.6875	2	-5	10
93805	KTLH	TLH	50907	TALLAHASSEE RGNL AP	30.39306	-84.3533	17	-5	10
12842	KTPA	TPA	12709	TAMPA INTL AP	27.96194	-82.5403	6	-5	8
12843	KVRB	VRB	30707	VERO BEACH INTL AP	27.651	-80.4199	9	-5	10
12844	KPBI	PBI	72109	WEST PALM BEACH INTL AP	26.6847	-80.0994	6	-5	10
53847	KNDZ	NA		WHITING FIELD NAS SOUTH	30.70444	-87.0231	54	-6	10
12876	KGIF	GIF	10509	WINTER HAVEN GILBERT AP	28.06222	-81.7542	44	-5	10
13869	KABY	ABY	53107	ALBANY SW GA RGNL AP	31.53556	-84.1944	58	-5	10
13870	KAMG	AMG	10609	ALMA BACON CO AP	31.5358	-82.5067	59	-5	10
13873	KAHN	AHN	51007	ATHENS BEN EPPS AP	33.948	-83.3275	239	-5	8
3888	KFTY	FTY	32007	ATLANTA FULTON CO AP	33.77917	-84.5214	256	-5	8
13874	KATL	ATL	32707	ATLANTA HARTSFIELD INTL AP	33.6301	-84.4418	308	-5	10
53863	KPDK	PDK	32107	ATLANTA PEACHTREE AP	33.875	-84.3022	303	-5	10
3820	KAGS	AGS	51209	AUGUSTA BUSH FLD AP	33.3644	-81.9633	40	-5	10
13837	KDNL	DNL	50409	AUGUSTA DANIEL FLD AP	33.46694	-82.0386	125	-5	10
13878	KSSI	SSI	30707	BRUNSWICK MALCOLM MCKINNON AP	31.1522	-81.3908	5	-5	10
53873	KVPC	VPC	32207	CARTERSVILLE AP	34.12306	-84.8486	228	-5	10
93842	KCSG	CSG	52407	COLUMBUS METRO AP	32.5161	-84.9422	119	-5	10
53838	KGVL	GVL	41207	GAINESVILLE GILMER AP	34.27194	-83.8303	387	-5	8
3813	KMCN	MCN	72407	MACON MIDDLE GA RGNL AP	32.6847	-83.6527	105	-5	10
53819	KFFC	FFC	20306	PEACHTREE CITY FALCON FLD	33.35528	-84.5669	243	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
93801	KRMG	RMG	51707	ROME R B RUSSELL AP	34.34778	-85.1611	195	-5	10
3822	KSAV	SAV	21209	SAVANNAH INTL AP	32.13	-81.21	14	-5	10
93845	KVLD	VLD	60707	VALDOSTA RGNL AP	30.7825	-83.2767	60	-5	10
22551	PHJR	NA		EWA KALAELOA AP	21.31667	-158.067	15	-10	10
21504	PHTO	ITO	62603	HILO INTL AP	19.7191	-155.053	12	-10	10
22521	PHNL	HNL	51209	HONOLULU INTL AP	21.324	-157.929	2	-10	8
22516	PHOG	OGG	62909	KAHULUI AP	20.89972	-156.429	16	-10	10
21510	PHKO	KOA	50809	KAILUA KONA KE-AHOLE AP	19.73556	-156.049	9	-10	10
22536	PHLI	LIH	61109	LIHUE WSO AP 1020.1	21.98389	-159.341	30	-10	10
22534	PHMK	MKK	61509	MOLOKAI AP	21.1545	-157.096	135	-10	10
94989	KAMW	AMW	102705	AMES MUNICIPAL AP	41.99056	-93.6189	291	-6	10
14931	KBRL	BRL	41707	BURLINGTON MUNI AP	40.78333	-91.1253	211	-6	10
14990	KCID	CID	41807	CEDAR RAPIDS MUNI AP	41.8833	-91.7166	265	-6	8
94982	KDVN	DVN	102005	DAVENPORT MUNI AP	41.61389	-90.5914	229	-6	10
14933	KDSM	DSM	41807	DES MOINES INTL AP	41.5338	-93.653	292	-6	10
94908	KDBQ	DBQ	42007	DUBUQUE RGNL AP	42.39778	-90.7036	322	-6	10
94971	KEST	EST	50207	ESTHERVILLE MUNICIPAL AP	43.40111	-94.7472	401	-6	10
14937	KIOW	IOW	102005	IOWA CITY MUNI AP	41.63278	-91.5431	198	-6	10
94991	KLWD	LWD	52907	LAMONI MUNICIPAL AP	40.6306	-93.9008	346	-6	10
94988	KMIW	MIW	102005	MARSHALLTOWN MUNICIPAL AP	42.11056	-92.9161	297	-6	10
14940	KMCW	MCW	51707	MASON CITY MUNI AP	43.1544	-93.3269	373	-6	10
14950	KOTM	OTM	61307	OTTUMWA INDUSTRIAL AP	41.1077	-92.4466	257	-6	10
14943	KSUX	SUX	43009	SIOUX CITY GATEWAY AP	42.3913	-96.3791	334	-6	10
14972	KSPW	SPW	110705	SPENCER MUNI AP	43.16444	-95.2017	407	-6	10
94910	KALO	ALO	62507	WATERLOO MUNI AP	42.5544	-92.4011	265	-6	10
24131	KBOI	BOI	10907	BOISE AIR TERMINAL	43.5666	-116.241	858	-7	10
24133	KBYI	BYI	110705	BURLEY MUNI AP	42.5416	-113.766	1266	-7	10
4114	KLLJ	LLJ	10203	CHALLIS AP	44.52278	-114.215	1534	-7	8
24145	KIDA	IDA	13007	IDAHO FALLS FANNING FLD	43.51639	-112.067	1441	-7	8

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
4110	KJER	JER	50406	JEROME CO AP	42.72667	-114.456	1223	-7	8
24149	KLWS	LWS	41107	LEWISTON NEZ PERCE CO AP	46.3747	-117.016	438	-8	8
94182	KMYL	MYL	60706	MCCALL AP	44.88889	-116.102	1528	-7	10
24154	KMLP	MLP	70306	MULLAN PASS VOR_DME	47.45694	-115.645	1837	-8	8
24156	KPIH	PIH	30607	POCATELLO RGNL AP	42.9202	-112.571	1357	-7	10
94194	KRXE	RXE	102005	REXBURG MADISON CO AP	43.83389	-111.804	1481	-7	10
94178	KTWF	TWF	71807	TWIN FALLS SUN VLY RGNL AP	42.48194	-114.487	1261	-7	10
94870	KCMI	CMI	51007	CHAMPAIGN WILLARD AP	40.03972	-88.2778	229	-6	8
4808	KARR	ARR	120602	CHICAGO AURORA MUNI AP	41.77	-88.4814	216	-6	8
14819	KMDW	MDW	61307	CHICAGO MIDWAY AP	41.78611	-87.7522	187	-6	10
94846	KORD	ORD	62707	CHICAGO OHARE INTL AP	41.995	-87.9336	202	-6	10
4838	KPWK	PWK	62607	CHICAGO PALWAUKEE AP	42.12083	-87.9047	194	-6	8
14880	KUGN	UGN	52407	CHICAGO WAUKEGAN RGNL AP	42.41667	-87.8667	216	-6	10
3887	KDEC	DEC	32007	DECATUR AP	39.83444	-88.8656	206	-6	10
13809	KLWV	LWV	110705	LAWRENCEVILLE INTL AP	38.76417	-87.6056	131	-6	10
53802	KMTO	MTO	102005	MATTOON COLES CO AP	39.47806	-88.2803	217	-6	8
14923	KMLI	MLI	41607	MOLINE QUAD CITY INTL AP	41.46528	-90.5233	180	-6	10
14842	KPIA	PIA	91806	PEORIA GTR PEORIA AP	40.6675	-89.6839	198	-6	10
93989	KUIN	UIN	92006	QUINCY RGNL AP	39.93694	-91.1919	234	-6	8
94822	KRFD	RFD	52207	ROCKFORD GTR ROCKFORD AP	42.1927	-89.093	223	-6	10
93822	KSPI	SPI	92506	SPRINGFIELD CAPITAL AP	39.8447	-89.6839	181	-6	10
94892	KDPA	DPA	62907	WEST CHICAGO DUPAGE AP	41.91444	-88.2464	230	-6	8
3893	KBMG	BMG	52507	BLOOMINGTON MONROE CO AP	39.13333	-86.6167	257	-5	8
93817	KEVV	EVV	92606	EVANSVILLE REGIONAL AP	38.0441	-87.5205	122	-6	8
14827	KFWA	FWA	92106	FT WAYNE INTL AP	40.9705	-85.2063	241	-5	10
14829	KGSH	GSH	121605	GOSHEN MUNI AP	41.5333	-85.7833	253	-5	10
53842	KEYE	EYE	100405	INDIANAPOLIS EAGLE CREEK AP	39.825	-86.2958	249	-5	10
93819	KIND	IND	52207	INDIANAPOLIS INTL AP	39.7318	-86.2788	241	-5	10
14835	KLAF	LAF	91406	LAFAYETTE PURDUE UNIV AP	40.41222	-86.9369	183	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
94895	KMIE	MIE	52907	MUNCIE DELAWARE CO AP	40.23417	-85.3936	286	-5	10
14848	KSBN	SBN	92706	S BEND AP	41.7072	-86.3163	236	-5	10
53866	KGEZ	GEZ	100405	SHELBYVILLE MUNI AP	39.57806	-85.8033	245	-5	10
3868	KHUF	HUF	11603	TERRE HAUTE HULMAN RGNL AP	39.45194	-87.3089	175	-5	8
4846	KVPZ	VPZ	110705	VALPARAISO PORTER CO MUNI AP	41.4525	-87.0058	232	-6	8
13981	KCNU	CNU	60706	CHANUTE MARTIN JOHNSON AP	37.67028	-95.4842	300	-6	10
13984	KCNK	CNK	11007	CONCORDIA MUNI AP	39.5514	-97.6508	448	-6	10
13985	KDDC	DDC	91306	DODGE CITY RGNL AP	37.7686	-99.9678	787	-6	8
13989	KEMP	EMP	40207	EMPORIA MUNI AP	38.32917	-96.1947	365	-6	10
23064	KGCK	GCK	12407	GARDEN CITY RGNL AP	37.92722	-100.725	878	-6	8
23065	KGLD	GLD	102705	GOODLAND RENNER FLD	39.36722	-101.693	1114	-7	10
93990	KHLC	HLC	20607	HILL CITY MUNI AP	39.37556	-99.8297	667	-6	10
13986	KHUT	HUT	12907	HUTCHINSON MUNI AP	38.06528	-97.8606	463	-6	10
3997	KLWC	LWC	100705	LAWRENCE MUNI AP	39.00778	-95.2117	254	-6	10
3936	KMHK	MHK	22107	MANHATTAN MUNI AP	39.1346	-96.6788	322	-6	8
93909	KIXD	IXD	12607	OLATHE JOHNSON CO AP	38.83167	-94.8897	327	-6	10
3967	KOJC	OJC	42007	OLATHE JOHNSON CO EXEC AP	38.85	-94.7392	326	-6	10
3998	KPPF	PPF	40307	PARSONS TRI CITY AP	37.32778	-95.5042	265	-6	8
93997	KRSL	RSL	90606	RUSSELL MUNI AP	38.87611	-98.8092	568	-6	10
13920	KFOE	FOE	90706	TOPEKA FORBES FLD	38.95028	-95.6639	325	-6	10
13996	KTOP	TOP	101702	TOPEKA MUNI AP	39.0725	-95.6261	267	-6	8
3974	KAAO	AAO	102005	WICHITA COLONEL JAMES JABARA A	37.74611	-97.2211	431	-6	10
3928	KICT	ICT	100605	WICHITA DWIGHT D EISENHOWER NA	37.6475	-97.43	403	-6	10
13932	KWLD	WLD	101305	WINFIELD STROTHER FLD AP	37.16806	-97.0369	351	-6	8
93808	KBWG	BWG	110705	BOWLING GREEN WARREN CO AP	36.9647	-86.4238	161	-6	8
93814	KCVG	CVG	42407	CINCINNATI NORTHERN KY AP	39.04306	-84.6717	265	-5	10
53841	KFFT	FFT	102005	FRANKFORT CAPITAL CITY AP	38.18472	-84.9033	238	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
3889	KJKL	JKL	102005	JACKSON JULIAN CARROLL AP	37.59139	-83.3144	416	-5	10
93820	KLEX	LEX	92706	LEXINGTON BLUEGRASS AP	38.0408	-84.6058	299	-5	10
3849	KLOZ	LOZ	110705	LONDON CORBIN AP	37.08722	-84.0769	362	-5	10
13810	KLOU	LOU	52307	LOUISVILLE BOWMAN FLD	38.22806	-85.6636	165	-5	10
93821	KSDF	SDF	61407	LOUISVILLE INTL AP	38.1811	-85.7391	149	-5	10
3816	KPAH	PAH	22207	PADUCAH BARKLEY RGNL AP	37.0563	-88.7744	126	-6	10
93915	KAEX	AEX	42607	ALEXANDRIA INTL AP	31.33472	-92.5586	26	-6	8
13970	KBTR	BTR	92606	BATON ROUGE RYAN AP	30.5372	-91.1469	20	-6	10
12884	KBVE	BVE	41807	BOOTHVILLE ASOS	29.333	-89.4075	1	-6	10
13976	KLFT	LFT	60107	LAFAYETTE RGNL AP	30.205	-91.9875	12	-6	8
3937	KLCH	LCH	41907	LAKE CHARLES RGNL AP	30.12472	-93.2283	3	-6	10
13942	KMLU	MLU	100908	MONROE REGIONAL AP	32.5155	-92.0405	24	-6	8
53915	KARA	ARA	61207	NEW IBERIA AP - ACADIANA RGNL	30.0375	-91.8839	7	-6	8
12916	KMSY	MSY	32307	NEW ORLEANS INTL AP	29.9933	-90.2511	1	-6	10
53917	KNEW	NEW	10803	NEW ORLEANS LAKEFRONT AP	30.0494	-90.0288	3	-6	8
53905	KDTN	DTN	41907	SHREVEPORT DWTN AP	32.54278	-93.745	55	-6	10
13957	KSHV	SHV	60707	SHREVEPORT RGNL AP	32.4472	-93.8244	77	-6	10
53865	KASD	ASD	30207	SLIDELL AP	30.34333	-89.8222	8	-6	10
3996	KTVR	TVR	60507	TALLULAH VICKSBURG AP	32.35	-91.0278	26	-6	10
14702	KBED	BED	52907	BEDFORD HANSCOM FLD	42.47	-71.2894	38	-5	10
54733	KBVY	BVY	100808	BEVERLY MUNI AP	42.58417	-70.9175	29	-5	10
14739	KBOS	BOS	100506	BOSTON LOGAN INTL AP	42.3606	-71.0106	4	-5	8
94624	KCQX	CQX	30107	CHATHAM MUNI AP	41.6875	-69.9933	17	-5	10
4780	KFIT	FIT	102005	FITCHBURG MUNI AP	42.55194	-71.7558	101	-5	10
94720	KHYA	HYA	43009	HYANNIS BARNSTABLE MUNI AP	41.66861	-70.28	14	-5	8
94723	KLWM	LWM	110408	LAWRENCE MUNI AP	42.71722	-71.1239	42	-5	10
14756	KACK	ACK	51909	NANTUCKET MEM AP	41.25306	-70.0608	10	-5	10
94726	KEWB	EWB	103008	NEW BEDFORD MUNI AP	41.67639	-70.9583	22	-5	8
54768	KAQW	AQW	110705	NORTH ADAMS HARRIMAN AP	42.7	-73.1667	200	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
54704	KOWD	OWD	111808	NORWOOD MEM AP	42.19083	-71.1736	15	-5	8
54756	KORE	ORE	102005	ORANGE MUNI AP	42.57	-72.2911	167	-5	10
14763	KPSF	PSF	102705	PITTSFIELD MUNI AP	42.42722	-73.2892	351	-5	10
54769	KPYM	PYM	102005	PLYMOUTH MUNI AP	41.90972	-70.7294	44	-5	10
54777	KTAN	TAN	100206	TAUNTON MUNI AP	41.87556	-71.0211	9	-5	10
94724	KMVY	MVY	51109	VINEYARD HAVEN AP	41.39306	-70.615	19	-5	10
14775	KBAF	BAF	100208	WESTFIELD BARNES MUNI AP	42.15778	-72.7161	80	-5	10
94746	KORH	ORH	32807	WORCESTER RGNL AP	42.2706	-71.8731	305	-5	10
93721	KBWI	BWI	92006	BALTIMORE WASH INTL AP	39.1666	-76.6833	48	-5	10
93706	KHGR	HGR	40307	HAGERSTOWN WASHINGTON CO AP	39.70778	-77.7297	213	-5	8
93786	KOXB	OXB	40507	OCEAN CITY MUNI AP	38.30833	-75.1239	4	-5	10
93720	KSBY	SBY	41107	SALISBURY WICOMICO RGNL AP	38.34056	-75.5103	15	-5	8
14605	KAUG	AUG	102705	AUGUSTA STATE AP	44.3155	-69.7972	107	-5	8
14606	KBGR	BGR	92706	BANGOR INTL AP	44.7978	-68.8185	45	-5	10
14607	KCAR	CAR	92602	CARIBOU MUNI AP	46.8705	-68.0173	190	-5	8
4836	KFVE	FVE	121605	FRENCHVILLE AROOSTOOK AP	47.28556	-68.3133	302	-5	10
54772	KIZG	IZG	91206	FRYEBURG E SLOPES AP	43.99056	-70.9475	136	-5	10
14610	KMLT	MLT	101305	MILLINOCKET MUNI AP	45.6477	-68.6925	124	-5	10
14764	KPWM	PWM	100606	PORTLAND INTL JETPORT	43.6497	-70.3002	14	-5	8
94623	KIWI	IWI	91306	WISCASSET AP	43.96361	-69.7117	13	-5	10
4847	KADG	ADG	100705	ADRIAN LENAWEE CO AP	41.86778	-84.0794	243	-5	10
94849	KAPN	APN	50407	ALPENA CO RGNL AP	45.0716	-83.5644	208	-5	10
94889	KARB	ARB	92408	ANN ARBOR MUNI AP	42.22278	-83.7444	253	-5	10
14815	KBTL	BTL	22007	BATTLE CREEK KELLOGG AP	42.3075	-85.2511	283	-5	10
94871	KBEH	BEH	121605	BENTON HARBOR AIRPORT ASOS	42.1256	-86.4284	196	-5	10
14822	KDET	DET	101206	DETROIT CITY AP	42.40917	-83.01	191	-5	10
94847	KDTW	DTW	60707	DETROIT METRO AP	42.2313	-83.3308	192	-5	10
14853	KYIP	YIP	92608	DETROIT WILLOW RUN AP	42.23333	-83.5333	237	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
14826	KFNT	FNT	91808	FLINT BISHOP INTL AP	42.9666	-83.7494	235	-5	10
4854	KGLR	GLR	100705	GAYLORD OTSEGO CO AP	45.01333	-84.7014	407	-5	10
94860	KGRR	GRR	22107	GRAND RAPIDS INTL AP	42.8825	-85.5239	245	-5	10
14858	KCMX	CMX	112602	HANCOCK HOUGHTON CO AP	47.16861	-88.4889	325	-5	8
4839	KBIV	BIV	111705	HOLLAND TULIP CITY AP	42.74611	-86.0967	206	-5	10
94814	KHTL	HTL	32907	HOUGHTON LK ROSCOMMON AP	44.3591	-84.6738	351	-5	10
94893	KIMT	IMT	110705	IRON MTN FORD AP	45.81833	-88.1144	338	-6	8
14833	KJXN	JXN	41607	JACKSON REYNOLDS FLD	42.2667	-84.4667	304	-5	8
94815	KAZO	AZO	83106	KALAMAZOO BATTLE CK INTL AP	42.23472	-85.5519	265	-5	10
14836	KLAN	LAN	90706	LANSING CAPITAL CITY AP	42.78028	-84.5789	256	-5	10
14840	KMKG	MKG	32007	MUSKEGON CO AP	43.17111	-86.2367	191	-5	10
14841	KPLN	PLN	100705	PELLSTON RGNL AP	45.5644	-84.7927	215	-5	8
94817	KPTK	PTK	42407	PONTIAC OAKLAND CO INTL AP	42.665	-83.4181	297	-5	10
14845	KMBS	MBS	53107	SAGINAW MBS INTL AP	43.53306	-84.0797	201	-5	10
14850	KTVC	TVC	51707	TRAVERSE CITY CHERRY CPTL AP	44.74083	-85.5825	188	-5	8
14910	KAXN	AXN	102705	ALEXANDRIA MUNI AP	45.8679	-95.3941	432	-6	10
94938	KBRD	BRD	102705	BRAINERD CROW WING CO AP	46.40472	-94.1308	372	-6	10
14913	KDLH	DLH	72507	DULUTH INTL AP	46.8369	-92.1833	437	-6	10
94931	KHIB	HIB	102005	HIBBING CHISHOLM HIBBING AP	47.38639	-92.8389	411	-6	10
14918	KINL	INL	90706	INTL FALLS INTL AP	48.5614	-93.3981	361	-6	10
14922	KMSP	MSP	91306	MINNEAPOLIS_ST PAUL AP	44.8831	-93.2289	266	-6	10
94960	KMIC	MIC	41807	MPLS CRYSTAL AP	45.06194	-93.3508	262	-6	10
94963	KFCM	FCM	42507	MPLS FLYING CLOUD AP	44.8321	-93.4705	276	-6	10
94967	KPKD	PKD	41807	PARK RAPIDS MUNI AP	46.90056	-95.0678	437	-6	10
14992	KRWF	RWF	102005	REDWOOD FALLS MUNI AP	44.5483	-95.0804	311	-6	10
14925	KRST	RST	53007	ROCHESTER INTL AP	43.9041	-92.4916	397	-6	10
14926	KSTC	STC	70306	ST CLOUD RGNL AP	45.5433	-94.0513	308	-6	10
14927	KSTP	STP	91206	ST PAUL DOWNTOWN AP	44.93194	-93.0556	213	-6	10
3935	KCGI	CGI	121605	CAPE GIRARDEAU MUNI AP	37.2252	-89.5705	102	-6	8

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
3945	KCOU	COU	62007	COLUMBIA RGNL AP	38.8169	-92.2183	272	-6	10
3963	KJEF	JEF	71107	JEFFERSON CITY MEM AP	38.59111	-92.1558	175	-6	10
13987	KJLN	JLN	92706	JOPLIN REGIONAL AIRPORT	37.1466	-94.5022	299	-6	10
13988	KMKC	MKC	91306	KANSAS CITY DOWNTOWN AP	39.1208	-94.5969	226	-6	8
3947	KMCI	MCI	91906	KANSAS CITY INTL AP	39.2972	-94.7306	306	-6	10
14938	KIRK	IRK	100705	KIRKSVILLE RGNL AP	40.09722	-92.5433	293	-6	10
53879	KLXT	LXT	41805	LEES SUMMIT MUNI AP	38.95972	-94.3714	304	-6	10
3975	KPOF	POF	121605	POPLAR BLUFF MUNI AP	36.7725	-90.3247	100	-6	8
3994	KDMO	DMO	81406	SEDALIA MEM AP	38.70417	-93.1833	274	-6	10
13995	KSGF	SGF	92006	SPRINGFIELD RGNL AP	37.2397	-93.3897	384	-6	10
53904	KSET	SET	102705	ST CHARLES CO AP	38.92861	-90.4281	133	-6	10
13993	KSTJ	STJ	30507	ST JOSEPH ROSECRANS AP	39.7736	-94.9233	249	-6	10
13994	KSTL	STL	92606	ST LOUIS LAMBERT INTL AP	38.7525	-90.3736	162	-6	10
3966	KSUS	SUS	43007	ST LOUIS SPRT OF S L AP	38.65722	-90.6558	141	-6	8
13997	KVIH	VIH	12407	VICHY ROLLA NATIONAL AP	38.13111	-91.7683	344	-6	10
53901	KUNO	UNO	101305	WEST PLAINS MUNI AP	36.87806	-91.9025	373	-6	10
13939	KGLH	GLH	13007	GREENVILLE ASOS	33.4825	-90.9853	39	-6	10
13978	KGWO	GWO	30607	GREENWOOD LEFLORE AP	33.4963	-90.0866	41	-6	10
93874	KGPT	GPT	30607	GULFPORT - BILOXI AP	30.4119	-89.0808	13	-6	10
13833	KHBG	HBG	22707	HATTIESBURG CHAIN MUNI AP	31.28194	-89.2531	46	-6	10
13927	KHKS	HKS	22807	JACKSON HAWKINS FLD	32.33667	-90.2214	104	-6	10
3940	KJAN	JAN	52207	JACKSON INTL AP	32.3205	-90.0777	101	-6	10
93919	KMCB	MCB	41607	MCCOMB_PIKE CO_JOHN E LEWIS AP	31.1827	-90.4708	126	-6	10
13865	KMEI	MEI	60607	MERIDIAN KEY FLD	32.3347	-88.7442	90	-6	10
53858	KPQL	PQL	32607	PASCAGOULA LOTT INTL AP	30.46361	-88.5319	5	-6	8
93862	KTUP	TUP	91708	TUPELO RGNL AP	34.2622	-88.7713	110	-6	10
94055	KBHK	BHK	102705	BAKER MUNI AP	46.3583	-104.25	906	-7	10
24033	KBIL	BIL	90506	BILLINGS INTL AP	45.8069	-108.542	1091	-7	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
24132	KBZN	BZN	42507	BOZEMAN GALLATIN FLD	45.788	-111.161	1349	-7	10
24135	KBTM	BTM	21306	BUTTE BERT MOONEY AP	45.9647	-112.501	1678	-7	10
24137	KCTB	CTB	20306	CUT BANK MUNI AP	48.6033	-112.375	1170	-7	10
24138	KDLN	DLN	13006	DILLON AP	45.2575	-112.554	1585	-7	10
94008	KGGW	GGW	41405	GLASGOW INTL AP	48.2138	-106.621	696	-7	10
24143	KGTF	GTF	32607	GREAT FALLS INTL AP	47.4733	-111.382	1117	-7	10
94012	KHVR	HVR	110705	HAVRE CITY CO AP	48.5428	-109.763	788	-7	10
24144	KHLN	HLN	53107	HELENA RGNL AP	46.6056	-111.964	1167	-7	10
24146	KGPI	GPI	91306	KALISPELL GLACIER AP	48.3042	-114.264	901	-7	8
24036	KLWT	LWT	41706	LEWISTOWN MUNI AP	47.0492	-109.458	1263	-7	10
24150	KLVM	LVM	102005	LIVINGSTON AP	45.6983	-110.441	1415	-7	10
24037	KMLS	MLS	102705	MILES CITY AP	46.4266	-105.883	800	-7	10
94017	KOLF	OLF	102005	WOLF POINT INTL AP	48.09444	-105.574	605	-7	10
3812	KAVL	AVL	101608	ASHEVILLE RGNL AP	35.43194	-82.5375	645	-5	8
93765	KMRH	MRH	20807	BEAUFORT MICHAEL J SMITH FLD	34.73361	-76.6606	2	-5	10
93783	KBUY	BUY	60507	BURLINGTON ALAMANCE AP	36.04667	-79.4769	187	-5	10
93729	KHSE	HSE	31907	CAPE HATTERAS AP	35.2326	-75.6219	3	-5	10
93785	KIGX	IGX	42905	CHAPEL HILL WILLIAMS AP	35.93333	-79.0642	152	-5	10
13881	KCLT	CLT	60209	CHARLOTTE DOUGLAS AP	35.2236	-80.9552	222	-5	10
13754	KNKT	NA		CHERRY POINT MCAS	34.88333	-76.8667	30	-5	10
13786	KECG	ECG	32607	ELIZABETH CITY CGAS	36.26056	-76.175	4	-5	10
93740	KFAY	FAY	60209	FAYETTEVILLE RGNL AP	34.99139	-78.8803	57	-5	10
53870	KAKH	AKH	112008	GASTONIA MUNI AP	35.19667	-81.1558	242	-5	10
3810	KHKY	HKY	110608	HICKORY FAA AP	35.7425	-81.3819	348	-5	8
93782	KMEB	MEB	60209	LAURINBURG MAXTON AP	34.79167	-79.3661	66	-5	10
13776	KLBT	LBT	41907	LUBERTION REGIONAL AP	34.608	-79.0591	37	-5	10
53872	KEQY	EQY	52009	MONROE AP	35.01694	-80.6206	204	-5	10
93719	KEWN	EWN	92206	NEW BERN CRAVEN CO AP	35.0677	-77.048	6	-5	8
13723	KGSO	GSO	63009	PIEDMONT TRIAD INTL AP	36.0969	-79.9432	271	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
13722	KRDU	RDU	70809	RALEIGH DURHAM INTL AP	35.8923	-78.7819	127	-5	10
93759	KRWI	RWI	51507	ROCKY MT WILSON AP	35.855	-77.8931	45	-5	8
13748	KILM	ILM	41307	WILMINGTON INTL AP	34.2675	-77.8997	10	-5	10
93807	KINT	INT	62509	WINSTON SALEM RYNLDS AP	36.13361	-80.2222	292	-5	8
24011	KBIS	BIS	50107	BISMARCK MUNI AP	46.7825	-100.757	503	-6	10
14914	KFAR	FAR	92606	FARGO HECTOR INTL AP	46.92528	-96.8111	274	-6	10
14916	KGFK	GFK	101702	GRAND FORKS INTL AP	47.9428	-97.1839	257	-6	8
94038	KHEI	HEI	100405	HETTINGER MUNI AP	46.01389	-102.655	824	-7	10
14919	KJMS	JMS	11007	JAMESTOWN MUNI AP	46.9258	-98.6691	455	-6	8
24013	KMOT	MOT	90606	MINOT INTL AP	48.2552	-101.273	507	-6	8
24012	KDIK	DIK	11707	THEODORE ROOSEVELT AP	46.7994	-102.797	786	-7	10
94014	KISN	ISN	40407	WILLISTON SLOULIN INTL AP	48.1738	-103.637	580	-6	10
24044	KAIA	AIA	92908	ALLIANCE MUNI AP	42.0573	-102.802	1198	-7	10
94946	KBBW	BBW	100705	BROKEN BOW MUNI AP	41.43333	-99.6333	771	-6	10
24017	KCDR	CDR	100708	CHADRON MUNI AP	42.8374	-103.098	1004	-7	10
94957	KFNB	FNB	91406	FALLS CITY BRENNER FLD	40.08028	-95.5919	299	-6	10
14935	KGRI	GRI	91906	GRAND ISLAND AP	40.9611	-98.3136	561	-6	8
94949	KHSI	HSI	110705	HASTINGS MUNI AP	40.6005	-98.4258	598	-6	8
24091	KIML	IML	60507	IMPERIAL MUNI AP	40.51	-101.62	996	-7	10
14939	KLNK	LNK	32607	LINCOLN MUNI AP	40.8508	-96.7475	363	-6	10
94040	KMCK	MCK	22107	MCCOOK MUNI AP	40.20639	-100.591	779	-6	10
14941	KOFK	OFK	102005	NORFOLK KARL STEFAN AP	41.9855	-97.4352	473	-6	10
24023	KLBF	LBF	102005	NORTH PLATTE RGNL AP	41.1213	-100.669	847	-6	8
14942	KOMA	OMA	91306	OMAHA EPPLEY AIRFIELD	41.3102	-95.8991	299	-6	10
94958	KODX	ODX	100405	ORD EVELYN SHARP FLD	41.62333	-98.9483	629	-6	8
24028	KBFF	BFF	22003	SCOTTSBLUFF HEILIG AP	41.8705	-103.593	1202	-7	8
24030	KSNY	SNY	92508	SIDNEY MUNI AP	41.0993	-102.986	1309	-7	10
94978	KTQE	TQE	102005	TEKAMAH MUNI AP	41.76361	-96.1778	313	-6	10
24032	KVTN	VTN	91306	VALENTINE MILLER FLD	42.8783	-100.55	789	-6	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
94700	KBML	BML	102705	BERLIN MUNI AP	44.57611	-71.1786	342	-5	10
14745	KCON	CON	102005	CONCORD MUNI AP	43.1952	-71.5011	105	-5	8
54770	KAFN	AFN	111705	JAFFREY MUNI AP	42.805	-72.0036	317	-5	10
94765	KLEB	LEB	11007	LEBANON MUNI AP	43.62639	-72.3047	182	-5	10
14710	KMHT	MHT	51309	MANCHESTER AP	42.93333	-71.4383	69	-5	10
54791	KDAW	DAW	102005	ROCHESTER SKYHAVEN AP	43.27806	-70.9222	96	-5	8
54728	KHIE	HIE	102005	WHITEFIELD MT WASHINGTON AP	44.3675	-71.545	320	-5	8
93730	KACY	ACY	112906	ATLANTIC CITY INTL AP	39.4494	-74.5672	18	-5	8
54743	KCDW	CDW	60909	CALDWELL ESSEX CO AP	40.87639	-74.2831	53	-5	8
13735	KMIV	MIV	92606	MILLVILLE MUNI AP	39.3667	-75.0667	21	-5	8
93780	KVAY	VAY	91206	MT HOLLY S JERSEY AP	39.94917	-74.8417	14	-5	10
14734	KEWR	EWR	70809	NEWARK INTL AP	40.6825	-74.1694	2	-5	10
54785	KSMQ	SMQ	100708	SOMERVILLE SOMERSET AP	40.62389	-74.6694	33	-5	10
54793	KFWN	FWN	92408	SUSSEX AP	41.20028	-74.6231	123	-5	10
94741	KTEB	TEB	51909	TETERBORO AP	40.85	-74.0614	2	-5	8
14792	KTTN	TTN	91708	TRENTON MERCER CO AP	40.27694	-74.8158	56	-5	8
23050	KABQ	ABQ	52207	ALBUQUERQUE INTL AP	35.0419	-106.616	1618	-7	10
93033	KCNM	CNM	51007	CARLSBAD CAVERN CITY AP	32.3335	-104.258	985	-7	10
23051	KCAO	CAO	50307	CLAYTON MUNI AIR PK	36.4486	-103.154	1512	-7	10
3027	KCQC	CQC	32007	CLINES CORNERS	35.00278	-105.663	2160	-7	10
23078	KDMN	DMN	92706	DEMING MUNI AP	32.26222	-107.721	1311	-7	10
23090	KFMN	FMN	40407	FARMINGTON RGNL AP	36.74361	-108.229	1675	-7	10
23081	KGUP	GUP	51507	GALLUP MUNI AP	35.5144	-108.794	1972	-7	10
23054	KLVS	LVS	42607	LAS VEGAS MUNI AP	35.65417	-105.142	2092	-7	10
23052	KRTN	RTN	31407	RATON MUNI CREWS AP	36.74139	-104.502	1934	-7	10
23009	KROW	ROW	41607	ROSWELL IND AIR PK	33.3075	-104.508	1112	-7	8
23049	KSAF	SAF	52307	SANTA FE CO MUNI AP	35.61694	-106.089	1923	-7	8
93045	KTCS	TCS	31907	TRUTH OR CONSEQUENCE AP	33.23667	-107.268	1470	-7	8
23048	KTCC	TCC	40507	TUCUMCARI MUNI AP	35.18222	-103.603	1234	-7	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
24121	KEKO	EKO	62807	ELKO RGNL AP	40.8288	-115.789	1533	-8	10
23154	KELY	ELY	121605	ELY YELLAND FLD AP	39.2952	-114.847	1909	-8	10
53123	KVGT	VGT	42607	LAS VEGAS AIR TERMINAL	36.21167	-115.196	670	-8	10
23169	KLAS	LAS	42507	LAS VEGAS MCCARRAN AP	36.0719	-115.163	664	-8	10
24172	KLOL	LOL	101305	LOVELOCK DERBY FLD	40.0681	-118.569	1189	-8	10
3160	KDRA	DRA	13006	MERCURY DESERT ROCK AP	36.6206	-116.028	985	-8	10
23185	KRNO	RNO	51507	RENO TAHOE INTL AP	39.4838	-119.771	1344	-8	10
23153	KTPH	TPH	20306	TONOPAH	38.0511	-117.09	1644	-8	10
24128	KWMC	WMC	111705	WINNEMUCCA MUNI AP	40.9017	-117.808	1309	-8	10
14735	KALB	ALB	90806	ALBANY AP	42.7431	-73.8092	95	-5	10
4725	KBGM	BGM	21307	BINGHAMTON GREATER AP	42.2068	-75.98	486	-5	8
14733	KBUF	BUF	60409	BUFFALO NIAGARA INTL AP	42.9408	-78.7358	218	-5	10
94704	KDSV	DSV	92408	DANSVILLE MUNI AP	42.57083	-77.7133	196	-5	10
14747	KDKK	DKK	91908	DUNKIRK CHAUTAUQUA AP	42.49333	-79.2722	203	-5	10
14748	KELM	ELM	42507	ELMIRA CORNING RGNL AP	42.15944	-76.8919	288	-5	10
54787	KFRG	FRG	51809	FARMINGDALE AP	40.73417	-73.4169	24	-5	10
54773	KFZY	FZY	102705	FULTON OSWEGO CO AP	43.34972	-76.3847	141	-5	8
14750	KGFL	GFL	102705	GLENS FALLS AP	43.35	-73.6167	98	-5	10
4781	KISP	ISP		ISLIP LONG IS MACARTHUR AP	40.79389	-73.1017	26	-5	8
94725	KMSS	MSS	111705	MASSENA INTL AP	44.93583	-74.8458	65	-5	10
4789	KMGJ	MGJ	71509	MONTGOMERY ORANGE AP	41.50917	-74.265	106	-5	10
94728	KNYC	NYC	91806	NEW YORK CNTRL PK TWR	40.77889	-73.9692	40	-5	10
94789	KJFK	JFK	63009	NEW YORK JFK INTL AP	40.63861	-73.7622	3	-5	10
14732	KLGA	LGA	70609	NEW YORK LAGUARDIA AP	40.77944	-73.8803	3	-5	10
54778	KPEO	PEO	110705	PENN YAN AP	42.6425	-77.0564	263	-5	10
64776	KPBG	PBG	110705	PLATTSBURGH INTL AP	44.65	-73.4667	71	-5	10
14757	KPOU	POU	91206	POUGHKEEPSIE DUTCHESS CO AP	41.6266	-73.8842	51	-5	8
14768	KROC	ROC	102008	ROCHESTER GTR INTL AP	43.1167	-77.6767	164	-5	10
64775	KRME	RME	32807	ROME GRIFFISS AIRFIELD	43.23389	-75.4117	147	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
94740	KSLK	SLK	111805	SARANAC RGNL AP	44.38528	-74.2067	501	-5	8
54790	KHWV	HWV	120308	SHIRLEY BROOKHAVEN AP	40.82167	-72.8689	20	-5	10
14771	KSYR	SYR	20107	SYRACUSE HANCOCK INTL AP	43.1111	-76.1038	126	-5	10
94790	KART	ART	110705	WATERTOWN INTL AP	43.9922	-76.0217	97	-5	8
54757	KELZ	ELZ	110705	WELLSVILLE MUNI AP	42.10944	-77.9919	647	-5	10
94745	KHPN	HPN	52209	WESTCHESTER CO AP	41.06694	-73.7075	116	-5	10
14719	KFOK	FOK	21109	WESTHAMPTN GABRESKI AP	40.84361	-72.6322	18	-5	10
14895	KCAK	CAK	70709	AKRON CANTON RGNL AP	40.91667	-81.4333	368	-5	10
14813	KAKR	AKR	100306	AKRON FULTON INTL AP	41.0375	-81.4642	318	-5	10
4857	KHZY	HZY	100606	ASHTABULA CO AP	41.77806	-80.6958	278	-5	10
93812	KLUK	LUK	42007	CINCINNATI LUNKEN AP	39.10333	-84.4189	149	-5	10
4853	KBKL	BKL	81808	CLEVELAND BURKE AP	41.5175	-81.6836	177	-5	8
14820	KCLE	CLE	52207	CLEVELAND HOPKINS INTL AP	41.405	-81.8528	235	-5	10
4804	KOSU	OSU	61107	COLUMBUS OHIO STATE UNIV AP	40.07806	-83.0781	276	-5	8
14821	KCMH	CMH	53007	COLUMBUS PORT COLUMBUS INTL AP	39.99139	-82.8808	247	-5	10
93815	KDAY	DAY	50907	DAYTON INTL AP	39.90611	-84.2186	305	-5	10
53859	KMGY	MGY	43007	DAYTON WRIGHT BROS AP	39.59361	-84.2264	290	-5	8
4851	KDFI	DFI	42407	DEFIANCE AP	41.3375	-84.4289	215	-5	10
4849	KLPR	LPR	120606	ELYRIA LORAIN CO AP	41.34611	-82.1794	241	-5	10
14825	KFDY	FDY	100506	FINDLAY AP	41.01361	-83.6686	244	-5	10
53855	KHAO	HAO	50807	HAMILTON BUTLER CO RGNL AP	39.36444	-84.5247	189	-5	10
53844	KLHQ	LHQ	61207	LANCASTER FAIRFIELD AP	39.75556	-82.6572	259	-5	10
4850	KAOH	AOH	91906	LIMA ALLEN COUNTY AP	40.7075	-84.0272	297	-5	10
14891	KMFD	MFD	52307	MANSFIELD LAHM MUNI AP	40.82028	-82.5178	395	-5	10
4855	KMNN	MNN	120606	MARION MUNI AP	40.61611	-83.0636	301	-5	10
4852	KPHD	PHD	90706	NEW PHILADELPHIA FLD	40.47194	-81.4236	272	-5	10
4858	KVTA	VTA	120606	NEWARK HEATH AP	40.02278	-82.4625	267	-5	10
94830	KTOL	TOL	12007	TOLEDO EXPRESS AP	41.58861	-83.8014	204	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
4848	KTDZ	TDZ	82008	TOLEDO METCALF FLD	41.56306	-83.4764	189	-5	10
13841	KILN	ILN	102005	WILMINGTON AIRBORNE PARK	39.42028	-83.8217	321	-5	10
4842	KBJJ	BJJ	100406	WOOSTER WAYNE CO AP	40.87306	-81.8867	336	-5	10
14852	KYNG	YNG	90808	YOUNGSTOWN RGNL AP	41.25444	-80.6739	360	-5	10
93824	KZZV	ZZV	30907	ZANESVILLE MUNI AP	39.94444	-81.8922	268	-5	8
3959	KBVO	BVO	21309	BARTLESVILLE F P FLD	36.7683	-96.0261	218	-6	10
3932	KCSM	CSM	12209	CLINTON-SHERMAN AP	35.3568	-99.2042	586	-6	10
3981	KFDR	FDR	21809	FREDERICK MUNI AP	34.21	-98.59	383	-6	10
13975	KGAG	GAG	12109	GAGE AP	36.2967	-99.7689	668	-6	10
53913	KGOK	GOK	72607	GUTHRIE MUNI AP	35.8517	-97.4142	326	-6	10
3030	KGUY	GUY	90606	GUYMON MUNI AP	36.68167	-101.505	950	-6	8
93986	KHBR	HBR	80107	HOBART MUNI AP	34.9894	-99.0525	474	-6	10
3950	KLAW	LAW	22509	LAWTON MUNI AP	34.5584	-98.4172	326	-6	10
93950	KMLC	MLC	43009	MCALESTER RGNL AP	34.8822	-95.783	235	-6	10
93953	KMKO	MKO	20907	MUSKOGEE DAVIS FLD	35.65667	-95.3614	184	-6	10
3954	KPWA	PWA	10709	OKLAHOMA CITY POST AP	35.53417	-97.6469	395	-6	8
13967	KOKC	OKC	11409	OKLAHOMA CITY WILL ROGERS AP	35.3889	-97.6006	392	-6	8
13969	KPNC	PNC	20609	PONCA CITY MUNI AP	36.73667	-97.1019	305	-6	10
3965	KSWO	SWO	12009	STILLWATER RGNL AP	36.1624	-97.0894	300	-6	10
13968	KTUL	TUL	42209	TULSA INTL AP	36.1994	-95.8872	198	-6	10
53908	KRVS	RVS	21209	TULSA R L JONES JR AP	36.03944	-95.9844	190	-6	10
94224	KAST	AST	32006	ASTORIA RGNL AP	46.1569	-123.883	3	-8	10
94281	KUAO	UAO	11503	AURORA STATE AP	45.24861	-122.769	60	-8	8
24130	KBKE	BKE	50806	BAKER CITY MUNI AP	44.8428	-117.809	1024	-8	10
94185	KBNO	BNO	50806	BURNS MUNI AP	43.595	-118.956	1262	-8	10
24221	KEUG	EUG	40507	EUGENE MAHLON SWEET AP	44.1278	-123.221	108	-8	10
4113	KHRI	HRI	41907	HERMISTON MUNI AP	45.82583	-119.261	196	-8	10
94236	KLMT	LMT	52507	KLAMATH FALLS INTL AP	42.14694	-121.724	1245	-8	10
94273	KMMV	MMV	40306	MCMINNVILLE MUNI AP	45.19472	-123.134	48	-8	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
24225	KMFR	MFR	40607	MEDFORD ROGUE VLY AP	42.3811	-122.872	395	-8	10
24162	KONO	ONO	50806	ONTARIO MUNI AP	44.02056	-117.013	668	-7	10
24155	KPDT	PDT	42607	PENDLETON E OR RGNL AP	45.6983	-118.855	453	-8	10
24229	KPDX	PDX	20107	PORTLAND INTL AP	45.5958	-122.609	6	-8	10
24242	KTTD	TTD	40407	PORTLAND TROUTDALE AP	45.55111	-122.409	9	-8	10
94261	KHIO	HIO	31607	PORTLAND-HILLSBORO AP	45.54056	-122.949	63	-8	10
24230	KRDM	RDM	32207	REDMOND ROBERTS FLD	44.2558	-121.139	928	-8	10
24231	KRBG	RBG	22706	ROSEBURG RGNL AP	43.23889	-123.355	158	-8	10
24232	KSLE	SLE	51507	SALEM MCNARY FLD	44.905	-123.001	62	-8	10
4201	KSPB	SPB	22706	SCAPPOOSE IND AP	45.77278	-122.861	16	-8	10
24235	KSXT	SXT	121605	SEXTON SUMMIT	42.6003	-123.364	1168	-8	10
14737	KABE	ABE	93008	ALLENTOWN INTL AP	40.65083	-75.4492	119	-5	8
14736	KAOO	AOO	82108	ALTOONA BLAIR CO AP	40.29639	-78.3203	451	-5	8
4751	KBFD	BFD	111705	BRADFORD RGNL AP	41.8	-78.6333	653	-5	8
54792	KFIG	FIG	102705	CLEARFIELD LAWRENCE AP	41.04667	-78.4117	462	-5	10
54786	KDYL	DYL	90208	DOYLESTOWN AP	40.33	-75.1225	119	-5	10
4787	KDUJ	DUJ	40207	DUBOIS JEFFERSON CO AP	41.17833	-78.8989	553	-5	8
14860	KERI	ERI	82108	ERIE INTL AP	42.08	-80.1825	223	-5	10
14751	KCXY	CXY	71207	HARRISBURG CPTL CY AP	40.21722	-76.8514	104	-5	8
4726	KJST	JST	82506	JOHNSTOWN CAMBRIA CO AP	40.31611	-78.8339	695	-5	8
54737	KLNS	LNS	81508	LANCASTER AP	40.12028	-76.2944	122	-5	8
14711	KMDT	MDT	82208	MIDDLETOWN HARRISBURG INTL AP	40.1962	-76.7724	95	-5	8
54789	KMPO	MPO	91908	MT POCONO MOUNTAINS AP	41.13889	-75.3794	578	-5	10
13739	KPHL	PHL	73009	PHILADELPHIA INTL AP	39.8683	-75.2311	3	-5	8
94732	KPNE	PNE	71207	PHILADELPHIA NE AP	40.08194	-75.0111	30	-5	10
14762	KAGC	AGC	40307	PITTSBURGH ALLEGHENY CO AP	40.35472	-79.9217	380	-5	8
94823	KPIT	PIT	72809	PITTSBURGH INTL AP	40.4846	-80.2144	367	-5	10
4843	KGKJ	GKJ	10606	PORT MEADVILLE AP	41.62639	-80.215	426	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
54782	KPTW	PTW	90408	POTTSTOWN LIMERICK AP	40.23833	-75.5572	90	-5	8
14712	KRDG	RDG	111908	READING SPAATZ FLD	40.36667	-75.9667	87	-5	8
14770	KSEG	SEG	81808	SELINSGROVE PENN VLY AP	40.82056	-76.8642	135	-5	10
14777	KAVP	AVP	32807	WILKES-BARRE INTL AP	41.3336	-75.7269	283	-5	10
14778	KIPT	IPT	53007	WILLIAMSPORT	41.2433	-76.9217	158	-5	8
93778	KTHV	THV	31505	YORK AP	39.91806	-76.8742	141	-5	10
11641	TJSJ	SJU	12709	SAN JUAN L M MARIN AP	18.4325	-66.0108	3	-4	10
14787	KUUU	UUU	92606	NEWPORT STATE AP	41.53333	-71.2833	43	-5	8
14765	KPVD	PVD	71709	PROVIDENCE T F GREEN AP	41.7219	-71.4325	18	-5	10
14794	KWST	WST	32107	WESTERLY STATE AP	41.34972	-71.7989	21	-5	8
93846	KAND	AND	102108	ANDERSON CO AP	34.4977	-82.7097	232	-5	10
13880	KCHS	CHS	61009	CHARLESTON INTL AP	32.8986	-80.0402	12	-5	10
53850	KCEU	CEU	100208	CLEMSON OCONEE CO AP	34.67194	-82.8864	265	-5	8
13883	KCAE	CAE	51209	COLUMBIA METRO AP	33.9419	-81.1181	69	-5	10
53867	KCUB	CUB	42809	COLUMBIA OWENS DWTN AP	33.97056	-80.9958	55	-5	8
13744	KFLO	FLO	41707	FLORENCE RGNL AP	34.1852	-79.7238	45	-5	10
13886	KGMU	GMU	102808	GREENVILLE DWTN AP	34.84611	-82.3461	309	-5	8
53874	KGRD	GRD	101508	GREENWOOD CO AP	34.24861	-82.1592	192	-5	10
3870	KGSP	GSP	110508	GRNVL SPART AP	34.8842	-82.2209	287	-5	8
93718	KCRE	CRE	42007	N MYRTLE BCH AP	33.81167	-78.7239	10	-5	10
53854	KOGB	OGB	42909	ORANGEBURG MUNI AP	33.46167	-80.8581	55	-5	10
53871	KUZA	UZA	51909	ROCK HILL YORK CO AP	34.98694	-81.0575	200	-5	10
14929	KABR	ABR	102705	ABERDEEN RGNL AP	45.4433	-98.413	395	-6	8
94032	KCUT	CUT	102005	CUSTER CO AP	43.73306	-103.611	1690	-7	10
14936	KHON	HON	42809	HURON RGNL AP	44.3981	-98.2231	390	-6	10
94950	KMHE	MHE	20306	MITCHELL MUNI AP	43.7743	-98.0384	396	-6	8
94052	KMBG	MBG	10907	MOBRIDGE MUNI AP	45.54639	-100.408	517	-6	10
24024	KPHP	PHP	51507	PHILIP AP	44.05111	-101.601	672	-7	10
24025	KPIR	PIR	111705	PIERRE RGNL AP	44.3813	-100.286	531	-6	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
94039	KIEN	IEN	102908	PINE RIDGE AP	43.02056	-102.518	1005	-7	10
24090	KRAP	RAP	92806	RAPID CITY REGIONAL AP	44.0433	-103.054	963	-7	10
14944	KFSD	FSD	60706	SIOUX FALLS FOSS FLD	43.5778	-96.7539	435	-6	10
14946	KATY	ATY	110705	WATERTOWN RGNL AP	44.9047	-97.1494	533	-6	8
94990	KICR	ICR	82108	WINNER WILEY FLD	43.39056	-99.8422	619	-6	10
13877	KTRI	TRI	42307	BRISTOL TRI CITY AP	36.4731	-82.4044	457	-5	10
13882	KCHA	CHA	32707	CHATTANOOGA LOVELL AP	35.0311	-85.2014	205	-5	10
3894	KCKV	CKV	41907	CLARKSVILLE OUTLAW AP	36.62389	-87.4194	171	-6	10
3847	KCSV	CSV	41007	CROSSVILLE MEM AP	35.9509	-85.0813	569	-6	8
13891	KTYS	TYS	41707	KNOXVILLE MCGHEE TYSON AP	35.8181	-83.9858	293	-5	10
13893	KMEM	MEM	100608	MEMPHIS INTL AP	35.0564	-89.9865	77	-6	10
13897	KBNA	BNA	40507	NASHVILLE INTL AP	36.11889	-86.6892	183	-6	10
53868	KOQT	OQT	32207	OAK RIDGE ASOS	36.0236	-84.2375	277	-5	10
13962	KABI	ABI	12909	ABILENE RGNL AP	32.4105	-99.6822	546	-6	10
12932	KALI	ALI	22707	ALICE INTL AP	27.74111	-98.0247	53	-6	10
23047	KAMA	AMA	82506	AMARILLO INTL AP	35.2295	-101.704	1098	-6	10
12976	KLBX	LBX	62107	ANGLETON BRAZORIA AP	29.10972	-95.4619	8	-6	10
53907	KGKY	GKY	22607	ARLINGTON MUNI AP	32.66361	-97.0939	188	-6	10
13904	KAUS	AUS	103008	AUSTIN BERGSTROM AP	30.1831	-97.6799	146	-6	10
13958	KATT	ATT	30807	AUSTIN-CAMP MABRY	30.3208	-97.7604	204	-6	10
3024	KBGD	BGD	90506	BORGER HUTCHINSON CO AP	35.695	-101.395	925	-6	10
12919	KBRO	BRO	40607	BROWNSVILLE INTL AP	25.9141	-97.423	7	-6	10
3999	KBMQ	BMQ	90308	BURNET MUNI AP	30.7406	-98.2354	393	-6	10
23007	KCDS	CDS	42607	CHILDRESS MUNI AP	34.4272	-100.283	595	-6	10
3904	KCLL	CLL	62309	COLLEGE STN	30.58917	-96.3647	93	-6	8
53902	KCXO	CXO	52609	CONROE MONTGOMERY CO AP	30.35667	-95.4139	71	-6	8
12924	KCRP	CRP	21306	CORPUS CHRISTI INTL AP	27.7742	-97.5122	13	-6	10
53912	KCRS	CRS	22207	CORSICANA CAMPBELL FLD	32.03111	-96.3989	136	-6	10
12947	KCOT	COT	22607	COTULLA LA SALLE CO AP	28.45667	-99.2183	145	-6	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
3927	KDFW	DFW	52709	DAL-FTW WSCMO AP	32.8978	-97.0189	171	-6	10
93042	KDHT	DHT	90506	DALHART FAA AP	36.0167	-102.55	1216	-6	10
13960	KDAL	DAL	52809	DALLAS FAA AP	32.8519	-96.8555	134	-6	10
3971	KRBD	RBD	30707	DALLAS REDBIRD AP	32.68083	-96.8681	201	-6	10
22010	KDRT	DRT	60707	DEL RIO INTL AP	29.3784	-100.927	304	-6	10
3991	KDTO	DTO	22707	DENTON MUNI AP	33.20611	-97.1989	197	-6	10
23044	KELP	ELP	111308	EL PASO INTL AP	31.81111	-106.376	1194	-7	10
23091	KFST	FST	50107	FT STOCKTON PECOS AP	30.91194	-102.917	917	-6	10
53909	KAFW	AFW	22007	FT WORTH ALLIANCE AP	32.97333	-97.3181	209	-6	10
13961	KFTW	FTW	21003	FT WORTH MEACHAM FLD	32.81917	-97.3614	209	-6	8
12923	KGLS	GLS	53007	GALVESTON SCHOLES FLD	29.2733	-94.8592	2	-6	8
12904	KHRL	HRL	50907	HARLINGEN RIO GRANDE AP	26.22806	-97.6542	10	-6	10
12962	KHDO	HDO	50207	HONDO MUNI AP	29.3601	-99.1742	280	-6	10
12975	KLVJ	LVJ	60607	HOUSTON CLOVER FLD	29.51889	-95.2417	13	-6	8
12918	KHOU	HOU	62707	HOUSTON HOBBY AP	29.63806	-95.2819	13	-6	10
53910	KDWH	DWH	73107	HOUSTON HOOKS MEM AP	30.0675	-95.5561	47	-6	10
12960	KIAH	IAH	61109	HOUSTON INTERCONT AP	29.98	-95.36	29	-6	10
12977	KSGR	SGR	62207	HOUSTON SUGARLAND MEM	29.62194	-95.6567	26	-6	8
53903	KUTS	UTS	61909	HUNTSVILLE MUNI AP	30.74389	-95.5861	106	-6	10
13973	KJCT	JCT	32807	JUNCTION KIMBLE CO AP	30.51083	-99.7664	524	-6	10
3901	KGGG	GGG	100808	LONGVIEW E TX RGNL AP	32.38472	-94.7117	111	-6	10
23042	KLBB	LBB	41107	LUBBOCK INTL AP	33.6656	-101.823	992	-6	10
93987	KLFK	LFK	112008	LUFKIN ANGELINA CO AP	31.23611	-94.7544	88	-6	8
12959	KMFE	MFE	51507	MCALLEN MILLER INTL AP	26.18389	-98.2539	30	-6	10
53914	KTKI	TKI	30507	MCKINNEY MUNI AP	33.19028	-96.5914	179	-6	10
23023	KMAF	MAF	41107	MIDLAND INTL AP	31.9475	-102.209	872	-6	8
93985	KMWL	MWL	22106	MINERAL WELLS AP	32.7816	-98.0602	283	-6	10
12971	KBAZ	BAZ	51305	NEW BRAUNFELS MUNI AP	29.7089	-98.0458	197	-6	10
3031	KODO	ODO	32907	ODESSA SCHLEMEYER FLD	31.92056	-102.387	906	-6	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
12935	KPSX	PSX	61107	PALACIOS MUNI AP	28.72472	-96.2536	4	-6	10
23055	KGDP	GDP	52207	PINE SPRINGS NP	31.83111	-104.809	1663	-6	6
12917	KBPT	BPT	62007	PORT ARTHUR SE TX AP	29.95056	-94.0206	5	-6	10
12957	KPIL	PIL	40307	PORT ISABEL CAMERON AP	26.16583	-97.3458	4	-6	10
12972	KRKP	RKP	30107	ROCKPORT ARANSAS CO AP	28.08361	-97.0464	7	-6	10
23034	KSJT	SJT	52307	SAN ANGELO MATHIS FLD	31.35167	-100.495	584	-6	10
12921	KSAT	SAT	102208	SAN ANTONIO INTL AP	29.5443	-98.4839	240	-6	10
12970	KSSF	SSF	90908	SAN ANTONIO STINSON AP	29.3389	-98.472	174	-6	10
53911	KTRL	TRL	30607	TERRELL MUNI AP	32.71	-96.2672	144	-6	8
13972	KTYR	TYR	103008	TYLER POUNDS FLD	32.35417	-95.4025	166	-6	10
12912	KVCT	VCT	22307	VICTORIA RGNL AP	28.8614	-96.9303	35	-6	10
13959	KACT	ACT	30207	WACO RGNL AP	31.61889	-97.2283	152	-6	10
13966	KSPS	SPS	32409	WICHITA FALLS MUNI AP	33.9786	-98.4928	310	-6	10
23040	KINK	INK	40307	WINKLER CO AP	31.7801	-103.202	856	-6	10
23159	KBCE	BCE	91906	BRYCE CANYON AP	37.70639	-112.146	2307	-7	10
93129	KCDC	CDC	120606	CEDAR CITY MUNI AP	37.7086	-113.094	1703	-7	8
94128	KLGU	LGU	110705	LOGAN CACHE AP	41.78722	-111.853	1356	-7	10
23176	KMLF	MLF	21306	MILFORD MUNI AP	38.41667	-113.017	1538	-7	10
93075	KCNY	CNY	50307	MOAB CANYONLAND AP	38.75	-109.763	1390	-7	10
93141	KPUC	PUC	110705	PRICE CARBON CO AP	39.60917	-110.755	1779	-7	10
24127	KSLC	SLC	52307	SALT LAKE CITY INTL AP	40.7781	-111.969	1288	-7	10
94030	KVEL	VEL	32707	VERNAL MUNI AP	40.44278	-109.513	1606	-7	10
93775	KOFP	OFP	32907	ASHLAND HANOVER CO MUNI AP	37.70806	-77.4344	63	-5	10
93736	KCHO	CHO	42307	CHARLOTTESVILLE AP	38.13861	-78.4531	188	-5	8
13728	KDAN	DAN	101508	DANVILLE RGNL AP	36.5728	-79.3361	174	-5	10
13733	KLYH	LYH	103008	LYNCHBURG RGNL AP	37.3208	-79.2067	287	-5	10
93741	KPHF	PHF	32007	NEWPORT NEWS INTL AP	37.13194	-76.4931	11	-5	10
13737	KORF	ORF	32707	NORFOLK INTL AP	36.9033	-76.1922	9	-5	10
13740	KRIC	RIC	32807	RICHMOND INTL AP	37.505	-77.3202	50	-5	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
13741	KROA	ROA	102308	ROANOKE RGNL AP	37.3169	-79.9741	358	-5	10
93773	KAKQ	AKQ	31307	WAKEFIELD MUNI AP	36.98389	-77.0072	33	-5	8
93739	KWAL	WAL	41107	WALLOPS ISLAND FLIGHT FAC	37.9372	-75.4708	14	-5	10
93738	KIAD	IAD	100306	WASHINGTON DC DULLES AP	38.9408	-77.4636	88	-5	10
13743	KDCA	DCA	92606	WASHINGTON REAGAN AP	38.8483	-77.0341	3	-5	8
94705	KMPV	MPV	102705	BARRE MONTPELIER AP	44.2035	-72.5623	343	-5	8
54781	KDDH	DDH	110705	BENNINGTON MORSE ST AP	42.89139	-73.2469	243	-5	10
14742	KBTV	BTV	92402	BURLINGTON INTL AP	44.4683	-73.1499	101	-5	8
54771	KMVL	MVL	110705	MORRISVILLE STOWE STATE AP	44.53444	-72.6144	225	-5	8
54740	KVSF	VSF	110705	SPRINGFIELD HARTNESS AP	43.34361	-72.5178	176	-5	8
24217	KBLI	BLI	40307	BELLINGHAM INTL AP	48.79389	-122.537	45	-8	10
94119	KDEW	DEW	91206	DEER PARK AP	47.97417	-117.428	668	-8	10
24220	KELN	ELN	71107	ELLENSBURG BOWERS FLD	47.03389	-120.53	535	-8	10
24141	KEPH	EPH	111705	EPHRATA MUNI AP	47.3078	-119.515	382	-8	10
24222	KPAE	PAE	32907	EVERETT SNOHOMISH AP	47.90778	-122.28	181	-8	10
94276	KFHR	FHR	121605	FRIDAY HARBOR AP	48.52222	-123.023	27	-8	8
94225	KHQM	HQM	41006	HOQUIAM BOWERMAN AP	46.9727	-123.93	4	-8	10
24110	KMWH	MWH	81507	MOSES LAKE GRANT CO AP	47.20778	-119.319	357	-8	10
24227	KOLM	OLM	51007	OLYMPIA AP	46.9733	-122.903	57	-8	8
94197	KOMK	OMK	62707	OMAK	48.46444	-119.517	396	-8	10
94266	KCLM	CLM	50806	PORT ANGELES INTL AP	48.12028	-123.498	79	-8	10
94129	KPUW	PUW	110705	PULLMAN MOSCOW RGNL AP	46.74389	-117.109	772	-8	10
94240	KUIL	UIL	52506	QUILLAYUTE STATE AP	47.9375	-124.555	56	-8	10
94248	KRNT	RNT	33007	RENTON MUNI AP	47.49333	-122.214	7	-8	8
24234	KBFI	BFI	51707	SEATTLE BOEING FLD	47.53028	-122.301	5	-8	8
24233	KSEA	SEA	51707	SEATTLE TACOMA INTL AP	47.4444	-122.314	113	-8	10
94227	KSHN	SHN	111705	SHELTON SANDERSON FLD	47.238	-123.141	83	-8	10
94176	KSFF	SFF	62607	SPOKANE FELTS FLD	47.68306	-117.321	595	-8	10
24157	KGEG	GEG	61407	SPOKANE INTL AP	47.6216	-117.528	717	-8	10

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
94274	KTIW	TIW	51607	TACOMA NARROWS AP	47.2675	-122.576	89	-8	8
24219	KDLS	DLS	62206	THE DALLES MUNI AP	45.6194	-121.166	72	-8	10
94298	KVUO	VUO	21306	VANCOUVER PEARSON AP	45.62083	-122.657	8	-8	8
24160	KALW	ALW	82207	WALLA WALLA RGNL AP	46.09472	-118.287	355	-8	10
94239	KEAT	EAT	121605	WENATCHEE PANGBORN AP	47.3977	-120.201	375	-8	10
24243	KYKM	YKM	80807	YAKIMA AIR TERMINAL	46.5683	-120.543	324	-8	10
94929	KASX	ASX	111705	ASHLAND KENNEDY MEM AP	46.54861	-90.9189	251	-6	10
94994	KOVS	OVS	100705	BOSCOBEL AP	43.15611	-90.6775	203	-6	10
14991	KEAU	EAU	32007	EAU CLAIRE RGNL AP	44.8665	-91.4879	270	-6	10
4840	KFLD	FLD	102005	FOND DU LAC CO AP	43.76944	-88.4908	246	-6	10
14898	KGRB	GRB	82605	GREEN BAY A S INTL AP	44.4794	-88.1366	209	-6	10
94973	KHYR	HYR	110705	HAYWARD MUNI AP	46.02611	-91.4442	367	-6	10
4845	KENW	ENW	32607	KENOSHA RGNL AP	42.595	-87.9381	225	-6	10
14920	KLSE	LSE	92006	LA CROSSE MUNI AP	43.8788	-91.2527	199	-6	10
14921	KLNR	LNR	50907	LONE ROCK TRI CO AP	43.21194	-90.1814	219	-6	10
14837	KMSN	MSN	51007	MADISON DANE RGNL AP	43.1405	-89.3452	264	-6	10
94985	KMFI	MFI	100405	MARSHFIELD MUNI AP	44.63806	-90.1875	383	-6	10
14839	KMKE	MKE	91406	MILWAUKEE MITCHELL AP	42.955	-87.9044	204	-6	10
94855	KOSH	OSH	110402	OSHKOSH WITTMAN AP	43.98444	-88.5569	238	-6	8
94818	KRAC	RAC	102705	RACINE BATTEN AP	42.76111	-87.8136	202	-6	10
4803	KRHI	RHI	100705	RHINELANDER ONEIDA AP	45.6314	-89.4823	495	-6	8
4841	KSBM	SBM	100405	SHEBOYGAN CO MEM AP	43.76944	-87.8506	227	-6	10
14897	KAUW	AUW	100405	WAUSAU ASOS	44.9288	-89.6277	366	-6	10
4826	KISW	ISW	100405	WISCONSIN RAPIDS ALEXANDER FLD	44.35917	-89.8369	311	-6	10
3872	KBKW	BKW	30907	BECKLEY RALEIGH CO AP	37.7836	-81.123	766	-5	10
3859	KBLF	BLF	100308	BLUEFIELD MERCER CO AP	37.2958	-81.2077	875	-5	8
13866	KCRW	CRW	91906	CHARLESTON YEAGER AP	38.3794	-81.59	277	-5	8
3802	KCKB	CKB	52207	CLARKSBURG BENEDUM AP	39.29556	-80.2289	361	-5	8

Surface WBAN	Surface Call	Ice Free Call	Ice Free Date	Name	Surface Latitude	Surface Longitude	Surface Elevation	UTC	Anemo Height (m)
13729	KEKN	EKN	92206	ELKINS RANDOLPH CO AP	38.8853	-79.8528	603	-5	8
3860	KHTS	HTS	12607	HUNTINGTON TRI STATE AP	38.365	-82.555	251	-5	8
13734	KMRB	MRB	31407	MARTINSBURG E WV RGNL AP	39.4019	-77.9844	163	-5	8
13736	KMGW	MGW	22807	MORGANTOWN HART FLD	39.64278	-79.9164	378	-5	8
3804	KPKB	PKB	50307	PARKERSBURG WOOD CO AP	39.2	-81.27	253	-5	10
14894	KHLG	HLG	22707	WHEELING OHIO CO AP	40.17639	-80.6472	359	-5	10
24164	KBPI	BPI	61107	BIG PINEY MARBLETON AP	42.58444	-110.108	2124	-7	10
94054	KBYG	BYG	90606	BUFFALO JOHNSON CO AP	44.38139	-106.721	1506	-7	10
24089	KCPR	CPR	41107	CASPER NATRONA CO AP	42.8977	-106.474	1619	-7	10
24018	KCYS	CYS	92606	CHEYENNE MUNI AP	41.15	-104.817	1868	-7	10
4111	KEVW	EVW	102005	EVANSTON BURNS FLD	41.27306	-111.031	2175	-7	10
94023	KGCC	GCC	12407	GILLETTE CAMPBELL AP	44.33944	-105.542	1327	-7	8
24048	KGEY	GEY	71807	GREYBULL S BIG HORN AP	44.51694	-108.082	1194	-7	10
24021	KLND	LND	111705	LANDER HUNT FLD AP	42.8154	-108.726	1704	-7	10
24022	KLAR	LAR	92806	LARAMIE AP	41.3167	-105.683	2215	-7	10
24057	KRWL	RWL	90408	RAWLINS AP	41.8025	-107.206	2053	-7	10
24061	KRIW	RIW	110705	RIVERTON RGNL AP	43.06417	-108.459	1660	-7	10
24027	KRKS	RKS	52907	ROCK SPRINGS AP	41.5944	-109.053	2055	-7	10
24029	KSHR	SHR	83006	SHERIDAN AP	44.7694	-106.969	1202	-7	10
94053	KTOR	TOR	91306	TORRINGTON MUNI AP	42.0613	-104.158	1280	-7	10
24062	KWRL	WRL	53107	WORLAND	43.96583	-107.951	1272	-7	10

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
13871	AIRWAYS,ASOS,COOP	AL	N	N	A	AL	Y	AL13871	72230	-6
13876	AIRSAMPLE,AIRWAYS,ASOS,COOP	AL	N	N	A	AL	Y	AL13876	72230	-6
53852	AIRWAYS,ASOS	AL	N	N	A	AL	Y	AL53852	72230	-6
13839	AIRWAYS,ASOS,COOP	AL	N	N	A	AL	Y	AL13839	72214	-6
53820	AIRWAYS,ASOS	AL	N	N	A	AL	Y	AL53820	72230	-6
3856	ASOS,COOP	AL	N	N	A	AL	Y	AL03856	72230	-6
13838	AIRWAYS,ASOS	AL	N	N	A	AL	Y	AL13838	72233	-6
13894	ASOS,COOP,UPPERAIR	AL	N	N	A	AL	Y	AL13894	72233	-6
13895	AIRSAMPLE,ASOS,COOP	AL	N	N	A	AL	Y	AL13895	72230	-6
13896	AIRWAYS,ASOS,COOP,USHCN	AL	N	N	A	AL	Y	AL13896	72230	-6
3878	ASOS	AL	N	N	A	AL	Y	AL03878	72230	-6
93806	AIRWAYS,ASOS,COOP	AL	N	N	A	AL	Y	AL93806	72230	-6
63872	ASOS,MILITARY	AL, GA	N	N	A	AL	Y	AL63872	72215	-6
26451	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26451	70273	-9
26409	AIRWAYS,ASOS	AK	Y	N	A	AK	Y	AK26409	70273	-9
25308	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25308	70398	-9
27502	ASOS,COOP	AK	Y	N	A	AK	Y	AK27502	70026	-9
26615	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26615	70219	-9
26533	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26533	70261	-9
26415	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26415	70261	-9
25624	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25624	70316	-9
26410	ASOS,COOP	AK	Y	N	A	AK	Y	AK26410	70273	-9
27406	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK27406	70261	-9
26411	ASOS,COOP	AK	Y	N	A	AK	Y	AK26411	70261	-9
26425	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26425	70261	-9
25323	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25323	71964	-9
25507	ASOS,COOP	AK	Y	N	A	AK	Y	AK25507	70273	-9
25506	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25506	70326	-9
25309	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25309	71964	-9

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
26523	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26523	70273	-9
25325	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25325	70398	-9
25503	ASOS,COOP	AK	Y	N	A	AK	Y	AK25503	70326	-9
25501	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25501	70350	-9
26616	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26616	70133	-9
26510	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26510	70231	-9
26435	ASOS	AK	Y	N	A	AK	Y	AK26435	70261	-9
26617	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26617	70200	-9
26412	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26412	70361	-9
27515	AIRWAYS,ASOS	AK	Y	N	A	AK	Y	AK27515	70026	-9
25331	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25331	70273	-9
26492	ASOS,COOP,WXSVC	AK	Y	N	A	AK	N	AK26492	70273	-9
25516	AIRWAYS,ASOS	AK	Y	N	A	AK	Y	AK25516	70350	-9
26438	AIRWAYS,ASOS	AK	Y	N	A	AK	Y	AK26438	70273	-9
25333	ASOS,COOP	AK	Y	N	A	AK	Y	AK25333	71964	-9
25335	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25335	71964	-9
25628	AIRWAYS,ASOS	AK	Y	N	A	AK	Y	AK25628	70308	-9
25713	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK25713	70308	-9
26528	ASOS,COOP	AK	Y	N	A	AK	Y	AK26528	70273	-9
26529	AIRWAYS,ASOS,COOP	AK	Y	N	A	AK	Y	AK26529	70231	-9
27503	AIRWAYS,ASOS	AK	Y	N	A	AK	Y	AK27503	70026	-9
53869	AIRWAYS,ASOS	AR	N	N	A	AR	Y	AR53869	72340	-6
53925	ASOS,COOP	AR	N	N	A	AR	Y	AR53925	72248	-6
93992	AIRWAYS,ASOS,COOP	AR	N	N	A	AR	Y	AR93992	72248	-6
93993	ASOS,COOP	AR	N	N	A	AR	Y	AR93993	72440	-6
53922	AIRWAYS,ASOS	AR	N	N	A	AR	Y	AR53922	72440	-6
13964	AIRSAMPLE,ASOS,COOP	AR	N	N	A	AR	Y	AR13964	72440	-6
13971	ASOS,COOP	AR	N	N	A	AR	Y	AR13971	72440	-6
3962	ASOS,COOP	AR	N	N	A	AR	Y	AR03962	72340	-6

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
3953	AIRWAYS,ASOS	AR	N	N	A	AR	Y	AR03953	72340	-6
13963	AIRWAYS,ASOS,COOP	AR	N	N	A	AR	Y	AR13963	72340	-6
53919	ASOS,COOP	AR	N	N	A	AR	Y	AR53919	72340	-6
53921	AIRWAYS,ASOS,COOP	AR	N	N	A	AR	N	AR53921	72340	-6
53918	AIRWAYS,ASOS	AR	N	N	A	AR	Y	AR53918	72440	-6
93988	ASOS	AR	N	N	A	AR	Y	AR93988	72340	-6
53920	AIRWAYS,ASOS	AR	N	N	A	AR	Y	AR53920	72340	-6
13977	ASOS,COOP	TX, AR	N	N	A	AR	Y	AR13977	72248	-6
93026	ASOS,COOP	AZ	N	Y	A	AZ	Y	AZ93026	72274	-7
3103	ASOS,COOP	AZ	N	Y	A	AZ	Y	AZ03103	72376	-7
3124	ASOS	AZ	N	Y	A	AZ	Y	AZ03124	72274	-7
3195	AIRWAYS,ASOS	AZ	N	Y	A	AZ	Y	AZ03195	72376	-7
93167	ASOS,COOP	AZ	N	Y	A	AZ	Y	AZ93167	72388	-7
3196	AIRWAYS,ASOS	AZ	N	Y	A	AZ	Y	AZ03196	72274	-7
3162	ASOS	AZ	N	Y	A	AZ	Y	AZ03162	72376	-7
3184	AIRWAYS,ASOS	AZ	N	Y	A	AZ	Y	AZ03184	72376	-7
23183	ASOS,COOP	AZ	N	Y	A	AZ	Y	AZ23183	72274	-7
23184	AIRWAYS,ASOS	AZ	N	Y	A	AZ	Y	AZ23184	72376	-7
93084	AIRWAYS,ASOS	AZ	N	Y	A	AZ	Y	AZ93084	72274	-7
3192	AIRWAYS,ASOS	AZ	N	Y	A	AZ	Y	AZ03192	72376	-7
93027	AIRWAYS,ASOS	AZ	N	Y	A	AZ	Y	AZ93027	72376	-7
23160	ASOS,COOP	AZ	N	Y	A	AZ	Y	AZ23160	72274	-7
3029	ASOS	AZ, NM	N	Y	A	AZ	Y	AZ03029	72365	-7
23194	ASOS,COOP	AZ	N	Y	A	AZ	Y	AZ23194	72376	-7
94299	ASOS	CA	N	Y	A	CA	Y	CA94299	72489	-8
24283	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA24283	72597	-8
23191	ASOS	CA	N	N	A	CA	Y	CA23191	72293	-8
23155	ASOS,COOP	CA	N	N	A	CA	Y	CA23155	72293	-8
23161	ASOS,COOP	CA	N	N	A	CA	Y	CA23161	72388	-8

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
23157	ASOS,COOP	CA	N	N	A	CA	Y	CA23157	72489	-8
23225	ASOS,COOP	CA	N	Y	A	CA	Y	CA23225	72489	-8
23158	ASOS,COOP	CA	N	Y	A	CA	Y	CA23158	72293	-8
23152	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA23152	72293	-8
23136	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA23136	72293	-8
3177	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA03177	72293	-8
3179	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA03179	72293	-8
23254	AIRWAYS,ASOS,COOP	CA	N	N	A	CA	Y	CA23254	72493	-8
24286	ASOS	CA	N	N	A	CA	Y	CA24286	72597	-8
3104	ASOS	CA	N	N	A	CA	Y	CA03104	72293	-8
23199	ASOS	CA	N	Y	A	CA	N	CA23199	72293	-8
93193	ASOS,COOP,USHCN	CA	N	N	A	CA	Y	CA93193	72493	-8
3166	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA03166	72293	-8
53119	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA53119	72493	-8
3167	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA03167	72293	-8
93228	AIRWAYS,ASOS,COOP	CA	N	N	A	CA	Y	CA93228	72493	-8
93115	ASOS	CA	N	N	A	CA	Y	CA93115	72293	-8
3144	ASOS	CA	N	Y	A	CA	Y	CA03144	72293	-8
3159	ASOS,COOP	CA	N	N	A	CA	Y	CA03159	72293	-8
23285	AIRWAYS,ASOS,COOP	CA	N	N	A	CA	Y	CA23285	72493	-8
23129	ASOS,COOP	CA	N	N	A	CA	Y	CA23129	72293	-8
23174	ASOS,COOP	CA	N	N	A	CA	Y	CA23174	72293	-8
93242	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA93242	72493	-8
93205	ASOS,COOP	CA	N	N	A	CA	Y	CA93205	72493	-8
23257	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA23257	72493	-8
23258	AIRWAYS,ASOS,COOP	CA	N	N	A	CA	Y	CA23258	72493	-8
23244	ASOS	CA	N	N	A	CA	Y	CA23244	72493	-8
24259	ASOS	CA	N	N	A	CA	Y	CA24259	72597	-8
23259	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA23259	72493	-8

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
24215	ASOS	CA	N	N	A	CA	N	CA24215	72597	-8
93227	AIRWAYS,ASOS,COOP	CA	N	N	A	CA	Y	CA93227	72493	-8
23179	ASOS,COOP,USHCN,WXSVC	CA	N	Y	A	CA	Y	CA23179	72388	-8
23230	ASOS,UPPERAIR	CA	N	N	A	CA	Y	CA23230	72493	-8
53121	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA53121	72293	-8
3102	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA03102	72293	-8
93210	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA93210	72489	-8
93110	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA93110	72293	-8
93138	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA93138	72293	-8
23182	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA23182	72293	-8
93209	ASOS,COOP	CA	N	N	A	CA	Y	CA93209	72493	-8
24216	ASOS,COOP	CA	N	N	A	CA	Y	CA24216	72597	-8
24257	ASOS,COOP,USHCN	CA	N	N	A	CA	Y	CA24257	72597	-8
3171	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA03171	72293	-8
23232	AIRWAYS,ASOS,COOP	CA	N	N	A	CA	Y	CA23232	72493	-8
23233	ASOS,COOP	CA	N	N	A	CA	Y	CA23233	72493	-8
23188	ASOS,COOP	CA	N	N	A	CA	Y	CA23188	72293	-8
3131	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA03131	72293	-8
23234	ASOS,COOP	CA	N	N	A	CA	Y	CA23234	72493	-8
23293	ASOS,COOP	CA	N	N	A	CA	Y	CA23293	72493	-8
93206	ASOS	CA	N	N	A	CA	Y	CA93206	72493	-8
23187	ASOS,COOP	CA	N	N	A	CA	N	CA23187	72293	-8
93184	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA93184	72293	-8
23190	AIRWAYS,ASOS,COOP	CA	N	N	A	CA	Y	CA23190	72293	-8
23273	ASOS,COOP	CA	N	N	A	CA	Y	CA23273	72493	-8
93197	ASOS	CA	N	N	A	CA	Y	CA93197	72293	-8
23213	AIRWAYS,ASOS,COOP	CA	N	N	A	CA	Y	CA23213	72493	-8
93230	ASOS,COOP	CA	N	Y	A	CA	Y	CA93230	72489	-8
23237	ASOS,COOP	CA	N	N	A	CA	Y	CA23237	72493	-8

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
23275	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA23275	72493	-8
93241	ASOS,COOP	CA	N	N	A	CA	Y	CA93241	72493	-8
23130	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA23130	72293	-8
23277	AIRWAYS,ASOS	CA	N	N	A	CA	Y	CA23277	72493	-8
24015	AIRWAYS,ASOS	CO	Y	N	A	CO	Y	CO24015	72469	-7
23061	ASOS,COOP,WXSVC	CO	N	Y	A	CO	Y	CO23061	72365	-7
93073	AIRWAYS,ASOS,COOP	CO	N	Y	A	CO	Y	CO93073	72476	-7
3026	AIRWAYS,ASOS	CO	N	Y	A	CO	Y	CO03026	72562	-7
93037	ASOS,COOP	CO	N	N	A	CO	Y	CO93037	72469	-7
93069	AIRWAYS,ASOS,COOP	CO	N	Y	A	CO	Y	CO93069	72476	-7
24046	AIRWAYS,ASOS,COOP	CO	N	Y	A	CO	Y	CO24046	72476	-7
93067	AIRWAYS,ASOS	CO	N	N	A	CO	Y	CO93067	72469	-7
3017	ASOS,COOP	CO	N	N	A	CO	Y	CO03017	72469	-7
93005	AIRWAYS,ASOS,COOP	CO	N	Y	A	CO	Y	CO93005	72476	-7
23066	ASOS,COOP	CO	N	Y	A	CO	Y	CO23066	72476	-7
23067	AIRWAYS,ASOS,COOP	CO	N	N	A	CO	Y	CO23067	72469	-7
3013	AIRWAYS,ASOS	CO	N	Y	A	CO	Y	CO03013	72451	-7
93009	ASOS	CO	Y	N	A	CO	Y	CO93009	72469	-7
93010	ASOS	CO	N	N	A	CO	N	CO93010	72469	-7
94050	AIRWAYS,ASOS,COOP	CO	N	Y	A	CO	Y	CO94050	72476	-7
93013	AIRWAYS,ASOS,COOP	CO	N	Y	A	CO	Y	CO93013	72476	-7
93058	ASOS,COOP,WXSVC	CO	N	N	A	CO	Y	CO93058	72469	-7
23070	AIRWAYS,ASOS,COOP	CO	N	N	A	CO	Y	CO23070	72469	-7
94702	ASOS,COOP	CT	Y	N	A	CT	Y	CT94702	72501	-5
54734	AIRWAYS,ASOS	CT	Y	N	A	CT	Y	CT54734	72501	-5
14707	ASOS	CT	Y	N	A	CT	Y	CT14707	72501	-5
14740	ASOS,COOP	CT	Y	N	A	CT	Y	CT14740	72501	-5
14752	ASOS	CT	Y	N	A	CT	Y	CT14752	72501	-5
54788	AIRWAYS,ASOS	CT	Y	N	A	CT	Y	CT54788	72501	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
14758	AIRWAYS,ASOS	CT	Y	N	A	CT	Y	CT14758	72501	-5
54767	AIRWAYS,ASOS	CT	Y	N	A	CT	Y	CT54767	72501	-5
13764	AIRWAYS,ASOS	DE	N	N	A	DE	Y	DE13764	72402	-5
13781	ASOS,COOP	DE	Y	N	A	DE	Y	DE13781	72403	-5
12832	ASOS	FL	N	N	A	FL	Y	FL12832	72214	-5
12818	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12818	72210	-5
13884	AIRWAYS,ASOS,COOP	FL	N	N	A	FL	Y	FL13884	72214	-6
12834	ASOS,COOP	FL	N	N	A	FL	Y	FL12834	72206	-5
53853	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL53853	72214	-6
12885	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12885	72202	-5
12849	AIRWAYS,ASOS,COOP	FL	N	N	A	FL	Y	FL12849	72202	-5
12835	AIRWAYS,ASOS,COOP,USHCN	FL	N	N	A	FL	Y	FL12835	72210	-5
12894	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12894	72210	-5
12895	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12895	72210	-5
12816	AIRWAYS,ASOS,COOP	FL	N	N	A	FL	Y	FL12816	72206	-5
92809	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL92809	72202	-5
53860	AIRWAYS,ASOS,COOP	FL	N	N	A	FL	Y	FL53860	72206	-5
13889	ASOS,COOP	FL	N	N	A	FL	Y	FL13889	72206	-5
12836	AIRSAMPLE,AIRWAYS,ASOS,COOP, USHCN	FL	N	N	A	FL	Y	FL12836	72201	-5
12819	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12819	72210	-5
12896	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12896	72201	-5
3818	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL03818	72214	-6
12838	ASOS,COOP	FL	N	N	A	FL	Y	FL12838	72210	-5
12839	ASOS,COOP	FL	N	N	A	FL	Y	FL12839	72202	-5
12888	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12888	72202	-5
12882	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12882	72202	-5
12897	AIRWAYS,ASOS,COOP	FL	N	N	A	FL	Y	FL12897	72202	-5
73805	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL73805	72214	-6

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
12841	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12841	72210	-5
12815	ASOS,COOP	FL	N	N	A	FL	Y	FL12815	72210	-5
12854	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12854	72210	-5
13899	AIRWAYS,ASOS,COOP,USHCN	FL	N	N	A	FL	Y	FL13899	72233	-6
12812	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12812	72210	-5
12871	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12871	72210	-5
92806	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL92806	72210	-5
12873	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12873	72210	-5
93805	ASOS,COOP,USHCN	FL	N	N	A	FL	Y	FL93805	72214	-5
12842	ASOS,COOP	FL	N	N	A	FL	Y	FL12842	72210	-5
12843	ASOS,COOP	FL	N	N	A	FL	Y	FL12843	72210	-5
12844	ASOS,COOP,UPPERAIR	FL	N	N	A	FL	Y	FL12844	72202	-5
53847	ASOS,MILITARY	FL	N	N	A	FL	Y	FL53847	72214	-6
12876	AIRWAYS,ASOS	FL	N	N	A	FL	Y	FL12876	72210	-5
13869	ASOS	GA	N	N	A	GA	Y	GA13869	72214	-5
13870	AIRWAYS,ASOS,COOP	GA	N	N	A	GA	Y	GA13870	72206	-5
13873	AIRSAMPLE,AIRWAYS,ASOS,COOP	GA	N	N	A	GA	Y	GA13873	72215	-5
3888	AIRWAYS,ASOS	GA	N	N	A	GA	Y	GA03888	72215	-5
13874	ASOS,COOP	GA	N	N	A	GA	Y	GA13874	72215	-5
53863	AIRWAYS,ASOS	GA	N	N	A	GA	Y	GA53863	72215	-5
3820	ASOS,COOP	GA, SC	N	N	A	GA	Y	GA03820	72208	-5
13837	ASOS	GA	N	N	A	GA	Y	GA13837	72208	-5
13878	ASOS,COOP	GA	N	N	A	GA	Y	GA13878	72206	-5
53873	ASOS	GA	N	N	A	GA	Y	GA53873	72215	-5
93842	AIRSAMPLE,AIRWAYS,ASOS,COOP	AL, GA	N	N	A	GA	Y	GA93842	72215	-5
53838	AIRWAYS,ASOS	GA	N	N	A	GA	Y	GA53838	72215	-5
3813	ASOS,COOP	GA	N	N	A	GA	Y	GA03813	72215	-5
53819	AIRWAYS,ASOS,UPPERAIR	GA	N	N	A	GA	Y	GA53819	72215	-5
93801	AIRWAYS,ASOS	GA	N	N	A	GA	Y	GA93801	72215	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
3822	ASOS,COOP,USHCN	GA	N	N	A	GA	Y	GA03822	72208	-5
93845	ASOS	GA	N	N	A	GA	Y	GA93845	72214	-5
22551	AIRWAYS,ASOS	HI	N	N	A	HI	Y	HI22551	91165	-10
21504	ASOS,COOP	HI	N	N	A	HI	Y	HI21504	91285	-10
22521	ASOS,COOP	HI	N	N	A	HI	Y	HI22521	91165	-10
22516	AIRWAYS,ASOS,COOP	HI	N	N	A	HI	Y	HI22516	91285	-10
21510	AIRWAYS,ASOS	HI	N	N	A	HI	Y	HI21510	91285	-10
22536	ASOS,COOP	HI	N	N	A	HI	Y	HI22536	91165	-10
22534	ASOS,COOP	HI	N	N	A	HI	Y	HI22534	91165	-10
94989	AIRWAYS,ASOS,COOP	IA	Y	N	A	IA	Y	IA94989	72558	-6
14931	ASOS	IA, IL	Y	N	A	IA	Y	IA14931	74455	-6
14990	AIRWAYS,ASOS,COOP	IA	Y	N	A	IA	Y	IA14990	74455	-6
94982	AIRWAYS,ASOS	IA	Y	N	A	IA	Y	IA94982	74455	-6
14933	ASOS,COOP	IA	Y	N	A	IA	Y	IA14933	72558	-6
94908	ASOS,COOP	IA	Y	N	A	IA	Y	IA94908	74455	-6
94971	AIRWAYS,ASOS,COOP	IA	Y	N	A	IA	Y	IA94971	72649	-6
14937	AIRWAYS,ASOS	IA	Y	N	A	IA	Y	IA14937	74455	-6
94991	AIRWAYS,ASOS,COOP	IA	Y	N	A	IA	Y	IA94991	72456	-6
94988	AIRWAYS,ASOS,COOP	IA	Y	N	A	IA	Y	IA94988	74455	-6
14940	ASOS,COOP	IA	Y	N	A	IA	Y	IA14940	72649	-6
14950	ASOS,COOP	IA	Y	N	A	IA	Y	IA14950	74455	-6
14943	AIRWAYS,ASOS,COOP	IA, NE	Y	N	A	IA	Y	IA14943	72558	-6
14972	AIRWAYS,ASOS	IA	Y	N	A	IA	Y	IA14972	72558	-6
94910	AIRWAYS,ASOS,COOP,WXSVC	IA	Y	N	A	IA	Y	IA94910	74455	-6
24131	ASOS,COOP	ID	N	N	A	ID	Y	ID24131	72681	-7
24133	ASOS,COOP	ID	N	Y	A	ID	Y	ID24133	72572	-7
4114	ASOS	ID	Y	N	A	ID	Y	ID04114	72681	-7
24145	AIRWAYS,ASOS	ID	N	Y	A	ID	Y	ID24145	72572	-7
4110	AIRWAYS,ASOS	ID	N	Y	A	ID	Y	ID04110	72681	-7

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
24149	AIRSAMPLE,ASOS,COOP,USHCN	WA, ID	N	N	A	ID	Y	ID24149	72786	-8
94182	AIRWAYS,ASOS	ID	Y	N	A	ID	Y	ID94182	72681	-7
24154	AIRWAYS,ASOS	MT, ID	Y	N	A	ID	N	ID24154	72786	-8
24156	ASOS,COOP	ID	N	Y	A	ID	Y	ID24156	72572	-7
94194	AIRWAYS,ASOS	ID	N	Y	A	ID	Y	ID94194	72572	-7
94178	AIRWAYS,ASOS	ID	N	Y	A	ID	Y	ID94178	72681	-7
94870	AIRWAYS,ASOS	IL	Y	N	A	IL	Y	IL94870	74560	-6
4808	AIRWAYS,ASOS	IL	Y	N	A	IL	Y	IL04808	74560	-6
14819	AIRWAYS,ASOS,WXSVC	IL	Y	N	A	IL	Y	IL14819	74560	-6
94846	ASOS,COOP,WXSVC	IL	Y	N	A	IL	Y	IL94846	74560	-6
4838	AIRWAYS,ASOS	IL	Y	N	A	IL	Y	IL04838	72645	-6
14880	ASOS	IL	Y	N	A	IL	Y	IL14880	72645	-6
3887	AIRWAYS,ASOS	IL	Y	N	A	IL	Y	IL03887	74560	-6
13809	AIRWAYS,ASOS	IL	Y	N	A	IL	Y	IL13809	74560	-6
53802	AIRWAYS,ASOS	IL	Y	N	A	IL	Y	IL53802	74560	-6
14923	ASOS,COOP,WXSVC	IL	Y	N	A	IL	Y	IL14923	74455	-6
14842	AIRSAMPLE,ASOS,COOP,WXSVC	IL	Y	N	A	IL	Y	IL14842	74560	-6
93989	ASOS,COOP	IL	N	N	A	IL	Y	IL93989	74455	-6
94822	ASOS,COOP,WXSVC	IL	Y	N	A	IL	Y	IL94822	74455	-6
93822	AIRSAMPLE,AIRWAYS,ASOS,COOP	IL	Y	N	A	IL	Y	IL93822	74560	-6
94892	AIRWAYS,ASOS	IL	Y	N	A	IL	Y	IL94892	74560	-6
3893	AIRWAYS,ASOS	IN	Y	N	A	IN	Y	IN03893	72426	-5
93817	AIRSAMPLE,AIRWAYS,ASOS,COOP	IN	N	N	A	IN	Y	IN93817	72327	-6
14827	AIRWAYS,ASOS,COOP	IN	Y	N	A	IN	Y	IN14827	72426	-5
14829	AIRWAYS,ASOS,COOP	IN	Y	N	A	IN	Y	IN14829	72632	-5
53842	AIRWAYS,ASOS	IN	Y	N	A	IN	Y	IN53842	72426	-5
93819	ASOS,COOP,WXSVC	IN	Y	N	A	IN	Y	IN93819	72426	-5
14835	AIRWAYS,ASOS,WXSVC	IN	Y	N	A	IN	Y	IN14835	74560	-5
94895	AIRWAYS,ASOS	IN	Y	N	A	IN	Y	IN94895	72426	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
14848	AIRWAYS,ASOS,COOP	IN	Y	N	A	IN	Y	IN14848	72632	-5
53866	AIRWAYS,ASOS	IN	Y	N	A	IN	Y	IN53866	72426	-5
3868	AIRWAYS,ASOS	IN	Y	N	A	IN	Y	IN03868	74560	-5
4846	AIRWAYS,ASOS	IN	Y	N	A	IN	Y	IN04846	74560	-6
13981	ASOS	KS	Y	N	A	KS	Y	KS13981	72456	-6
13984	ASOS,COOP	KS	Y	N	A	KS	Y	KS13984	72456	-6
13985	ASOS,COOP,WXSVC	KS	N	N	A	KS	Y	KS13985	72451	-6
13989	AIRWAYS,ASOS,BASIC,COOP, WXSVC	KS	N	N	A	KS	Y	KS13989	72456	-6
23064	AIRWAYS,ASOS	KS	N	N	A	KS	Y	KS23064	72451	-6
23065	ASOS,COOP,WXSVC	KS	N	Y	A	KS	Y	KS23065	72562	-7
93990	AIRWAYS,ASOS	KS	Y	N	A	KS	Y	KS93990	72451	-6
13986	ASOS	KS	N	N	A	KS	Y	KS13986	74646	-6
3997	AIRWAYS,ASOS,COOP	KS	Y	N	A	KS	Y	KS03997	72456	-6
3936	AIRWAYS,ASOS,COOP,WXSVC	KS	Y	N	A	KS	Y	KS03936	72456	-6
93909	AIRWAYS,ASOS	KS	N	N	A	KS	Y	KS93909	72456	-6
3967	AIRWAYS,ASOS	KS	N	N	A	KS	Y	KS03967	72456	-6
3998	AIRWAYS,ASOS	KS	Y	N	A	KS	Y	KS03998	72456	-6
93997	ASOS	KS	N	N	A	KS	Y	KS93997	72451	-6
13920	AIRWAYS,ASOS	KS	N	N	A	KS	Y	KS13920	72456	-6
13996	ASOS,COOP,WXSVC	KS	N	N	A	KS	Y	KS13996	72456	-6
3974	ASOS	KS	N	N	A	KS	Y	KS03974	74646	-6
3928	ASOS,COOP,WXSVC	KS	N	N	A	KS	Y	KS03928	74646	-6
13932	AIRWAYS,ASOS	KS	N	N	A	KS	Y	KS13932	74646	-6
93808	AIRWAYS,ASOS,COOP,USHCN	KY	N	N	A	KY	Y	KY93808	72327	-6
93814	ASOS,COOP	OH, KY	Y	N	A	KY	Y	KY93814	72426	-5
53841	AIRWAYS,ASOS	KY	Y	N	A	KY	Y	KY53841	72426	-5
3889	ASOS,COOP	KY	N	N	A	KY	Y	KY03889	72426	-5
93820	ASOS,COOP,WXSVC	KY	Y	N	A	KY	Y	KY93820	72426	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
3849	AIRWAYS,ASOS,COOP	KY	N	N	A	KY	Y	KY03849	72426	-5
13810	AIRWAYS,ASOS	KY	Y	N	A	KY	Y	KY13810	72327	-5
93821	ASOS,COOP,WXSVC	KY	Y	N	A	KY	Y	KY93821	72327	-5
3816	AIRSAMPLE,ASOS,COOP,WXSVC	KY	Y	N	A	KY	Y	KY03816	72327	-6
93915	AIRWAYS,ASOS	LA	N	N	A	LA	Y	LA93915	72240	-6
13970	ASOS,COOP,USHCN	LA	N	N	A	LA	Y	LA13970	72233	-6
12884	ASOS,COOP	LA	N	N	A	LA	N	LA12884	72233	-6
13976	AIRWAYS,ASOS,COOP,USHCN	LA	N	N	A	LA	Y	LA13976	72240	-6
3937	ASOS,COOP,UPPERAIR	LA	N	N	A	LA	Y	LA03937	72240	-6
13942	ASOS,COOP	LA	N	N	A	LA	Y	LA13942	72248	-6
53915	AIRWAYS,ASOS,COOP	LA	N	N	A	LA	Y	LA53915	72240	-6
12916	ASOS,COOP	LA	N	N	A	LA	Y	LA12916	72233	-6
53917	AIRWAYS,ASOS,COOP	LA	N	N	A	LA	Y	LA53917	72233	-6
53905	AIRWAYS,ASOS,COOP	LA	N	N	A	LA	Y	LA53905	72248	-6
13957	AIRSAMPLE,ASOS,COOP	LA	N	N	A	LA	Y	LA13957	72248	-6
53865	ASOS,COOP	LA	N	N	A	LA	Y	LA53865	72233	-6
3996	AIRWAYS,ASOS,COOP	MS, LA	N	N	A	LA	Y	LA03996	72235	-6
14702	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA14702	74494	-5
54733	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA54733	74494	-5
14739	ASOS,COOP	MA	Y	N	A	MA	Y	MA14739	74494	-5
94624	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA94624	74494	-5
4780	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA04780	72518	-5
94720	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA94720	74494	-5
94723	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA94723	74389	-5
14756	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA14756	74494	-5
94726	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA94726	74494	-5
54768	AIRWAYS,ASOS,COOP	VT, MA	Y	N	A	MA	Y	MA54768	72518	-5
54704	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA54704	74494	-5
54756	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA54756	72518	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
14763	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA14763	72518	-5
54769	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA54769	74494	-5
54777	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA54777	74494	-5
94724	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA94724	74494	-5
14775	AIRWAYS,ASOS	MA	Y	N	A	MA	Y	MA14775	72518	-5
94746	ASOS,COOP	MA	Y	N	A	MA	Y	MA94746	72518	-5
93721	ASOS,COOP	MD	N	N	A	MD	Y	MD93721	72403	-5
93706	AIRWAYS,ASOS	PA, MD	N	N	A	MD	Y	MD93706	72403	-5
93786	AIRWAYS,ASOS	MD	N	N	A	MD	Y	MD93786	72402	-5
93720	AIRWAYS,ASOS,COOP	MD	N	N	A	MD	Y	MD93720	72402	-5
14605	ASOS,COOP	ME	Y	N	A	ME	Y	ME14605	74389	-5
14606	AIRWAYS,ASOS,COOP	ME	Y	N	A	ME	Y	ME14606	74389	-5
14607	ASOS,COOP	ME	Y	N	A	ME	Y	ME14607	72712	-5
4836	AIRWAYS,ASOS	ME	Y	N	A	ME	Y	ME04836	72712	-5
54772	AIRWAYS,ASOS	ME, NH	Y	N	A	ME	Y	ME54772	74389	-5
14610	AIRWAYS,ASOS	ME	Y	N	A	ME	Y	ME14610	72712	-5
14764	ASOS,COOP,USHCN	ME	Y	N	A	ME	Y	ME14764	74389	-5
94623	AIRWAYS,ASOS	ME	Y	N	A	ME	Y	ME94623	74389	-5
4847	AIRWAYS,ASOS	MI	Y	N	A	MI	Y	MI04847	72632	-5
94849	ASOS,COOP	MI	Y	N	A	MI	Y	MI94849	72634	-5
94889	AIRWAYS,ASOS	MI	Y	N	A	MI	Y	MI94889	72632	-5
14815	AIRWAYS,ASOS	MI	Y	N	A	MI	Y	MI14815	72632	-5
94871	ASOS,COOP	MI	Y	N	A	MI	Y	MI94871	72645	-5
14822	ASOS	MI	Y	N	A	MI	Y	MI14822	72632	-5
94847	ASOS,COOP,WXSVC	MI	Y	N	A	MI	Y	MI94847	72632	-5
14853	AIRWAYS,ASOS	MI	Y	N	A	MI	Y	MI14853	72632	-5
14826	ASOS,COOP,WXSVC	MI	Y	N	A	MI	Y	MI14826	72632	-5
4854	AIRWAYS,ASOS	MI	Y	N	A	MI	Y	MI04854	72634	-5
94860	ASOS,COOP,WXSVC	MI	Y	N	A	MI	Y	MI94860	72632	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
14858	ASOS	MI	Y	N	A	MI	Y	MI14858	72645	-5
4839	AIRWAYS,ASOS	MI	Y	N	A	MI	Y	MI04839	72634	-5
94814	ASOS,COOP,WXSVC	MI	Y	N	A	MI	Y	MI94814	72634	-5
94893	AIRWAYS,ASOS	WI, MI	Y	N	A	MI	Y	MI94893	72645	-6
14833	ASOS,COOP	MI	Y	N	A	MI	Y	MI14833	72632	-5
94815	AIRWAYS,ASOS,WXSVC	MI	Y	N	A	MI	Y	MI94815	72632	-5
14836	ASOS,COOP,WXSVC	MI	Y	N	A	MI	Y	MI14836	72632	-5
14840	ASOS,COOP,WXSVC	MI	Y	N	A	MI	Y	MI14840	72634	-5
14841	AIRWAYS,ASOS,COOP	MI	Y	N	A	MI	Y	MI14841	72634	-5
94817	AIRWAYS,ASOS	MI	Y	N	A	MI	Y	MI94817	72632	-5
14845	ASOS	MI	Y	N	A	MI	Y	MI14845	72632	-5
14850	AIRWAYS,ASOS,COOP	MI	Y	N	A	MI	Y	MI14850	72634	-5
14910	AIRWAYS,ASOS,COOP	MN	Y	N	A	MN	Y	MN14910	72649	-6
94938	AIRWAYS,ASOS	MN	Y	N	A	MN	Y	MN94938	72649	-6
14913	AIRWAYS,ASOS,COOP,WXSVC	MN	Y	N	A	MN	Y	MN14913	72747	-6
94931	ASOS	MN	Y	N	A	MN	Y	MN94931	72747	-6
14918	ASOS,COOP,WXSVC	MN	Y	N	A	MN	Y	MN14918	72747	-6
14922	ASOS,COOP,USHCN,WXSVC	MN	Y	N	A	MN	Y	MN14922	72649	-6
94960	AIRWAYS,ASOS,COOP	MN	Y	N	A	MN	Y	MN94960	72649	-6
94963	ASOS,COOP	MN	Y	N	A	MN	Y	MN94963	72649	-6
94967	AIRWAYS,ASOS	MN	Y	N	A	MN	Y	MN94967	72747	-6
14992	ASOS,COOP	MN	Y	N	A	MN	Y	MN14992	72649	-6
14925	ASOS,COOP,WXSVC	MN	Y	N	A	MN	Y	MN14925	72649	-6
14926	AIRWAYS,ASOS,COOP,WXSVC	MN	Y	N	A	MN	Y	MN14926	72649	-6
14927	ASOS,COOP	MN	Y	N	A	MN	Y	MN14927	72649	-6
3935	AIRWAYS,ASOS,COOP,WXSVC	MO	N	N	A	MO	Y	MO03935	74560	-6
3945	AIRWAYS,ASOS,COOP	MO	Y	N	A	MO	Y	MO03945	72440	-6
3963	AIRWAYS,ASOS	MO	Y	N	A	MO	Y	MO03963	72440	-6
13987	ASOS,COOP	MO	N	N	A	MO	Y	MO13987	72440	-6

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
13988	AIRWAYS,ASOS,COOP	KS, MO	Y	N	A	MO	Y	MO13988	72456	-6
3947	AIRWAYS,ASOS,COOP	MO	Y	N	A	MO	Y	MO03947	72456	-6
14938	ASOS	MO	Y	N	A	MO	Y	MO14938	74455	-6
53879	AIRWAYS,ASOS	MO	Y	N	A	MO	Y	MO53879	72456	-6
3975	AIRWAYS,ASOS	MO	N	N	A	MO	Y	MO03975	72340	-6
3994	AIRWAYS,ASOS	MO	N	N	A	MO	Y	MO03994	72440	-6
13995	ASOS,COOP,WXSVC	MO	N	N	A	MO	Y	MO13995	72440	-6
53904	AIRWAYS,ASOS	IL, MO	N	N	A	MO	Y	MO53904	74560	-6
13993	AIRWAYS,ASOS,COOP	KS, MO	Y	N	A	MO	Y	MO13993	72456	-6
13994	ASOS,COOP,WXSVC	MO	N	N	A	MO	Y	MO13994	74560	-6
3966	AIRWAYS,ASOS	MO	N	N	A	MO	Y	MO03966	74560	-6
13997	ASOS,COOP	MO	N	N	A	MO	Y	MO13997	72440	-6
53901	AIRWAYS,ASOS	MO	N	N	A	MO	Y	MO53901	72440	-6
13939	ASOS,COOP	MS	N	N	A	MS	Y	MS13939	72235	-6
13978	AIRWAYS,ASOS,COOP	MS	N	N	A	MS	Y	MS13978	72235	-6
93874	AIRWAYS,ASOS,COOP	MS	N	N	A	MS	Y	MS93874	72233	-6
13833	ASOS,COOP	MS	N	N	A	MS	Y	MS13833	72233	-6
13927	ASOS,COOP	MS	N	N	A	MS	Y	MS13927	72235	-6
3940	ASOS,COOP,UPPERAIR	MS	N	N	A	MS	Y	MS03940	72235	-6
93919	AIRWAYS,ASOS,COOP	MS	N	N	A	MS	Y	MS93919	72233	-6
13865	ASOS,COOP	MS	N	N	A	MS	Y	MS13865	72235	-6
53858	AIRWAYS,ASOS	MS	N	N	A	MS	Y	MS53858	72233	-6
93862	AIRSAMPLE,ASOS,COOP	MS	N	N	A	MS	Y	MS93862	72230	-6
94055	ASOS,COOP	MT	Y	N	A	MT	Y	MT94055	72662	-7
24033	ASOS,COOP	MT	N	Y	A	MT	Y	MT24033	72672	-7
24132	AIRWAYS,ASOS,COOP	MT	N	Y	A	MT	Y	MT24132	72776	-7
24135	ASOS,COOP	MT	N	Y	A	MT	Y	MT24135	72776	-7
24137	ASOS,COOP,USHCN	MT	Y	N	A	MT	Y	MT24137	72776	-7
24138	AIRWAYS,ASOS,COOP	MT	N	Y	A	MT	Y	MT24138	72776	-7

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
94008	ASOS,COOP,USHCN	MT	N	Y	A	MT	Y	MT94008	72768	-7
24143	ASOS,COOP,USHCN	MT	Y	N	A	MT	Y	MT24143	72776	-7
94012	ASOS,COOP	MT	Y	N	A	MT	Y	MT94012	72776	-7
24144	ASOS,COOP,USHCN	MT	N	Y	A	MT	Y	MT24144	72776	-7
24146	ASOS,COOP,USHCN	MT	Y	N	A	MT	Y	MT24146	72776	-7
24036	AIRWAYS,ASOS,COOP	MT	N	Y	A	MT	Y	MT24036	72776	-7
24150	ASOS,COOP	MT	N	Y	A	MT	Y	MT24150	72776	-7
24037	ASOS,COOP,USHCN	MT	N	Y	A	MT	Y	MT24037	72768	-7
94017	AIRWAYS,ASOS	MT	N	Y	A	MT	Y	MT94017	72768	-7
3812	ASOS,COOP,WXSVC	NC	N	N	A	NC	Y	NC03812	72317	-5
93765	ASOS	NC	N	N	A	NC	Y	NC93765	72305	-5
93783	AIRWAYS,ASOS	NC	N	N	A	NC	Y	NC93783	72317	-5
93729	ASOS,COOP,USHCN	NC	N	N	A	NC	Y	NC93729	72305	-5
93785	AIRWAYS,ASOS	NC	N	N	A	NC	Y	NC93785	72317	-5
13881	ASOS,COOP	NC	N	N	A	NC	Y	NC13881	72317	-5
13754	ASOS	NC	N	N	A	NC	Y	NC13754	72305	-5
13786	AIRWAYS,ASOS	NC	N	N	A	NC	Y	NC13786	72305	-5
93740	AIRWAYS,ASOS	NC	N	N	A	NC	Y	NC93740	72317	-5
53870	AIRWAYS,ASOS	NC, SC	N	N	A	NC	Y	NC53870	72317	-5
3810	AIRWAYS,ASOS,COOP	NC	N	N	A	NC	Y	NC03810	72317	-5
93782	AIRWAYS,ASOS	NC	N	N	A	NC	Y	NC93782	72317	-5
13776	AIRWAYS,ASOS,COOP	NC	N	N	A	NC	Y	NC13776	72317	-5
53872	AIRWAYS,ASOS	NC	N	N	A	NC	Y	NC53872	72317	-5
93719	AIRWAYS,ASOS,COOP	NC	N	N	A	NC	Y	NC93719	72305	-5
13723	ASOS,COOP	NC	N	N	A	NC	Y	NC13723	72317	-5
13722	ASOS,COOP	NC	N	N	A	NC	Y	NC13722	72317	-5
93759	ASOS	NC	N	N	A	NC	Y	NC93759	72305	-5
13748	ASOS,COOP	NC	N	N	A	NC	Y	NC13748	72305	-5
93807	AIRWAYS,ASOS	NC	N	N	A	NC	Y	NC93807	72317	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
24011	ASOS,COOP,WXSVC	ND	Y	N	A	ND	Y	ND24011	72764	-6
14914	AIRWAYS,ASOS,COOP	ND, MN	Y	N	A	ND	Y	ND14914	72659	-6
14916	AIRWAYS,ASOS,COOP,WXSVC	ND	Y	N	A	ND	Y	ND14916	72659	-6
94038	AIRWAYS,ASOS	ND	Y	N	A	ND	Y	ND94038	72662	-7
14919	ASOS,COOP,WXSVC	ND	Y	N	A	ND	Y	ND14919	72659	-6
24013	AIRWAYS,ASOS,COOP	ND	Y	N	A	ND	Y	ND24013	72764	-6
24012	ASOS,COOP,WXSVC	ND	Y	N	A	ND	Y	ND24012	72764	-7
94014	ASOS,COOP,WXSVC	ND	Y	N	A	ND	Y	ND94014	72768	-6
24044	AIRWAYS,ASOS,COOP	NE	Y	N	A	NE	Y	NE24044	72662	-7
94946	AIRWAYS,ASOS	NE	N	N	A	NE	Y	NE94946	72562	-6
24017	ASOS,COOP	NE	N	Y	A	NE	Y	NE24017	72662	-7
94957	ASOS	NE	Y	N	A	NE	Y	NE94957	72456	-6
14935	AIRWAYS,ASOS,COOP,WXSVC	NE	Y	N	A	NE	Y	NE14935	72558	-6
94949	AIRWAYS,ASOS,COOP	NE	N	N	A	NE	Y	NE94949	72558	-6
24091	ASOS	NE	Y	N	A	NE	Y	NE24091	72562	-7
14939	AIRWAYS,ASOS,COOP	NE	N	N	A	NE	Y	NE14939	72558	-6
94040	AIRWAYS,ASOS	NE	Y	N	A	NE	Y	NE94040	72562	-6
14941	ASOS,COOP,WXSVC	NE	N	Y	A	NE	Y	NE14941	72558	-6
24023	ASOS,COOP,WXSVC	NE	Y	N	A	NE	Y	NE24023	72562	-6
14942	AIRWAYS,ASOS,COOP	IA, NE	Y	N	A	NE	Y	NE14942	72558	-6
94958	ASOS,COOP	NE	Y	N	A	NE	Y	NE94958	72562	-6
24028	ASOS,COOP,WXSVC	NE	Y	N	A	NE	Y	NE24028	72662	-7
24030	AIRWAYS,ASOS,COOP	NE	Y	N	A	NE	Y	NE24030	72562	-7
94978	AIRWAYS,ASOS	NE	Y	N	A	NE	Y	NE94978	72558	-6
24032	ASOS,COOP,WXSVC	NE	N	Y	A	NE	Y	NE24032	72562	-6
94700	AIRWAYS,ASOS	NH	Y	N	A	NH	Y	NH94700	74389	-5
14745	ASOS,COOP	NH	Y	N	A	NH	Y	NH14745	74389	-5
54770	AIRWAYS,ASOS	NH	Y	N	A	NH	Y	NH54770	72518	-5
94765	AIRWAYS,ASOS	VT, NH	Y	N	A	NH	Y	NH94765	72518	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
14710	AIRWAYS,ASOS	NH	Y	N	A	NH	Y	NH14710	74389	-5
54791	AIRWAYS,ASOS	ME, NH	Y	N	A	NH	Y	NH54791	74389	-5
54728	AIRWAYS,ASOS	NH	Y	N	A	NH	Y	NH54728	74389	-5
93730	ASOS,COOP	NJ	N	N	A	NJ	Y	NJ93730	72402	-5
54743	ASOS	NJ	Y	N	A	NJ	Y	NJ54743	72501	-5
13735	AIRWAYS,ASOS,COOP	NJ	Y	N	A	NJ	Y	NJ13735	72402	-5
93780	AIRWAYS,ASOS	NJ	Y	N	A	NJ	Y	NJ93780	72501	-5
14734	ASOS,COOP	NJ	Y	N	A	NJ	Y	NJ14734	72501	-5
54785	ASOS	NJ	Y	N	A	NJ	Y	NJ54785	72501	-5
54793	ASOS	NJ	Y	N	A	NJ	Y	NJ54793	72501	-5
94741	ASOS	NJ	Y	N	A	NJ	Y	NJ94741	72501	-5
14792	AIRWAYS,ASOS	PA, NJ	Y	N	A	NJ	Y	NJ14792	72501	-5
23050	AIRSAMPLE,AIRWAYS,ASOS,COOP	NM	N	Y	A	NM	Y	NM23050	72365	-7
93033	AIRWAYS,ASOS,COOP,WXSVC	NM	N	Y	A	NM	Y	NM93033	72265	-7
23051	AIRSAMPLE,ASOS,COOP,USHCN	NM	N	Y	A	NM	Y	NM23051	72363	-7
3027	ASOS	NM	N	Y	A	NM	N	NM03027	72365	-7
23078	AIRWAYS,ASOS	NM	N	Y	A	NM	Y	NM23078	72364	-7
23090	AIRWAYS,ASOS	NM	N	Y	A	NM	Y	NM23090	72365	-7
23081	ASOS,COOP	NM	N	Y	A	NM	Y	NM23081	72365	-7
23054	ASOS	NM	N	Y	A	NM	Y	NM23054	72365	-7
23052	ASOS	NM	N	Y	A	NM	Y	NM23052	72365	-7
23009	AIRSAMPLE,AIRWAYS,ASOS,COOP, USHCN	NM	N	Y	A	NM	Y	NM23009	72364	-7
23049	AIRWAYS,ASOS	NM	N	Y	A	NM	Y	NM23049	72365	-7
93045	ASOS	NM	N	Y	A	NM	Y	NM93045	72364	-7
23048	ASOS	NM	N	Y	A	NM	Y	NM23048	72363	-7
24121	AIRWAYS,ASOS,COOP,USHCN	NV	N	Y	A	NV	Y	NV24121	72582	-8
23154	AIRWAYS,ASOS,COOP	NV	N	Y	A	NV	Y	NV23154	72582	-8
53123	ASOS	NV	N	Y	A	NV	Y	NV53123	72388	-8

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
23169	AIRSAMPLE,AIRWAYS,ASOS,COOP	NV	N	Y	A	NV	Y	NV23169	72388	-8
24172	ASOS,COOP	NV	N	Y	A	NV	Y	NV24172	72489	-8
3160	ASOS,COOP	NV	N	Y	A	NV	Y	NV03160	72388	-8
23185	ASOS,COOP,USHCN	NV	N	Y	A	NV	Y	NV23185	72489	-8
23153	AIRWAYS,ASOS,COOP	NV	N	Y	A	NV	Y	NV23153	72388	-8
24128	AIRWAYS,ASOS,COOP,USHCN	NV	N	Y	A	NV	Y	NV24128	72582	-8
14735	ASOS,COOP,USHCN	NY	Y	N	A	NY	Y	NY14735	72518	-5
4725	AIRWAYS,ASOS,COOP,USHCN	NY	Y	N	A	NY	Y	NY04725	72518	-5
14733	ASOS,COOP,USHCN	NY	Y	N	A	NY	Y	NY14733	72528	-5
94704	ASOS	NY	Y	N	A	NY	Y	NY94704	72528	-5
14747	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY14747	72528	-5
14748	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY14748	72528	-5
54787	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY54787	72501	-5
54773	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY54773	72528	-5
14750	ASOS,COOP	NY	Y	N	A	NY	Y	NY14750	72518	-5
4781	AIRWAYS,ASOS,COOP	NY	Y	N	A	NY	Y	NY04781	72501	-5
94725	ASOS	NY	Y	N	A	NY	Y	NY94725	71722	-5
4789	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY04789	72501	-5
94728	ASOS,COOP,USHCN	NY, NJ	Y	N	A	NY	N	NY94728	72501	-5
94789	ASOS,COOP	NY	Y	N	A	NY	Y	NY94789	72501	-5
14732	ASOS,COOP	NY	Y	N	A	NY	Y	NY14732	72501	-5
54778	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY54778	72528	-5
64776	ASOS	NY	Y	N	A	NY	Y	NY64776	72518	-5
14757	ASOS,COOP	NY	Y	N	A	NY	Y	NY14757	72501	-5
14768	ASOS,COOP,USHCN	NY	Y	N	A	NY	Y	NY14768	72528	-5
64775	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY64775	72518	-5
94740	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY94740	72518	-5
54790	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY54790	72501	-5
14771	ASOS,COOP,USHCN	NY	Y	N	A	NY	Y	NY14771	72518	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
94790	ASOS,COOP	NY	Y	N	A	NY	Y	NY94790	71722	-5
54757	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY54757	72528	-5
94745	AIRWAYS,ASOS,COOP	NY, CT	Y	N	A	NY	Y	NY94745	72501	-5
14719	AIRWAYS,ASOS	NY	Y	N	A	NY	Y	NY14719	72501	-5
14895	ASOS,COOP	OH	Y	N	A	OH	Y	OH14895	72520	-5
14813	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH14813	72520	-5
4857	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH04857	72520	-5
93812	AIRWAYS,ASOS,COOP	OH, KY	Y	N	A	OH	Y	OH93812	72426	-5
4853	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH04853	72520	-5
14820	ASOS,COOP	OH	Y	N	A	OH	Y	OH14820	72520	-5
4804	ASOS	OH	Y	N	A	OH	Y	OH04804	72426	-5
14821	ASOS,COOP	OH	Y	N	A	OH	Y	OH14821	72426	-5
93815	ASOS,COOP	OH	Y	N	A	OH	Y	OH93815	72426	-5
53859	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH53859	72426	-5
4851	ASOS,COOP	OH	Y	N	A	OH	Y	OH04851	72632	-5
4849	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH04849	72632	-5
14825	ASOS,COOP	OH	Y	N	A	OH	Y	OH14825	72426	-5
53855	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH53855	72426	-5
53844	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH53844	72426	-5
4850	AIRWAYS,ASOS,COOP	OH	Y	N	A	OH	Y	OH04850	72426	-5
14891	ASOS,COOP	OH	Y	N	A	OH	Y	OH14891	72426	-5
4855	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH04855	72426	-5
4852	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH04852	72520	-5
4858	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH04858	72426	-5
94830	ASOS,COOP	OH	Y	N	A	OH	Y	OH94830	72632	-5
4848	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH04848	72632	-5
13841	ASOS	OH	Y	N	A	OH	Y	OH13841	72426	-5
4842	AIRWAYS,ASOS	OH	Y	N	A	OH	Y	OH04842	72520	-5
14852	ASOS,COOP	OH	Y	N	A	OH	Y	OH14852	72520	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
93824	ASOS,COOP	OH	Y	N	A	OH	Y	OH93824	72520	-5
3959	AIRWAYS,ASOS,COOP,USHCN	OK	N	N	A	OK	Y	OK03959	74646	-6
3932	ASOS,COOP	OK	N	N	A	OK	Y	OK03932	72357	-6
3981	ASOS,COOP	OK	N	N	A	OK	Y	OK03981	72357	-6
13975	AIRWAYS,ASOS,COOP	OK	N	N	A	OK	Y	OK13975	72451	-6
53913	AIRWAYS,ASOS,COOP	OK	N	N	A	OK	Y	OK53913	72357	-6
3030	ASOS	OK	N	Y	A	OK	Y	OK03030	72363	-6
93986	AIRWAYS,ASOS,COOP,USHCN	OK	N	N	A	OK	Y	OK93986	72357	-6
3950	AIRWAYS,ASOS,COOP	OK	N	N	A	OK	Y	OK03950	72357	-6
93950	AIRWAYS,ASOS,COOP	OK	N	N	A	OK	Y	OK93950	72357	-6
93953	AIRWAYS,ASOS	OK	N	N	A	OK	Y	OK93953	72357	-6
3954	AIRWAYS,ASOS	OK	N	N	A	OK	Y	OK03954	72357	-6
13967	ASOS,COOP	OK	N	N	A	OK	Y	OK13967	72357	-6
13969	ASOS,COOP	OK	N	N	A	OK	Y	OK13969	74646	-6
3965	ASOS,COOP	OK	N	N	A	OK	Y	OK03965	74646	-6
13968	AIRSAMPLE,ASOS,COOP	OK	N	N	A	OK	Y	OK13968	74646	-6
53908	AIRWAYS,ASOS	OK	N	N	A	OK	Y	OK53908	74646	-6
94224	ASOS,COOP,USHCN	OR	N	N	A	OR	Y	OR94224	72694	-8
94281	AIRWAYS,ASOS	OR	N	N	A	OR	Y	OR94281	72694	-8
24130	AIRWAYS,ASOS,COOP,USHCN	OR	Y	N	A	OR	Y	OR24130	72681	-8
94185	ASOS,COOP	OR	N	N	A	OR	Y	OR94185	72681	-8
24221	ASOS,COOP	OR	N	N	A	OR	Y	OR24221	72694	-8
4113	AIRWAYS,ASOS	OR	N	N	A	OR	Y	OR04113	72786	-8
94236	AIRWAYS,ASOS,COOP	OR	N	N	A	OR	Y	OR94236	72597	-8
94273	AIRWAYS,ASOS	OR	N	N	A	OR	Y	OR94273	72694	-8
24225	ASOS,COOP	OR	N	N	A	OR	Y	OR24225	72597	-8
24162	AIRWAYS,ASOS,COOP	OR	N	N	A	OR	Y	OR24162	72681	-7
24155	ASOS,COOP	OR	N	N	A	OR	Y	OR24155	72786	-8
24229	ASOS,COOP	WA, OR	N	N	A	OR	Y	OR24229	72694	-8

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
24242	AIRWAYS,ASOS	WA, OR	N	N	A	OR	Y	OR24242	72694	-8
94261	AIRWAYS,ASOS	OR	N	N	A	OR	Y	OR94261	72694	-8
24230	AIRWAYS,ASOS,COOP	OR	N	N	A	OR	Y	OR24230	72694	-8
24231	AIRWAYS,ASOS	OR	N	N	A	OR	Y	OR24231	72597	-8
24232	ASOS,COOP	OR	N	N	A	OR	Y	OR24232	72694	-8
4201	AIRWAYS,ASOS	OR	N	N	A	OR	Y	OR04201	72694	-8
24235	ASOS,COOP	OR	N	N	A	OR	N	OR24235	72597	-8
14737	ASOS,COOP,USHCN	PA	Y	N	A	PA	Y	PA14737	72501	-5
14736	ASOS	PA	Y	N	A	PA	Y	PA14736	72403	-5
4751	AIRWAYS,ASOS,COOP	PA	Y	N	A	PA	Y	PA04751	72528	-5
54792	ASOS	PA	Y	N	A	PA	Y	PA54792	72520	-5
54786	AIRWAYS,ASOS	PA	Y	N	A	PA	Y	PA54786	72501	-5
4787	ASOS,COOP	PA	Y	N	A	PA	Y	PA04787	72520	-5
14860	ASOS,COOP,USHCN	PA	Y	N	A	PA	Y	PA14860	72520	-5
14751	AIRWAYS,ASOS	PA	Y	N	A	PA	Y	PA14751	72403	-5
4726	ASOS	PA	Y	N	A	PA	Y	PA04726	72520	-5
54737	AIRWAYS,ASOS	PA	Y	N	A	PA	Y	PA54737	72403	-5
14711	AIRWAYS,ASOS,COOP	PA	Y	N	A	PA	Y	PA14711	72403	-5
54789	AIRWAYS,ASOS	PA	Y	N	A	PA	Y	PA54789	72501	-5
13739	ASOS,COOP	PA, NJ	Y	N	A	PA	Y	PA13739	72403	-5
94732	ASOS	PA	Y	N	A	PA	Y	PA94732	72501	-5
14762	AIRWAYS,ASOS,WXSVC	PA	N	N	A	PA	Y	PA14762	72520	-5
94823	ASOS,COOP	PA	Y	N	A	PA	Y	PA94823	72520	-5
4843	AIRWAYS,ASOS	PA	Y	N	A	PA	Y	PA04843	72520	-5
54782	AIRWAYS,ASOS	PA	Y	N	A	PA	Y	PA54782	72403	-5
14712	ASOS	PA	Y	N	A	PA	Y	PA14712	72403	-5
14770	AIRWAYS,ASOS	PA	Y	N	A	PA	Y	PA14770	72403	-5
14777	ASOS,COOP	PA	Y	N	A	PA	Y	PA14777	72518	-5
14778	ASOS,COOP,USHCN	PA	Y	N	A	PA	Y	PA14778	72528	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
93778	AIRWAYS,ASOS	PA	Y	N	A	PA	Y	PA93778	72403	-5
11641	ASOS,COOP	PR	N	N	A	PR	Y	PR11641	78526	-4
14787	AIRWAYS,ASOS	RI	Y	N	A	RI	Y	RI14787	74494	-5
14765	ASOS,COOP,USHCN	RI	Y	N	A	RI	Y	RI14765	74494	-5
14794	AIRWAYS,ASOS	CT, RI	Y	N	A	RI	Y	RI14794	72501	-5
93846	AIRWAYS,ASOS,COOP	SC	N	N	A	SC	Y	SC93846	72215	-5
13880	ASOS,COOP,UPPERAIR	SC	N	N	A	SC	Y	SC13880	72208	-5
53850	ASOS	SC	N	N	A	SC	Y	SC53850	72215	-5
13883	ASOS,COOP	SC	N	N	A	SC	Y	SC13883	72208	-5
53867	AIRWAYS,ASOS	SC	N	N	A	SC	Y	SC53867	72208	-5
13744	AIRWAYS,ASOS,COOP	SC	N	N	A	SC	Y	SC13744	72208	-5
13886	AIRWAYS,ASOS	SC	N	N	A	SC	Y	SC13886	72215	-5
53874	ASOS	SC	N	N	A	SC	Y	SC53874	72208	-5
3870	ASOS,COOP,USHCN	SC	N	N	A	SC	Y	SC03870	72317	-5
93718	ASOS,COOP	SC	N	N	A	SC	Y	SC93718	72208	-5
53854	AIRWAYS,ASOS	SC	N	N	A	SC	Y	SC53854	72208	-5
53871	AIRWAYS,ASOS	SC	N	N	A	SC	Y	SC53871	72317	-5
14929	ASOS,COOP,USHCN,WXSVC	SD	Y	N	A	SD	Y	SD14929	72659	-6
94032	ASOS	SD	N	Y	A	SD	Y	SD94032	72662	-7
14936	ASOS,COOP,WXSVC	SD	Y	N	A	SD	Y	SD14936	72659	-6
94950	AIRWAYS,ASOS,COOP	SD	Y	N	A	SD	Y	SD94950	72659	-6
94052	AIRWAYS,ASOS	SD	Y	N	A	SD	Y	SD94052	72764	-6
24024	ASOS	SD	N	Y	A	SD	Y	SD24024	72662	-7
24025	AIRWAYS,ASOS,COOP,USHCN	SD	N	Y	A	SD	Y	SD24025	72659	-6
94039	AIRWAYS,ASOS	SD, NE	N	Y	A	SD	Y	SD94039	72662	-7
24090	ASOS,COOP	SD	N	Y	A	SD	Y	SD24090	72662	-7
14944	ASOS,COOP,WXSVC	SD	Y	N	A	SD	Y	SD14944	72558	-6
14946	AIRWAYS,ASOS,COOP,USHCN	SD	Y	N	A	SD	Y	SD14946	72659	-6
94990	AIRWAYS,ASOS	SD	N	Y	A	SD	Y	SD94990	72562	-6

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
13877	AIRSAMPLE,ASOS,COOP	TN	N	N	A	TN	Y	TN13877	72318	-5
13882	AIRSAMPLE,AIRWAYS,ASOS,COOP, WXSVC	TN, GA	N	N	A	TN	Y	TN13882	72215	-5
3894	ASOS,COOP	KY, TN	N	N	A	TN	Y	TN03894	72327	-6
3847	ASOS,COOP	TN	N	N	A	TN	Y	TN03847	72327	-6
13891	AIRSAMPLE,AIRWAYS,ASOS,COOP	TN	N	N	A	TN	Y	TN13891	72215	-5
13893	AIRSAMPLE,ASOS,COOP	TN	N	N	A	TN	Y	TN13893	72340	-6
13897	AIRSAMPLE,ASOS,COOP	TN	N	N	A	TN	Y	TN13897	72327	-6
53868	AIRSAMPLE,ASOS,COOP	TN	N	N	A	TN	N	TN53868	72327	-5
13962	AIRSAMPLE,ASOS,COOP	TX	N	Y	A	TX	Y	TX13962	72249	-6
12932	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX12932	72251	-6
23047	ASOS,COOP	TX	N	Y	A	TX	Y	TX23047	72363	-6
12976	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX12976	72251	-6
53907	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX53907	72249	-6
13904	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX13904	72251	-6
13958	ASOS,COOP	TX	N	N	A	TX	N	TX13958	72249	-6
3024	AIRWAYS,ASOS	TX	N	Y	A	TX	Y	TX03024	72363	-6
12919	AIRWAYS,ASOS,COOP,UPPERAIR	TX	N	Y	A	TX	Y	TX12919	72250	-6
3999	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX03999	72249	-6
23007	AIRWAYS,ASOS,COOP	TX	N	Y	A	TX	Y	TX23007	72363	-6
3904	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX03904	72249	-6
53902	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX53902	72240	-6
12924	ASOS,COOP,USHCN,WXSVC	TX	N	N	A	TX	Y	TX12924	72251	-6
53912	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX53912	72249	-6
12947	ASOS,COOP	TX	N	N	A	TX	Y	TX12947	72251	-6
3927	ASOS,COOP	TX	N	N	A	TX	Y	TX03927	72249	-6
93042	ASOS,COOP	TX	N	Y	A	TX	Y	TX93042	72363	-6
13960	ASOS,COOP	TX	N	N	A	TX	Y	TX13960	72249	-6
3971	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX03971	72249	-6

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
22010	ASOS,COOP	TX	N	Y	A	TX	Y	TX22010	72261	-6
3991	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX03991	72249	-6
23044	ASOS,COOP,USHCN	TX	N	Y	A	TX	Y	TX23044	72364	-7
23091	AIRWAYS,ASOS	TX	N	Y	A	TX	Y	TX23091	72265	-6
53909	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX53909	72249	-6
13961	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX13961	72249	-6
12923	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX12923	72240	-6
12904	AIRWAYS,ASOS	TX	N	Y	A	TX	Y	TX12904	72250	-6
12962	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX12962	72261	-6
12975	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX12975	72240	-6
12918	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX12918	72240	-6
53910	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX53910	72240	-6
12960	ASOS,COOP	TX	N	N	A	TX	Y	TX12960	72240	-6
12977	ASOS,COOP	TX	N	N	A	TX	Y	TX12977	72240	-6
53903	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX53903	72240	-6
13973	AIRWAYS,ASOS	TX	N	Y	A	TX	Y	TX13973	72261	-6
3901	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX03901	72248	-6
23042	AIRWAYS,ASOS,COOP	TX	N	Y	A	TX	Y	TX23042	72363	-6
93987	ASOS,COOP	TX	N	N	A	TX	Y	TX93987	72248	-6
12959	AIRWAYS,ASOS,COOP,WXSVC	TX	N	Y	A	TX	Y	TX12959	72250	-6
53914	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX53914	72249	-6
23023	ASOS,COOP,WXSVC	TX	N	Y	A	TX	Y	TX23023	72265	-6
93985	ASOS,COOP	TX	N	Y	A	TX	Y	TX93985	72249	-6
12971	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX12971	72251	-6
3031	AIRWAYS,ASOS	TX	N	Y	A	TX	Y	TX03031	72265	-6
12935	ASOS,COOP	TX	N	N	A	TX	Y	TX12935	72251	-6
23055	ASOS	TX	N	Y	A	TX	N	TX23055	72364	-6
12917	ASOS,COOP	TX	N	N	A	TX	Y	TX12917	72240	-6
12957	AIRWAYS,ASOS	TX	N	Y	A	TX	Y	TX12957	72250	-6

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
12972	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX12972	72251	-6
23034	AIRSAMPLE,ASOS,COOP	TX	N	Y	A	TX	Y	TX23034	72265	-6
12921	ASOS,COOP,USHCN	TX	N	N	A	TX	Y	TX12921	72251	-6
12970	AIRWAYS,ASOS,COOP	TX	N	N	A	TX	Y	TX12970	72251	-6
53911	AIRWAYS,ASOS	TX	N	N	A	TX	Y	TX53911	72249	-6
13972	AIRWAYS,ASOS,COOP,WXSVC	TX	N	N	A	TX	Y	TX13972	72248	-6
12912	ASOS,COOP	TX	N	N	A	TX	Y	TX12912	72251	-6
13959	ASOS,COOP	TX	N	N	A	TX	Y	TX13959	72249	-6
13966	ASOS,COOP	TX	N	N	A	TX	Y	TX13966	72357	-6
23040	ASOS,COOP,WXSVC	TX	N	Y	A	TX	Y	TX23040	72265	-6
23159	ASOS	UT	N	Y	A	UT	Y	UT23159	72376	-7
93129	AIRWAYS,ASOS,COOP	UT	N	Y	A	UT	Y	UT93129	72388	-7
94128	AIRWAYS,ASOS	UT	N	Y	A	UT	Y	UT94128	72572	-7
23176	ASOS	UT	N	Y	A	UT	Y	UT23176	72572	-7
93075	AIRWAYS,ASOS,COOP	UT	N	Y	A	UT	Y	UT93075	72476	-7
93141	AIRWAYS,ASOS	UT	N	Y	A	UT	Y	UT93141	72572	-7
24127	ASOS,COOP	UT	N	Y	A	UT	Y	UT24127	72572	-7
94030	ASOS,COOP	UT	N	Y	A	UT	Y	UT94030	72476	-7
93775	AIRWAYS,ASOS	VA	Y	N	A	VA	Y	VA93775	72403	-5
93736	AIRWAYS,ASOS	VA	N	N	A	VA	Y	VA93736	72403	-5
13728	ASOS,COOP	VA, NC	N	N	A	VA	Y	VA13728	72317	-5
13733	ASOS,COOP	VA	N	N	A	VA	Y	VA13733	72318	-5
93741	ASOS	VA	N	N	A	VA	Y	VA93741	72402	-5
13737	ASOS,COOP,USHCN	VA	N	N	A	VA	Y	VA13737	72402	-5
13740	ASOS,COOP	VA	N	N	A	VA	Y	VA13740	72403	-5
13741	ASOS,COOP	VA	N	N	A	VA	Y	VA13741	72318	-5
93773	AIRWAYS,ASOS	VA	N	N	A	VA	Y	VA93773	72402	-5
93739	ASOS,COOP	VA	N	N	A	VA	Y	VA93739	72402	-5
93738	ASOS,COOP	VA	N	N	A	VA	Y	VA93738	72403	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
13743	ASOS,COOP	MD, DC, VA	N	N	A	VA	Y	VA13743	72403	-5
94705	AIRWAYS,ASOS,COOP	VT	Y	N	A	VT	Y	VT94705	72518	-5
54781	AIRWAYS,ASOS	VT, NY	Y	N	A	VT	Y	VT54781	72518	-5
14742	ASOS,COOP,USHCN	VT	Y	N	A	VT	Y	VT14742	72518	-5
54771	AIRWAYS,ASOS	VT	Y	N	A	VT	Y	VT54771	72518	-5
54740	AIRWAYS,ASOS	VT	Y	N	A	VT	Y	VT54740	72518	-5
24217	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA24217	72797	-8
94119	AIRWAYS,ASOS,COOP	WA	Y	N	A	WA	Y	WA94119	72786	-8
24220	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA24220	72786	-8
24141	AIRWAYS,ASOS,COOP	WA	Y	N	A	WA	Y	WA24141	72786	-8
24222	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA24222	72797	-8
94276	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA94276	72797	-8
94225	ASOS,COOP	WA	N	N	A	WA	Y	WA94225	72797	-8
24110	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA24110	72786	-8
24227	ASOS,COOP	WA	N	N	A	WA	Y	WA24227	72797	-8
94197	ASOS	WA	Y	N	A	WA	N	WA94197	71203	-8
94266	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA94266	72797	-8
94129	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA94129	72786	-8
94240	ASOS,COOP	WA	N	N	A	WA	Y	WA94240	72797	-8
94248	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA94248	72797	-8
24234	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA24234	72797	-8
24233	ASOS,COOP	WA	N	N	A	WA	Y	WA24233	72797	-8
94227	ASOS,COOP	WA	N	N	A	WA	Y	WA94227	72797	-8
94176	AIRWAYS,ASOS	WA	Y	N	A	WA	Y	WA94176	72786	-8
24157	AIRSAMPLE,ASOS,COOP,USHCN	WA	Y	N	A	WA	Y	WA24157	72786	-8
94274	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA94274	72797	-8
24219	AIRWAYS,ASOS,COOP	WA, OR	Y	N	A	WA	Y	WA24219	72694	-8
94298	AIRWAYS,ASOS	WA, OR	N	N	A	WA	Y	WA94298	72694	-8

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
24160	AIRWAYS,ASOS	WA	N	N	A	WA	Y	WA24160	72786	-8
94239	ASOS,COOP	WA	Y	N	A	WA	Y	WA94239	72786	-8
24243	ASOS,COOP	WA	N	N	A	WA	Y	WA24243	72694	-8
94929	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI94929	72649	-6
94994	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI94994	74455	-6
14991	ASOS,COOP,WXSVC	WI	Y	N	A	WI	Y	WI14991	72649	-6
4840	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI04840	72645	-6
14898	ASOS,COOP,WXSVC	WI	Y	N	A	WI	Y	WI14898	72645	-6
94973	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI94973	72649	-6
4845	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI04845	72645	-6
14920	AIRWAYS,ASOS,COOP	WI, MN	Y	N	A	WI	Y	WI14920	74455	-6
14921	ASOS	WI	Y	N	A	WI	Y	WI14921	74455	-6
14837	ASOS,COOP,WXSVC	WI	Y	N	A	WI	Y	WI14837	72645	-6
94985	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI94985	72645	-6
14839	AIRSAMPLE,ASOS,COOP	WI	Y	N	A	WI	Y	WI14839	72645	-6
94855	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI94855	72645	-6
94818	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI94818	72645	-6
4803	AIRWAYS,ASOS,COOP	WI	Y	N	A	WI	Y	WI04803	72645	-6
4841	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI04841	72645	-6
14897	ASOS,COOP	WI	Y	N	A	WI	Y	WI14897	72645	-6
4826	AIRWAYS,ASOS	WI	Y	N	A	WI	Y	WI04826	72645	-6
3872	ASOS,COOP	WV	Y	N	A	WV	Y	WV03872	72318	-5
3859	ASOS,COOP	WV, VA	Y	N	A	WV	Y	WV03859	72318	-5
13866	ASOS,COOP	WV	Y	N	A	WV	Y	WV13866	72318	-5
3802	AIRWAYS,ASOS	WV	Y	N	A	WV	Y	WV03802	72520	-5
13729	ASOS,COOP	WV	Y	N	A	WV	Y	WV13729	72520	-5
3860	ASOS,COOP	WV, OH, KY	N	N	A	WV	Y	WV03860	72426	-5
13734	ASOS,COOP,USHCN	WV	N	N	A	WV	Y	WV13734	72403	-5

Surface WBAN	Type Station	State tif file	snow	arid	moisture	state	Airport	AERMOD File	Upper Air WMO	Upper Air UTC
13736	AIRWAYS,ASOS,COOP	WV	Y	N	A	WV	Y	WV13736	72520	-5
3804	ASOS,COOP	WV	Y	N	A	WV	Y	WV03804	72520	-5
14894	AIRWAYS,ASOS	OH, WV	N	N	A	WV	Y	WV14894	72520	-5
24164	AIRWAYS,ASOS	WY	N	Y	A	WY	Y	WY24164	72672	-7
94054	AIRWAYS,ASOS	WY	N	Y	A	WY	Y	WY94054	72672	-7
24089	AIRWAYS,ASOS,COOP,WXSVC	WY	N	Y	A	WY	Y	WY24089	72672	-7
24018	ASOS,COOP,USHCN,WXSVC	WY	Y	N	A	WY	Y	WY24018	72469	-7
4111	AIRWAYS,ASOS	WY, UT	N	Y	A	WY	Y	WY04111	72572	-7
94023	AIRWAYS,ASOS	WY	N	Y	A	WY	Y	WY94023	72662	-7
24048	AIRWAYS,ASOS	WY	N	Y	A	WY	Y	WY24048	72672	-7
24021	ASOS,COOP,WXSVC	WY	N	Y	A	WY	Y	WY24021	72672	-7
24022	ASOS,COOP,USHCN	WY	Y	N	A	WY	Y	WY24022	72469	-7
24057	ASOS,COOP	WY	N	Y	A	WY	Y	WY24057	72672	-7
24061	AIRWAYS,ASOS,WXSVC	WY	Y	N	A	WY	Y	WY24061	72672	-7
24027	AIRWAYS,ASOS,AWOS,COOP, USHCN,WXSVC	WY	N	Y	A	WY	Y	WY24027	72672	-7
24029	ASOS,COOP	WY	N	Y	A	WY	Y	WY24029	72672	-7
94053	AIRWAYS,ASOS,COOP	WY	N	N	A	WY	Y	WY94053	72662	-7
24062	AIRWAYS,ASOS	WY	Y	N	A	WY	Y	WY24062	72672	-7

Appendix 4

Dispersion Model Receptor Revisions and Additions

Dispersion Model Receptor Revisions and Additions for the Metal Coil Coatings Source Category

To estimate ambient concentrations for evaluating long-term exposures, the HEM-3 model uses the geographic centroids of census blocks (currently utilizing the 2010 Census) as dispersion model receptors. The census block centroids are generally good surrogates for where people live within a census block. A census block generally encompasses about 40 people or 10-15 households. However, in cases where a block centroid is located on industrial property, or where a census block is large and the centroid less likely to be representative of the block's residential locations, the block centroid may not be an appropriate surrogate.

Census block centroids that are on facility property can sometimes be identified by their proximity to emission sources. In cases where a census block centroid was within 300 meters of any emission source, we viewed aerial images of the facility to determine whether the block centroid was likely located on facility property. The selection of the 300-meter distance reflects a compromise between too few and too many blocks identified as being potentially on facility property. Distances smaller than 300 meters would identify only block centroids very near the emission sources and could exclude some block centroids that are still within facility boundaries, particularly for large facilities. Distances significantly larger than 300 meters would identify many block centroids that are outside facility boundaries, particularly for small facilities. Where we confirmed a block centroid on facility property, we moved the block centroid to a location that best represents the residential locations in the block.

In addition, census block centroids for blocks with large areas may not be representative of residential locations. Risk estimates based on such centroids can be understated if there are residences nearer to a facility than the centroid, and overstated if the residences are farther from the facility than the centroid. To avoid understating the maximum individual risk associated with a facility, in some cases we relocated block centroids, or added dispersion model receptors other than the block centroid. We examined aerial images of all large census blocks within one kilometer of any emission source. Experience from previous risks characterizations show that in most cases the MIR is generally located within 1 km of the facility boundary. If the block centroid did not represent the residential locations, we relocated it to better represent them. If residential locations could not be represented by a single receptor (that is, the residences were spread out over the block), we added additional receptors for residences nearer to the facility than the centroid.

For this source category, the table below contains each census block for which we changed the centroid location because it was on facility property or was otherwise not representative of the residential locations in the block. The table also contains the locations of additional receptors that were included to represent residential locations nearer to the facility than the block centroid.

Revised Census Block Centroid Locations and Additional Receptors for the Metal Coil Coatings Source Category

Block No	FacilityID	NewLat	NewLong	Issue	Action
	01033110005979824	34.763754°	-87.571884°	Large Block	Add Receptor
010730139021005	01073110006548566	33.447153°	-86.973484°	On-site/ Centroid not representative of residences	Move Centroid
050930107004082	05093110028304957	35.927854°	-89.759227°	On-site/ Centroid not representative of residences	Move Centroid
060710021101057	06071110002647743	34.091425°	-117.579734°	On-site/ Centroid not representative of residences	Move Centroid
060710021101069	06071110002647743	34.090563°	-117.584177°	On-site/ Centroid not representative of residences	Move Centroid
	12069110001969576	28.872765°	-81.777358°	Large Block	Add Receptor
170318205012000	17031110000433102	41.764444°	-87.817275°	On-site/ Centroid not representative of residences	Move Centroid
171194009043009	17119110003064567	38.740960°	-90.124869°	On-site/ Centroid not representative of residences	Move Centroid
171194041002273	17119110003064567	38.740360°	-90.136553°	On-site/ Centroid not representative of residences	Move Centroid
	18059110000396535	39.789646°	-85.740933°	Large Block	Add Receptor
	18059110000396535	39.790057°	-85.738116°	Large Block	Add Receptor
181279800011022	18127110000747862	41.623173°	-87.184533°	On-site/ Centroid not representative of residences	Move Centroid
	18141110000399453	41.689880°	-86.453270°	Large Block	Add Receptor
	18145110021290413	39.581691°	-85.862631°	Large Block	Add Receptor
	18145110021290413	39.581560°	-85.857402°	Large Block	Add Receptor
191630110002002	19163110000416014	---	---	No Residences	Remove Centroid
	21091110000380935	37.929475°	-86.782222°	Large Block	Add Receptor
	21091110000380935	37.923977°	-86.769961°	Large Block	Add Receptor
	21141110000605364	36.944829°	-86.942926°	Large Block	Add Receptor
280490111031128	28049110000376815	32.174094°	-90.248582°	On-site/ Centroid not representative of residences	Move Centroid
280490111031133	28049110000376815	32.176281°	-90.245559°	On-site/ Centroid not representative of residences	Move Centroid
280490111031122	28049110000376815	---	---	No Residences	Remove Centroid
	28049110000376815	32.170378°	-90.259025°	Large Block	Add Receptor
	28049110000376879	32.170460°	-90.259059°	Large Block	Add Receptor
	29510110000441255	38.581448°	-90.251724°	Large Block	Add Receptor

340350536041000	34035110000858670	40.605437°	-74.699536°	On-site/ Centroid not representative of residences	Move Centroid
	34035110000858670	40.599850°	-74.694052°	Large Block	Add Receptor
360150110002040	36015110046535344	42.006989°	-76.611434°	On-site/ Centroid not representative of residences	Move Centroid
	36015110046535344	42.007421°	-76.616108°	Large Block	Add Receptor
390950056002039	39095110000384218	41.715243°	-83.527610°	On-site/ Centroid not representative of residences	Move Centroid
390950056002040	39095110000384218	---	---	No Residences	Remove Centroid
391199116001031	39119110000384487	---	---	No Residences	Remove Centroid
391290212001023	39129110000381836	39.742640°	-82.965466°	On-site/ Centroid not representative of residences	Move Centroid
	39129110000381836	39.724143°	-82.967704°	Large Block	Add Receptor
391290212001025	39129110000381836	39.724107°	-82.959675°	On-site/ Centroid not representative of residences	Move Centroid
391559211003017	39155110000389963	41.249241°	-80.817209°	On-site/ Centroid not representative of residences	Move Centroid
391730208003005	39173110000383353	41.600642°	-83.511786°	On-site/ Centroid not representative of residences	Move Centroid
420810117005038	42081110000584163	41.179067°	-76.857232°	On-site/ Centroid not representative of residences	Move Centroid
	42081110000584163	41.183835°	-76.844671°	Large Block	Add Receptor
	42081110000584163	41.181793°	-76.852443°	Large Block	Add Receptor
	45047110016994118	34.247251	-82.054168	Large Block	Add Receptor
	45047110016994118	34.245539	-82.057218	Large Block	Add Receptor
	45047110016994118	34.243905	-82.05963	Large Block	Add Receptor
	45047110016994118	34.240607	-82.063691	Large Block	Add Receptor
	45047110016994118	34.255855	-82.054028	Large Block	Add Receptor
	45047110016994118	34.245363	-82.07548	Large Block	Add Receptor
	45047110016994118	34.238316	-82.070229	Large Block	Add Receptor
	45047110016994118	34.258609	-82.062555	Large Block	Add Receptor
511210207004046	51121110000343012	37.186380°	-80.388271°	On-site/ Centroid not representative of residences	Move Centroid
	54009110000344716	40.388960°	-80.604087°	Large Block	Add Receptor
	55133110000858652	43.104122°	-88.481883°	Large Block	Add Receptor

Appendix 5:
Analysis of data on short-term emission rates relative to long-term emission rates

Analysis of data on short-term emission rates relative to long-term emission rates

Ted Palma
Roy Smith
EPA/OAQPS/SBAG

1. Introduction

1.1. The problem

The process of listing hazardous air pollutants (HAPs) provided by the Clean Air Act (CAA, section 112(b)(2)) explicitly includes acute toxicity as a listing criterion. For this reason, in addition to chronic exposures, EPA considers acute exposures in risk-based decision-making for the HAP regulatory program. Estimating acute exposures via dispersion modeling requires input data on hourly meteorological conditions (available for most areas of the US) and short-term emission rates of individual facilities (almost universally absent from the National Emissions Inventory (NEI), the Toxic Release Inventory (TRI), and state emission databases).

Lacking short-term emission rates, we must estimate peak short-term rates based on annual average rates, which are available. For Risk and Technology Review (RTR) rulemakings, we have assumed that the 1-hour emission rate for each facility could exceed the annual average hourly emission rate by as much as tenfold, and further assumed that this tenfold emission spike could coincide with worst-case meteorological conditions and the presence of a human receptor at the facility boundary, as a means of screening for potentially significant acute exposures.

In a consultation on the "RTR Assessment Plan", a panel of the EPA's Science Advisory Board (SAB), several reviewers questioned the appropriateness of the factor of ten; some even suggested that this tenfold assumption may underestimate actual maximum short-term emissions for some facilities, and thereby also underestimate maximum acute risks. The SAB recommended an analysis of available short-term emissions data for HAP to test this assumption. This analysis responds to that SAB recommendation and attempts to test the protectiveness of the tenfold assumption using a database of "event emissions" collected from facilities in the Houston-Galveston area, to compare events representative of HAP releases to long-term release rates. We welcome comments from the public on the methods used and the conclusions reached by this analysis.

2. Methods

2.1. Texas Commission on Environmental Quality event emissions database

The Texas Commission on Environmental Quality (TCEQ) collects emissions data using online reporting required of any facility releasing 100 pounds or more of a listed chemical (primarily ozone-forming VOCs) during a non-routine event. The TCEQ data are intended to improve the state's knowledge of how short-term releases affect tropospheric ozone levels in that area. The database we utilized in our analysis was a subset of the TCEQ data covering emission events that

occurred in an eight-county area in eastern Texas during a 756-day period between January 31, 2003 and February 25, 2005.

The complete emissions event data were obtained in April 2007 from Cynthia Folsom Murphy, a research scientists with the University of Texas at Austin (UTA) Center for Energy and Environmental Resources. The data were provided in four Excel spreadsheets generated from an original MS Access file. We used these Excel files to reconstruct a MS Access database in order to facilitate selection of a representative subset of records for this analysis.

Although some of the released substances were HAPs, this was incidental to the database's primary purpose of enhancing the TCEQ's knowledge of photochemical activity. Thus, more than 80% of the released mass was ethene and propene, neither of which are HAPs. The database included release events caused by accidents, equipment failures, maintenance, startup, and shutdown. It also contained facility names, information on amounts of individual compounds released. To provide a basis for comparing the event releases with "typical" emissions, the UTA staff included total VOC emissions data for each facility for calendar year 2004, obtained from the EPA Toxic Release Inventory (TRI). The database did *not* contain any records for facilities that did not experience any reportable events during this period.

2.2. Data filtering

Because the event release data were intended for modeling short-term releases of ozone-producing VOCs, the database includes releases from accidents (which are regulated under section 112(r) of the CAA and are therefore not considered in residual risk assessments) and releases of light hydrocarbon compounds that are not HAPs and are much more volatile than most HAPs. This intent of this analysis, on the other hand, was to evaluate short-term releases of HAPs due to normal process variability or scheduled startups, shutdowns, and maintenance, relative to long-term release rates. Because the full emission events database was not representative of likely HAP emissions normally considered under the residual risk program, we filtered the release data as follows in an attempt to improve its representativeness:

1. Hydrocarbons of C5 or less were dropped, except that all HAPs (including non-VOCs) were retained regardless of molecular structure;

2. Accidental releases were dropped, but all others (including startup, shutdown, and maintenance) were retained;

3. Only facilities whose long-term VOC releases exceeded 0.068 tons per day (25 tons per year) were retained, to approximate the population of facilities likely to be subject to residual risk standards (i.e., major facilities);

4. A few release records had to be dropped because their facility numbers did not link to any facility in the database;

5. A few facilities had to be dropped because the database did not include their 2004 TRI VOC release information.

2.3. Analysis

Annual VOC emissions and emission event release data were both converted to lb/hr. In order to conform to our atmospheric dispersion models, which estimate ambient concentrations for periods of 1 hour or more, amounts released during events shorter than 1 hour were assigned to the whole hour. For example, a release of 100 lb in ten minutes was converted to 100 lb/hr. Events longer than 1 hour were converted normally, e.g., a release of 100 lb in 120 minutes was converted to 50 lb/hr. The event release rates for individual compounds were summed, yielding a total release rate for each event. This total release rate for each event was divided by the annual VOC release rate for the facility to derive the ratio of peak-to-mean emission rate for the event.

3. Results and Discussion

3.1. Database filtering

The original database contained 505 individual contaminants, including multiple redundancies. These redundancies did not affect this analysis, so we did not resolve them. After filtering out light, non-HAP, VOCs, 317 contaminants remained (Table 1).

The database contained release records for 150 unique facilities. Of these, 48 facilities (Table 2) were major VOC emitters that reported releases of at least one of the contaminants in Table 1.

The database contained 3641 individual release events reported by the original 150 facilities. Of these, 319 events involved a Table 1 contaminant released by a Table 2 facility during startup, shutdown, or maintenance. For evaluating short-term releases for residual risk assessments, these 319 events comprise the most representative subset of the full database.

3.2. Descriptive statistics

For this subset of emission events, ratios of event release rate to long-term release rate varied from 0.00000004 to 74. Distribution statistics appear in Tables 3 and 4. The 99^{th} percentile ratio was 9 (i.e., an event release rate nine times the long-term average). Only 3 ratios exceeded our default assumption of 10, and of these only one exceeded 11. The full cumulative probability density of the ratios is shown in Figure 1.

Figure 2 shows the relationship between ratio and event duration. As expected, the ratio declined as duration increased. Only 18 events lasted less than 2 hours, but these events produced the three highest ratios. Figure 3 is a similar ratio vs. duration plot, but with duration as a percentage of total time. Only 35 events exceeded 1% of the total period covered by the database. Figure 4 shows the relationship between ratio and total amount released, and suggests that the highest ratios were produced by facilities whose routine VOC emissions were relatively small. Thus, the events themselves also tended to be relatively small in absolute terms.

3.3. Discussion

These results suggest that the tenfold ratio assumption for short-term releases is protective, and that the facilities for which it may underestimate event releases may tend to be smaller emitters.

However, this analysis is limited in the following ways by the nature of the database and the filtering that we applied:

1. The only long-term release data available from the database were total VOC emissions for 2004. Ideally, we would have preferred to have routine release rates for each individual contaminant. However, retrieving these data from other sources and linking them to this database was not feasible.

2. Removing VOCs that are not representative of HAPs, and comparing the releases against all VOCs, would tend to underestimate the true ratios. This effect could be quantitatively large.

3. Retaining HAPs that are not VOCs (such as toxic metals), and including them in the total to be compared against all VOCs, would tend to overestimate the true ratios. The size of this effect is not known, but seems likely to be less than for (2) above.

4. The database contains only facilities that had at least one release event during the reporting period. The number of facilities in the statistical population that did not experience an event is not known. The lack of data for these facilities (whose ratios in this analysis would have been zero) would cause the descriptive statistics to be skewed toward an overestimate. The size of this effect is unknown.

Table 1. Event emissions in the Houston-Galveston area. Representative contaminants included in the analysis, selected because they are either HAPs or VOCs with more than 5 carbon atoms. (These data were retrieved directly from the original database, which included multiple redundancies that did not affect the analysis and were left intact.)

Contaminant	*HAP*	*CAS*	*SAROAD*
2-Methyloctane	No	3221-61-2	90008
2-Methylpentane	No	107-83-5	43229
2-methylhexane	No	591-76-4	43263
2-Methylpentane	No	107-83-5	43229
2,2,3-Trimethylpentane	No	564-02-3	
2,2,4-Trimethylpentane	Yes	540-84-1	43250
dimethyl butane	No	75-83-2	43291
2,3-Dimethylbutane	No	79-29-8	43276
2,3,4-Trimethylpentane	No	565-75-3	43252
2,3-Dimethylbutane	No	79-29-8	43276
2,4-Dimethylpentane	No	108-08-7	43247
2-methylheptane	No	592-27-8	43296
2-methylhexane	No	591-76-4	43263
2-Methylpentane	No	107-83-5	43229
3-Methylhexane	No	589-34-4	43295
3-Methylpentane	No	96-14-0	43230
3-Methylhexane	No	589-34-4	43295
3-Methylpentane	No	96-14-0	43230
3-Methylheptane	No	589-81-1	43253
3-Methylhexane	No	589-34-4	43295
3-Methylpentane	No	96-14-0	43230

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Contaminant	*HAP*	*CAS*	*SAROAD*
Acetaldehyde	Yes	75-07-0	43503
Acetic Acid	No	64-19-7	43404
Acetonitrile	Yes	75-05-8	70016
Acetophenone	Yes	98-86-2	
Acrolein	Yes	107-02-8	43505
Acrylic acid	Yes	79-10-7	43407
Acrylonitrile	Yes	107-13-1	43704
alkylphenol	No	none	
Benzene	Yes	71-43-2	45201
Benzo[a]anthracene	Yes	56-55-3	46716
Benzo[a]pyrene	Yes	50-32-8	46719
Benzo[b]fluoranthene	Yes	205-99-2	46717
Biphenyl	Yes	92-52-4	45226
Butanol	No	35296-72-1	
Butyl Acrylate	No	141-32-2	43440
t-Butyl Alcohol	No	75-65-0	43309
butylcyclohexane	No	1678-93-9	90101
Butyraldehyde	No	123-72-8	43510
C9 Aromatics	No	none	
Naphthalene	Yes	91-20-3	46701
Nonane	No	111-84-2	43235
C9+	No	none	
Carbon tetrachloride	Yes	56-23-5	43804
Carbonyl Sulfide	Yes	463-58-1	43933
Chloral	No	75-87-6	
Trichloromethane	Yes	67-66-3	43803
Chlorothalonil	No	1897-45-6	
Petroleum	No	8002-05-9	
Petroleum	No	8002-05-9	
Cumene	Yes	98-82-8	45210
Cyclohexane	No	110-82-7	43248
Cyclohexanol	No	108-93-0	43317
Cyclohexanone	No	108-94-1	43561
Cyclohexanone	No	108-94-1	43561
Decane	No	124-18-5	43238
Decane	No	124-18-5	43238
1,2-Dichloroethane	No	107-06-2	43815
Diethylbenzene (mixture)	No	25340-17-4	45106
Methyl Ether	No	115-10-6	43350
Dimethylcyclohexane	No	27195-67-1	98059
Dimethylcyclopentane	No	28729-52-4	90064

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Contaminant	*HAP*	*CAS*	*SAROAD*
Dimethylcyclopentane	No	28729-52-4	90064
Dimethyl formamide	Yes	68-12-2	43450
Dimethylhexane	No	28777-67-5	90067
Dimethyl pentane	No	38815-29-1	90063
Epichlorohydrin	Yes	106-89-8	43863
Ethyl Alcohol	No	64-17-5	43302
Ethyl Acrylate	Yes	140-88-5	43438
Ethyl Alcohol	No	64-17-5	43302
Ethyl Benzene	Yes	100-41-4	45203
Ethyl Chloride	Yes	75-00-3	43812
Ethylcyclohexane	No	1678-91-7	43288
ethylacetylene	No	107-00-6	43281
Ethyl Benzene	Yes	100-41-4	45203
Ethylene Oxide	Yes	75-21-8	43601
ethylmethylbenzene	No	25550-14-5	45104
formaldehyde	Yes	50-00-0	43502
Furfural	No	98-01-1	45503
straight-run middle distillate	No	64741-44-2	
Gasoline	No	86290-81-5	
Gasoline	No	86290-81-5	
Heavy Olefins	No	none	
n-Heptane	No	142-82-5	43232
n-Heptane	No	142-82-5	43232
Heptylene	No	25339-56-4	
hexane	Yes	110-54-3	43231
hexane	Yes	110-54-3	43231
2-Methylpentane	No	107-83-5	43229
hexane	Yes	110-54-3	43231
Hexene	No	25264-93-1	43289
Indeno[1,2,3-cd]pyrene	Yes	193-39-5	46720
Isobutyraldehyde	No	78-84-2	43511
2-Methyl-1-propanol	No	78-83-1	43306
2-Methyl-1-propanol	No	78-83-1	43306
Isobutyraldehyde	No	78-84-2	43511
Isoheptanes (mixture)	No	31394-54-4	43106
2-Methylpentane	No	107-83-5	43229
2,2,4-Trimethylpentane	No	540-84-1	43250
2,2,4-Trimethylpentane	No	540-84-1	43250
Isopar E	No		
Isoprene	No	78-79-5	43243
2-Propanol	No	67-63-0	43304

Table 1. Event emissions in the Houston-Galveston area. Representative contaminants included in the analysis, selected because they are either HAPs or VOCs with more than 5 carbon atoms. (These data were retrieved directly from the original database, which included multiple redundancies that did not affect the analysis and were left intact.)

Contaminant	*HAP*	*CAS*	*SAROAD*
2-Propanol	No	67-63-0	43304
Cumene	Yes	98-82-8	45210
Isopropylcyclohexane	No	696-29-7	90128
Diisopropyl ether	No	108-20-3	85005
Kerosene	No	64742-81-0	
Methyl ethyl ketone	No	78-93-3	43552
Methyl isobutenyl ketone	Yes	141-79-7	
Methanol	Yes	67-56-1	43301
Methyl Acetylene	No	74-99-7	43209
Cresol	Yes	1319-77-3	45605
Methyl Chloride	Yes	74-87-3	43801
methyl cyclohexane	No	108-87-2	43261
Methyl ethyl ketone	No	78-93-3	43552
Iodomethane	No	74-88-4	86025
Methyl Mercaptan	No	74-93-1	43901
methyl cyclohexane	No	108-87-2	43261
Methylcyclopentane	No	96-37-7	43262
2-Methyldecane	No	6975-98-0	98155
Methylheptane	No	50985-84-7	90045
2-methylheptane	No	592-27-8	43296
2-Methyl nonane	No	871-83-0	90047
Tert-butyl methyl ether	No	1634-04-4	43376
meta-xylene	No	108-38-3	45205
Nonane	No	111-84-2	43235
Naphtha	No	8030-30-6	45101
Naphthalene	Yes	91-20-3	46701
Naphtha	No	8030-30-6	45101
Naphthalene	No	91-20-3	46701
Butyl acetate	No	123-86-4	43435
Butyraldehyde	No	123-72-8	43510
Nonane	No	111-84-2	43235
Nonane	No	111-84-2	43235
Octadecene	No	27070-58-2	
n-Octane	No	111-65-9	43233
Octene (mixed isomers)	No	25377-83-7	
ortho-xylene	No	95-47-6	45204
Parathion	Yes	56-38-2	
4-Aminohippuric Acid	No	61-78-9	
Phenol	Yes	108-95-2	45300
Silicone	No	63148-62-9	
Naphtha	No	8030-30-6	45101

Table 1. Event emissions in the Houston-Galveston area. Representative contaminants included in the analysis, selected because they are either HAPs or VOCs with more than 5 carbon atoms. (These data were retrieved directly from the original database, which included multiple redundancies that did not affect the analysis and were left intact.)

Contaminant	*HAP*	*CAS*	*SAROAD*
Naphtha	No	8030-30-6	45101
Polyethylene	No	9002-88-4	
Poly(Isobutylene)	No	9003-27-4	
Chloromethyl pivalate	No	18997-19-8	
Process fuel gas	No	none	
Propionic Acid	No	79-09-4	43405
Propylene oxide	No	75-56-9	43602
para-xylene	No	106-42-3	45206
Styrene	Yes	100-42-5	45220
Sulfolane	No	126-33-0	
t-Butyl Alcohol	No	75-65-0	43309
t-Butyl Alcohol	No	75-65-0	43309
tert-butyl hydroperoxide	No	75-91-2	
Toluene	Yes	108-88-3	45202
Aqualyte(TM), LSC cocktail	No	25551-13-7	45107
1,3,4-Trimethylbenzene	No	95-63-6	45208
trimethylcyclopentane	No	30498-64-7	98058
trimethylpentane	No	29222-48-8	90092
Undecane	No	1120-21-4	43241
Vinyl acetate	Yes	108-05-4	43453
Vinyl acetate	Yes	108-05-4	43453
Vinyl chloride	Yes	75-01-4	43860
vinyl resin	No	none	
Vinylcyclohexane	No	695-12-5	
xylenes	Yes	1330-20-7	45102
xylenes	Yes	1330-20-7	45102
meta-xylene	Yes	108-38-3	45205
ortho-xylene	Yes	95-47-6	45204
para-xylene	Yes	106-42-3	45206
Mineral spirits	No	64475-85-0	43118
Propylene glycol	No	57-55-6	43369
Vinyl chloride	Yes	75-01-4	43860
1-Decene	No	872-05-9	90014
2-Ethyl-1-hexanol	No	104-76-7	43318
2-Pyrrolidone	No	616-45-5	
Aromatic	No	none	
Decene	No	25339-53-1	90014
2-N,N-Dibutylaminoethanol	No	102-81-8	86007
Diisopropanolamine	No	110-97-4	86004
N,N-Dimethylethanolamine	No	108-01-0	84004
trifluoroethane	No	27987-06-0	

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Contaminant	*HAP*	*CAS*	*SAROAD*
2,2'-Oxybisethanol	No	111-46-6	43367
Hydrocarbons	No	none	
Methyl Formate	No	107-31-3	43430
Isopropylamine	No	75-31-0	86014
n-Butanol	No	71-36-3	43305
Polypropylene glycol ether	No		
N-Vinyl-2-Pyrrolidinone	No	88-12-0	
1,1-Di(t-Amylperoxy) Cyclohexane	No	15667-10-4	
1,2,3-Trimethyl-4-ethylbenzene	No	none	
2-Methyldecane	No	6975-98-0	98155
2-methylheptane	No	592-27-8	43296
2-Methyl nonane	No	871-83-0	90047
2,5-Dimethylhexane-2,5-dihydroperoxide	No	3025-88-5	
Butyl ether	No	142-96-1	43372
1,2-Dichloroethane	Yes	107-06-2	43815
Hydrindene	No	496-11-7	98044
Methylheptane	No	50985-84-7	90045
methyl methacrylate	No	80-62-6	43441
Naphtha	No	8030-30-6	45101
hexane	Yes	110-54-3	43231
tert-amyl hydroperoxide	No	3425-61-4	
1,3,4-Trimethylbenzene	No	95-63-6	45208
n-Butanol	No	71-36-3	43305
2-Butoxy ethanol	Yes	111-76-2	43308
hexane	Yes	110-54-3	43231
cycloheptane	No	291-64-5	43115
n-Heptane	No	142-82-5	43232
n-Octane	No	111-65-9	43233
Hexyl Carbitol	No	112-59-4	
Nonene	No	27215-95-8	
Silane, ethenyltrimethoxy	No	2768-02-7	
tetrahydrofuran	No	109-99-9	70014
Vinyl chloride	Yes	75-01-4	43860
Methyl Formate	No	107-31-3	43430
Phenyl ether	No	101-84-8	
phosgene	Yes	75-44-5	
1,2-Dichloroethane	No	107-06-2	43815
2-Butoxy ethanol	Yes	111-76-2	43308
Gasoline	No	86290-81-5	
1-Tridecanol	No	112-70-9	

Table 1. Event emissions in the Houston-Galveston area. Representative contaminants included in the analysis, selected because they are either HAPs or VOCs with more than 5 carbon atoms. (These data were retrieved directly from the original database, which included multiple redundancies that did not affect the analysis and were left intact.)

Contaminant	*HAP*	*CAS*	*SAROAD*
1,2,4-Trichlorobenzene	Yes	120-82-1	45208
2-(2-Butoxyethoxy)ethanol	Yes	112-34-5	43312
2,3,4-trihydroxybenzophenone Ester	No	1143-72-2	
Methyl n-amyl ketone	No	110-43-0	43562
4,4-Cyclohexylidenebis[phenol]	No	843-55-0	
Anisole	No	100-66-3	
2-Butoxy ethanol	Yes	111-76-2	43308
Cresol-Formaldehyde novolac Resin	No	proprietary	
Decane	No	124-18-5	43238
gamma-Butyrolactone	No	96-48-0	
Dimethyl pentane	No	38815-29-1	90063
Dodecyl Benzenesulfonic Acid	No	27176-87-0	
Ethanol Amine	No	141-43-5	43777
ethyl lactate	No	687-47-8	
Hexamethyldisilazane	No	999-97-3	
Methyl ethyl ketone	No	78-93-3	43552
Cresol	Yes	1319-77-3	45605
Naphthalene Sulfonic Acid Resin	No		
Naphthalene Sulfonic Acid Resin	No		
n-Butanol	No	71-36-3	43305
Decane	No	124-18-5	43238
1-Methyl-2-pyrrolidinone	No	872-50-4	70008
Pentyl Ester Acetic Acid	No		
Phenol Formaldehyde Resin, Novolac	No		
Phenol Formaldehyde Resin, Novolac	No		
Propylene Glycol Monomethyl Ether	No	107-98-2	70011
Pyrocatechol	No	120-80-9	
Carbon Disulfide	Yes	75-15-0	43934
Hexene	No	592-41-6	43245
VOC	No	none	
Methacrylic acid	No	79-41-4	84009
Methyl 3-hydroxybutyrate	No	1487-49-6	
t-Butyl Alcohol	No	75-65-0	43309
methyl valeraldehyde	No	123-15-9	
Butyl Methacrylate	No	97-88-1	85008
dipropyl ether	No	111-43-3	
n-Propanol	No	71-23-8	43303
Propyl propionate	No	106-36-5	86052

Table 1. Event emissions in the Houston-Galveston area. Representative contaminants included in the analysis, selected because they are either HAPs or VOCs with more than 5 carbon atoms. (These data were retrieved directly from the original database, which included multiple redundancies that did not affect the analysis and were left intact.)

Contaminant	*HAP*	*CAS*	*SAROAD*
1,2-Epoxybutane	Yes	106-88-7	
Methylamine	No	74-89-5	
1,1-Dimethylcyclohexane	No	590-66-9	
1,1-Dimethylcyclopentane	No	1638-26-2	
2-Methylpentane	No	107-83-5	43229
dimethyl butane	No	75-83-2	43291
2,3,3-Trimethylpentane	No	560-21-4	
2,3-Dimethylhexane	No	584-94-1	
2,3-Dimethylpentane	No	565-59-3	
2,4-Dimethylhexane	No	589-43-5	
2,5-Dimethyl-hexane	No	592-13-2	
2-Butoxy ethanol	Yes	111-76-2	43308
2-mercaptoethanol	No	60-24-2	
Bisphenol A	No	80-05-7	
straight-run middle distillate	No	64741-44-2	
4-Vinylcyclohexene	No	100-40-3	
straight-run middle distillate	No	64741-44-2	
Allyl alcohol	No	107-18-6	
xylenes	Yes	1330-20-7	45102
Naphthalene	Yes	91-20-3	46701
3-Methylethylcyclohexane	No		
VOC	No	none	
Gasoline	No	86290-81-5	
Butyl ether	No	142-96-1	
dimethyl butane	No	75-83-2	
Dodecene	No	25378-22-7	
Styrene	Yes	100-42-5	45220
tetrahydrofuran	No	109-99-9	70014
hexane	Yes	110-54-3	43231
2-Propanol	No	67-63-0	43304
liquified petroleum gas	No	68476-85-7	
Methyl acetylene propadiene	No		
methyl isobutyl ketone	Yes	108-10-1	
Methyl n-amyl ketone	No	110-43-0	43562
Methylpentane	No	43133-95-5	
Tert-butyl methyl ether	Yes	1634-04-4	43376
Toluene	Yes	108-88-3	45202
Mineral oil	No	8012-95-1	
Gasoline	No	86290-81-5	
2,2-Dimethylpropane	No	463-82-1	43222
n-propylbenzene	No	103-65-1	

Table 1. Event emissions in the Houston-Galveston area. Representative contaminants included in the analysis, selected because they are either HAPs or VOCs with more than 5 carbon atoms. (These data were retrieved directly from the original database, which included multiple redundancies that did not affect the analysis and were left intact.)

Contaminant	*HAP*	*CAS*	*SAROAD*
propylcyclohexane	No	1678-92-8	
n-Octane	No	111-65-9	43233
ortho-xylene	No	95-47-6	45204
Gasoline	No	86290-81-5	
propylenimine	No	75-55-8	
Gasoline	No	86290-81-5	
Technical White Oil	No		
Total Alkylate - non-speciated	No		
Trichloroethylene	Yes	79-01-6	
Di(2-ethylhexyl) peroxydicarbonate	No	16111-62-9	
trimethylcyclopentane	No	30498-64-7	98058
Ultraformate	No		
4-Vinylcyclohexene	No	100-40-3	

Table 2. Event emissions in the Houston-Galveston area. Major emitters reporting at least one release event of a representative substance.	
Company Name	*2004 VOC Emission Rate (lb/h)*
ATOFINA PETROCHEMICALS LA PORTE PLANT	47.88
BALL METAL BEVERAGE CONTAINER CONROE FACILITY	24.18
BASF FREEPORT SITE	46.47
BELVIEU ENVIRONMENTAL FUELS	112.3
BOC GROUP CLEAR LAKE BOC GASES PLANT	9.52
BP AMOCO CHEMICAL CHOCOLATE BAYOU PLANT	130.4
BP AMOCO CHEMICAL PASADENA PLANT	36.92
BP AMOCO POLYMERS	57.18
BP PRODUCTS NORTH AMERICA TEXAS CITY	737.4
BP TEXAS CITY CHEMICAL PLANT B	112.2
CELANESE BAY CITY PLANT	17.12
CELANESE CLEAR LAKE PLANT	53.11
CELANESE PASADENA PLANT	5.934
CHEVRON PHILLIPS CEDAR BAYOU PLANT	105.3
CHEVRON PHILLIPS CHEMICAL SWEENY COMPLEX	106.7
CHEVRON PHILLIPS HOUSTON CHEMICAL COMPLEX	215.7
CROWN BEVERAGE PACKAGING	18.05
CROWN CENTRAL PETROLEUM PASADENA PLANT	114.3
CROWN CORK & SEAL	18.10
DEER PARK LIQUID STORAGE TERMINAL	124.8
DOW CHEMICAL LA PORTE SITE	5.902
DOW TEXAS OPERATIONS FREEPORT	203.2
E I DUPONT DE NEMOURS AND COMPANY - LA PORTE PLANT	51.30
EQUISTAR CHEMICALS CHANNELVIEW COMPLEX	275.4
EQUISTAR CHEMICALS CHOCOLATE BAYOU COMPLEX	84.87
EQUISTAR CHEMICALS LA PORTE COMPLEX	90.97
EXXON MOBIL CHEMICAL BAYTOWN OLEFINS PLANT	84.73
EXXONMOBIL CHEMICAL BAYTOWN CHEMICAL PLANT	313.7
EXXONMOBIL CHEMICAL MONT BELVIEU PLASTICS PLANT	40.64
GOODYEAR HOUSTON CHEMICAL PLANT	85.68
ISP TECHNOLOGIES TEXAS CITY PLANT	22.12
KANEKA TEXAS CORPORATION	20.55
KINDER MORGAN LIQUID TERMINALS PASADENA	913.9
KINDER MORGAN LIQUIDS TERMINALS	132.7
LBC HOUSTON BAYPORT TERMINAL	12.83
LYONDELL CHEMICAL BAYPORT PLANT	30.04
LYONDELL CHEMICAL CHANNELVIEW	74.15
MARATHON ASHLAND PETROLEUM TEXAS CITY REFINERY	111.8
MOBIL CHEMICAL HOUSTON OLEFINS PLANT	26.29
MORGANS POINT PLANT	31.03
PASADENA PLANT	13.40

Table 2. Event emissions in the Houston-Galveston area. Major emitters reporting at least one release event of a representative substance.

Company Name	*2004 VOC Emission Rate (lb/h)*
SHELL OIL DEER PARK	405.2
SOLUTIA CHOCOLATE BAYOU PLANT	53.09
STOLTHAVEN HOUSTON TERMINAL	7.347
SWEENY COMPLEX	157.1
UNION CARBIDE TEXAS CITY OPERATIONS	174.4
VALERO REFINING TEXAS CITY	260.1
WHARTON GAS PLANT	7.552

Table 3. Frequency distribution for ratio of event emission rate to long-term emission rate

Bin	*Frequency*	*Cumulative Frequency*
1.00E-08	0	0
3.16E-08	0	0
1.00E-07	2	2
3.16E-07	1	3
1.00E-06	0	3
3.16E-06	2	5
1.00E-05	1	6
3.16E-05	2	8
1.00E-04	5	13
3.16E-04	9	22
1.00E-03	15	37
3.16E-03	28	65
1.00E-02	33	98
3.16E-02	41	139
1.00E-01	59	198
3.16E-01	38	236
1.00E+00	33	269
3.16E+00	31	300
1.00E+01	16	316
3.16E+01	2	318
1.00E+02	1	319
3.16E+02	0	319

Table 4. Statistics for ratio of event emission rate to long-term emission rate	
Statistic for Ratio	*Value*
Median	0.043923
75th %ile	0.342655
90th %ile	2.204754
95th %ile	3.344422
96th %ile	3.400832
97th %ile	3.8126
98th %ile	4.790098
99th %ile	8.973897
Max	74.37138
Average	0.815352

Figure 1. Cumulative probability density for ratio of event to routine emission rates.

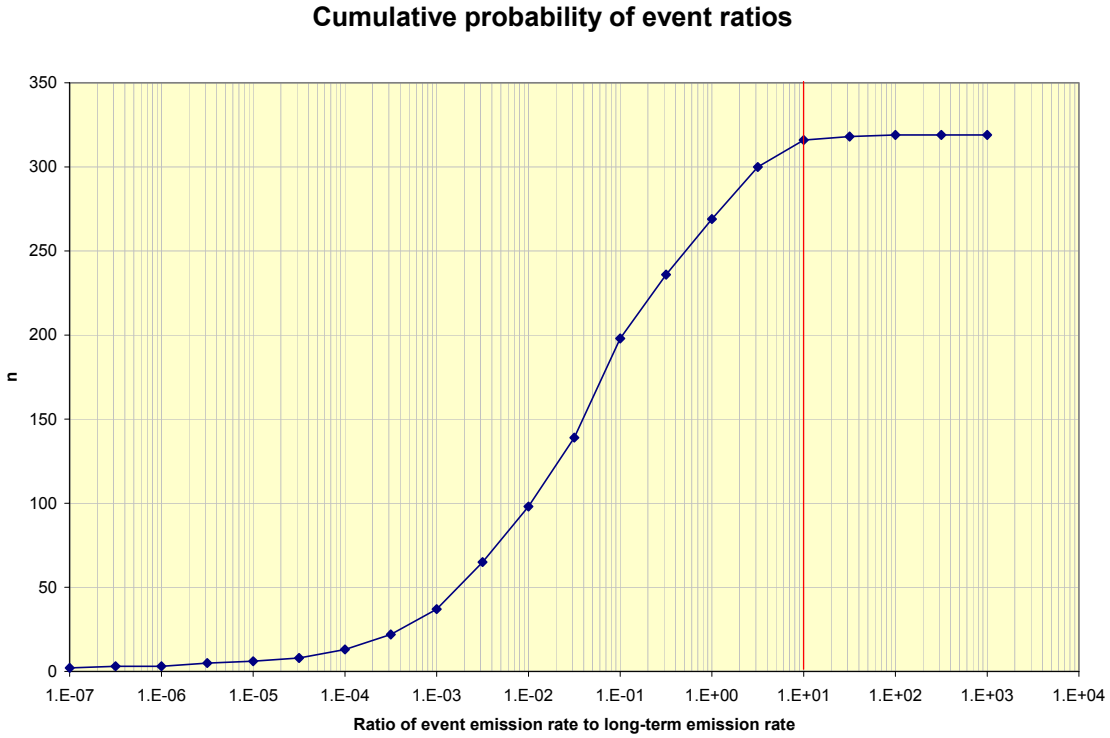


Figure 2. Relationship between ratio of event to duration emission rate and emission duration.

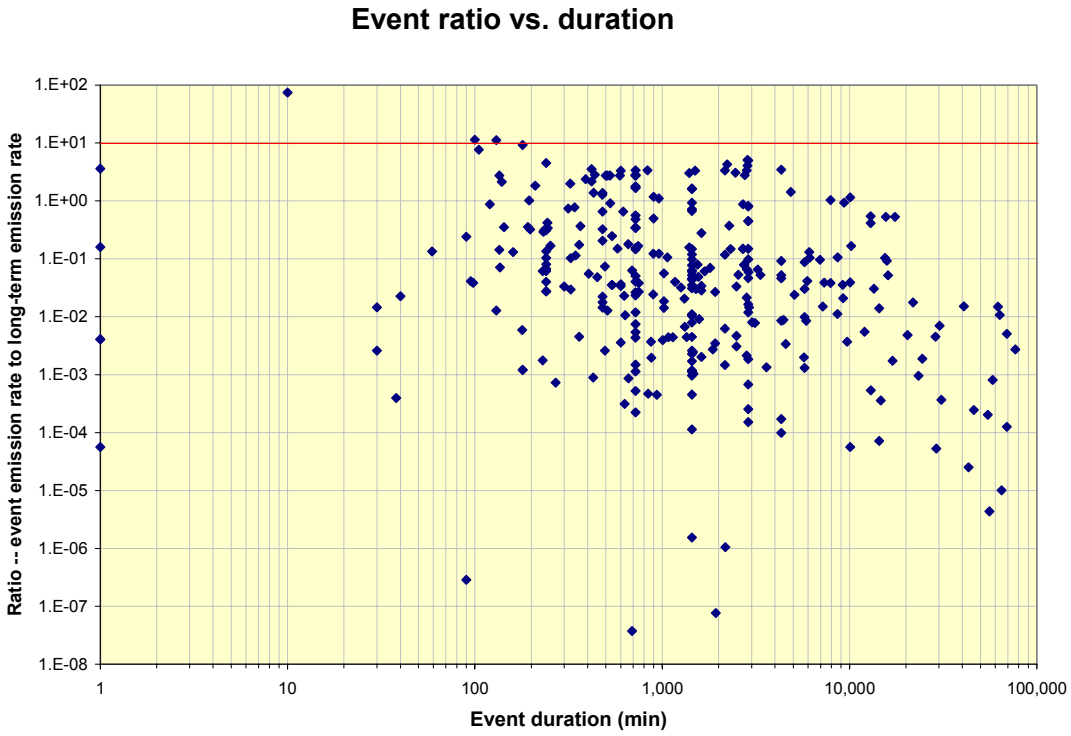


Figure 3. Relationship between ratio of event to duration emission rate and emission duration, as percentage of total time.

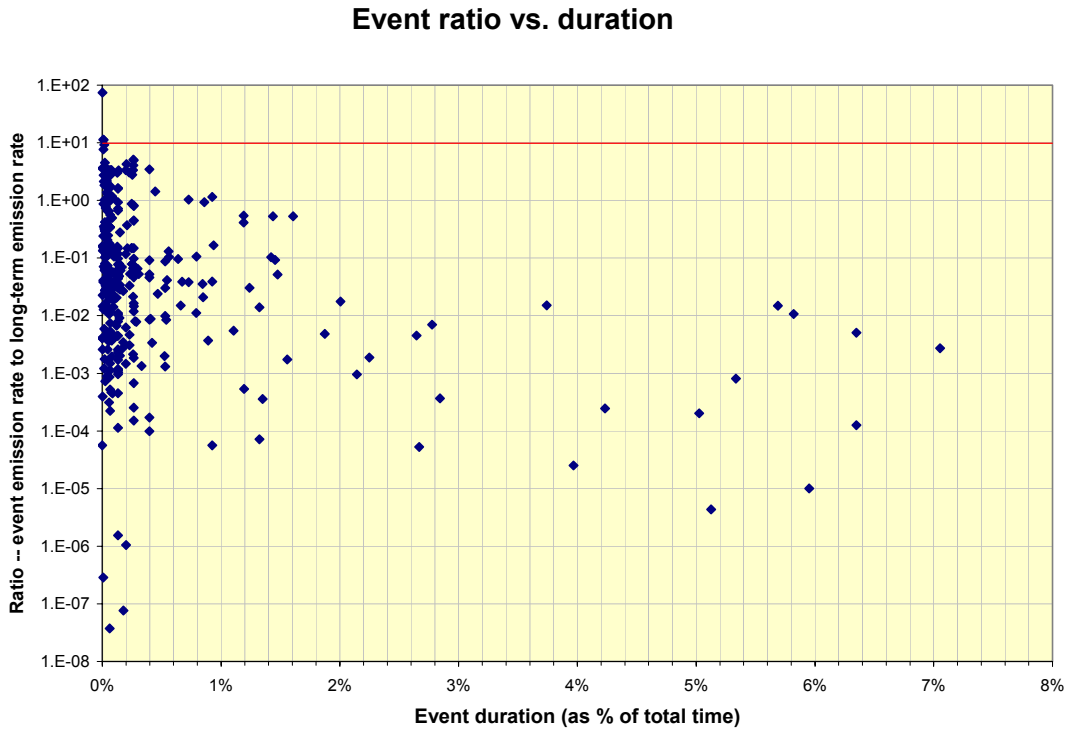
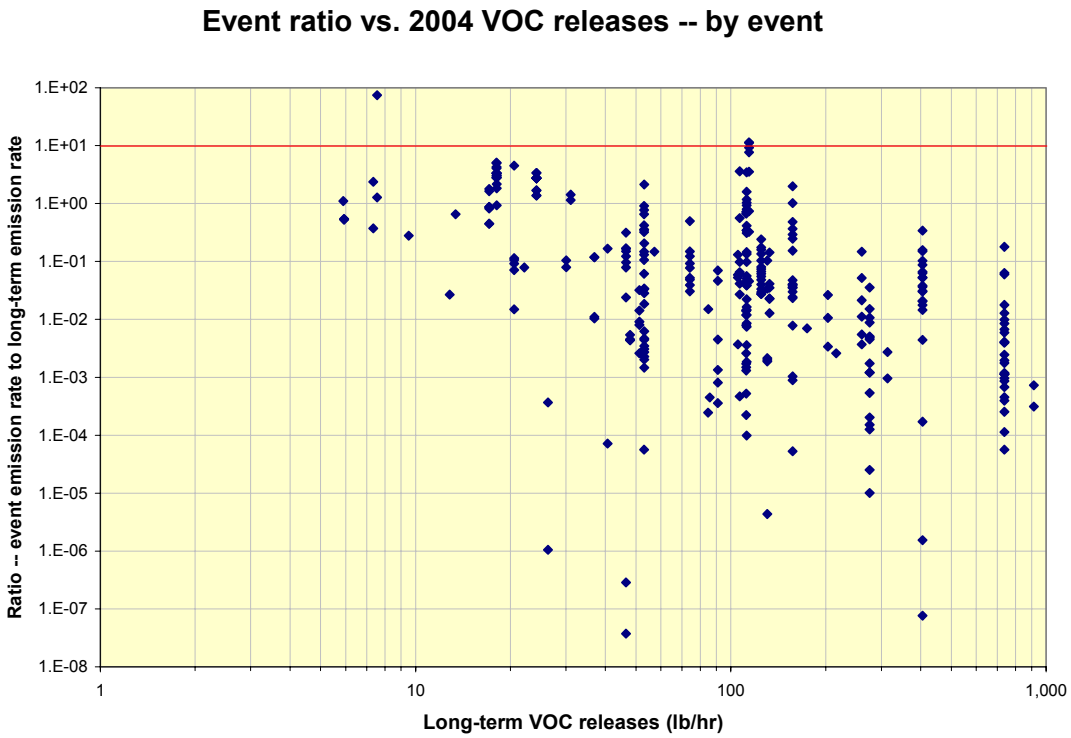


Figure 4. Relationship between ratio of event to duration emission rate and total amount emitted during the event.



Appendix 6
Technical Support Document for the TRIM-Based Multipathway Tiered Screening Methodology for RTR

Technical Support Document for the TRIM-Based Multipathway Tiered Screening Methodology for RTR

July 2018

Prepared For:

U.S. Environmental Protection Agency
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Contents

Exhibits v

Acronyms vii

1. Introduction 1
   - 1.1 Purpose of RTR Multipathway Screens 1
   - 1.2 Overview of Multimedia Ingestion Screening Methods 2
   - 1.3 Tiered Approach 3
   - 1.4 Chemicals of Potential Concern 5
   - 1.5 Report Organization 7

2. Tier 1 Screen 7
   - 2.1 Overview of Tier 1 7
   - 2.2 Conceptual Exposure Scenario 11
     - 2.2.1 Approach to Development of the Tier 1 Scenario 12
     - 2.2.2 Fate and Transport Modeling (TRIM.FaTE) 14
     - 2.2.3 Exposure Modeling and Risk Characterization 15
     - 2.2.4 Implementation of Risk-based Equivalency Factors for POM and Dioxin Congeners 16
   - 2.3 Description of Environmental Modeling Scenario 22
     - 2.3.1 Chemical Properties 22
     - 2.3.2 Spatial Layout 22
     - 2.3.3 Watershed and Water Body Parameterization 26
     - 2.3.4 Meteorology 29
     - 2.3.5 Aquatic Food Web 31
     - 2.3.6 Using TRIM.FaTE Media Concentrations 32
   - 2.4 Description of Human Exposure and Risk Estimates 33
     - 2.4.1 Calculating Concentrations in Farm Food Chain Media 34
     - 2.4.2 Ingestion Exposure 34
     - 2.4.3 Calculating Risk 37
     - 2.4.4 Summary of Tier 1 Assumptions 38
   - 2.5 Evaluation of Screening Scenario 42
     - 2.5.1 Arsenic 43
     - 2.5.2 Cadmium Compounds 46
     - 2.5.3 Mercury Compounds 48
     - 2.5.4 Dioxins 50
     - 2.5.5 Polycyclic Aromatic Hydrocarbons and Other Polycyclic Organic Matter 53
     - 2.5.6 Summary 56

3. Tier 2 Screen 56
   - 3.1 Overview of Approach 57
     - 3.1.1 Tier 2 Environmental Assumptions 57
     - 3.1.2 Tier 2 Exposure Assumptions 57

3.1.3 Implementation of Tier 2 58
3.2 Library of Tier 2 Screening Threshold Emission Rates 60
3.2.1 Meteorological Data 61
3.2.2 Locations of Lakes and Farms/Gardens 63
3.2.3 Gardener Exposure Scenario 66
3.2.4 Development of Library of Tier 2 Screening Threshold Emission Rates, REFs, and Mixing Height Refinements 67
3.3 Implementing the Tier 2 Multipathway Screen 68
3.3.1 Accounting for Sustainable Fishing 71
3.3.2 Refined-fisher Scenario 75
3.3.3 Considering Inhalation Risks at Hypothetical Garden Locations 76
3.3.4 Outputs 77
4. Tier 3 Screen 78
4.1 Overview of Approach 79
4.2 Lake Screen 79
4.3 Farmer Scenario Evaluation 81
4.4 Gardener Scenario Evaluation 81
4.5 Plume-rise Screen 82
4.6 Screen Using Hourly Time-series Meteorological Data and Effective Release Heights 83
5. References 84
Attachment A. TRIM.FaTE Inputs A-1
Attachment B. Multimedia Ingestion Risk Methodology Used for RTR Exposure and Risk Estimates ..... B-i
Attachment C. Dermal Risk Screening C-1
Attachment D. Summary of TRIM.FaTE Parameters Considered for Inclusion in Tier 2 Assessment D-1
Attachment E. Analysis of Lake Size and Sustainable Fish Population E-1

EXHIBITS

Exhibit 1. Overview of Ingestion Exposure and Risk Screening Evaluation Method 2

Exhibit 2. Conceptual Decision Tree for Evaluating Non-Inhalation Exposures for PB-HAPs 4

Exhibit 3. OAQPS PB-HAP Compounds 6

Exhibit 4. Conceptual Decision Tree for Tier 1 Evaluation of Multimedia Ingestion Exposures to PB-HAPs 10

Exhibit 5. Screening Threshold Emission Rates for Multimedia Ingestion Exposures 10

Exhibit 6. Overview of Multimedia Ingestion Risk Calculations for RTR 15

Exhibit 7. Exposure, Toxicity, and Risk Equivalency Factors Relative to BaP for POM Congeners Currently Evaluated in the Screens 18

Exhibit 8. Relationship between Ingestion Exposure and K_{ow} for POM Chemicals 19

Exhibit 9. LogK_{ow} Values for POM Chemicals Congeners 20

Exhibit 10. Exposure and Toxicity Equivalency Factors Relative to TCDD for Modeled Dioxin Congeners 21

Exhibit 11. TRIM.FaTE Surface Parcel Layouts 24

Exhibit 12. Summary of Key Meteorological Parameter Inputs 29

Exhibit 13. Aquatic Biota Parameter Values for the TRIM.FaTE Screening Scenario 32

Exhibit 14. Spatial Considerations—TRIM.FaTE Results Selected for Calculating Farm Food Chain and Fish Media Concentrations and Receptor Exposures 33

Exhibit 15. Summary of Ingestion Exposure Pathways 34

Exhibit 16. Overview of Exposure Factors Used for RTR Tier 1 Ingestion Screen 35

Exhibit 17. Dose-response Values for PB-HAPs in RTR Ingestion Screening Scenario 38

Exhibit 18. Summary of RTR Tier 1 Screening Scenario Assumptions 39

Exhibit 19. Estimated Media Contributions to Arsenic Ingestion Exposures and Lifetime Cancer Risks 46

Exhibit 20. Estimated Media Contributions to Cadmium Ingestion Exposures and HQs 48

Exhibit 21. Estimated Media Contributions to Methyl Mercury Ingestion Exposures and HQ 50

Exhibit 22. Estimated Media Contributions to Dioxin Ingestion Exposures 52

Exhibit 23. Estimated Media Contributions to POM Ingestion Exposures 55

Exhibit 24. Estimated Contributions of Modeled Food Types to Additional POM Chemical Ingestion Exposures 56

Exhibit 25. Basic Process for Implementing the Tier 2 Multipathway Screen 60

Exhibit 26. The Locations of Meteorological Stations Used in RTR Modeling, and Locations of NATA 2011 Point-Source Facilities for Reference 62

Exhibit 27. Values for Meteorological Parameters Used to Develop the Tier 2 Screening Threshold Emission Rates and REFs 63

Exhibit 28. Distances Used to Develop the Tier 2 REFs and Screening Threshold Emission Rates 64

Exhibit 29. TRIM.FaTE Surface Layouts for the Tier 2 Multipathway Screen, Using Alternative Distances Between the Facility and the Fishable Lake or Farm/Garden (base layout is the same as Tier 1 shown in Exhibit 11 and not shown here) 65

Exhibit 30. Ingested Media for Farmer and Gardener Scenarios 66

Exhibit 31. Estimated Maximum Fish-ingestion Rate (g/d) Associated with Sustainable Fishing 74

Exhibit 32. Example of Source Category Summary Results Output from Tier 2 Tool 77

Exhibit 33. Example of Facility-level Results Output from Tier 2 Tool 78

Exhibit 34. Example of the Refined-fisher Output for Facility and PB-HAP from Tier 2 Tool 78

Exhibit 35. Example of Lake Removed from Screening—Likely Evaporated or Drained 80

Exhibit 36. Example of Lake Removed from Screening—Likely an Industrial Lake 81

ACRONYMS

3MRA	Multimedia, Multipathway, and Multireceptor Risk Assessment Modeling System
ADAF	age-dependent adjustment factor
ADD	average daily dose
AERMOD	American Meteorological Society/Environmental Protection Agency Regulatory Model
AT	averaging time
BAF	bioaccumulation factor
BaP	benzo[a]pyrene
BC	benthic carnivore (fish)
BCF	bioconcentration factor
BI	benthic invertebrate
BMF	biomagnification factor
BO	benthic omnivore (fish)
BSAF	biota-sediment accumulation factor
BW	body weight
CSF	cancer slope factor
DDE	dichlorodiphenyldichloroethylene
ED	exposure duration
EEF	exposure equivalency factor
EF	exposure frequency
EPA	U.S. Environmental Protection Agency
ER	emission rate
ESRI	Environmental Systems Research Institute
FC	fraction contaminated
HAP	hazardous air pollutant
HHRAP	Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities
HQ	hazard quotient
IR	ingestion rate
IRIS	Integrated Risk Information System
LADD	lifetime average daily dose
MACT	maximum achievable control technology
MVP	minimum viable population
NAAQS	National Ambient Air Quality Standards
NATA	National Air Toxics Assessment
NCDC	National Climatic Data Center
NEI	National Emissions Inventory
OAQPS	Office of Air Quality Planning and Standards (U.S. EPA)
ORD	Office of Research and Development (U.S. EPA)
PAH	polycyclic aromatic hydrocarbon
PB	persistent and bioaccumulative
PB-HAP	persistent and bioaccumulative hazardous air pollutant
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzodioxin
PCDF	polychlorinated dibenzofuran
POM	polycyclic organic matter
REF	risk equivalency factor
RfD	reference dose
RGP	reactive gaseous phase

RTR	Risk and Technology Review
RZ	root zone
SAB	Science Advisory Board
SV	screening value
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin, termed “dioxin” in this report
TEF	toxic equivalency factor
TL	trophic level
TL2	trophic level 2
TL3	trophic level 3
TL3.5	between trophic level 3 and 4
TL4	trophic level 4
TPY	short tons per a year
TRIM	Total Risk Integrated Methodology
TRIM.FaTE	TRIM’s Fate, Transport, and Ecological Exposure module
TSD	Technical Support Document
USGS	U.S. Geological Survey
USLE	universal soil loss equation
WBAN	Weather Bureau-Army-Navy
WCC	water-column carnivore (fish)
WCH	water-column herbivore (fish)
WCO	water-column omnivore (fish)

1. Introduction

Section 112 of the Clean Air Act (CAA) directs the U.S. Environmental Protection Agency (EPA) to assess the risk remaining (i.e., residual risk) from emissions of hazardous air pollutants (HAPs) following the implementation of maximum achievable control technology (MACT) standards for emission sources. These risk assessments are a major component of EPA's Risk and Technology Review (RTR) program. One aspect of human health that EPA must consider under RTR is the potential for health effects resulting from exposures to HAPs via non-inhalation pathways, namely ingestion and dermal exposure. These non-inhalation human health risks are considered in combination with estimated inhalation human health risks, potential ecological impacts, and other factors to support RTR decisions. This report documents the technical bases and methods used for RTR non-inhalation human health risk screens.

This section introduces the reader to the Total Risk Integrated Methodology (TRIM)-Based Multipathway Tiered Screening Methodology. It describes the purpose of the RTR program and this Technical Support Document (TSD, Section 1.1) and provides an overview of the multipathway screening approach (Section 1.2). This section also provides an overview of the tiered implementation of the screen (Section 1.3), the chemicals that are evaluated in the RTR mulipathway screen (Section 1.4), and the organization of the remainder of the TSD (Section 1.5). The subsequent main sections, 2 through 4 of this report, describe Tiers 1 through 3 of the screen in greater detail. References are listed in Section 5.

1.1 Purpose of RTR Multipathway Screens

As noted above, Section 112 of the CAA directs EPA to assess the residual risk from emissions of HAPs following the implementation of MACT standards for emission sources. Facilities are grouped into source categories, and each source category is evaluated independently. As part of this program, EPA considers additional emission controls for a source category if the current MACT does not provide an "ample margin of safety" to protect human health.

EPA's Office of Air Quality Planning and Standards (OAQPS) has identified specific persistent and bioaccumulative HAPs (PB-HAPs) for which it must consider all possible routes of exposure—inhalation, ingestion, and dermal. EPA must evaluate potential ingestion and dermal exposures to PB-HAPs deposited from air to ground-level surfaces, considering subsequent transport and fate of those chemicals in the environment. For PB-HAPs, exposures via ingestion have been shown to be much higher than exposures via dermal absorption (see Attachment C).

EPA OAQPS has developed an iterative, tiered approach to screen exposure and risk specifically for multimedia ingestion of PB-HAPs for its RTR program. The iterative, tiered screening approach described in this document allows EPA to efficiently gauge the largest potential exposures and health risks from non-inhalation exposure to emitted PB-HAPs in a source category. If the conclusion of a screen is that exposures and health risks above levels of concern cannot be ruled out, EPA can conduct refined, complex, site-specific modeling of potential risks (which is not discussed in this document).

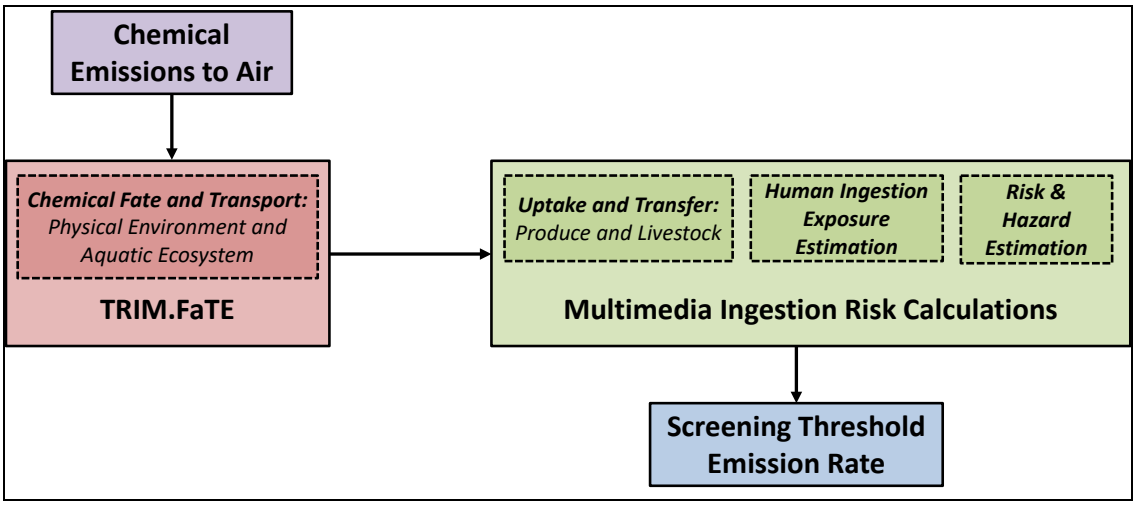
EPA evaluates human inhalation exposures to HAPs separately using other tools. For each source category, EPA OAQPS considers risks to humans from ingestion exposures along with risks from inhalation, potential ecological and other environmental impacts, and other factors when deciding if regulatory action is needed.

1.2 Overview of Multimedia Ingestion Screening Methods

The screening approach and tools are summarized in Exhibit 1 and described below:

1. We use TRIM.FaTE—the Fate, Transport, and Ecological Exposure module of TRIM—to model fate and transport of air emissions of PB-HAPs using a base emission rate of 1 g/d. This modeling includes chemical partitioning across soil, water, and other environmental media (including fish). TRIM.FaTE outputs include chemical concentrations in fish (mg/kg wet weight), soil (µg/g dry weight), and water (mg/L), and deposition rates for chemicals from air via wet and dry deposition.

Exhibit 1. Overview of Ingestion Exposure and Risk Screening Evaluation Method



2. We use TRIM.FaTE outputs (e.g., chemical air deposition and environmental media concentrations) as inputs to multimedia ingestion risk calculations that include ingestion of PB-HAPs in locally raised foods (e.g., produce, livestock, and dairy products). The multimedia risk estimation methods are based on EPA's *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (U.S. EPA 2005a).
3. The calculated chemical concentrations in the ingested media, along with food ingestion rates and other exposure factors, are used to estimate ingestion exposures from the selected media for hypothetical human receptors. Specifically, estimates are made of average daily doses (ADDs) for the non-carcinogenic chemicals assessed (i.e., for cadmium and mercury) and lifetime ADDs (LADDs) for the carcinogenic chemicals (i.e., for arsenic, dioxins/furans [abbreviated in this document as dioxins]), and polycyclic organic matter [POM]).
4. Chemical-specific lifetime cancer risk or chronic noncancer hazard (expressed as a hazard quotient [HQ]) are estimated for each PB-HAP at a modeled emission rate of 1 g/d.
5. For each PB-HAP, based on the estimated cancer risk or HQ at the 1 g/d emission rate, we determine the emission rate at which the excess lifetime cancer risk equals 1-in-one million or the non-cancer HQ equals 1. These emission rates are termed "screening threshold emission rates".
6. We then compare a facility's PB-HAP emission rate to the screening threshold emission rate for each PB-HAP emitted (e.g., a facility's actual cadmium emission rate would be compared

to the screening threshold emission rate for cadmium). The resulting ratio of a facility's actual emission rate to the screening threshold emission rate is termed a "screening value" or SV.

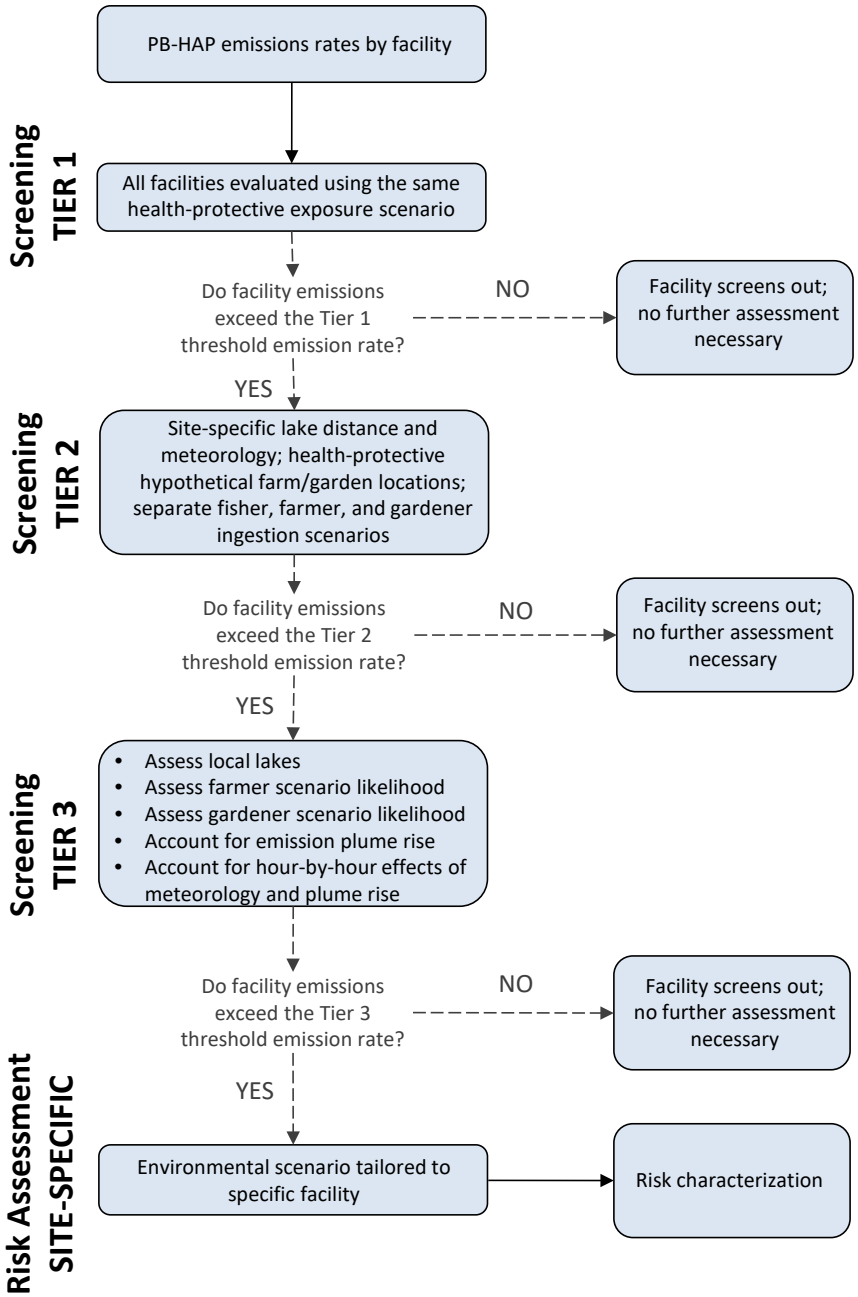
1.3 Tiered Approach

EPA developed the tiered approach to screen out PB-HAP emissions unlikely to pose health risks above levels of concern, allowing the Agency focus on facilities and chemicals of greatest concern within a source category. Sensitivity analyses and model testing revealed that the spatial layout of the modeled domain (e.g., distance to a fishable lake) and the meteorological data used (or a combination of these two factors) influence estimated chemical concentrations in air, soil, water, sediment, and fish more than physical/chemical parameter values of the PB-HAPs. As discussed in detail below, the Tier 1 assumptions about meteorological data and lake location are refined with more site-specific data in subsequent tiers. In addition, if a facility does not screen out, we further evaluate the surrounding land use to determine if the exposure scenario is realistic, and if not, we remove the exposure scenario from evaluation. For example, if a farmer scenario does not screen out, and we determine that exposures are in an urban setting and it is unlikely that a full-scale farming operation will operate in the vicinity, we will remove the farmer exposure scenario. The iterative approach is divided into three tiers of increasing refinement as illustrated in Exhibit 2 and described below.

- Tier 1 compares facility-specific PB-HAP emissions to the screening threshold emission rates for a hypothetical scenario in which an individual eats locally caught fish; consumes only home grown produce, livestock, and livestock products (e.g., eggs, meat, dairy products); and incidentally ingests local soil. The ingestion rate for each ingested medium was set to an upper percentile value. This approach overestimates total chemical exposure for a single hypothetical individual, but it will not miss an important exposure pathway. The screening scenario represents a "worst-case" ingestion exposure that is unlikely to be exceeded at any actual facility evaluated for the RTR program. For a facility, if the emission rate of each PB-HAP is less than the Tier 1 screening threshold emission rate (i.e., if the SVs are less than or equal to 2, when rounded to one significant figure), no additional multipathway screening is done. If, however, the emission rate of any PB-HAP exceeds the Tier 1 screening threshold emission rate (i.e., an SV of 2 or more, when rounded to one significant figure), the facility can be evaluated further in Tier 2.
- In Tier 2, the actual location of each modeled facility is used to refine some assumptions associated with the environmental scenario. Combinations of meteorological conditions, lake locations, and farming locations were systematically varied, and TRIM.FaTE and multimedia exposure algorithms were used to calculate screening threshold emission rates for PB-HAPs for each combination, or bin (see Section 3.2). For each facility, an algorithm identifies the predefined bin that most closely matches the local weather conditions and relative location(s) of fishable lakes for that facility. Multiple hypothetical farming locations also are evaluated for each facility. The facility's emissions are compared to the screening threshold emission rates for the best-match bin for each PB-HAP to determine SVs. Unlike Tier 1, which considers combined ingestion of fish, farm foods, and soil, Tier 2 separately screens a hypothetical person consuming fish and a hypothetical person consuming farm foods and soil (i.e., the Tier 1 ingestion scenario is disaggregated into separate hypothetical subsistence farmer and subsistence fisher exposure scenarios). In addition, a gardener exposure scenario is added in Tier 2 to represent exposures to individuals who garden and eat eggs from home-raised chickens, but who do not raise animals

for meat or dairy ingestion. Hypothetical gardeners and hypothetical farmers are evaluated at identical locations (and thus use the same calculated media concentrations). The gardener's ingestion rates for consumed media depend upon whether or not the facility is located in an urban or rural area (lower ingestion rates are assumed for gardeners in urban areas compared with rural areas, see: Section 3.2.3). If the resulting SVs are all less than 2 (when rounded to one significant figure), no additional screening is needed. Facilities with SVs greater than or equal to 2 (when rounded to one significant figure) for one or more PB-HAPs, for any of the exposure scenarios, can be further analyzed in Tier 3.

Exhibit 2. Conceptual Decision Tree for Evaluating Non-Inhalation Exposures for PB-HAPs



- In Tier 3, further site-specific refinements are included in the screen.
  - To further evaluate a fisher scenario SV exceedance (i.e., an SV of 2 or greater), nearby lakes are examined more closely for suitability for fishing; unsuitable lakes are removed from the lake database, and the facility is rescreened (using Tier 2 methods) with the revised lake database.
  - To further evaluate a farmer SV exceedance, EPA uses census data, aerial imagery, and other available data to further assess the likelihood of subsistence farmer operations within 50 km of the facility. If, based on the additional analysis and review, it is determined that no subsistence farming operations are in the area, then the farmer scenario is not used in Tier 3 and only gardener SVs are reported.
  - To further evaluate a gardener exceedance, EPA will examine information such as Census data, aerial imagery, and land-use data to determine the likelihood that people reside at the location of the gardener exceedance. If EPA determines that people likely reside there, then the Tier 2 gardener SV will be retained in Tier 3. Otherwise, EPA will report the next-highest gardener SV at a location where EPA determines that people likely reside.
  - Each of the next two refinements (i.e., plume-rise and hourly weather data) can result in different Tier 3 screening threshold emission rates. Facilities having emissions that exceed the refined screening threshold emission rates for Tier 3 (i.e., SVs are 2 or more as described previously) may require additional analysis.

If, based on results of the screens, a risk assessor concludes that there is a reasonable probability that individual humans could be adversely affected by the facility emissions, a refined site-specific multipathway assessment can be performed. The land parcels are defined using geographic features around a facility that define the magnitude of runoff and erosion. The lake parcels follow the general shapes of the actual lakes. Important site-specific data likely would include emission release height and plume buoyancy, hourly meteorology (e.g., wind flow, temperature, mixing height, and precipitation), surface compartments based on watershed and terrain data, location of local farms/gardens and water bodies, types of land use, soil properties, erosion and runoff rates with slope features, surface water and sediment properties, water transfer rates, and aquatic ecosystem information. If available, other site-specific information could be included (e.g., crops grown, local fish ingestion rates, typical growing season).

1.4 Chemicals of Potential Concern

EPA's assessment of multipathway human exposures for RTR focuses on PB-HAPs[1] that OAQPS has identified as candidates for multipathway risk assessments. OAQPS developed a list of 14 chemicals and chemical groups considered to be PB-HAPs using two criteria:

- Their presence on three EPA lists of persistent, bioaccumulative, and toxic substances, and
- A semiquantitative ranking of toxicity and bioaccumulation potential of the entire list of HAPs.

[1]Although POM (polycyclic organic matter) is the name of the HAP listed in the Clean Air Act, most of the POM chemicals are "polycyclic aromatic hydrocarbons" or PAHs.

The list's development and utility in hazard identification for multipathway risk assessment are explained further in Chapter 14 and Appendix D of Volume I of EPA's *Air Toxics Risk Assessment (ATRA) Reference Library* (U.S. EPA 2004a). Exhibit 3, below, presents the 14 PB-HAP chemicals and chemical groups, with the addition of arsenic, which was not in the original list (see Exhibit endnote). TRIM.FaTE is not configured to evaluate risk for all PB-HAPs on the list. Currently, TRIM.FaTE includes chemical-specific parameter values required for modeling exposure and risk for four of the 14 PB-HAPs (as indicated in Exhibit 3) plus arsenic. These five PB-HAPs are the focus of the RTR multimedia screen because, based on current emissions, bioaccumulation potential, and toxicity considerations, they are expected to pose the vast majority of the non-inhalation risks to humans from air emissions at sources subject to residual risk provisions of the Clean Air Act.[2]

The five PB-HAPs assessed under RTR include:

- Arsenic compounds,
- Cadmium compounds,
- Chlorinated dibenzodioxins and furans (dioxins),
- Mercury compounds, and
- Polycyclic organic matter (POM[3]).

Exhibit 3. OAQPS PB-HAP Compounds

PB-HAP Compound[a]	Addressed by Screening Scenario?
Arsenic[b]	Yes
Cadmium compounds	Yes
Chlordane	No
Chlorinated dibenzodioxins and furans	Yes
DDE (1,1-dichloro-2,2-bis(p-chlorophenyl) ethylene)	No
Heptachlor	No
Hexachlorobenzene	No
Hexachlorocyclohexane (all isomers)	No
Lead compounds	No
Mercury compounds	Yes
Methoxychlor	No
Polychlorinated biphenyls	No
Polycyclic organic matter (POM)	Yes

[2]Potential impacts on human health from non-inhalation exposures to lead are evaluated for RTR using the National Ambient Air Quality Standard (NAAQS) for lead, which takes into account multipathway risks. Non-inhalation exposures to the other nine PB-HAPs not addressed by the modeling scenario discussed in this report will be evaluated on an individual facility or source category basis as appropriate.

[3]Although POM (polycyclic organic matter) is the name of the HAP listed in the Clean Air Act, most of the POM chemicals are "polycyclic aromatic hydrocarbons" or PAHs. Throughout this document, PAH and POM generally are used interchangeably. There are, however, instances where the discussion is specific to one or the other group of chemicals; for example, when discussing regulatory chemical groups or properties that are specific to a specific chemical class.

PB-HAP Compound[a]	Addressed by Screening Scenario?
Toxaphene	No
Trifluralin	No

[a]Source of list: U.S. EPA (2004a).
[b]Arsenic was not in the OAQPS initial list of PB-HAPs because its bioaccumulation potential is limited. It was recently added, however, because it is carcinogenic at very low doses, is persistent in the environment, and is emitted from many source categories. We refer to it as one of five PB-HAPs in the RTR multipathway assessment.

1.5 Report Organization

The remainder of this document is organized into four sections. Section 2 describes the Tier 1 screen, including the spatial layout of the hypothetical facility environment, the Tier 1 exposure scenario, and derivation of Tier 1 screening threshold emission rates.

Section 3 describes use of readily available site-specific information to refine estimates of screening threshold emission rates for Tier 2, and other aspects of the Tier 2 assessment. Section 4 discusses additional refinements that can be applied, sequentially, in Tier 3. References are listed in Section 5.

2. Tier 1 Screen

EPA's multimedia risk screen for RTR focuses on PB-HAPs that OAQPS identified as candidates for multimedia ingestion risk assessments (Section 1.4). Sources that are "screened out" at Tier 1 are assumed to pose no risks to human health. For sources that do not pass the Tier 1 screen, more refined screens, up to and including site-specific assessments, can be conducted as appropriate.

Using a worst-case hypothetical screening scenario for Tier 1 minimizes the chance that a facility that actually poses a risk to human health screens out. However, the scenario is not so biased that it never screens out facilities. An abundance of "false positives" would not help EPA focus on the facilities and PB-HAPs with emissions of actual concern.

This section describes the technical basis for Tier 1 of EPA's human multimedia ingestion screen of PB-HAP emissions from RTR sources. Specifically, the scenarios, models, configurations, and inputs used to derive screening threshold emission rates are described in detail in four subsections:

- Section 2.1 presents an overview of how screening is conducted in Tier 1, the chemicals and exposure scenario evaluated in Tier 1, and the models and methods used to conduct the screen.
- Sections 2.2 and 2.3 present technical descriptions of the hypothetical environmental setting, the exposure scenario used in Tier 1, and the models used in the screen.
- Section 2.4 provides a brief discussion of the screening threshold emission rates for each of the chemicals assessed.

Finally, Section 2.5 provides evaluations of the screening scenario for each of the modeled chemicals or chemical groups.

2.1 Overview of Tier 1

An ideal screening approach strikes a balance between being health-protective—to ensure that risks above levels of concern are identified (i.e., no false negatives)— and being accurate—to

minimize false positives (i.e., results suggesting that additional analysis is required when, in fact, the actual risk is low). Typically, gains in accuracy in environmental modeling (i.e., reductions of both false positives and false negatives) require additional resources.

The Tier 1 hypothetical watershed includes a farm homestead and a fishable lake near the facility, which are assumed to be the primary food sources for exposed individuals. The spatial layout and other physical aspects of the modeled domain configuration are health-protective; i.e., designed to maximize PB-HAP chemical concentrations in the food sources. The many environmental and chemical-specific properties governing fate and transport of PB-HAPs are parameterized with either conservative (i.e., health-protective) values or central-tendency values. Health-protective values (e.g., upper percentile values from a national distribution) are used for parameters that most influence exposure and risk. Including central-tendency values for the remaining parameters should help limit the number of false positives. False positives (i.e., results that suggest more assessment is required when in fact the actual risk is low) waste resources by leading to additional, unnecessary analysis. The Tier 1, TRIM-FaTE-based, multipathway fate and transport modeling scenario, or "Tier 1 scenario," is used to determine the Tier 1 screening threshold emission rates for comparison with individually reported facility emissions. The Tier 1 scenario includes the Tier 1 spatial layout for a hypothetical watershed and the assumptions and input values for a health-protective exposure and risk screen. The Tier 1 scenario is a static configuration that calculates a Tier 1 screening threshold emission rate for each of the five PB-HAP chemical groups.

The Tier 1 approach for evaluating multimedia ingestion exposures to PB-HAPs for RTR is diagrammed in Exhibit 4. Air toxics emitted by a source under consideration are reviewed to determine, first, whether emissions are reported for any of the five PB-HAPs of concern for non-inhalation pathways. If such emissions are reported, the emission rates are compared to Tier 1 screening threshold emission rates derived for each PB-HAP as described in this section. A screening threshold emission rate is the rate that corresponds to a cancer risk of 1-in-one million or an HQ of 1. Exhibit 5 presents those rates for the five PB-HAP groups.[4]

As depicted in Exhibit 4, the final decision point in the Tier 1 evaluation for a given facility has two possible outcomes:

- Emissions are equal to or less than the threshold emission rate of concern and therefore the facility screens out from further evaluation (SV ≤1); or
- Emissions are above the threshold emission rate of concern (SV >1), and risks from ingestion exposures cannot be ruled out (the facility does not screen out).

Conceptually, a threshold level for the RTR multipathway screening evaluation could be obtained by back-calculating the emission rate that results in the specified cancer risk or HQ, taking into account the exposure and fate and transport calculations included in the model. Because the models used in this assessment are not designed to run "backwards," the rates instead were derived from regression equations established following a series of TRIM.FaTE and exposure/risk model runs spanning a wide range of emission rates for each chemical. The estimated screening threshold emission rates are verified by performing model runs using the

[4]For chemicals known to cause both cancer and chronic noncancer impacts, *and* for which acceptable quantitative dose-response values are available for both cancer and noncancer endpoints, the endpoint that results in the lower screening threshold emission rate is used for screening (i.e., the screening threshold emission rate will be based on the effect that occurs at the lower exposure level). For the set of PB-HAPs for which screening threshold emission rates have been derived, arsenic and chlorinated dibenzo-dioxins and -furans cause both types of effects. Because the cancer dose-response value at a risk of 1-in-one million is lower than that for the noncancer reference toxicity dose, the screening threshold emission rate is based on the cancer endpoint.

estimated screening threshold emission rate to confirm that the emission rates result in a cancer risk of 1-in-one million or an HQ of 1.0.

Actual risks for each screening threshold emission rate would be lower than the levels of concern in nearly all circumstances, given the health protective nature of the Tier 1 scenario configuration.

Tier 1 screening threshold emission rates were developed individually for elemental and divalent mercury. Both were based on the lower of the screening threshold emission rates associated with multimedia ingestion exposures to divalent mercury and methyl mercury.[5] Only emissions of divalent mercury are screened because the sum of elemental mercury emissions across all National Emission Inventory (NEI) facilities is less than the elemental mercury screening threshold emission rate. Moreover, elemental mercury has a high vapor pressure and generally remains in air; deposited only during precipitation events and rapidly revolatilizing.

[5]Note that TRIM.FaTE models the transformation of mercury within the environment; thus, emissions of both divalent and elemental mercury will result in multipathway exposures to elemental mercury, methyl mercury, and divalent mercury.

Exhibit 4. Conceptual Decision Tree for Tier 1 Evaluation of Multimedia Ingestion Exposures to PB-HAPs

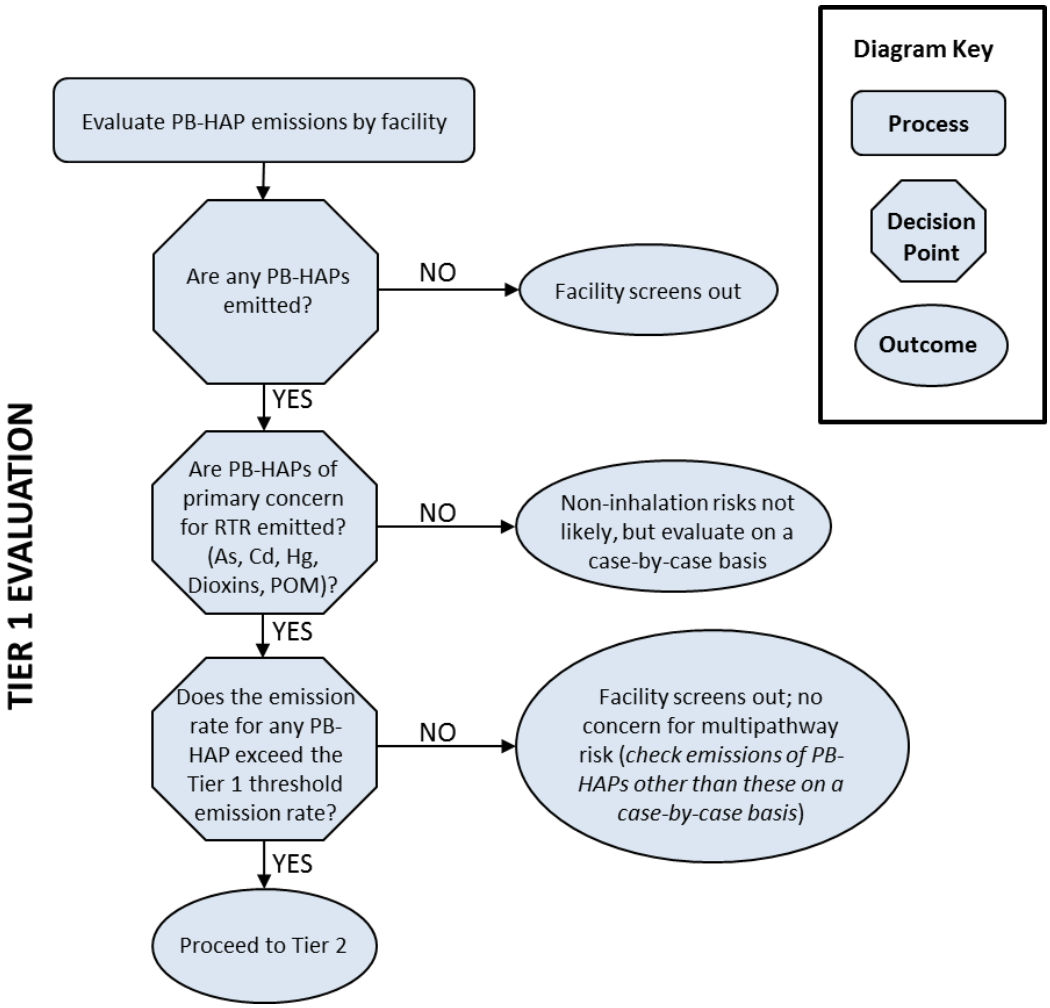


Exhibit 5. Screening Threshold Emission Rates for Multimedia Ingestion Exposures

Chemical	Screening Threshold Emission Rate (TPY)	Basis of Threshold (Type of Health Endpoint)
Arsenic	2.08E-04	Cancer
Cadmium	2.38E-03	Noncancer
Mercury (as divalent mercury emissions)	1.46E-04	Noncancer
POM (as benzo(a)pyrene equivalents)[a]	3.72E−03	Cancer
Dioxins (as 2,3,7,8-TCDD equivalents)[a]	6.02E-10	Cancer

TPY = U.S. short tons per year.
[a]See Section 2.2.4 for a discussion on the derivation of equivalent emissions.

2.2 Conceptual Exposure Scenario

A conceptual model for exposure pathways describes the movement of chemicals from the point of release to the points where exposure occurs. An exposure model generally includes several elements:

- Release to the environment (i.e., emissions);
- A receiving medium (e.g., air);
- Transport processes within and between media;
- Transformation to other chemicals via one or more physical, chemical, or biological processes;
- Continued tracking of a transformed chemical, if of concern (e.g., methyl mercury), or loss of chemical from the modeling domain via degradation;
- Estimates of chemical concentrations in human exposure media (e.g., air, foods, soils); and
- Human uptake of chemicals from those media by specific routes of exposure (i.e., inhalation, ingestion, dermal absorption).

PB-HAPs can persist in the environment for many years and, therefore, can build up in soils and lakes (sediments) and accumulate in biota, including fish, fruits and vegetables, and animal products (e.g., meat, dairy, eggs). For this reason, ingestion of foods grown near facilities that release PB-HAPs to air can be an important source of exposure.

Previously, to assess risks from hazardous waste combustion facilities, EPA identified several hypothetical receptor scenarios, noting that the scenarios can be appropriate for a broad range of situations where emissions to air are evaluated. The scenarios are described in EPA's *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities,* or HHRAP (U.S. EPA 2005a). In HHRAP, EPA recommends assessing several hypothetical receptors: a Farmer, Farmer Child, Resident, Resident Child, Fisher, Fisher Child, Acute Receptor, and Nursing Infant. These receptors are distinguished by their pathways of exposure and contact rates (e.g., food ingestion rates, hand-to-mouth soil ingestion, skin surface area). EPA further notes in HHRAP that some exposure settings might warrant including additional exposure pathways, such as fish ingestion by the Farmer.

For the RTR Tier 1 screen, ingestion exposure is estimated for a single hypothetical receptor who ingests both locally caught fish and home raised or produced farm foods. The ingestion exposure scenario for the PB-HAP Tier 1 screen includes several media:

- soil,
- farm-grown fruits and vegetables,
- farm-raised beef,
- dairy products from local farm-raised cows,
- farm-raised poultry and eggs,
- farm-raised pork,
- locally caught fish, and

- for children less than 1 year old, breast milk from a woman exposed via the media listed above (for dioxins only).[6]

As discussed in detail in Section 2.4.2, aside from ingestion of breast milk, ingestion exposure for all other media is assessed for adults and several age categories of children.

Other non-inhalation exposures possible for PB-HAPs discussed in HHRAP include using surface water or groundwater as a drinking water source and dermal exposure to chemicals in surface water and in soils; however, those exposure pathways are not evaluated for RTR. First, farmers are unlikely to use untreated surface water for drinking (or other household water uses).[7] HHRAP also recommends that exposure to groundwater not be evaluated because EPA found that groundwater is an insignificant exposure pathway for airborne combustion emissions (U.S. EPA 2005a). In addition, based on numerous evaluations of groundwater concentrations developed during RTR evaluations using TRIM.FaTE, we have confirmed that exposure from groundwater ingestion is a small fraction of overall exposure. Dermal absorption of deposited PB-chemicals that are originally airborne generally is relatively minor compared with other exposures (U.S. EPA 2006, Cal/EPA 2012). Preliminary calculations of estimated dermal exposure and risk of PB-HAPs, presented in Attachment C, show that the dermal exposure route is not a significant risk pathway relative to ingestion exposures. In addition, HHRAP recommends that dermal exposure not be assessed because available data indicate that the contribution to overall risk from dermal exposure to soils typically is small (U.S. EPA 2005a).

2.2.1 Approach to Development of the Tier 1 Scenario

The TRIM-based Tier 1 scenario described in this document does not represent any particular facility. The Tier 1 scenario is hypothetical and designed to estimate screening threshold emission rates that are health protective for any potential exposure situation that might plausibly be encountered in the United States. A range of conditions was assessed when conceptualizing and developing the screening scenario. The final configuration was chosen so that for a given individual human, any potential long-term exposures for any given geographic region would be unlikely to exceed those estimated for the Tier 1 configuration.

The development and application of the Tier 1 scenario for residual risk evaluations considered EPA's technical and policy guidelines presented in the *Residual Risk Report to Congress* (U.S. EPA 1999); Volumes I and II of the *Air Toxics Risk Assessment Reference Library* (U.S. EPA 2004a, 2005a); and other EPA publications (e.g., U.S. EPA 2003a, 2005a).The scenario described herein is the culmination of assessments completed since 2005. It allows an efficient and scientifically defensible screen of multipathway human health risk and provides a solid baseline from which to perform Tier 2 and Tier 3 screens, as described in Sections 3 and 4, respectively. All attributes of this scenario should not be considered "final," however. Some will continue to evolve based on feedback from the scientific community and Agency reviewers, on lessons learned as the scenario is further applied for RTR, and on future changes in legislated requirements.

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[6]Breast milk ingestion is an important exposure pathway for lipophilic compounds like dioxins. Breast milk does not contribute meaningfully to exposures to the other PB-HAPs assessed. See Section 2.4.2.2 below and Attachment B, Section B.3.4 for full discussions of infant exposures via breast milk ingestion.

[7]An exception to this generality would be reservoirs used for drinking water supplies, although treatment facilities would remove some proportion of PB-HAPs prior to water distribution. Such a situation might be worthy of additional analysis, if warranted for a given assessment (e.g., several facilities close to a reservoir).

2.2.1.1 Modeling Framework

The approach for multimedia ingestion risk screening and evaluation for RTR can be divided into four steps as shown in Exhibit 1:

1. Model the fate and transport of PB-HAPs emitted to air including partitioning to soil, water, and other environmental media (including fish[8]);
2. Estimate uptake of PB-HAPs by farm-grown foods (e.g., produce, livestock, dairy products) from soil and air and calculate the resulting concentrations in each food category;
3. Estimate human ingestion of PB-HAPs in farm-grown foods and in fish and through incidental ingestion of soils; and
4. Calculate estimates of lifetime cancer risk or chronic noncancer HQs, as appropriate, for each PB-HAP and compare these to selected evaluation criteria.

As shown in Exhibit 1, EPA's TRIM.FaTE model provides multimedia fate and transport modeling. Subsequent uptake of chemicals into farm foods and human ingestion exposures and risk is estimated using the multimedia exposure algorithms.

EPA's TRIM was conceived as a comprehensive modeling framework for evaluating risks from air toxics, and the TRIM system was designed to address each of the four steps involved in screening ingestion risk (Exhibit 1).[9] TRIM.FaTE—the fate and transport module—is available for application. EPA has completed some development activities for TRIM.Expo-Ingestion and TRIM.Risk-Human Health, two additional modules that cover the other three steps. Software development, however, is not yet complete for these modules. Thus, the RTR screening approach uses separate multimedia exposure and risk calculations to estimate PB-HAP concentrations in farm-grown foods, average daily ingestion doses, and cancer risks and chronic noncancer HQs. TRIM.FaTE plus the exposure and risk algorithms that are used are conceptually identical to the ingestion exposure and risk assessments that TRIM is intended to cover.

TRIM.FaTE outputs that are used as inputs to exposure and risk calculations include:

- PB-HAP concentrations in air,
- Air-to-surface deposition rates for PB-HAPs in both particle and vapor phases,
- PB-HAP concentrations in fish tissue, and
- PB-HAP concentrations in surface soil and root zone soil.

Using the exposure and risk algorithms, the RTR screening approach then estimates chemical concentrations in crop products based on deposition from air and uptake from soils, ingestion of PB-HAPs by farm animals via plant and soil ingestion and transfer to livestock products that are consumed by humans (e.g., eggs, milk, meat), and ingestion of PB-HAPs through these media by humans at various age groups from toddlers to adults (breast milk ingestion is also

[8]Concentrations in fish calculated by TRIM.FaTE are used to estimate ingestion exposures for humans consuming fish (except for arsenic, as provided below). Modeling of fish concentrations is therefore discussed herein as part of the TRIM.FaTE fate and transport modeling. TRIM.FaTE media concentration outputs are used to calculate the uptake of PB-HAPs into all other biotic media assumed to be ingested as part of the second step of the modeling framework.

[9]Information about the current status of TRIM modules and comprehensive documentation of modules developed thus far can be accessed on EPA's Fate, Exposure, and Risk Analysis website (https://www.epa.gov/fera/total-risk-integrated-methodology-trim-trimfate).

considered for infants for dioxins). The screening approach sums cancer risks across different age-groups to calculate a total lifetime cancer risk and calculates HQs for each age group for noncancer effects.

2.2.1.2 Model Configuration and Parameterization

The Tier 1 scenario is intended to minimize the chance that EPA would underestimate potential human multimedia ingestion risks. Although the health-protective approach likely overestimates risk for any given facility, it is appropriate for an initial screen. As in the 2006 preliminary multipathway screening for RTR (U.S. EPA 2006), exposures are modeled for a hypothetical farm homestead and fishable lake located adjacent to an emissions source. The hypothetical individual for which exposures are calculated derives all foods and soil from potentially contaminated adjacent locations and food and soil ingestion rates are from the upper ends of a nationally representative distribution of values (e.g., from EPA's 2011 *Exposure Factors Handbook*).

The physical/chemical environment represented in the screening scenario was parameterized with two types of values: typical and health-protective. In general, the spatial layout and the components of the scenario that influence air concentrations and deposition rates (which in turn affect PB-HAP concentrations in all other media) are defined or set to be health protective. Properties of environmental media are set with either typical or health protective values, as further discussed below and provided in Attachment A.

Calculated TRIM.FaTE concentrations generally are more sensitive to attributes of the spatial layout and the meteorological data than to other attributes of the scenario. For example, the dominant wind direction influences the direction of greatest deposition of emissions from a source, thereby driving estimated concentrations of PB-HAPs in soil, water, and biota. In contrast, relatively large changes in soil characteristics within the range of possible values (e.g., organic carbon content, water content) typically result in relatively small changes in media concentrations. Thus, health-protective values for meteorological data and a spatial layout that maximizes PB-HAP concentrations in the farm and lake are used for TRIM.FaTE in Tier 1.

2.2.2 Fate and Transport Modeling (TRIM.FaTE)

In developing the Tier 1 scenario, Version 3.6.2 of TRIM.FaTE was used to model the fate and transport of emitted PB-HAPs and to estimate concentrations in relevant environmental media. Additional information about TRIM.FaTE, including support documentation, software, and the TRIM.FaTE public reference library, is available at https://www.epa.gov/fera/total-risk-integrated-methodology-trim-trimfate.

The two main components of the fate and transport modeling are (1) the modeled domain, including the meteorological data, and (2) the environmental and chemical-specific properties associated with fate and transport through the environment.

Algorithms used to model mercury species and POM are described in Volume II of the *TRIM.FaTE Technical Support Document* (U.S. EPA 2002a). A comprehensive evaluation of the performance of TRIM.FaTE for modeling mercury was documented in Volumes I and II of the T*RIM.FaTE Evaluation Report* (U.S. EPA 2002b, 2005b). Algorithms specific to the fate and transport of chlorinated dibenzo-dioxin and -furan congeners are documented in the third volume of the *TRIM.FaTE Evaluation Report* (U.S. EPA 2004b).

Since 2005, the TRIM.FaTE master library was updated to include cadmium and, most recently, arsenic. In general, many of the algorithms and properties that are used to model mercury

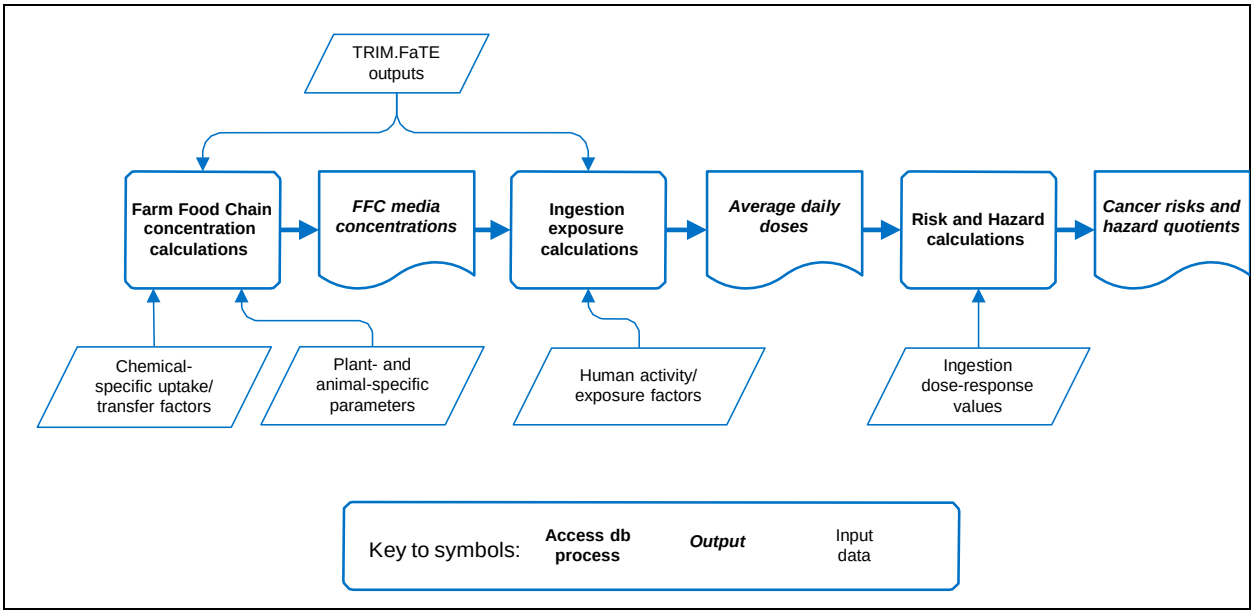
(except for the mercury transformation algorithms) are also applicable to cadmium and arsenic, although different empirical data are used for chemical-specific parameters values. Comprehensive technical documentation of TRIM.FaTE algorithms specific to cadmium and arsenic have not yet been compiled; however, all chemical-specific properties used by TRIM.FaTE to model cadmium and arsenic (as well as POM, mercury, and dioxins) are documented in Attachment A.

Based on a thorough 2011 evaluation of TRIM.FaTE performance in modeling the aquatic food web, a zooplankton compartment was added to TRIM.FaTE's aquatic compartment to facilitate comparison of TRIM.FaTE results for organic chemicals to those from other aquatic food-web models which include zooplankton separately from phytoplankton. Performance of the model was then recalibrated for mercury comparing the ratio and concentrations of inorganic and methylated mercury in each component of the aquatic food web with data from field studies. Parameterization of the TRIM.FaTE scenario used for RTR screening is described in more detail in Section 2.3.

2.2.3 Exposure Modeling and Risk Characterization

The algorithms that calculate chemical concentrations in farm-grown foods and ingestion exposures for hypothetical individuals are from EPA's Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities, or HHRAP (U.S. EPA 2005a). An overview of the input data and flow of computations for these calculations is presented in Exhibit 6. This exhibit demonstrates the general relationships between the relevant TRIM.FaTE outputs (i.e., chemical concentrations in environmental media and fish) and the calculations of ingestion exposure and risk. Additional discussion of exposure and risk calculations for the Tier 1 scenario is presented in Section 2.4 and Attachment B, and all inputs required by these calculations are documented in Attachment B.

Exhibit 6. Overview of Multimedia Ingestion Risk Calculations for RTR



Two of the five PB-HAPs for which screening threshold emission rates have been developed for RTR—POM and dioxins—are chemical groups comprising numerous individual compounds. The members of these groups as reported in NEI include both specific chemicals and groups

containing multiple chemicals. For example, for POM, emissions reported in NEI include various species, such as benz[a]anthracene, 2-methylnaphthalene, and chrysene, as well as non-specific entries, such as "PAH, total."[10] The constituents included in the POM and dioxin PB-HAP categories are grouped because they have a similar mode of toxic action and because they share attributes of environmental behavior.

2.2.4 Implementation of Risk-based Equivalency Factors for POM and Dioxin Congeners

To facilitate application of the multimedia ingestion screening methods for RTR, reported emissions of POM and dioxins are compared with a single reference (or index) chemical for each group: benzo[a]pyrene for POM and 2,3,7,8-TCDD for dioxins. These reference chemicals are both relatively well-studied and among the most toxic compounds within each group.

Derivation of equivalency factors begins with the basic relationship used to characterize health risk:

$$Risk \propto (Exposure\ Concentration) \times (Toxicity)$$

For a given air pollutant, exposure concentration increases in direct proportion to increases in air emissions of that substance. Furthermore, risks of toxic effects increase linearly with concentration. Consequently, emissions of one substance (e.g., chrysene) can be scaled to the risk equivalent of the emissions of the reference compound (e.g., benzo(a)pyrene or BaP) by multiplying by relative toxicity equivalency factors (TEFs) and relative exposure equivalency factors (EEFs). Using the dioxin group as an example, and 2,3,7,8-TCDD as the reference compound, scaling emissions follows Equation 1:

$$Emiss_{Dioxin(i):TCDD} = Emiss_{Dioxin(i)} \times EEF_{Dioxin(i):TCDD} \times TEF_{Dioxin(i):TCDD} \qquad \textbf{Eqn. 1}$$

where:

$Emiss_{Dioxin(i):TCDD}$	=	Risk-weighted emissions of Dioxin(i) (weighted according to cancer risk relative to 2,3,7,8-TCDD for oral exposures)
$Emiss_{Dioxin(i)}$	=	Emission rate of Dioxin(i)
$EEF_{Dioxin(i):TCDD}$	=	Exposure equivalency factor accounting for the ratio of final Dioxin(i) exposure dose compared with initial Dioxin(i) emissions relative to the final BaP dose compared with initial 2,3,7,8-TCDD emissions
$TEF_{Dioxin(i):TCDD}$	=	Toxicity equivalency factor accounting for the toxicity of Dioxin(i) relative to the toxicity of 2,3,7,8-TCDD via ingestion

After all the emissions of all congeners of Dioxins (*i… z*) have been converted to TCDD-equivalent emissions, they can be summed to a total TCDD-equivalent emissions rate. This TCDD-equivalent emission rate is compared with the TCDD screening threshold emission rate to develop a screening value (to determine if there is a possibility of adverse health effects.)

[10]Throughout this document, PAH and POM generally are used interchangeably because the majority of the POM chemicals are also considered PAHs. There are, however, instances where the discussion is specific to one or the other group of chemicals; for example when discussing regulatory chemical groups or characteristics that are specific to a particular chemical class.

The oral TEF for each POM and dioxin compound is based on the compound's oral toxicity relative to the oral toxicity of the index chemical for the group. The oral TEFs for POMs and dioxins were obtained from previous EPA analyses (U.S. EPA 2008b and 2017a, respectively). For POM, oral toxicity values for individual compounds have been derived following the same approach used to develop inhalation toxicity values (U.S. EPA 2017a). For dioxins, oral TEFs are those published by U.S. EPA (2008b), which were adopted from the values developed for the World Health Organization for its 2005 TEF reevaluation (van den Berg et al. 2006). Refer to Attachment B, Section B.4 for more information on these values.

The EEFs are calculated for each individual chemical that is modeled. TRIM.FaTE is configured for 14 POM and 17 dioxin congeners. For these chemicals, EEFs were calculated using the Tier 1 screening scenario described in this document. A release rate of 1 g/sec was modeled for each of the congeners. A chemical's EEF equals the ratio of its exposure concentration or dose to that of the index chemical for the group, also modeled with an emission rate of 1 g/sec. Emissions of several additional POM chemicals, however, are reported in the NEI. The determination of EEFs for these chemicals, and chemical groups, are discussed in the subsections below.

The product of the EEF and TEF for a given substance is called its "risk equivalency factor" (REF) for the purposes of RTR evaluations. POM (or dioxin) emissions from a facility can be quickly evaluated by summing the products of chemical-specific REFs and chemical-specific emission rates.

2.2.4.1 Calculation of Exposure, Toxicity, and Risk Equivalency Factors for POM Congeners

A total of 29 POM chemicals are reported by facilities in the NEI. Of those, 14 are configured in TRIM.FaTE; that is, values for all chemical-specific parameters required by TRIM.FaTE (e.g., solubility, vapor pressure, K_{ow}, Henry's law constant) for its multimedia transport and fate algorithms are included in the TRIM.FaTE library for the chemicals. Chemical and physical parameter values for the remaining 22 POMs or POM groups are not included.

The calculated EEFs, TEFs, and REFs for the 14 POM congeners that are configured in TRIM.FaTE, plus the 22 others, are shown in Exhibit 7. To determine appropriate exposure surrogates for chemicals not parameterized in TRIM.FaTE, EPA evaluated the relationships between chemical-specific properties (e.g., K_{ow}, Henry's law constant), intermediate modeled values (e.g., deposition, soil concentration), and exposures in terms of lifetime average daily dose (LADD), where the average daily doses (ADDs) for the youngest two age groups were adjusted by the age-dependent adjustment factors (ADAFs) to account for the mutagenic mode of action of PAHs (U.S. EPA 2005c,d,e). The correlation between K_{ow} and LADD is stronger than any other chemical-specific property and a power regression was developed to estimate LADD based on congener-specific K_{ow}. Based on this analysis, total LADD (age-adjusted) for each congener is calcualated based on the congener's K_{ow} and the power regression of the modeled PAHs, as provided in Exhibit 8. Exhibit 8 shows that, in general, as K_{ow} increases, so too does exposure.

Exhibit 7. Exposure, Toxicity, and Risk Equivalency Factors Relative to BaP for POM Congeners Currently Evaluated in the Screens

PB-HAP[a]	Fully Characterized for TRIM.FaTE Modeling?[b]	Tier 1 Exposure-equivalency Factor (EEF)	Toxic-equivalency Factor (TEF)[c]	Tier 1 Risk-equivalency Factor (REF)
7,12-Dimethylbenz[a]Anthracene	Yes	3.9	250	967
Dibenzo[a,i]pyrene	No	31.3	10	313
3-Methylcholanthrene	No	4.2	22	93.3
Dibenzo[a,h]anthracene	Yes	7.6	1	7.6
Benzo[b+k]fluoranthene	No	17.7	0.1	1.8
Benzo[b]fluoranthene	Yes	10.8	0.1	1.1
Benzo[a]pyrene	Yes	1	1	1
Indeno[1,2,3-c,d]pyrene	Yes	4.5	0.1	0.4
Benzo[j]fluoranthene	No	4.1	0.1	0.4
Benzofluoranthenes	No	8.1	0.05	0.4
PAH, total	No	5.1	0.05	0.3
Polycyclic organic matter	No	5.1	0.05	0.3
Benz[a]anthracene/Chrysene	No	5.1	0.05	0.3
Benzo[k]fluoranthene	Yes	17.7	0.01	0.2
Benzo(e)pyrene	No	4.4	0.05	0.2
Benzo(ghi)perylene	Yes	4.2	0.05	0.2
Retene	No	3.6	0.05	0.2
Benzo(a)fluoranthene	No	2.1	0.05	0.1
Dibenz[a,j]acridine	No	0.7	0.1	0.07
Perylene	No	1.1	0.05	0.05
Benzo(g,h,i)fluoranthene	No	0.5	0.05	0.03
Benzo(c)phenanthrene	No	0.5	0.05	0.03
Benz[a]anthracene	Yes	0.09	0.1	0.01
Chrysene	Yes	0.3	0.001	0.0003
2-Acetylaminofluorene	No	0.003	1	0.003
Fluoranthene	Yes	0.03	0.05	0.002
Acenaphthylene	Yes	0.02	0.05	0.001
beta-Chloronaphthalene	No	0.02	0.05	0.001
Fluorene	Yes	0.02	0.05	0.001
Acenaphthene	Yes	0.02	0.05	0.001
1-Methylnaphthalene	No	0.01	0.05	0.0006
2-Methylnaphthalene	Yes	0.01	0.05	0.0005

PB-HAP[a]	Fully Characterized for TRIM.FaTE Modeling?[b]	Tier 1 Exposure-equivalency Factor (EEF)	Toxic-equivalency Factor (TEF)[c]	Tier 1 Risk-equivalency Factor (REF)
Carbazole	No	0.008	0.02	0.0002
Anthracene	No	0.04	0	0
Phenanthrene	No	0.04	0	0
Pyrene	No	0.1	0	0

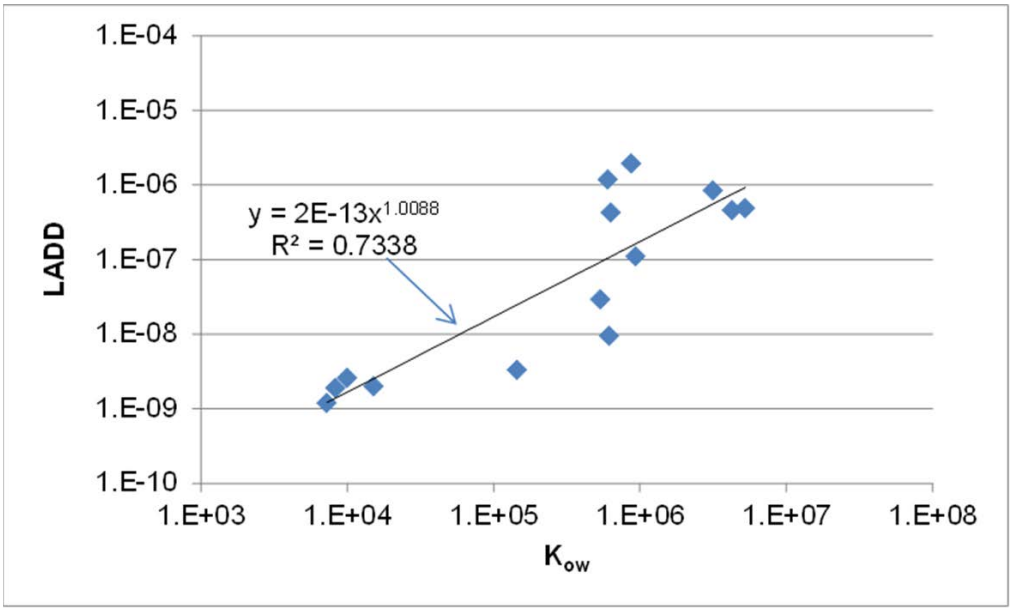
Notes: Rounding artifacts present. HAP = hazardous air pollutant; PB-HAP = persistent and bioaccumulative HAP; TRIM.FaTE = Total Risk Integrated Methodology (Fate and Transport Ecological model); POM = polycyclic organic matter; BaP = benzo(a)pyrene; RTR = Risk and Technology Review program; K_{ow} = octanol-water partition coefficient; PAH = polycyclic aromatic hydrocarbon

[a]Naphthalene is not included in the POM category for the RTR multipathway (i.e., non-inhalation) analyses. Naphthalene is listed individually as a HAP under section 112(b) of the Clean Air Act. POM also is listed as a HAP under section 112(b) and is defined as organic compounds with more than one benzene ring and a boiling point greater than or equal to 100°C (see http://www.epa.gov/ttn/atw/orig189.html). Although naphthalene is a POM as defined in the Clean Air Act, unlike the other POM chemicals modeled in the multipathway assessment, naphthalene remains primarily (>98–99%) in vapor phase at ambient temperatures; thus it disperses far away from a facility in air with negligible local deposition. Given its volatility (solid phase sublimates to vapor phase at ambient temperatures), it does not accumulate in localized environmental media over time (ATSDR 2005). Additionally, based on a log K_{ow} of 3.29, it has a low affinity for lipids compared with other POMs. For these reasons, EPA does not consider naphthalene to be a persistent and bioaccumulative POM; inhalation is the only pathway of concern for RTR assessment of naphthalene.

[b]Some POM congeners are not fully characterized in TRIM.FaTE (with their chemical properties, partition coefficients, etc.) and so cannot be modeled directly. As discussed in the text, EEFs for these uncharacterized POM congeners are estimated based on K_{ow}.

[c]Sources: U.S. EPA (2017a); professional judgment.

Exhibit 8. Relationship between Ingestion Exposure and K_{ow} for POM Chemicals



For POM reported as undefined groups (i.e., "PAH, total" and "Polycyclic Organic Matter"), EPA assigned K_{ow} values near the upper end of the range of all of the K_{ow} values, corresponding to an exposure near the upper end of the range ($\log K_{ow}$ = 6.5) (see Exhibit 9). This assignment is assumed to be health protective and is unlikely to under predict exposure.

For POM chemicals that are not fully parameterized in TRIM.FaTE ("No" in Exhibit 7), we use the regression equation in Exhibit 8 with the K_{ow} values listed in Exhibit 9 to extrapolate the EEFs in Exhibit 7. Thus, all POM have EEF, TEF, and REF values relative to BaP.

Exhibit 9. LogK_{ow} Values for POM Chemicals Congeners

Chemical	LogK_{ow} (K_{ow})	Source
1-Methylnaphthalene	3.87 (7.41E+03)	Mackay et al. 2006[a]
2-Chloronaphthalene	4.14 (1.38E+04)	Mackay et al. 2006[a]
3-Methylcholanthrene	6.42 (2.63E+06)	Mackay et al. 2006[a]
Anthracene	4.45 (2.82E+04)	Mackay et al. 2006[a]
Benzo[e]pyrene	6.44 (2.75E+06)	Mackay et al. 2006[b]
Carbazole	3.72 (5.25E+03)	U.S. EPA 2012a (EPI Suite)[a]
Dibenzo[a,i]pyrene	7.28 (1.91E+07)	U.S. EPA 2012a (EPI Suite, estimate)
Dibenz[a,j]acridine	5.63 (4.27E+05)	U.S. EPA 2012a (EPI Suite)[c]
PAH, total[d]	6.50 (3.16E+06)	EPA assigned
Perylene	5.82 (6.61E+05)	Mackay et al. 2006[a]
Phenanthrene	4.46 (2.88E+04)	Mackay et al. 2006[a]
Polycyclic Organic Matter[d]	6.50 (3.16E+06)	EPA assigned
Pyrene	4.88 (7.59E+04)	Mackay et al. 2006[a]
Retene	6.35 (2.24E+06)	U.S. EPA 2012a (EPI Suite, estimate)
2-Acetylaminofluorene	3.28 (1.91E+03)	Montgomery 2007[e]
Benzo[c]phenanthrene	5.52 (3.31E+05)	U.S. EPA 2012a (EPI Suite, estimate)
Benzofluoranthenes	6.70 (5.01E+06)	U.S. EPA 2012a (EPI Suite, estimate)
Benzo[j]fluoranthene	6.40 (2.51E+06)	Mackay et al. 2006[f]
Benzo[g,h,i]fluoranthene	5.52 (3.31E+06)	U.S. EPA 2012a (EPI Suite, estimate)
Benzo[a]fluoranthene	6.11 (1.29E+06)	U.S. EPA 2012a (EPI Suite, estimate)

[a]Original source is Hansch et al. 1995.
[b]Original source is Sangster 1993.
[c]Original source is Helweg et al. 1997.
[d]For POMs reported as unspeciated groups (i.e., "PAH, total" and "Polycyclic Organic Matter") EPA assigned surrogates with K_{ow} values near the upper end of the range of all of the K_{ow} values, corresponding to an exposure near the upper end of the range (logK_{ow} = 6.5). This assignment is assumed to be health protective and likely will not under predict exposure.
[e]Original source is Mercer et al. 1990.
[f]Original source is Bayona et al. 1991.

One PAH that is not evaluated for ingestion exposure is naphthalene, which is listed individually as a HAP under Section 112(b) of the Clean Air Act. POM also is listed as a HAP under Section 112(b) and is defined as organic compounds with more than one benzene ring and a boiling point greater than or equal to 100 °C (see http://www.epa.gov/ttn/atw/orig189.html). While naphthalene is a POM, as defined in the Clean Air Act, unlike the other POM chemicals modeled in the multipathway assessment, at ambient temperatures, naphthalene remains in

vapor phase; generally only 2–3 percent of naphthalene emitted to air deposits to ground level (ATSDR 2005). Naphthalene in other environmental media is short-lived due to its tendency to volatilize. Thus, it does not build up in soils, sediments, water, or biota over time (ATSDR 2005). With a logK_{ow} of 3.29, naphthalene has a moderate affinity for lipids and can accumulate in some tissues over the short-term; however, it is rapidly exhaled or metabolized to other readily eliminated chemicals. For these reasons, EPA is not including naphthalene in its multipathway risk assessment.

2.2.4.2 Calculation of Scaling Factors for Dioxin Congeners

The calculated EEFs, TEFs, and REFs for the 17 dioxin congeners that are chlorinated in the lateral 2, 3, 7, and 8 positions are presented in Exhibit 10.

Exhibit 10. Exposure and Toxicity Equivalency Factors Relative to TCDD for Modeled Dioxin Congeners

PB-HAP	Tier 1 Exposure-equivalency Factor (EEF)	Toxic-equivalency Factor (TEF)[a]	Tier 1 Risk-equivalency Factor (REF)
1,2,3,4,6,7,8,9-Octochlorodibenzofuran	0.2	0.0003	0.00007
1,2,3,4,6,7,8,9-Octochlorodibenzo-p-dioxin	0.3	0.0003	0.00008
1,2,3,4,6,7,8-Heptachlorodibenzofuran	0.2	0.01	0.002
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	1.0	0.01	0.01
1,2,3,4,7,8,9-Heptachlorodibenzofuran	0.2	0.01	0.002
1,2,3,4,7,8-Hexachlorodibenzofuran	0.3	0.1	0.03
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	1.6	0.1	0.2
1,2,3,6,7,8-Hexachlorodibenzofuran	0.5	0.1	0.05
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	1.0	0.04[a]	0.04
1,2,3,7,8,9-Hexachlorodibenzofuran	0.5	0.1	0.05
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin	1.1	0.04[a]	0.04
1,2,3,7,8-Pentachlorodibenzofuran	0.4	0.03	0.01
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	3.6	1	3.6
2,3,4,6,7,8-Hexachlorodibenzofuran	0.5	0.1	0.05
2,3,4,7,8-Pentachlorodibenzofuran	0.4	0.3	0.1
2,3,7,8-Tetrachlorodibenzofuran	0.1	0.1	0.01
2,3,7,8-Tetrachlorodibenzo-p-dioxin	1	1	1

Notes: Rounding artifacts present; HAP = hazardous air pollutant; PB-HAP = persistent and bioaccumulative HAP; TCDD = tetrachlorodibenzo-p-dioxin; IRIS = EPA's Integrated Risk Information System; CSF = cancer slope factor.
[a]Sources: van den Berg et al. (2006), except for 1,2,3,6,7,8-hexachlorodibenzo-p-dioxin and 1,2,3,7,8,9-hexachlorodibenzo-p-dioxin, which are calculated based on the ratio of the IRIS-based CSF for the respective congener to the IRIS-based CSF for 2,3,7,8-TCDD (available at U.S. EPA 2017b)

As provided in Exhibit 10, WHO TEFs from van den Berg et al. (2006) are used except for two congeners for which EPA's IRIS program has developed a CSF—1,2,3,6,7,8-hexachlorodibenzo-p-dioxin and 1,2,3,7,8,9-hexachlorodibenzo-p-dioxin. Collectively across RTR assessments that EPA has conducted in recent years, these two congeners together constitute roughly 4 percent of total dioxin emissions from point sources. When the dioxin emissions are weighted by TEFs (to calculate TEQs), the two congeners constitute about 4 percent of the total dioxin TEQ emissions from point sources using TEF=0.1 from van den Berg et al. (2006) and about 2 percent using TEF=0.04 derived from the IRIS-based CSF. Therefore, the impact of changing the TEFs of the two congeners is small.

Some facilities report dioxins as "Dioxins, Total, without Individual Isomers Reported," "Dioxins," or as "2,3,7,8-TCDD TEQ," and in these cases, we do not adjust or scale the emissions. That is, we assume that they behave like 2,3,7,8-TCDD in the environment. We assume that the toxicity of unspecified total "Dioxins" equals that of the same quantity of 2,3,7,8-TCDD. This approach could be improved by obtaining information on the speciation of dioxin emissions for each facility or data that might allow calculation of an average speciation profile that could be applied to all facilities in a source category.

2.3 Description of Environmental Modeling Scenario

As described in Section 2.2.1.2, the physical configuration of the RTR Screening Scenario was designed to encompass the upper end of possible long-term PB-HAP exposures. Values for environmental and chemical-specific properties were selected to be either health protective or central-tendency. Scenario configuration and parameterization, the rationale for selecting particular property values, and model uncertainties are presented in the sections that follow. Comprehensive documentation of TRIM.FaTE property values for the Tier 1 screening scenario is provided in Attachment A.

2.3.1 Chemical Properties

The chemical-specific chemical/physical properties that TRIM.FaTE requires to simulate transport and fate through multiple environmental media (e.g., Henry's law constant, molecular weight, melting point) were obtained from peer-reviewed and standard chemical reference sources. Numerous other chemical-specific properties also depend on the particular abiotic or biotic compartment type; those properties are discussed generally in the sections that follow and are documented in Attachment A.

2.3.2 Spatial Layout

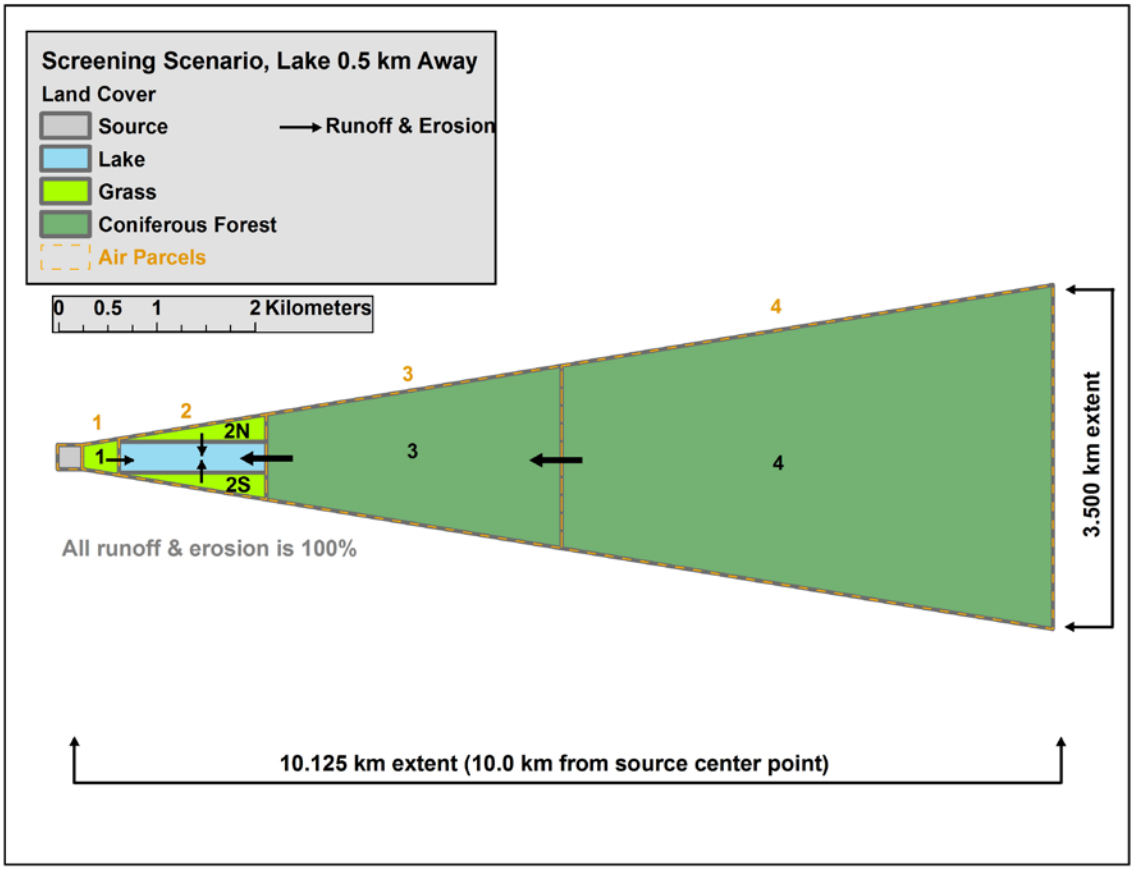
The Tier 1 spatial layout for TRIM.FaTE, provided in Exhibit 11, represents a farm homestead with a fishable lake (and its surrounding watershed) located near a facility emissions source. The source parcel is a square with sides of 250 m to represent a relatively small-to-medium facility at the fence line. With a predominant wind direction toward the east, the modeled layout is generally symmetric about an east-west line and is wedge-shaped to reflect Gaussian dispersion of the emissions plume.

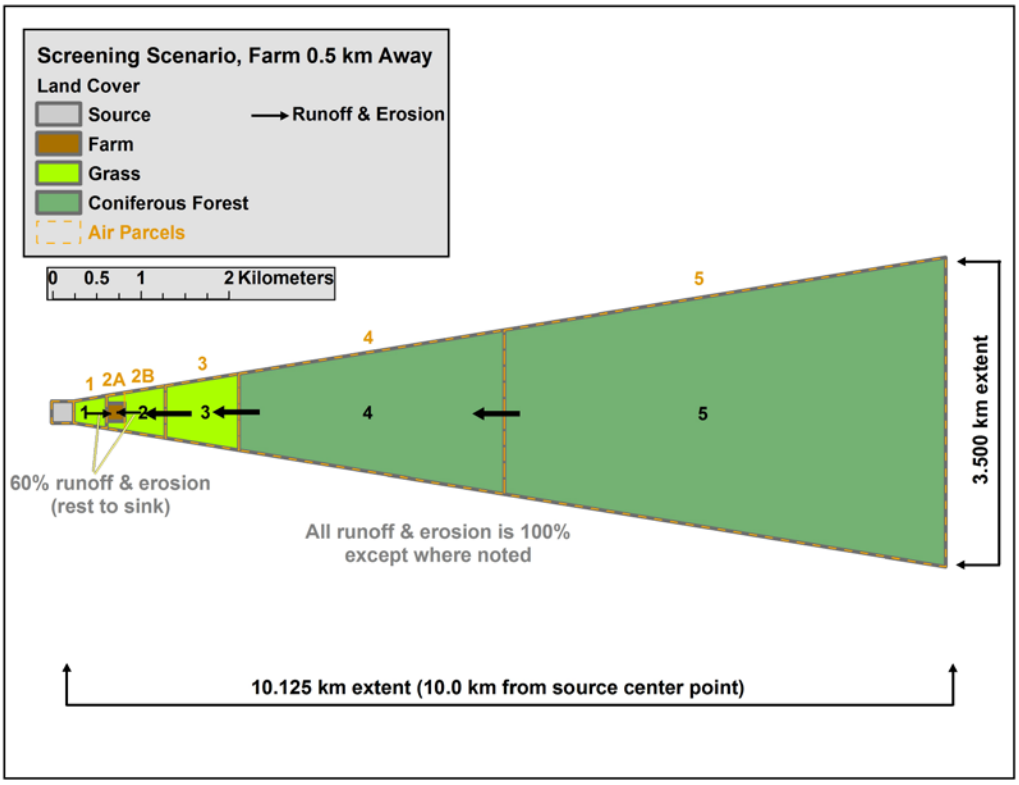
The modeled wedge extends 10 km downwind (eastward) from the facility. Air dispersion modeling indicates that the maximum air concentration and deposition rates occur relatively close to the facility (probably within a few hundred meters, with the exact location varying with stack height and other parameters), which is well within a 10-km radius. TRIM.FaTE modeling also indicates that at 10 km downwind of a source, deposition rates for the PB-HAPs are expected to be lower by about two orders of magnitude than deposition rates at a few hundred meters of the source. Extending the modeling layout beyond 10-km downwind would increase

the amount of deposition "captured" by the modeled watershed, but the incremental chemical mass expected to accumulate in the watershed diminishes rapidly with distance. Moreover, the additional mass deposited beyond 10 km is expected to cause a negligible increase in total ingestion exposure.[11] Given these conditions, a downwind length of 10 km is appropriate for the screening scenario.

[11]Mass deposited at the outer edge of the watershed is expected to result in only a very small increase in estimated exposure via fish consumption by increasing the chemical mass transported to the lake through erosion and runoff. The TRIM.FaTE runs supporting Tier 1 (discussed in this section) indicate that average chemical deposition rates at the parcel farthest from the emission source (e.g., parcel 5 in the farm layout of Exhibit 11, which is 5–10 km from the emission source) are between 1 and 2 orders of magnitude smaller than those within 1 km of the source (e.g., parcels 1 and 2 in the farm layout of Exhibit 11), depending on the chemical. The large distance from the eastern edge of the watershed to the lake or farm attenuates transport of chemical mass by erosion and runoff, dampening the effect of including additional deposition beyond 10 km.

Exhibit 11. TRIM.FaTE Surface Parcel Layouts


Screening Scenario, Lake 0.5 km Away
Land Cover
Source
Runoff & Erosion
Lake
Grass
Coniferous Forest
Air Parcels
0 0.5 1 2 Kilometers
1
2
2N
2S
3
4
All runoff & erosion is 100%
3.500 km extent
10.125 km extent (10.0 km from source center point)


Screening Scenario, Farm 0.5 km Away
Land Cover
Source
Runoff & Erosion
Farm
Grass
Coniferous Forest
Air Parcels
0 0.5 1 2 Kilometers
1
2A
2B
3
4
5
60% runoff & erosion
(rest to sink)
All runoff & erosion is 100%
except where noted
3.500 km extent
10.125 km extent (10.0 km from source center point)

The north-south width of the wedge-shaped watershed was set based on the observed behavior of chemicals emitted to the ambient air. If meteorological stability is known or can be assumed, the lateral spread of the plume (σ_y, measured from the centerline) at a certain distance from the source can be estimated using the Pasquill-Gifford curves. Turner (1970) derived the equations for these curves, which can be found in the Industrial Source Complex 3 Dispersion Model Manual (among other sources).[12] For a relatively neutral atmosphere (stability class D), σ at 10 km is about 550 m using this estimation. In a Gaussian distribution, about 99.6 percent of the plume spread area is contained within 3σ of the median line. Therefore, the plume σ was set at 3 times 550 m, or approximately 1.75 km north and south from the centerline at a distance of 10 km. The total plume width at 10 km is twice that or 3.5 km. These dimensions were used to define the dimensions of the overall air and surface parcel layouts for the screening scenario.

The area of each parcel would encompass similar chemical mass (i.e., larger area for a parcel farther from the source would encompass a similar total chemical mass because concentrations per unit area would be lower than for a smaller parcel closer to the source with higher concentrations per unit area).

The depth of the surface soil compartments was set to 1 cm, except for the farm parcel, for which the depth was set to 20 cm to simulate the effect of tillage. Characteristics of the soil layers (e.g., organic carbon content, air and water content, and subsoil depth) generally were set to represent typical or national averages as summarized by McKone et al. (2001).

The the air parcel layouts mirror those of the surface parcel layout, except that the air parcels over the lake and farm encompass the areas north and south of the lake and the farm.

2.3.3 Watershed and Water Body Parameterization

Properties associated with the watershed soil and lake determine how pollutants in the system are transported through and accumulate in various media compartments. These properties describe the physical characteristics of the environmental media included in the modeled region, as well as the assumed connections and relationships between media types and modeled spatial components that in turn affect chemical transport via water runoff, ground infiltration, deposition of suspended sediments in the water column, and other processes. This section discusses the selection of values used for key properties of the soil, water, and sediment compartments. Also discussed are chemical properties related to watershed and water body processes (chemical-specific compartment properties in TRIM.FaTE) and the configuration of terrestrial plants included in the scenario.

2.3.3.1 Water Balance

Water-related properties of the lake and related watershed characteristics (e.g., runoff rates from each surface soil compartment) were set so that a simplified water balance is achieved. Although TRIM.FaTE maintains chemical mass balance, the model does not calculate or maintain media mass balances (e.g., for water) except where specified in certain formulas. For the Tier 1 scenario, parameter values were set to satisfy two equations to balance water volume. The first equation (Equation 2) maintains a balance of water entering and leaving the terrestrial portion of the scenario:

[total precipitation] = [evapotranspiration] + [total runoff] ***Eqn. 2***

[12] http://www.epa.gov/scram001/userg/regmod/isc3v2.pdf

In Equation 2, total runoff is equal to the sum of overland runoff to the lake and seepage to the lake via groundwater. Evapotranspiration represents that water released to air from plants in vegetated parcels.

Equation 3 describes the volumetric balance of transfers of water to and from the lake:

[total runoff] + [direct precipitation to the lake] = ***Eqn. 3***

[evaporation from the lake surface] + [outflow from the lake]

Note that TRIM.FaTE uses all these properties with the exception of evapotranspiration, which is part of the water balance calculation outside TRIM.FaTE. The water characteristics assumed for the Tier 1 scenario are meant to represent a relatively wet and moderately warm location in the United States (USGS 1987). Following are the assumptions for this scenario:

- 35 percent of the total precipitation leaves the scenario through evapotranspiration.
- 25 percent of total precipitation infiltrates into the groundwater and eventually flows into the lake.
- 40 percent of total precipitation contributes to overland runoff.

For these calculations, the source facility parcel was considered to be outside the watershed and therefore was not included in the water balance. The evaporation rate from the lake was assumed to be 700 mm/year based on data reported by Morton (1986) for various lakes. The runoff rate was defined to be both spatially and temporally constant (i.e., it is not linked to precipitation events) throughout the modeled domain. Based on these assumptions, the outflow of water from the lake is about 18 million m^3/year, which translates to a volumetric turnover rate of about 12.2 lake volumes per year.

Other quantitative water-body and watershed characteristics TRIM.FaTE uses are listed in Attachment A.

2.3.3.2 Erosion and Runoff

Erosion and runoff are important surface transport algorithms for modeling chemical transport in TRIM.FaTE. Model input parameters for erosion include (1) parcel-specific erosion rates (in kg/m^2-day); and (2) inter-parcel erosion links (directing erosion to a specific parcel or parcels). Model input parameters for runoff include (1) parcel-specific runoff rates (in m^3/m^2-day); and (2) inter-parcel runoff links (as defined above). TRIM.FaTE uses those properties for chemical transport only; movement of soil and water into and out of parcels are assumed to balance so that there is no net change.

To establish soil erosion and runoff rates into the lake and onto the farm parcel, mean values, as estimated or measured in several studies, were used (Bajracharya et al., 1998; Gaspar et al., 2013; Schimmack et al., 2002; Young et al., 2014). Separate sites and measurement methods across the studies were treated as distinct observations, for a total of eight mean deposition rates to represent a distribution of values for varying landscapes. Use of mean values from multiple data sets limits the influence of extreme measured values within any one data set. Combining observations from different sites and measurement methods effectively combines variability and uncertainty distributions. To ensure an appropriate level of health protectiveness in this context, the 90th percentile of the mean soil deposition rate was used in the RTR screening scenario for all chemicals and for all tiers. This corresponds to a soil deposition rate

of 3 mm/year onto the farm parcel, which is achieved in the Tier 1 layout by setting erosion and runoff onto the farm parcel from neighboring parcels at 60 percent. Runoff and erosion patterns were exactly aligned, instead of setting distinct values for the two processes. For the lake only scenario, 100 percent of the erosion and runoff from neighboring parcels enters the lake. This assumption is both health protective and physically plausible in terms watershed dynamics and and based on the lake flush rate it implies.

2.3.3.3 Sediment Balance

A simplified balance of sediment transfers between the watershed and the lake also was maintained for the scenario via parameterization of sediment-related properties. As with water, TRIM.FaTE does not internally balance sediment mass; calculations external to TRIM.FaTE balanced sediment gains and losses to set relevant parameter values. The sediment balance maintained is described by Equation 4, where terms represent mass of sediment:

[total surface soil transfers to the lake via erosion] = ***Eqn. 4***

[removal of sediment from the water column via outflow] + [sediment burial]

The second term (removal of sediment from the water column via outflow) is represented in TRIM.FaTE by the lake flush (or turnover) rate. The third term (sediment burial) is the transfer of sediment from the unconsolidated benthic sediment to the consolidated sediment layer below.

To maintain the sediment balance, erosion rates were calculated for each surface soil compartment using the universal soil loss equation (USLE, Wischmeier and Smith 1978), assuming a relatively high rate of erosion. The total suspended sediment concentration in the lake is assumed to remain constant, and the flush rate of the lake (calculated via the water balance approach described above) allows an estimate of sediment removal from the modeling domain via lake water outflow. The difference between these sediment fluxes equals the sediment burial rate, which is the rate at which sediment particles in the unconsolidated benthic sediment layer are transported to the consolidated sediment, where the particles can no longer freely interact with the water column.

In TRIM.FaTE, the consolidated sediment layer is represented with a sediment sink; as with all sinks in TRIM.FaTE, chemical mass sorbed to buried sediment that is transported to the sink cannot be returned to the modeling domain. TRIM.FaTE calculates burial rate as the difference between user-specified values for sediment deposition velocity (from the water column to the benthic sediment) and sediment resuspension velocity (back into the water column from the top, unconsolidated benthic sediment layer). TRIM.FaTE keeps a constant volume of particles in the unconsolidated sediment layer. The density of solid particles is the same for both particles suspended in the water column and for benthic sediments; therefore, the mass of solid particles in the sediment is also constant.

For the Tier 1 scenario described here, the average sediment delivery rate (i.e., transfer of sediment mass from watershed surface soil to the lake due to erosion) for the entire watershed was estimated to be about 0.0036 kg/m^2-day, based on calculations using the USLE. The HHRAP documentation notes that the USLE equation sometimes overestimates sediment loading to a lake from the surrounding watershed (U.S. EPA 2005a). For the Tier 1 scenario,

however, this possible bias is appropriate because it is health protective.[13] Surface soil compartments adjacent to the lake are linked directly to the lake for the purposes of estimating erosion and runoff transfers (see layout in Exhibit 11). Erosion and runoff from the *source* parcel move directly to a sink and, therefore, do not enter the Tier 1 scenario lake. The overland transport of sediment to the lake from Parcel 4 also occurs via a direct link; however, in reality, the overland runoff and erosion would be attenuated by the intervening soil parcels. That attenuation is simulated by using a lower sediment delivery ratio in the USLE as applied to Parcel 4.

Using the calculated surface soil erosion rates for the scenario, the total average daily sediment load to the lake from the watershed is about 16,600 kg/day. About 15 percent of this load is removed from the lake via outflow of suspended sediments (based on a calculated flush rate of 12.2 volume turnovers per year), with the remainder of the sediment input to the lake eventually transferred to the sediment burial sink.

2.3.4 Meteorology

Meteorological properties used in TRIM.FaTE algorithms include air temperature, air mixing height, wind speed and direction, and precipitation rate. These properties, which can vary significantly among geographic locations, and seasonally and hourly for a single location, greatly influence the chemical concentrations predicted in media of interest. Because the screening scenario is intended to be generally applicable to any U.S. location, and to minimize the frequency of false negatives, a health protective configuration was used. The meteorology of the screening scenario was defined to ensure that (when used in combination with the selected spatial layout) the maximum exposures that might be encountered for the scenarios of interest would be encompassed (i.e., consumption of home grown farm foodstuffs and self-caught fish, with all farm foods and fish obtained from locations receiving chemicals emitted from the local source). Ensuring that the meteorological parameters were not overly protective of health, such as *always* having the wind blow toward the location of interest, however, was also important to avoid too many false positives.

The meteorological data for the screening scenario are intended to represent a location with a low wind speed, a wind direction primarily over the simulated watershed, a low mixing height, and a relatively high amount of total precipitation falling on the watershed. The values used were based on the distribution of values for U.S. locations as specified in Exhibit 12, but an artificial data set was compiled for this screen and not linked to any real location (for example, temporally variable meteorological parameters were changed only on a daily basis). Using a daily time step instead of an hourly time step substantially reduces required model run time. Meteorological inputs are summarized in Exhibit 12.

Exhibit 12. Summary of Key Meteorological Parameter Inputs

Parameter	Selected Value	Justification
Air temperature	Constant at 298 Kelvin	Recommended default value listed in HHRAP (U.S. EPA 2005a). Value is similar to the mean maximum daily temperatures in May and September in much of the U.S. mid-Atlantic, mid-West, and Great Plains, according to 1981–2010 climatology.[a]

[13]Based on sensitivity analysis, a higher erosion rate will both increase surface water concentrations and decrease surface soil concentrations; the relative impact on resulting concentrations, however, will be proportionally greater in the waterbody.

Parameter	Selected Value	Justification
Mixing height	Constant at 226 m	Value is ~5th percentile of median hourly mixing heights recorded at 824 meteorological stations across the United States from January 1–December 31, 2016.
Wind direction	Blows from source parcel into scenario domain (west to east) 3 days per week (roughly 43 percent of the week); during other times does not blow into domain	Wind blowing toward a location of interest (e.g., toward a lake or farm) will move more emitted chemical mass over the location of interest than wind blowing in other directions. For much of the U.S. mid-Atlantic and western regions, the wind generally blows eastward.[a] Among the NOAA 1981–2010 normalized wind-vector data, the average wind direction had a strong eastward component at over one-third of the stations.[b] For the hypothetical RTR scenario, a more extreme example of this pattern is represented by conditions in Yakima, Washington, where the wind blows eastward approximately 40 percent of the time (review of wind direction data compiled by the National Weather Service; NCDC 1995). This pattern is approximated in the RTR scenario with a configuration in which the modeled domain is downwind of the source 3 out of every 7 days.
Horizontal wind speed	Constant at 1.6 m/sec	Set to ~5th percentile of median hourly wind speeds, partitioned by eight wind directions, recorded at 824 meteorological stations across the United States from January 1–December 31, 2016.
Precipitation frequency	Precipitation occurs 3 days per week (roughly 43 percent of the week); wind direction blows into domain 2 of these days (roughly 29 percent of the week)	Two-thirds of the total precipitation occurs when the domain is downwind of the modeled source. This pattern approximates that for rainy U.S. locations, where precipitation occurs 35–40 percent of the time (Holzworth 1972). These locations include parts of the U.S. Northeast and Northwest, according to 1961-1990 climatology.[c]
Total Precipitation	1.47 m/yr	1.47 m/yr approximates the 95th percentile of annual average precipitation for 824 meteorological stations across the United States. Where available (813 meteorological stations), annual precipitation is the 30-year normal value[b]; where normal values were unavailable, annual average precipitation was calculated from precipitation measured at the station from January 1–December 31, 2016.

[a]National Oceanic and Atmospheric Administration U.S. Climate Atlas for 1981-2010; https://www.ncdc.noaa.gov/climateatlas/.
[b]National Oceanic and Atmospheric Administration 1981–2010 Climate Normals; https://www.ncdc.noaa.gov/data-access/land-based-station-data/land-based-datasets/climate-normals/1981-2010-normals-data.
[c]A graphical view of U.S. rainfall for 1981–2010 climate normals was available for precipitation amount but not precipitation frequency. Instead, we used a map of precipitation frequency based on 1961–1990 climate normals; https://www.ncdc.noaa.gov/cgi-bin/climaps/climaps.pl. Regional patterns of rainfall frequency may have changed between 1961–1990 and 1981–2010.

The sensitivity of modeled PB-HAPs to changes in these meteorological variables was tested. Lower wind speeds and mixing heights affected concentrations the most. Lower wind speeds should increase localized pollutant deposition onto the soil and lower mixing heights reduces the volume of air in which emissions are mixed and diluted. The wind speed and mixing height used for the screening scenario were 1.6 m/s and 226 m, respectively, approximating the 5th percentile values among 824 meteorological stations in the contiguous United States.

2.3.5 Aquatic Food Web

The lake aquatic food web is an important part of the screening scenario because chemical concentrations modeled in fish are used to calculate human ingestion exposure and risks associated with eating contaminated local fish. A biokinetic approach to modeling bioaccumulation in fish is used in the RTR screening scenario for all chemicals except arsenic, for which water-biota and sediment-biota bioaccumulation factors were used instead.

For the biokinetic approach, primary producers (first trophic level) in the TRIM.FaTE lake are algae and macrophytes in the water column and detritus in the sediments (the latter simulated as sediment particles). Algae are represented as a phase in the water column and macrophytes are represented in a single but separate compartment. Zooplankton (another compartment) feed on algae in the water column, while benthic invertebrates (a separate compartment) consume detritus that settles to the sediment compartment. In the water column, small young-of-the-year fish and minnows that feed on zooplankton and phytoplankton are represented by a single water-column herbivore (WCH) compartment. The small fish are in turn consumed by larger or "pan" fish (e.g., bluegills, white perch), represented by a single water-column omnivore (WCO) compartment, which are in turn consumed by the top consumers (e.g., gar, pickerel) represented as a single water-column carnivore (WCC) compartment. The invertebrates in the sediments of the benthic environment support bottom-feeding fish, or benthic omnivores (BO), of small to moderate size, which in turn are consumed by large bottom-feeding fish (e.g., catfish) in the benthic carnivore (BC) compartment. For TRIM.FaTE to provide reasonable predictions of the distribution of a chemical mass (and thereby chemical concentrations) across biotic and abiotic compartments in aquatic systems, the biomass of the biotic compartments must represent all biota in the system and the distribution of biomass among trophic groups (or compartments) must be as realistic as possible.

To support the development of a relatively generic freshwater aquatic ecosystem in which to model bioaccumulation in fish, a literature search, review, and analysis was conducted (ICF 2005). As expected, the diversity of species and food webs across U.S. aquatic ecosystems is substantial, reflecting the wide range of sizes, locations, and physical/chemical attributes of both flowing (rivers, streams) and low-flow water bodies (ponds, lakes, reservoirs). In general, lentic bodies of water (lakes and ponds) can accumulate higher levels of contaminants in both sediments and biota than lotic systems (rivers, streams). That initial research suggested that a lake of approximately 50 hectares (ha) or 120 acres could support high trophic level predatory fish (i.e., WCC).

The RTR Tier 1 scenario includes a 47-hectare (116-acre) lake, given the lake parcel shape and overall size of the defined watershed in the screening scenario. The fish types, biomass, diet fractions, and average individual body weights for the Tier 1 scenario are listed in Exhibit 13. The total biomass for all fish compartments was assumed to be 5.7 grams wet weight per square meter based on Kelso and Johnson (1991) for clear-water lakes in Ontario. That assumption yields health protective (i.e., higher) estimates of chemical concentrations in fish than would the assumption of higher standing fish biomass and fish productivity for lakes characteristic of warmer climates.

For arsenic, freshwater fish bioaccumulation factors (BAFs) and biota-sediment accumulation factors (BSAFs) were identified from the literature (see Attachment A).

Exhibit 13. Aquatic Biota Parameter Values for the TRIM.FaTE Screening Scenario

TRIM.FaTE Compartment	Organisms in Compartment	Biomass		Diet	Avg. Body Weight (kg)
		Areal density ($g_{wet\ weight}/m^2$)	Fraction of Total Fish Biomass		
Algae[a]	green algae, diatoms, blue-green algae	7.95	–	Autotrophic	–
Zooplankton	water fleas, rotifers, protozoans	6.36	–	100% algae[a]	5.7E−8
Macrophyte	hydrilla, milfoil	600	–	–	–
Water column planktivore/ herbivore (WCH)	young-of-the-year, minnows	2.0	0.35	100% zooplankton	0.025
Water column omnivore (WCO)	bluegill, white perch	0.5	0.08	100% water column planktivore	0.25
Water column carnivore (WCC)	largemouth bass, walleye	0.2	0.035	100% water column omnivore	2.0
Benthic invertebrates (BI)[b]	aquatic insect larvae, crustaceans	20	–	detritus in sediments	0.000255
Benthic omnivore (BO)	small catfish, rock bass	2.0	0.351	100% benthic invert.	0.25
Benthic carnivore (BC)	large catfish, sculpins	1.0	0.175	50% benthic invert. 50% benthic omniv.	2.0
Total Fish Biomass [c]		**5.7**	1.00	–	–

Acronyms and abbreviations: Avg. = average; invert. = invertebrate; omniv. = omnivore; % = percent.
[a]Algae is modeled as a phase of surface water in TRIM.FaTE (i.e., surface water has three phases: aqueous, particulate, and algal).
[b]Benthic invertebrates include aquatic insects (e.g., nymphs of mayflies, caddisflies, dragonflies, and other species that emerge from the water when they become adults), crustaceans (e.g., amphipods, crayfish), and mollusks (e.g., snails, mussels).
[c]Total fish biomass does not include algae, macrophytes, zooplankton, or benthic invertebrates.

2.3.6 Using TRIM.FaTE Media Concentrations

The Tier 1 scenario TRIM.FaTE outputs include average PB-HAP concentrations in air and air deposition rates for each year and for each air parcel of the model scenario. In each surface parcel of the scenario, TRIM.FaTE models wet and dry deposition of chemicals to surface soil compartments (and surface water). From surface soils, for each parcel TRIM.FaTE estimates transport of chemicals downward through root-zone and vadose–zone soils as well as runoff and erosion to the lake. For each air parcel, air concentrations are provided. For the lake, the multimedia risk screening approach uses TRIM-FaTE-estimated concentrations in the BC and WCC fish compartments along with adult human fish-ingestion rates to estimate an adult's exposure via local fish consumption. For the farm ingestion exposure calculations, the RTR multimedia screening approach calculates direct exposures via incidental soil ingestion and indirect exposures via transfers from soil and air into various types of produce, livestock, and animal products, which then are ingested by humans.

To ensure health protective calculations, the locations (i.e., parcels) with the highest chemical concentrations predicted by TRIM.FaTE provide the input data for the multimedia exposure calculations. For Tier 1, we assume that the highest air concentrations and deposition rates

occur in the parcels closest to the source. For the farm, those locations also receive the majority of chemical from the rest of the simulated watershed by erosion and runoff. The assumptions are summarized in Exhibit 14.

Exhibit 14. Spatial Considerations—TRIM.FaTE Results Selected for Calculating Farm Food Chain and Fish Media Concentrations and Receptor Exposures

TRIM.FaTE Output Used in Human Exposure Calculations	Representative Compartment	Layout[a]
Concentration in air, for uptake by plants via vapor transfer	Air compartment in air Parcel 2A (air over tilled soil)	Farm (bottom of Exhibit 11)
Deposition rates, for uptake by farm produce	Deposition to surface soil compartment in surface Parcel Farm (tilled soil)	Farm (bottom of Exhibit 11)
Concentration in surface soil, for incidental ingestion by humans and farm animals	Surface soil compartment in surface Parcel Farm (tilled soil)	Farm (bottom of Exhibit 11)
Concentration in soil, for uptake by farm produce and animal feed	Surface soil compartment in surface Parcel Farm (tilled soil)	Farm (bottom of Exhibit 11)
Concentration in fish consumed by fisher	Water column carnivore compartment in lake (50 percent of fish consumed) and benthic carnivore in lake (50 percent of fish consumed)	Lake (top of Exhibit 11)

[a]The Tier 1 screening scenario is based on the combination of exposures from soil, farm produce, and farm animals (from the farmer scenario, spatial layout shown at the bottom of Exhibit 11) and from fish (from the fisher scenario, spatial layout shown at the top of Exhibit 11). Both the farm and the lake are located 0.5 km from the facility.

TRIM.FaTE can output "instantaneous" chemical concentrations at the end of a short, user-specified time step (e.g., 1 hour, 4 hour, 1 day) and also can be configured to calculate temporal averages (e.g., annual averages). For the Tier 1 scenario, TRIM.FaTE results are saved for each 24-hour period, because wind direction and precipitation input to TRIM.FaTE change on a daily (not hourly) basis. The annual average concentration equals the average of the 365 daily estimates. The simulation runs for 50 years, and the concentrations at the end of year 50 are used to estimate human exposures (i.e., we do not use earlier or time-weighted concentrations for PB-HAPs in soils and fish over the duration of the facility operation to estimate human exposures).

For arsenic, cadmium, POM, and dioxins, TRIM-FaTE-estimated concentrations in environmental media are close to steady state (i.e., almost constant from year to year) by year 50. Although mercury concentrations are continuing to increase by year 50 in the screening scenario, the rate of increase in mercury concentrations in soils and fish is much slower by year 50 than in the first 3–4 decades

2.4 Description of Human Exposure and Risk Estimates

This section describes the approach for estimating chemical concentrations in farm food products (Section 2.4.1); estimating human exposures associated with ingestion of those products, incidental ingestion of soil, ingestion of fish, and infant consumption of breast milk (Section 2.4.2); and characterizing screening-level human health ingestion risks (Section 2.4.3). The multimedia risk screening approach uses calculates partitioning of PB-HAPs into farm produce using TRIM.FaTE-estimated chemical concentrations in soil, air concentrations, and wet and dry chemical deposition rates. It also computes total ingestion exposure as described in this section. Attachment B describes the multimedia exposure and risk calculations further. Section 2.4.4 summarizes the Tier 1 assumptions.

2.4.1 Calculating Concentrations in Farm Food Chain Media

As discussed above and shown Exhibit 6, the RTR multimedia risk screening approach estimates PB-HAPs concentrations in farm food products, including:

- exposed and protected fruit,
- exposed and protected vegetables,
- root vegetables,
- beef,
- dairy products,
- pork, and
- poultry and eggs.

PB-HAP concentrations in these products are calculated with algorithms from HHRAP (U.S. EPA 2005a). HHRAP also provides plant- and animal-specific parameter values that can be used to calculate media concentrations, including chemical-specific transfer factors.

2.4.2 Ingestion Exposure

The multimedia risk screening approach estimates average daily doses (ADDs) of ingested chemical, normalized to body weight, for the exposure pathways listed in Exhibit 15.

For the Tier 1 scenario described here, exposure characteristics that would result in a highly health protective estimate of total exposure were selected. The ingestion rate for each medium was set at high-end values (equal to the 90th percentile values for all food types except for fish, which was set at a 99th percentile value). All media are from locations receiving the highest rate of deposition from the modeled source. Although this approach could overestimate total chemical exposure for an individual (i.e., total food ingestion rate is extremely high with an upper percentile rate for each food type), it avoids underestimating exposure for any single farm food type. The exposure characteristics selected for the Tier 1 scenario are summarized in Exhibit 16.

Exhibit 15. Summary of Ingestion Exposure Pathways

Ingestion Exposure Pathway	Medium Ingested	Intermediate Exposure Pathway – Farm Animals[a]	Environmental Uptake Route	
			Medium	Process[b]
Incidental ingestion of soil	Untilled surface soil	N/A	Surface soil	Deposition; transfer via erosion and runoff[c]
Consumption of fish	Fish from local water body	N/A	Fish tissue	Direct uptake from water and consumption of food compartments modeled in TRIM.FaTE[c]
Consumption of breast milk, infants only[d]	Breast milk	N/A	Breast milk	Contaminant ingested by mother partitions to breast milk
Consumption of produce	Aboveground produce, exposed fruits and vegetables	N/A	Air Air RZ soil	Deposition on leaves/plants Vapor transfer to leaves Root uptake

Ingestion Exposure Pathway	Medium Ingested	Intermediate Exposure Pathway – Farm Animals[a]	Environmental Uptake Route	
			Medium	Process[b]
	Above- and belowground produce, protected fruits and vegetables	N/A	RZ soil	Root uptake
Consumption of farm animals and related food products	Beef	Ingestion of forage	Air Air RZ soil	Direct deposition on plant Vapor transfer to plant Root uptake
		Ingestion of silage		
		Ingestion of grain	RZ soil	Root uptake
		Ingestion of soil	Surface soil	Ingestion while grazing
	Dairy (milk)[a]	Ingestion of forage	Air Air RZ Soil	Direct deposition on plant Vapor transfer to plant Root uptake
		Ingestion of silage		
		Ingestion of grain	RZ Soil	Root uptake
		Ingestion of soil	Surface soil	Ingestion while grazing
	Pork	Ingestion of silage	Air Air RZ soil	Direct deposition on plant Vapor transfer to plant Root uptake
		Ingestion of grain	RZ soil	Root uptake
		Ingestion of soil	Surface soil	Ingestion from surface
	Poultry	Ingestion of grain	RZ soil	Root uptake
		Ingestion of soil	Surface soil	Ingestion while foraging on grains spread on ground
	Poultry (eggs)[a]	Ingestion of grain	RZ soil	Root uptake
		Ingestion of soil	Surface soil	Ingestion while foraging

Abbreviations: N/A = not applicable; RZ = root-zone.
[a]Calculation of intermediate exposure concentrations were required only for the farm animal/animal product ingestion pathways.
[b]Process by which HAP enters medium ingested by humans.
[c]Modeled in TRIM.FaTE.
[d]The infant consumption of breast milk pathway is discussed in Section 2.4.2.2.

Exhibit 16. Overview of Exposure Factors Used for RTR Tier 1 Ingestion Screen[a,b]

Exposure Factor	Selection for Screen
Age group evaluated	Infants under 1 year (breast milk only) Children 1–2 years of age Children 3–5 years of age Children 6–11 years of age Children 12–19 years of age Adult (20–70 years)
Body weight (BW; varies by age)	Weighted mean of national distribution (U.S. EPA 2008a, 2011a; see Exhibit B-15 in Appendix B).
Ingestion rate (IR) for farm produce and animal products other than fish (varies by age and food type)	90th percentile of distribution of consumers who produce own food (see Exhibit B-17 in Appendix B); values from Chapter 13 of EPA's 2011 *Exposure Factors Handbook* not adjusted for proportion of those surveyed who did not eat food type during the week covered by the survey.

Exposure Factor	Selection for Screen
Ingestion rate for fish	For adults, 99th percentile *as-prepared* ingestion rate representative of subsistence fisher woman. For children, based on 99th percentile, *as-prepared*, consumer-only, national ingestion rates – adjusted (see Exhibit B-18 in Appendix B).
Exposure frequency (EF)	350 days/year (i.e., 2 weeks away from home per year) (U.S. EPA 2005a).
Exposure duration (ED)	For carcinogens: 70-yr lifetime. For noncancer effects: varies by chemical (i.e., whether effect occurs during critical window in development or effect requires chronic exposure (i.e., more than 7 years of a human lifespan).
Fraction contaminated (FC) (could vary by media consumed)[c]	1.0 (i.e., all ingested fish and farm food products and soils are from most contaminated parcel).
Cooking loss[d]	Assumed to be "typical"; varies by food product (see Attachment B). Cooking losses were not considered for fish consumption because ingestion rates are "as prepared" values.
Food preparation/cooking adjustment factor for fish[e]	Arsenic = 1.5 Cadmium = 1.5 Mercury = 1.5 Dioxin = 0.7 PAH = 1.0

[a]Data for exposure characteristics are presented in Attachment B. Exposure parameter values were based on data obtained primarily from the *Exposure Factors Handbook* (U.S. EPA 2011a). See Attachment B for details.
[b]Exposure factor inputs are used in calculating ADD estimates for each exposure pathway. ADD equations for each pathway evaluated in this screen are provided in Attachment B.
[c]"Fraction contaminated" represents the fraction of food product that is from the contaminated parcels in the screening scenario. Because ingestion rates reflect intake of home produced foods, a fraction contaminated of 1.0 is used.
[d]Cooking loss inputs were included to simulate the amount of a food product that is not ingested due to loss during preparation or cooking, or after cooking.
[e]Because "as consumed," fish consumption rates are used with whole-fish concentrations, adjustment factors are applied to the fish tissue concentrations to reflect changes in concentrations due to cooking. See Attachment B, Section B.6.4.4 for additional discussion.

2.4.2.1 Calculating Average Daily Doses

The multimedia risk screening approach calculates chemical-specific ADDs normalized to body weight (mg PB-HAP per kg of body weight per day). Equations used to calculate ADDs were adapted from the algorithms in EPA's *Multimedia, Multipathway, and Multireceptor Risk Assessment (3MRA) Modeling System* (U.S. EPA 2003a), with the exception of values for exposure factors, which were updated using EPA's 2011 *Exposure Factors Handbook*. The ingestion exposure modeling approach in 3MRA is conceptually similar to that presented in HHRAP and frequently used in risk assessments (Equation 5).

$$ADD_{(y,i)} = \left(\frac{C_{(i)} \times IR_{(y,i)} \times FC_{(i)} \times ED_{(y)}}{BW_{(y)} \times AT_{(y)}} \right) \left(\frac{EF_{(y)}}{365 \text{ days}} \right) \qquad \textbf{Eqn. 5}$$

where:

$ADD_{(y,i)}$ = Average daily dose for age group *y* from food type or ingestion medium *i* (mg chemical/kg body weight-day)

$C_{(i)}$ = Concentration of chemical in food type *i* harvested from the contaminated area (mg chemical/kg food or mg food/L water)

$IR_{(y,i)}$ = Ingestion rate for age group *y* of food type *i* (kg/day or L/day)

$FC_{(i)}$ = Fraction of food type *i* that was harvested from contaminated area (unitless)

$ED_{(y)}$ = Exposure duration for age group *y* (years)

$BW_{(y)}$ = Body weight for age group *y* (kg)

$AT_{(y)}$ = Averaging time for calculation of daily dose (years) for age group *y*, set equal to ED

$EF_{(y)}$ = Annual exposure frequency for age group *y* (days)

A discussion of exposure dose estimation and the equations to calculate ADDs for each ingestion pathway are provided in Attachment B.

2.4.2.2 Infant Ingestion of Breast Milk

A nursing mother exposed to contaminants by ingestion can pass the contaminants to her infant through breast milk (ATSDR 1998). The nursing infant's exposure is estimated from chemical concentrations in breast milk, which are estimated based on the mother's chemical intake rates.

Reports of bioaccumulation of lipophilic compounds, such as polychlorinated biphenyls (PCBs), polychlorinated dibenzofurans (PCDFs), and dioxins (PCDDs), are prevalent in the scientific literature. Due to their high lipophilicity, these compounds partition almost exclusively to the milk fat of breast milk rather than the aqueous phase (U.S. EPA 1998). PCBs, PCDFs, and PCDDs are frequently reported as contaminants in breast milk at concentrations resulting in higher daily doses to infants than their mothers received (Trapp et al. 2008). Other organic compounds, with lower octanol-water partition coefficients, such as phenol, benzene, halobenzenes, and PAHs, are found in both the fat and the aqueous phases of breast milk. Heavy metals, such as arsenic, lead, cadmium, and mercury, have been found in the aqueous phase of the breast milk. For dioxins, a substantial proportion of an individual's total lifetime cancer risk could result from breast feeding in the first year of life; therefore, dioxins are the only PB-HAP evaluated for infant exposures via breast milk for RTR at this time. This approach is consistent with EPA's HHRAP (U.S. EPA 2005a).

This pathway is included in computing total exposure of a person to dioxins over their lifetime for developing the screening threshold emission rate for dioxins. In the absence of congener-specific data, all dioxin congeners were assumed to accumulate in breast milk to the same degree as 2,3,7,8-TCDD.

2.4.3 Calculating Risk

The multimedia risk screening approach calculates excess lifetime cancer risk and non-cancer hazard (expressed as the hazard quotient or HQ) using the calculated ADDs and oral cancer slope factors (CSFs) and toxicity reference doses (RfDs), respectively. The CSFs and RfDs for the PB-HAPs included in the RTR tiered screening approach are presented in Exhibit 17 and are discussed in more detail in Attachment B.

The equations used to estimate cancer risk and non-cancer hazard also are provided in Attachment B. Exposure and risk estimation follows the age-groupings EPA recommends for estimating cancer risks for each life-stage (U.S. EPA 2005c), and total lifetime cancer risk is the sum of those age-specific cancer risks. The approach also conforms with EPA guidance on estimating cancer risks for chemicals with a demonstrated mutagenic mode of action, applying, as appropriate, age-adjustment factor to account for the higher sensitivity of developing children to mutagens compared with adults (U.S. EPA 2005c,d,e).

Exhibit 17. Dose-response Values for PB-HAPs in RTR Ingestion Screening Scenario

PB-HAP	CSF ([mg/kg-day]$^{-1}$)	Source	RfD (mg/kg-day)	Source
Inorganics				
Arsenic compounds (as As)[a]	1.5	IRIS	not critical health endpoint	
Cadmium compounds (as Cd)[a,b]	not available		1E−3	IRIS
Elemental mercury[c]	not available		not available	
Divalent mercury[c]	not available		3E−4	IRIS
Methyl mercury[a]	not available		1E−4	IRIS
Organics				
Benzo[a]pyrene (BaP)[a,d]	1.0	IRIS	not critical health endpoint and no RfD	
2,3,7,8-TCDD[a]	1.5E+5	ORD	not critical health endpoint	

Notes: CSF = cancer slope factor; RfD = reference dose; IRIS = EPA's Integrated Risk Information System; ORD = EPA's Office of Research and Development; TCDD = tetrachlorodibenzo-p-dioxin; POM = polycyclic organic matter; BaP = benzo(a)pyrene; PB-HAP = persistent and bioaccumulative hazardous air pollutant.

[a]Source: U.S. EPA Integrated Risk Information System (see U.S. EPA 2017b).

[b]RfD for cadmium in food (not water).

[c]Exposure to elemental mercury is not assessed in the multipathway screening due to limited information on oral dose-response. Exposure to divalent mercury is not assessed in the multipathway screening due to its higher (i.e., less stringent) RfD and lower bioaccumulation potential in the ingested food products in the screening, relative to methyl mercury.

[d]EPA considers BaP to be a mutagenic carcinogen (IRIS).

Accordingly, estimated individual cancer risks for BaP (and other POM), which has a mutagenic mode of carcinogenesis, were adjusted upward to account for the stronger mutagenic potency of these compounds during childhood, as specified by EPA in its supplemental guidance for cancer risk assessment (U.S. EPA 2005c). Specifically, cancer potency for BaP (and all POM) is assumed to be tenfold greater for the first 2 years of life and threefold greater for the next 14 years (U.S. EPA 2005c,e). The cancer potency adjustment for chemicals with a mutagenic mode of action is discussed in Attachment B, Section B.5.1.

2.4.4 Summary of Tier 1 Assumptions

As emphasized previously, the screening scenario created for evaluating PB-HAP emissions from RTR facilities is intended to be health protective to prevent underestimating risk. The scenario also is intended to avoid grossly overestimating risk to the point where no emissions screen out. The degree to which the scenario is health protective overall depends on the combination of parameters for which "upper-end" percentile or health protective values are used instead of nationwide mean values. Exhibit 18 summarizes influential parameter values for this scenario and indicates the likely degree of health protectiveness associated with each. Although this summary does not quantify overall health-protective bias, it does demonstrate qualitatively that the scenario generally overestimates exposure and therefore is unlikely to screen out facilities that might pose risks to human health.

Exhibit 18. Summary of RTR Tier 1 Screening Scenario Assumptions

Characteristic	Value	Neutral or Health Protective?	Comments on Assumptions
General Spatial Attributes			
Farm location	375 m from source; generally downwind	Health Protective	Location influences soil and air concentrations and deposition rates used to calculate chemical levels in farm produce.
Lake location	375 m from source; generally downwind	Health Protective	Location influences contamination levels in fish.
Surface soil properties	Typical values or national averages	Neutral	Based on existing EPA documentation and other references.
Size of farm parcel	About 4 ha	Health Protective	Relatively small parcel size results in higher chemical concentration (i.e., not "diluted" by averaging with less contaminated areas farther from the source).
Size of lake	47 ha; about 3 m average depth	Health Protective	Lake is just large enough to support an aquatic ecosystem with high trophic-level fish. The higher water content of larger or deeper lakes would provide more dilution of chemicals received.
Meteorological Inputs			
Total precipitation	1.47 m/yr	Health Protective	Reflects particularly rainy areas of the United States (see Exhibit 12 for source). Higher precipitation rates result in more wet deposition over the modeled domain.
Precipitation frequency (with respect to impacted farm/lake)	Two thirds of total precipitation fall on farm/lake and watershed	Health Protective	Most precipitation occurs when the farm/lake are downwind of the source (see Exhibit 12 for additional justification).
Wind direction	Farm/lake are downwind 40 percent of the time	Health Protective	Reflects areas of the United States with particularly persistent wind flows (see Exhibit 12 for additional justification). Farm/lake located in the predominantly downwind direction. Chemical deposition over the watershed increases when winds blow from the facility into the watershed.
Wind speed	1.6 m/sec	Health Protective	Reflects areas of the United States with particularly low wind speeds (see Exhibit 12 for source). Slower wind speeds lead to more chemical deposition closer to the facility (i.e., over the farm/lake).
Air temperature	298 K	Neutral	Recommended default value listed in HHRAP (U.S. EPA 2005a). See Exhibit 12 for additional context.

Characteristic	Value	Neutral or Health Protective?	Comments on Assumptions
Mixing height	226 m	Health Protective	Reflects areas of the United States with particularly low mixing heights (see Exhibit 12 for source). Lower mixing heights decrease the volume of air in which chemical mixing occurs, resulting in higher chemical concentrations in air and higher chemical deposition rates to the watershed.
Watershed and Water Body Characteristics			
Evaporation of lake surface water	700 mm/yr	Neutral	Based on sensitivity analyses, value is not expected to under- or over-estimate concentration in surface water.
Surface runoff into lake, onto farm	Equal to 40 percent of total precipitation	Health Protective	Based on typical water flow in wetter U.S. locations; higher runoff results in greater transfer of chemical to lake/farm.
Surface water turnover rate in lake	About 12 turnovers per year	Neutral	Consistent with calculated water balance; reasonable in light of published values for small lakes. Might overestimate flushing rate if water inputs are also overestimated. Note that after evapotranspiration, remaining water volume added via precipitation is assumed to flow into or through lake.
Soil erosion from surface soil into lake	Varies by parcel; ranges from 0.002 to 0.005 kg/m^2-day	Health Protective	Erosion rates calculated using the universal soil loss equation (USLE); input parameter values were selected to favor higher erosion rates (i.e., to move more chemical from the watershed into the lake). Might underestimate erosion for locations with steeper slopes or more exposed soils. .
Soil erosion from surface soil onto farm	About 0.003 kg/m^2-day	Health Protective	Erosion rates calculated using the USLE; inputs parameter values were selected to favor higher erosion rates. Might underestimate erosion for locations susceptible to high erosion rates; might overestimate erosion for locations where a farm is not an erosion sink in the watershed. Higher erosion increases concentration in soil (and farm foods).
Aquatic food web structure and components	Multilevel; includes large, upper trophic-level fish	Health Protective	Inclusion of upper trophic-level fish and absence of large-bodied herbivore/detritivore fish favor higher bioaccumulation of chemicals in consumed fish. Linear food-chains (instead of more realistic food webs) maximizes concentration of bioaccumulative chemicals in higher trophic-level fish.

Characteristic	Value	Neutral or Health Protective?	Comments on Assumptions
Parameters for Estimating Concentrations in Farm Food Products			
Fraction of plants and soil ingested by farm animals that is contaminated	1.0 (all food and soil from contaminated areas)	Health Protective	Assumes all livestock feed sources (including grains and silage) are derived from land parcel with highest chemical concentrations.
Soil- and air-to-plant transfer factors for produce and related parameters	Typical (see Attachment B for details)	Neutral	Obtained from peer-reviewed and standard EPA reference sources.
Biotransfer factors for efficiency of uptake by animal of chemical in food/soil	Typical (see Attachment B for details)	Neutral	Obtained from peer-reviewed and standard EPA reference sources.
Bioavailability of chemicals in soil (for soil ingested by animals)	1.0 (relative to bioavailability of chemical in plant matter)	Health Protective	Probably overestimates bioavailability in soil; many chemicals are less bioavailable in soil than in plants.
Human Ingestion Exposure Assumptions			
Combined ingestion of farm food chain media and fish	High-end, subsistence ingestion rates for both (i.e., 90th to 99th percentile)	Health Protective	Assuming combined high-end consumption, consistent with subsistence farming and fishing, likely overestimates exposure to any single individual.
Ingestion rates for all farm produce/ livestock products	Person obtains all food from local farm; 90th percentile ingestion rate for each of 10 foods	Health Protective	All food from contaminated farm; total food ingestion rate (across 10 food categories) for individual exceeds expected body weight-normalized ingestion rates (prevents underestimating any individual food type).
Fish ingestion rate	Adult: 373 g/day Child age groups: 1–2: 108 g/day 3–5: 159 g/day 6–11: 268 g/day 12–19: 331 g/day	Health Protective	The adult rate, the 99th percentile value for adult females from Burger (2002), is considered representative of subsistence fishers. Rates for children are based on the 99th percentile, consumer-only fish ingestion rates from EPA's 2002c *Estimated Per capita Fish Consumption in the United States*. Rates were adjusted to represent the age groups used in the screening scenario. See Attachment B for a detailed discussion.
Exposure frequency	Consumption of contaminated food items occurs 350 days/yr	Health Protective	All meals from local farm or fish products, except for two weeks per year when consumer is elsewhere.
Body weight	Mean of national distribution	Neutral	Note that this does not affect the body-weight-normalized ingestion rates for produce and animal products.

Characteristic	Value	Neutral or Health Protective?	Comments on Assumptions
Chemical-Specific Characteristics			
General chemical properties used in fate and transport modeling (Henry's law, K_{ow}, etc.)	Depends on chemical	Neutral	Obtained from peer-reviewed sources; intended to be representative of typical behavior and characteristics.
"General" physical properties (plant matter density, aquatic life biomass, algal growth rate, etc.)	Varies	Neutral	Obtained from peer-reviewed sources; intended to be representative of typical behavior and characteristics.
Dose-response values		Neutral to Health Protective	Values used are those determined to be appropriate for risk assessment by OAQPS; values are developed to be health protective.

Tier 1 screening threshold emission rates were calculated by conducting iterative model simulations using the screening scenario described above to determine emission rates for arsenic, cadmium, mercury, dioxins, and POM that correspond to a cancer risk of 1-in-one million or a chronic non-cancer HQ of 1. Given the generally health protective nature of the scenario inputs, these screening threshold emission rates are appropriate for a Tier 1 screen.

The Tier 1 screening approach is, by design, generic and health protective. It was constructed for quick application to a large number of facilities in a source category with the least chance of returning false negatives for risk potential. Once the Tier 1 screen is complete, however, facilities whose emissions exceed the emission screening threshold emission rate for any PB-HAP can be scrutinized further.

2.5 Evaluation of Screening Scenario

For a given source category, all facilities are reviewed to determine if emissions of any of five PB-HAPs are reported. If any facility emits one or more of the PB-HAPs, the Tier 1 screen is applied. Facility emissions of each PB-HAP are compared with the Tier 1 screening threshold emission rate to determine the resulting SV. In Tier 1, the magnitude of an SV has a limited implication for relative risk. For example, exceeding the screening threshold emission rate by a factor of 60 for dioxins does not imply an actual cancer risk of 60-in-one million. Rather, an SV of 60 implies that it is highly unlikely that the actual risk would exceed 60-in-one million, and likely would be much lower.

The Tier 1 methods evaluate congener-specific differences in fate and transport and in toxicity for dioxins and POMs. The final results are reported in 2,3,7,8-TCDD equivalents and benzo[a]pyrene equivalents, respectively.

The screening scenario developed for assessing multipathway human health risk for EPA's Risk and Technology Review has been subjected to a series of evaluations. As described previously, the major PB-HAP categories of concern for this assessment are arsenic (Section 2.5.1), cadmium compounds (Section 2.5.2), mercury compounds (Section 2.5.3), dioxins (Section 2.5.4), and POM (Section 2.5.5). The scenario evaluations focused on assessing the behavior

of these HAP categories in the environment, accumulation of these chemicals in fish and farm food products, and the exposure pathways and chemicals that contributed most to human risks.

2.5.1 Arsenic

Arsenic is a natural component of the earth's crust and is found naturally in minerals, most often as a compound with sulfide (HSDB 2009). Some of the largest anthropogenic sources of arsenic to air are nonferrous metal mining and smelting, pesticide application, coal combustion, wood combustion, and waste incineration (ATSDR, 2007). Inorganic arsenic has two stable oxidation states, +3 and +5 (arsenite and arsenate, respectively).

2.5.1.1 Behavior in the Environment

Depending on its chemical form and source, arsenic can undergo a variety of transformations, including oxidation or reduction, ligand exchange, precipitation out of solution when arsenate or arsenite combine with iron, sulfur, or chloride in water, and biotransformation to or from organic forms. Arsenic released to air from sources evaluated by the RTR program is primarily in particulate form as highly soluble oxides (pentavalent arsenate or As(V) as the arsenite ions $H_2AsO_4^-$ and $HAsO_4^{2-}$; trivalent arsenite or As(III) as arsenous acid, H_3AsO_3).Trivalent arsenite predominates in releases to air from industrial processes. Compounds detected in air include arsenic trisulfide from coal combustion, organic arsines from oil combustion, and arsenic trichloride from waste incineration (ATSDR 2007).

Arsenic is found in soil as a result of natural processes and anthropogenic sources including ash residue from power plants, smelting facilities, mining wastes, and industrial waste. Arsenic is found in mixtures of mineral phases (e.g., co-precipitates, sorbed to soil particles). Arsenic adsorbs to particulate matter in soils and sediments and tends to concentrate in the upper layers of soil. Iron content strongly affects arsenic adsorption to soil particles (ATSDR 2007). Arsenic has low to moderate mobility in clay soils, in which particles are small and total particle surface area high, and much higher mobility in loamy and sandy soils, for which particle sizes are larger, with less surface area per particle and per unit weight solid material.

Potential volatilization of arsenic from soil depends on its original form when deposited from air. Some microorganisms can methylate some inorganic arsenicals, with a proportion of dimethyl and trimethyl arsenic volatilizing to air. Soil particles adsorb other arsenic compounds, depending on iron oxide and organic carbon content, limiting bioavailability and future volatilization (HSDB 2009).

Arsenic exists primarily in the pentavalent form under oxidizing conditions, such as found in surface water, and in the trivalent form in reducing conditions, such as found in groundwater (ATSDR 2007). Arsenic transport in groundwater is determined by the chemical form of arsenic and adsorption is based on the other materials present in the aquifer, as well as the pH of the water (ATSDR 2007). Arsenic strongly sorbs onto sediments, and bacteria and fungi methylate arsenic compounds to form dimethyl and trimethylarsines (HSDB 2009).

Bioaccumulation of arsenic in plants and organisms in water depends on factors including type of water body, organism type, status in the food chain, concentration, and route of uptake. Bioconcentration of arsenic occurs primarily in algae and invertebrates; however, bottom feeders and predatory fish might accumulate arsenic from ingestion of sediments along with prey. Arsenic does not appear to biomagnify in the aquatic food chain (ATSDR 2007). In a study of bioaccumulation data for fish and invertebrates, bioconcentration factor (BCF) values ranged from 0.048 to 1,390 (U.S. EPA 2003b).

2.5.1.2 Arsenic Speciation Modeling Approach

Although inorganic arsenic exists in the environment as two predominant species—trivalent arsenic (arsenite) and pentavalent arsenic (arsenate)—with distinct characteristics, the modeling approach in TRIM.FaTE aggregated the two species for several reasons:

- EPA's Integrated Risk Information System (IRIS) provides one oral reference dose for inorganic arsenic (i.e., an undifferentiated species). Although some investigators report that trivalent arsenic is more toxic than pentavalent, the studies are in the context of aquatic toxicity to fish and invertebrates, not human ingestion toxicity.
- All sources of arsenic biotransfer factors (which represent the ratio of chemical concentration in produce to the chemical concentration in soil) and bioaccumulation factors (which represent the ratio of chemical concentration in various aquatic trophic levels to the chemical concentration in water) that we reviewed report values for total inorganic arsenic, without specifying oxidation state.
- NEI emissions data usually are reported in unspeciated terms, like "total inorganic arsenic." Estimating speciation for those emissions would require substantial research or simplifying assumptions.
- Other EPA programs also model a single inorganic arsenic species (e.g., the recent *Risk Assessment of Spent Foundry Sands In Soil-Related Applications,* U.S. EPA 2014c).

Modeling a single form of arsenic was implemented in both TRIM.FaTE and the multimedia exposure and risk calculations:

- For parameters for which different values are available for trivalent and pentavalent arsenic, the more health-protective value is used. If we could not predict which value would be more protective a priori, the screening approach was performed with each value, and the value that resulted in higher risk was chosen.
- Organic arsenic is not explicitly modeled. Ignoring potential methylation of inorganic arsenic in soils and sediments is a health-protective assumption because inorganic arsenic has been considered more toxic than organic forms (ATSDR 2007), although some recent studies suggest that a portion of ingested organic arsenic might be converted back to inorganic forms in animals (Carlin et al. 2005; U.S. EPA 2003b). For farm products and livestock, biotransfer factors are reported for total arsenic only. Similarly, BAF and BSAF factors for fish are based on studies primarily of trivalent or unspecified inorganic arsenic in water and sediments, respectively. The BAF and BSAF factors are presumably based on total arsenic in the fish compared with dissolved inorganic arsenic in the environmental medium (U.S. EPA 2003b).

2.5.1.3 Arsenic Aquatic Bioaccumulation Modeling Approach

In modeling transfers of arsenic through the aquatic food web, empirical BAFs and BSAFs were used instead of the biokinetic approach, which is used for the other PB-HAPs in TRIM.FaTE. Fish tissue concentrations of arsenic are calculated as the product of water column and sediment arsenic concentrations (from TRIM.FaTE) and the empirical BAFs and BSAFs, respectively, for freshwater. The screening approach, therefore, estimates arsenic concentrations in fish in much the same way as in produce.

The approach can save time and effort because several arsenic-specific parameter values (e.g., gill uptake rate, metabolic transformations, absorption efficiency across the gut, form-specific

elimination rates) are needed to implement the TRIM.FaTE aquatic food chain. Moreover, those arsenic values would need to be coded in the TRIM.FaTE Java library for the aquatic invertebrate and fish compartments. Although the empirical BAF approach is not mass-balanced (i.e., does not remove the arsenic transferred to fish compartments from the sediment and water-column compartments), we believe that the approach is adequate for the RTR screen for three reasons: (1) total chemical mass in the fish compartments typically is small compared with chemical mass in the surface water and sediment compartments; (2) RTR models focus on concentrations at year 50, by which time simulated environmental concentrations are typically close to steady-state; and (3) in past applications, the biokinetic food-chain models have been calibrated using measured chemical concentrations in algae, zooplankton, and fish at different trophic levels.

Arsenic concentrations tend to be the same or lower at successive trophic levels (e.g., it biodiminishes instead of bioaccumulates; Williams et al. 2006). BAF values for arsenic also tend to decrease with increasing water concentrations, indicating some physiological regulation by fish (Williams et al. 2006).The BAF/BSAF-based approach developed for arsenic, can be applied to other chemicals in future.

2.5.1.4 Arsenic—Bioavailability in Soils Ingested by Humans

At the screening level, contaminants ingested with foods and with soils are assumed to be 100 percent bioavailable (i.e., all of the chemical ingested is absorbed from the exposure medium). Although unlikely to be true (e.g., a few percent is expected to be eliminated with feces, and some might be excreted in bile), the assumption is health protective and close to accurate for many organic chemicals. Inorganic chemicals, on the other hand, might or might not be well absorbed, particularly from soils with minerals and organic carbon to which inorganic chemicals adsorb. For example, the RfD for cadmium is higher (less stringent) for its ingestion with food (1E-03) than cadmium ingested with water (5E-04), meaning it is less bioavailable in food than in water. For arsenic, the CSF is based on ingestion in drinking water. EPA's Superfund Program has therefore investigated arsenic bioavailability from incidental ingestion of arsenic-contaminated soils compared with its bioavailability in water to assist in setting target concentrations in soils for site remediation.

Swine have served as an in vivo animal model by which to evaluate the bioavailability of chemicals in soils for many years (primates also have been used on occasion; U.S. EPA 2012b). In the past decade, in vivo animal models have been extended to mice. Examining the results across species, EPA developed an estimate of the bioavailability of arsenic in soils to mammals, finding the upper 95th percentile value to approximate 0.60. The upper 95th percentile values for swine, monkeys, and mice were estimated as 0.609, 0.327, and 0.502, respectively (U.S. EPA 2012b). Thus, the value of 0.60 is likely to be a health-protective value for humans and is used to estimate exposure from ingestion of soil.

2.5.1.5 Arsenic Concentrations in Ingestible Products

Most non-inhalation exposure to arsenic outside of occupational settings is through dietary intake. Arsenic in agricultural soils is largely immobile, and remains in the upper soil levels (ATSDR 2007). Most plants would accumulate arsenic initially released to air from stationary facilities by uptake by the roots from the soil or by arsenic deposition on the leaves. Larsen et al. (1992) found that arsenic emitted from burning arsenic-treated wood was taken up by kale from arsenic deposited on leaves and that arsenic in potatoes and carrots came from both atmospheric deposition to leaves and root uptake. In general, arsenic accumulation by plants depends on the form(s) of arsenic in the environment and the species of plant. In the United States, seafood (i.e., marine and estuarine fish and shellfish), meat, and rice have been

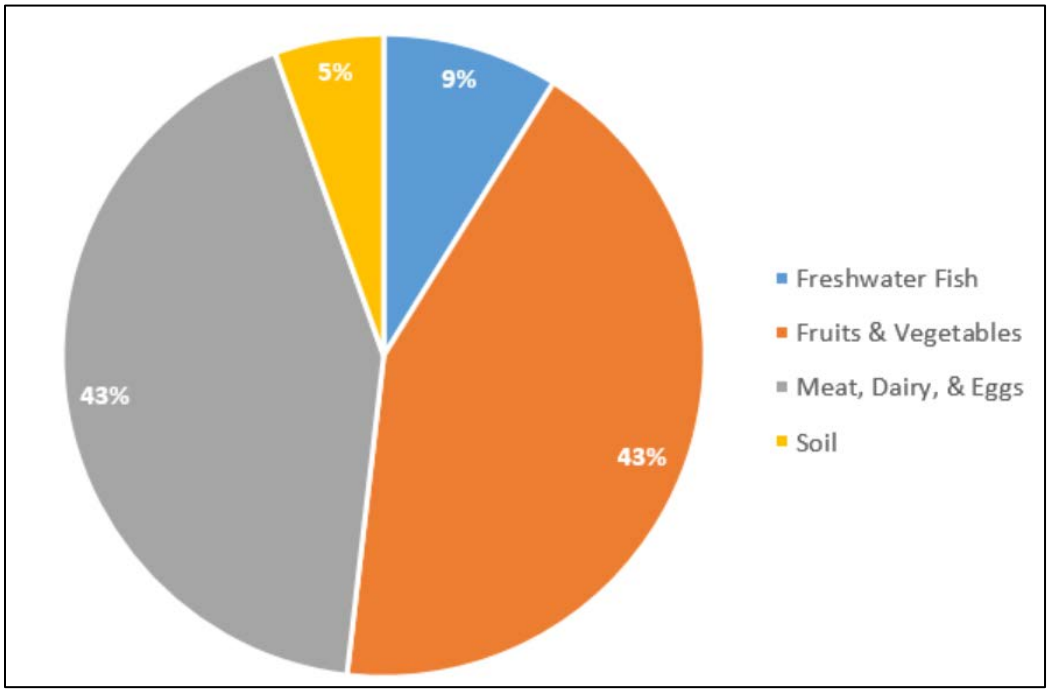
reported to contain the highest levels of arsenic. Arsenic has also been detected at low levels in other foods (ATSDR 2007).

For the RTR screening scenario, the relative arsenic concentration estimates were consistent with relative concentrations reported for soils, produce, and other farm-grown meat products.

2.5.1.6 Lifetime Average Daily Dose (LADD)

Exhibit 19 presents the contribution of the various ingested media to overall arsenic exposure for different age groups. Ingestion of freshwater fish contributes approximately 9 percent of total exposure, whereas direct soil ingestion contributes 5 percent. The remaining exposure is fairly evenly distributed across farm products such as fruit, vegetables, dairy, eggs, and meat. That distribution appears reasonable, given that data (U.S. EPA 2012b; Williams et al. 2006) indicate that freshwater fish accumulate substantially less arsenic from water than marine or estuarine fish species.

Exhibit 19. Estimated Media Contributions to Arsenic Ingestion Exposures and Lifetime Cancer Risks



2.5.2 Cadmium Compounds

Some of the largest anthropogenic sources of cadmium to air are facilities that process, mine, or smelt cadmium-zinc ores or cadmium-zinc-lead ores, coal- and oil-fired boilers, other urban and industrial facilities, phosphate fertilizer manufacturing facilities, road dust, and municipal sewage sludge incinerators (ATSDR 2008). Cadmium has one stable oxidation (or valence) state, +2.

2.5.2.1 Behavior in the Environment

Once emitted to air, cadmium in or on small airborne particles can travel long distances before being deposited; however, most cadmium released to air is found in soils near facilities that released it (ATSDR 2008).

The mobility of cadmium in soil depends strongly on soil pH, clay content, and availability of organic matter—factors that determine whether the cadmium is dissolved or adsorbed in surface soil. In general, cadmium adsorbs to soil particles in the surface layers of the soil profile, but to a lesser degree than many other heavy metals (HSDB 2005). Cadmium also binds strongly to organic matter, rendering the metal relatively immobile in highly organic soils. Nonetheless, some plant species absorb cadmium efficiently via their roots, thus providing an entry point for cadmium into the terrestrial food chain (ATSDR 2008).

Cadmium in air can enter surface waters directly via wet and dry deposition and indirectly from runoff and erosion of cadmium deposited to soil. Most cadmium compounds entering the water column are quickly removed through adsorption to suspended particles or algae, with eventual sedimentation. Cadmium that remains in the water column is expected to exist primarily as dissolved cations, which are readily bioavailable to aquatic organisms.

Freshwater fish accumulate cadmium primarily through direct uptake of dissolved cadmium through the gills, but also can accumulate cadmium ingested with their foods (Reinfelder et al. 1998; Chen et al. 2000; Saiki et al. 1995). Although some biomagnification of cadmium has been reported for aquatic food chains in saltwater systems, bioaccumulation in freshwater systems occurs mainly at lower trophic levels (Chen et al. 2000), primarily in phytoplankton and zooplankton and in filter-feeding macroinvertebrates (e.g., bivalves; Croteau et al. 2005). Biomagnification factors (BMFs) of less than 1 generally have been reported for fish at higher trophic levels, indicating that cadmium concentrations can biodiminish in fish from one trophic level to the next (Chen et al. 2000; Mason et al. 2000).

For the RTR screening scenario, the partitioning behavior modeled in TRIM.FaTE was consistent with monitoring data for cadmium in the environment.

2.5.2.2 Concentrations in Foods

Most exposure to cadmium outside of occupational settings is through dietary intake. Available data indicate that cadmium accumulates in plants, aquatic organisms, and terrestrial animals, offering multiple ingestion exposure pathways (ATSDR 2008). Measured cadmium levels in foods vary based on type of food, agricultural and cultivation practices, atmospheric deposition rates, characteristics of environmental media, and presence of other anthropogenic pollutants. Meat and fish generally contain lower amounts of cadmium overall, but cadmium can be highly concentrated in certain organ meats, such as kidney and liver (ATSDR 2008). In a study of cadmium concentrations in 14 food groups (including prepared foods), meat, cheese, and fruits generally contained low levels of cadmium (ATSDR 2008).

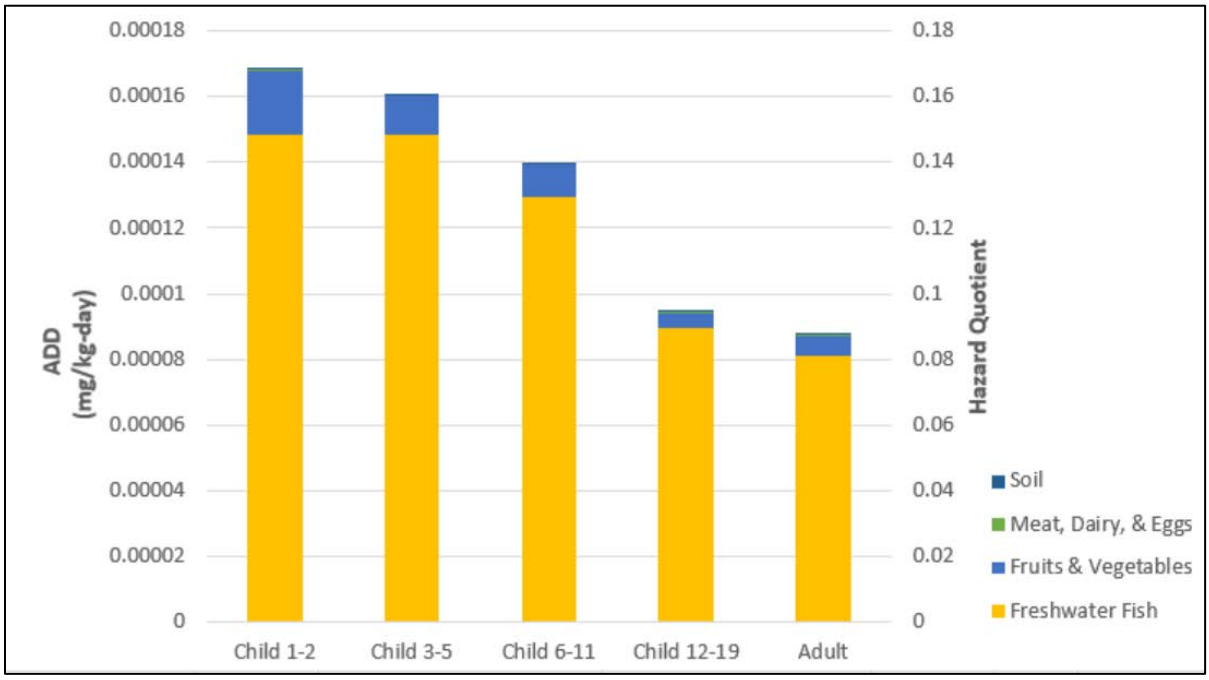
The cadmium concentrations estimated with the RTR screening approach were consistent with values reported in the literature. The products with higher reported cadmium levels in the literature also contained the higher modeled concentrations.

2.5.2.3 Average Daily Dose

Exhibit 20 presents the average daily dose (ADD) received through each of the ingested media, by age category, at a unit emission rate of 1 gram cadmium per day. This chart can be used to evaluate the relative contributions of ingested media to the chemical HQ. (Using the cadmium screening threshold emission rate would simply change the y-axis ADD and HQ labels; the media contributions relative to each other would be unchanged). Fish ingestion dominates risk for cadmium across all age categories, accounting for over 90 percent of the ADD for all groups. The combined contribution from all other ingested media accounts for less than 10 percent of the total ADD for all age groups. Most of the additional exposure was from ingestion of fruits and

vegetables. The highest ADD is for children aged 1–2 years because of their high food ingestion rate relative to body weight; thus, the exposure corresponding to this group determines the screening threshold emission rate for cadmium (i.e., the rate at which the HQ for this age category equals 1.0).

Exhibit 20. Estimated Media Contributions to Cadmium Ingestion Exposures and HQs



2.5.3 Mercury Compounds

Some of the largest anthropogenic sources of mercury to air are facilities that process, mine, or smelt mercury ores; industrial/commercial boilers; fossil fuel combustion activities (primarily coal); cement production facilities; other urban and industrial facilities; and medical and municipal waste incinerators (ATSDR 1999). These facilities can emit a mixture of elemental and divalent mercury, mostly in the gaseous phase, with some divalent forms particle-bound (U.S. EPA 1997).

2.5.3.1 Behavior in the Environment

Once emitted to air, mercury form and valence state can change in the atmosphere. Elemental mercury (Hg_2^0) vapor is the most prevalent species of mercury in the atmosphere at ambient temperatures. Due to its high vapor pressure, generally more than 98 percent of elemental mercury remains in the atmosphere, where it is dispersed long distances on a global scale.

Divalent mercury can sorb to particles, but a large proportion is in vapor phase and quite reactive with surfaces. Divalent mercury therefore deposits from air relatively quickly, moreso in its reactive gaseous phase (RGP) than in particulate phase (Landis et al. 2004; Cohen 2005). Thus, divalent mercury deposits locally via wet, dry, or reactive deposition, where it adsorbs tightly to soil particles (U.S. EPA 1997). Divalent mercury also deposits to surface waters, where it sorbs to particulate or dissolved organic carbon (Driscoll et al. 2007). Divalent mercury in soil also can be methylated by microbes or reduced to elemental mercury, which volatilizes back into the atmosphere. Most divalent mercury from atmospheric deposition will remain in the soil

profile, however, in the form of inorganic compounds bound to soil organic matter (ATSDR 1999). Although complexing with organic matter and several minerals in soil significantly limits further aqueous mercury transport (e.g., via leaching or runoff), the tendency of mercury to form these complexes depends on soil conditions such as pH, temperature, and soil humic content (U.S. EPA 1997). For example, mercury strongly adsorbs to humic materials and sesquioxides in soil at pH > 4 and in soils with high iron and aluminum content (ATSDR 1999). More mercury in soil is likely to reach surface waters via erosion than via runoff or leaching.

Mercury also deposits to lakes directly from air. Once in the water body, microbes can methylate divalent mercury, particularly in the sediments. In addition, divalent and methyl mercury can be further reduced to elemental mercury, which can volatilize to the atmosphere. Solid forms of inorganic mercury compounds could adsorb to particulates in the water column or partition to the sediment bed (U.S. EPA 1997).

The solubility of mercury in water depends on the species and form of mercury present as well as properties of the water such as pH and chloride ion concentration (ATSDR 1999). Low pH favors methylation of mercury in the water column and sediments, typically performed by sulfur-reducing bacteria in anaerobic conditions (e.g., anaerobic layer of sediments). Methyl mercury is typically of greatest concern because it readily bioaccumulates *and* biomagnifies in aquatic organisms. Once ingested by fish, methylmercury distributes to all tissues and binds to proteins, thereby sequestering large amounts in muscle.

A considerable amount (25–60 percent) of both divalent mercury compounds and methyl mercury is strongly bound to particulates in the water column (U.S. EPA 1997). The remaining mercury is dissolved. Most of the elemental mercury produced as a result of reduction of divalent mercury volatilizes back into the atmosphere.

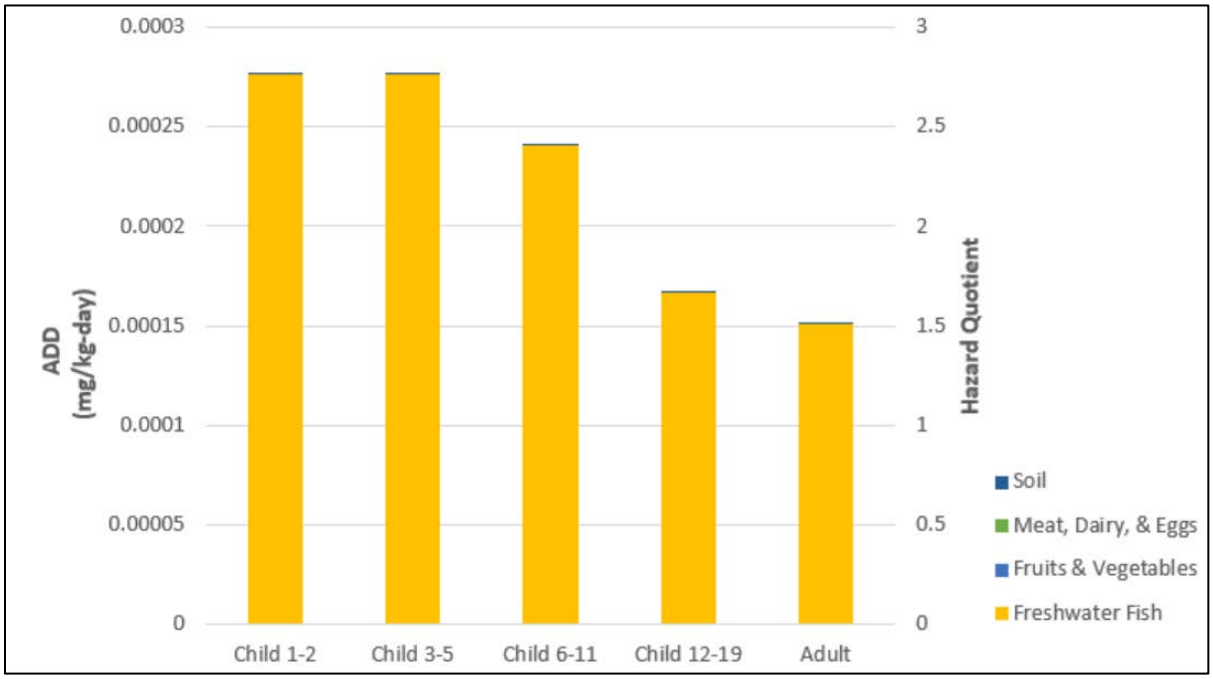
For the screening scenario, the partitioning behavior modeled in TRIM.FaTE generally was consistent with trends noted in the literature. Divalent mercury was the most prevalent species in modeled surface soil, surface water, and sediment compartments, while methyl mercury was the dominant species in fish.

2.5.3.2 Concentrations in Foods

Available data indicate that mercury bioaccumulates in plants, aquatic organisms, and terrestrial animals, providing multiple ingestion exposure pathways (U.S. EPA 1997; ATSDR 1999). Low levels of mercury are found in plants, with leafy vegetables containing higher concentrations than potatoes, grains, legumes, and other vegetables and fruits (ATSDR 1999; U.S. EPA 1997). Cattle demethylate mercury in the rumen and, therefore, store very little of the mercury they ingest by foraging or consuming silage or grain. Thus, mercury content in meat and cow's milk is relatively low (ATSDR 1999). Concentrations of methyl mercury in fish are generally highest in larger, older fish at the higher trophic levels (U.S. EPA 1997).

Although data on mercury in foods other than fish are not abundant in the literature, estimated relative mercury concentrations across food types are generally consistent with available environmental monitoring data. The ingested media that most influenced the mercury HQs in the model are presented in Exhibit 21. As shown, the dominant exposure pathway for all age groups is ingestion of fish. In top trophic level fish, methyl mercury accounts for more than 95 percent of total mercury.

Exhibit 21. Estimated Media Contributions to Methyl Mercury Ingestion Exposures and HQ



2.5.3.3 Average Daily Dose

Exhibit 21 presents the ADD received through each of the ingested media, by age category, at a unit emission rate of 1 gram divalent mercury per day. This chart can be used to judge the relative contribution of ingested media to the methyl mercury HQ. (Using the divalent mercury screening threshold emission rate would simply change the y-axis ADD and HQ labels; the media contributions relative to each other would be unchanged). As shown, fish is the dominant exposure pathway across all age categories, accounting for nearly 100 percent of the ADD for each group. The combined contribution of all other ingested media accounts for less than 1 percent of the total ADD for all age groups. The high degree of exposure to methyl mercury through fish ingestion is attributed to the ease with which this compound bioaccumulates and biomagnifies in fish and to the health protective ingestion assumptions used in the screening scenario. The highest ADD corresponds to children aged 1–2 years; thus, the exposure corresponding to this group is used to determine the screening threshold emission rate for mercury.

2.5.4 Dioxins

Incineration and combustion processes are believed to be the primary sources for emissions of chlorinated dioxins (ATSDR 1998). The five stationary source categories that generate the vast majority of 2,3,7,8-TCDD emissions in the United States are municipal waste incineration, medical waste incineration, hazardous waste kilns from Portland cement manufacturing, secondary aluminum smelting, and biological incineration. Forest fires and agricultural field burning also account for a large proportion of dioxins in soils and in ambient air (ATSDR 1998).

2.5.4.1 Behavior in the Environment

Dioxins emitted to the atmosphere can be transported long distances in vapor form or bound to small particulates, depositing to soils and water bodies primarily via precipitation events in

otherwise pristine locations far from all sources. Although airborne dioxins are susceptible to wet and dry deposition, most dioxins emitted to the atmosphere through incineration/combustion processes that vent from tall stacks are not deposited close to the source (ATSDR 1998).

In soil, dioxins strongly adsorb to organic matter and show very little vertical movement, particularly in soils with a high organic carbon content (ATSDR 1998). Most dioxins deposited in soil are expected to remain in the subsurface soil profile, with erosion of contaminated soil particles the only significant mechanism for transport to water bodies. Dioxin volatilization from the soil surface can contribute to plant uptake via foliage because of the very high bioaccumulation potential for TCDD (and presumably the other dioxins/furans) by plant leaves from air (Trapp 1995).

Dry deposition of dioxins from the atmosphere to water bodies is another important transport process. Because of their hydrophobic nature, most dioxins entering the water column are expected to adsorb to suspended organic particles or partition to bed sediment, which appears to be the primary environmental sink for this chemical group (U.S. EPA 2004c). Although dioxins bound to aquatic sediment particles eventually become buried in consolidated sediments, some resuspension and remobilization of congeners can occur if sediments are disturbed (e.g., by benthic organisms; ATSDR 1998).

Bioaccumulation factors in fish are high as a result of the lipophilic nature of chlorinated dioxins. Although the processes by which freshwater fish accumulate dioxins are not well understood, both fish and invertebrates bioaccumulate congeners that have partitioned to sediment or have become suspended in water (U.S. EPA 2004c). Because most dioxins in the aquatic environment are adsorbed to suspended particles, however, direct uptake from the water is unlikely to be the primary route of exposure for most aquatic organisms at higher trophic levels (ATSDR 1998). At lower trophic levels, the primary route of exposure appears to be through uptake of water in contaminated sediment pores, and the primary route of exposure in the higher trophic levels appears to be through food chain transfer. Following ingestion, some fish can slowly metabolize certain congeners, such as 2,3,7,8-TCDD, and release the polar metabolites in bile. This process ultimately might limit bioaccumulation at higher trophic levels (ATSDR 1998).

For the RTR screening scenario, the partitioning behavior modeled in TRIM.FaTE was consistent with the behavior of 2,3,7,8-TCDD expected in the natural environment. Dioxins also readily partition into breast milk, which has a high fat content, due to the lipophilic nature of these compounds.

2.5.4.2 Concentrations in Foods

The primary source of non-inhalation exposure to dioxins outside of occupational settings is through dietary intake (ATSDR 1998). Available data indicate that dioxins concentrate in plants, aquatic organisms, and animals, offering multiple ingestion exposure pathways. Actual congener levels in foods, however, can vary based on type of food, agricultural and cultivation practices, atmospheric deposition rates, characteristics of environmental media, and presence of other anthropogenic pollutants. Dioxins appear to enter the terrestrial food chain primarily through vapor-phase uptake by plant foliage, which then can be consumed by larger animals. Another major source of animal exposure to dioxins is through ingestion of contaminated soil.

Observed trends indicate that meat, dairy, and fish consumption are the dominant exposure pathways for environmental dioxins, comprising 90 percent of dioxin dietary intake (ATSDR 1998). Consistent with the literature, the modeled concentration of 2,3,7,8-TCDD in the fish compartment for the screening scenario was at least one order of magnitude greater than

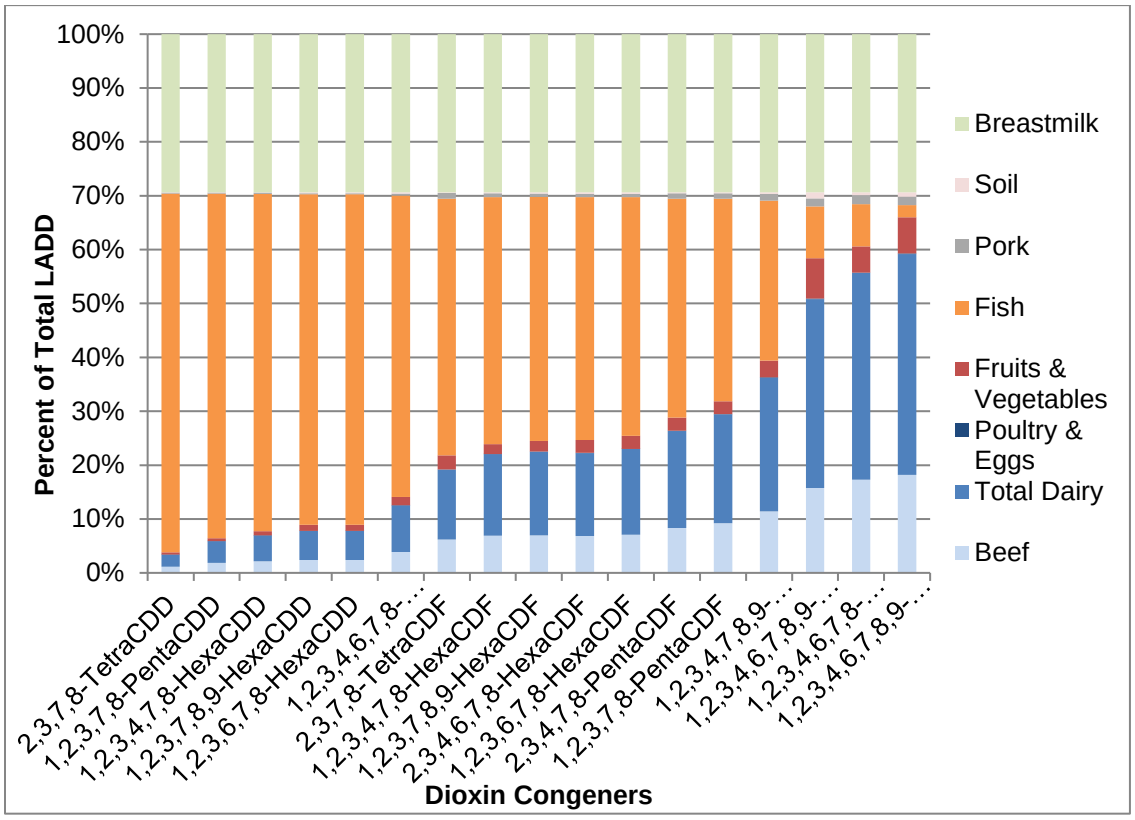
concentrations in the other compartments. Among the compartments with the lowest concentrations were fruits and vegetables, which do not readily accumulate 2,3,7,8-TCDD.

Ingestion of breast milk during infancy and fish ingestion contribute to over 97 percent of lifetime dioxin exposure for 2,3,7,8-TCDD-equivalents in the screening scenario. Daily intakes of 2,3,7,8-TCDD from cow's milk, produce, and fish have been reported in the literature to comprise 27 percent, 11 percent, and 10 percent, respectively, of the total daily intake in the general population (ATSDR 1998). Some studies note that specific subpopulations, such as subsistence farmers and fishers, however, might have very different exposure profiles in which fish, meat, and dairy drive congener exposure (ATSDR 1998). Given the subsistence diet modeled in the RTR screening scenario, the high exposure from consumption of fish is appropriate within the context of this screen.

2.5.4.3 Lifetime Average Daily Dose (LADD)

The contributions of the various ingested media to the lifetime average daily dose (LADD) (and thus lifetime cancer risk) for the modeled dioxin congeners are presented in Exhibit 22. Based on the models and assumptions used, exposure via breast milk ingestion during the first year of life accounts for approximately 30 percent of the lifetime exposure for all congeners, while exposure via ingestion of fish, soil, and the various farm food chain media varies across congeners largely because of differences in physiochemical properties that drive environmental transport processes (e.g., K_{ow}, molecular weight). Some differences are also likely due to different biological half-lives of congeners in plants and animals. The relative contribution of farm-raised livestock and produce to total congener exposure is higher for the more highly chlorinated dioxins, which are less soluble in water.

Exhibit 22. Estimated Media Contributions to Dioxin Ingestion Exposures



2.5.5 Polycyclic Aromatic Hydrocarbons and Other Polycyclic Organic Matter

PAHs can enter the atmosphere as a result of a variety of combustion processes, both natural and anthropogenic. Stationary emission sources account for approximately 80 percent of total annual anthropogenic PAH emissions (ATSDR 1995). Although the primary source of stationary source PAH emissions is thought to be residential wood burning, other processes such as power generation; incineration; coal tar, coke, and asphalt production; and petroleum catalytic cracking are also major contributors (ATSDR 1995).

2.5.5.1 Behavior in the Environment

PAHs and other POM emitted to the atmosphere can travel long distances in vapor form or attached to small particles, or they can deposit relatively close to an emission source via wet or dry deposition onto surface waters, soils, and vegetation. In the atmosphere, PAHs are found primarily in the particle-bound phase, and atmospheric residence time and transport distances are highly influenced by climatic conditions and the size of the particles to which they are bound (ATSDR 1995). Lower molecular weight PAHs are more volatile than higher molecular weight compounds, and the smallest PAH, two-ringed naphthalene, is highly volatile and remains largely (e.g., 98%) in vapor phase in air. Thus, naphthalene is not evaluated as a PB-HAP.

As a result of sustained input from anthropogenic sources, PAHs are ubiquitous in soil. High molecular weight PAHs, such as benzo(a)pyrene, strongly adsorb to organic carbon in soil, which limits the mobility of these compounds following deposition to soil (ATSDR 1995).

Most PAHs enter the water column directly through atmospheric deposition (ATSDR 1995). Following deposition onto surface waters, approximately two-thirds of PAHs adsorb strongly to sediment and suspended particles, while only small amounts of the smaller molecules revolatilize back to the atmosphere (ATSDR 1995). Aquatic organisms can accumulate PAHs via uptake from water, sediment, or food. Although fish and other organisms readily take up PAHs from contaminated food (e.g., aquatic insects, other benthic invertebrates, smaller fish), biomagnification in fish does not occur because fish can rapidly metabolize PAHs (ATSDR 1995). As a result, concentrations of PAHs in fish have generally been observed to decrease with increasing trophic levels (ATSDR 1995). Sediment-dwelling invertebrates can accumulate PAHs via filter feeding, consumption of detritus, and direct uptake from sediment pore water (ATSDR 1995).

For the screening scenario, the partitioning behavior of benzo(a)pyrene is generally consistent with trends reported in the literature.

2.5.5.2 Concentrations in Food

The primary source of non-inhalation exposure to benzo(a)pyrene, other PAHs, and other POM outside of occupational settings is through dietary intake. Exposure concentrations depend on the origin of the food (higher values are often recorded at contaminated sites). Moreover, because PAHs are created by combustion of organic materials, cooking foods at high temperatures (e.g., grilling or broiling to a surface char) or smoking meats can create PAHs in and on foods from any location.

PAHs can bioaccumulate in aquatic organisms and terrestrial animals through uptake of contaminated water, soil, and food. These compounds are readily metabolized by vertebrates, however, so bioaccumulation in fish and livestock generally is considered to be insignificant (ATSDR 1995). Plants can accumulate vapor-phase PAHs through open leaf stomata during the

day, and some particulate-phase POM deposited to leaf surfaces might transfer through the leaf cuticle. Monitored PAH concentrations in plants, however, tend to be below detection limits.

PAH concentrations in meat can vary widely, from below detection levels to high concentrations, particularly in smoked meats. Similar concentrations have been reported for fish, with smoked fish concentrations sometimes quadruple those found in terrestrial animals. Because PAH concentrations are highest in products that are smoked or grilled, most of the available data for benzo(a)pyrene in food is for meat and fish products preserved or prepared using these processes. It is possible that concentrations of PAHs in foods grown near facilities that emit PAHs to air are lower than in foods grilled or smoked.

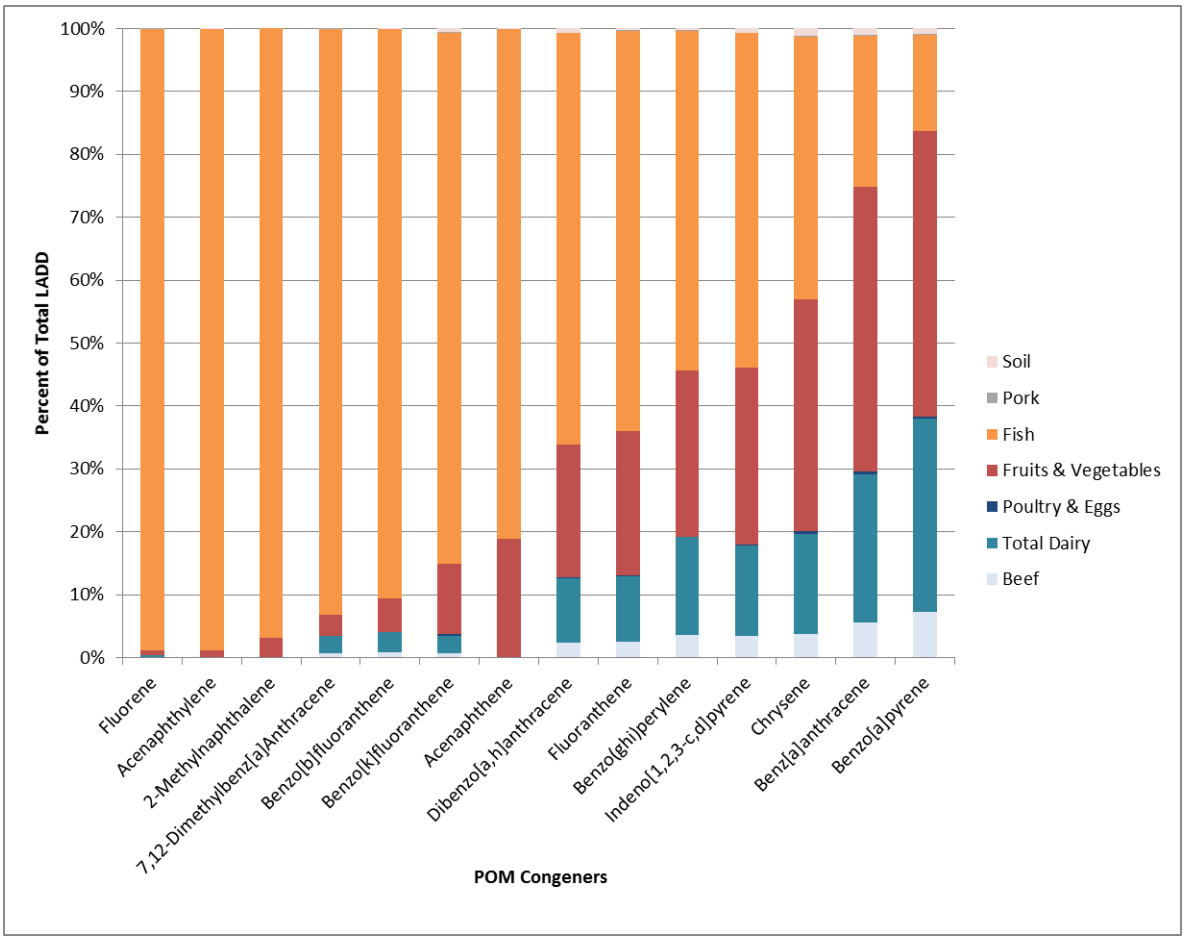
For the RTR screening scenario, estimated BaP concentrations in foods grown near a facility are generally lower than the reported ranges for BaP in cooked or smoked fish and meat products and generally are predicted to be near or below likely detection limits.

2.5.5.3. Lifetime Average Daily Dose

The contributions of the various ingested media to the LADD (and thus lifetime cancer risk) for various PAHs frequently reported in NEI for RTR source categories and that are fully parameterized in TRIM.FaTE are presented in Exhibit 23. As shown, the contribution of different ingested media to total ingestion of each compound varies, although fish, beef, dairy, fruits, and vegetables comprise between 90 and 99 percent of exposure for different PAHs.

This variability can be accounted for in part by differences in the physiochemical properties that drive the environmental fate and transport processes of these PAHs (e.g., K_{ow}, molecular weight, chemical structure), differences in the PAH-specific half-life in abiotic media, and the degree to which the PAHs are metabolized by plants and animals. The variability across exposure pathways is consistent with information provided in the literature. The PAHs with lower molecular weights tend to be more volatile and more soluble in water than the PAHs with higher molecular weights; hence fish ingestion is a more important pathway for lighter-weight PAHs.

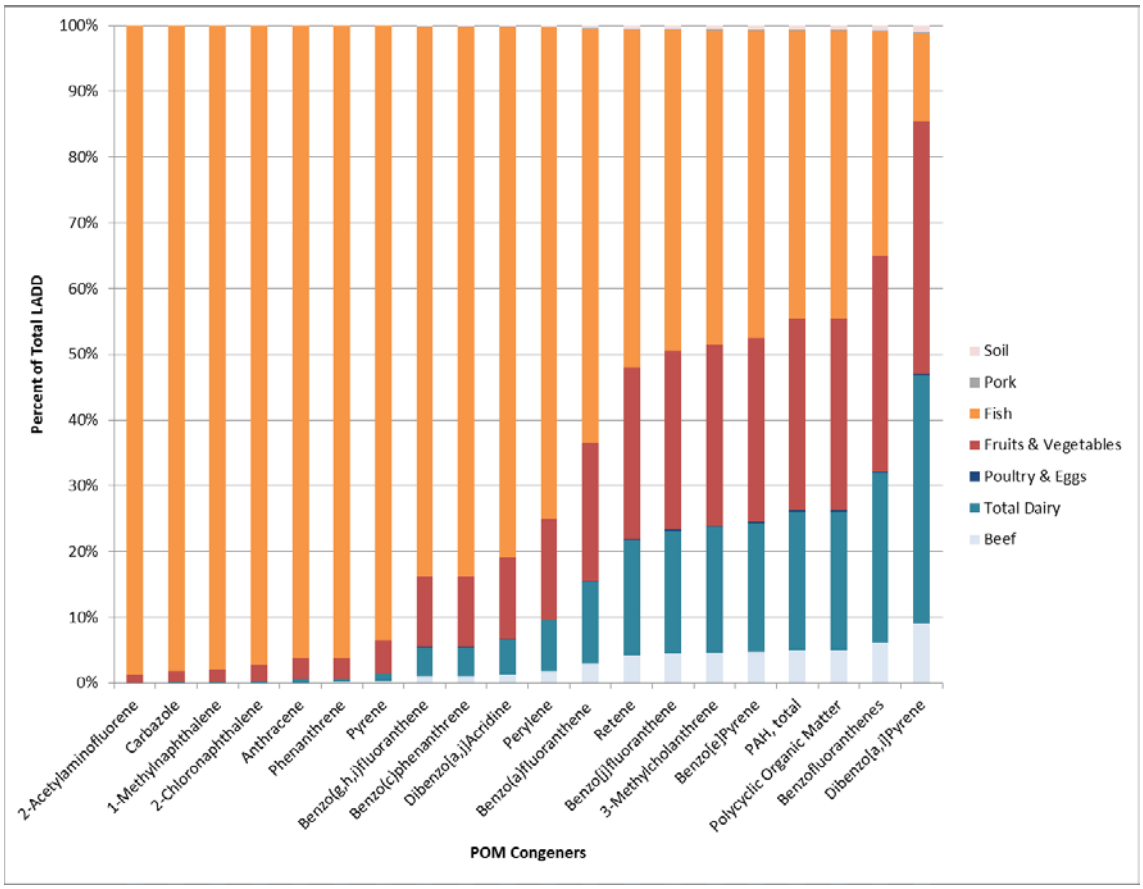
Exhibit 23. Estimated Media Contributions to POM Ingestion Exposures[a]



[a]POM chemicals that are fully parameterized in TRIM.FaTE.

Exhibit 24 shows the contribution of the various ingested media to the LADD (and thus lifetime cancer risk) for each of the additional 22 POM chemicals evaluated for RTR, but not fully parameterized in TRIM.FaTE. The same trend with lower to higher molecular weight POM observed in Exhibit 23 is evident in Exhibit 24.

Exhibit 24. Estimated Contributions of Modeled Food Types to Additional POM Chemical Ingestion Exposures[a]



[a]POM chemicals not fully parameterized in TRIM.FaTE (note that benz(a)anthracene/chrysene and benzo(b+k)fluoranthene are not provided in this exhibit because benz(a)anthracene/chrysene is modeled as "polycyclic organic matter" and benzo(b+k)fluoranthene is modeled as benzo(k)fluoranthene) for RTR screens due to data limitations).

2.5.6 Summary

Trends in the predominant media contributing to exposure for the PB-HAP categories assessed to date by EPA's RTR Program are generally consistent with trends in measured data and partitioning behavior reported in the literature. This assessment reveals that fish ingestion is a major route of exposure for cadmium, mercury, dioxins, and the lower molecular weight POM chemicals. For arsenic, farm-raised produce and livestock also contribute to ingestion exposures, with freshwater fish ingestion accounting for a comparatively small percentage of total exposure. For the lipophilic organic PB-HAPs (i.e., dioxins and PAHs), farm-raised food products also contribute substantially to ingestion exposures, with beef and dairy contributing significantly to the LADD.

3. Tier 2 Screen

This section describes the Tier 2 screening methods and assumptions. Section 3.1 provides an overview and compares and contrasts the Tier 2 approach with the Tier 1 approach. Construction of the library of Tier 2 screening threshold emission rates is presented in Section 3.2. Finally, Section 3.3 describes implementation of Tier 2: the one-time set of runs to define the library of emission thresholds values, and the facility-specific implementation of Tier 2 screens.

3.1 Overview of Approach

In Tier 2, some of the conservative assumptions in the Tier 1 screen are replaced with more site-specific information. These fall into two general categories: environmental assumptions and exposure assumptions. The remainder of this section describes in detail the Tier 2 environmental assumptions (Section 3.1.1), exposure assumptions (Section 3.1.2), and implementation of Tier 2 (Section 3.1.3).

3.1.1 Tier 2 Environmental Assumptions

In Tier 2, location-specific data on five environmental variables comprising two general types of data are evaluated:

- Meteorological characteristics: (1) the fraction of time the wind blows toward each farm and lake (based on wind direction), (2) wind speed, (3) precipitation rate, and (4) air mixing height; and
- (5) Distance from facility: Locations of farms/gardens and fishable lake(s) relative to the facility[14] (including the absence of a fishable lake).

Those inputs were selected for Tier 2 modifications based on:

- Relative influence on estimated risks,
- Ease of implementation in TRIM.FaTE (e.g., can the parameter be modified as a user input, or must the model code be modified and tested?), and
- Ease of obtaining reliable parameter values more representative of specific locations.

A series of TRIM.FaTE simulations was performed that systematically varied the values used in the screening scenario for four of the five selected variables listed above (i.e., lake location, wind speed, precipitation rate, and mixing height). Wind direction affects only whether the chemical mass advects toward the farms and lakes, so the effect of site-specific wind directions can be evaluated outside of the TRIM.FaTE simulations in eight octants. The values of each of the four variables were varied independently from one another (i.e., other variable values held constant). The values (i.e., four to six different values for each variable, including the original, Tier 1 scenario values) were selected using statistics on U.S. meteorological data or professional judgment to capture the expected range in the data. Four to six values per variable resulted in a reasonable number of total runs and condition combinations.

For distance from the facility, lakes and farms/gardens were modeled at five different distances. In implementing Tier 2, actual locations of fishable lakes near the facility are determined and these locations are used in selecting the appropriate distance from the facility. For the fisher scenario, the Tier 2 screen allows for aggregate contributions from multiple facilities located near actual lakes.

3.1.2 Tier 2 Exposure Assumptions

In Tier 2, several aspects of the exposure scenario are reevaluated. Although subsistence fisher and farmer scenarios still are evaluated (if needed based on the results of the Tier 1 screen), the ingestion is disaggregated into two exposure scenarios that represent a subsistence farmer

[14]The lake size also changes with each lake distance allowing for a constant ratio between watershed and erosion area compared with lake area within the TRIM.FaTE modeling structure.

(who ingests fruits and vegetables, meat and dairy products, eggs, and soil) and a subsistence fisher (who ingests only fish).

In addition, another exposure scenario is introduced in Tier 2: a gardener scenario. In many settings, it is unrealistic to assume that a farm, that provides all exposure media for the farmer scenario, can exist in the area surrounding a facility. The gardener scenario often will provide a more realistic exposure scenario where the gardener grows and eats fruits and vegetables from a home garden and eats eggs from home-raised hens, but does not produce and eat meat or dairy products. The gardener also is assumed to ingest the same high-end amount of soil as a farmer. The gardener scenario is further defined as either an urban gardener or a rural gardener based on Census data for the location being assessed. If the census block closest to the facility is in a Census-defined urbanized area based on population density, the area around the facility is considered urban (otherwise, it is considered rural). The rural gardener consumes produce at the same 90th percentile ingestion rates as the famer; however, the urban gardener consumes less food from the garden (i.e., mean ingestion rates) because they likely also consume store-bought produce and have a smaller footprint in which to grow produce. The gardener scenario uses the same media concentration data that are developed for the farmer scenario.

Tier 2 differs in two additional ways from the Tier 1 exposure scenario:

- Incidental soil ingestion and farm-food ingestion is evaluated at each of 40 locations around the facility (five distances and eight directions), not just at one location close to the source. The results reported for Tier 2 for the farmer and the gardener are from the location with the maximum SV (i.e., which corresponds to the area with maximum deposition).
- Fish are harvested at a sustainable rate based on lake size, so that the fisher might need to fish from multiple lakes to meet the subsistence fish ingestion rate.

3.1.3 Implementation of Tier 2

The overall implementation of the Tier 2 multipathway screen is illustrated in Exhibit 25. The steps on the left in Exhibit 25, which are discussed in Section 3.2, need only to be conducted once. The “one-time” steps include running 64 combinations of meteorological parameters through TRIM.FaTE for each of five separate lake-distance scenarios and five separate farm/garden-distance scenarios (i.e., distance from the facility). Resultant concentrations are processed assuming separate fisher, farmer, and gardener (both rural and urban gardener) ingestion scenarios. These runs result in Tier 2 REFs and screening threshold emission rates for all Tier 2 exposure scenarios.

The steps on the right in Exhibit 25 are conducted for each facility using a Microsoft® Access™ tool developed for RTR screens. For each facility, the tool identifies the same meteorology station used in RTR inhalation assessments, and records the values for the four relevant meteorological parameters at that station. The tool also computes distances from the facility to real lakes in the RTR lake dataset within a user-specified distance (default is 50 km) of the facility and matches the lakes to their respective directional “octant” relative to the facility. These five parameter values become the set of facility-specific inputs for Tier 2.

The lake dataset for RTR is based on U.S. Geological Survey (USGS) database, which includes information on location and surface area, use or type designation, and name (if available) for all lakes in the United States. The dataset consists of hundreds of thousands of water bodies classified as “Lake/Pond” or “Reservoir” but not designated for disposal, evaporation, or treatment. To focus on lakes that can support fishing of upper trophic level fish, a minimum lake

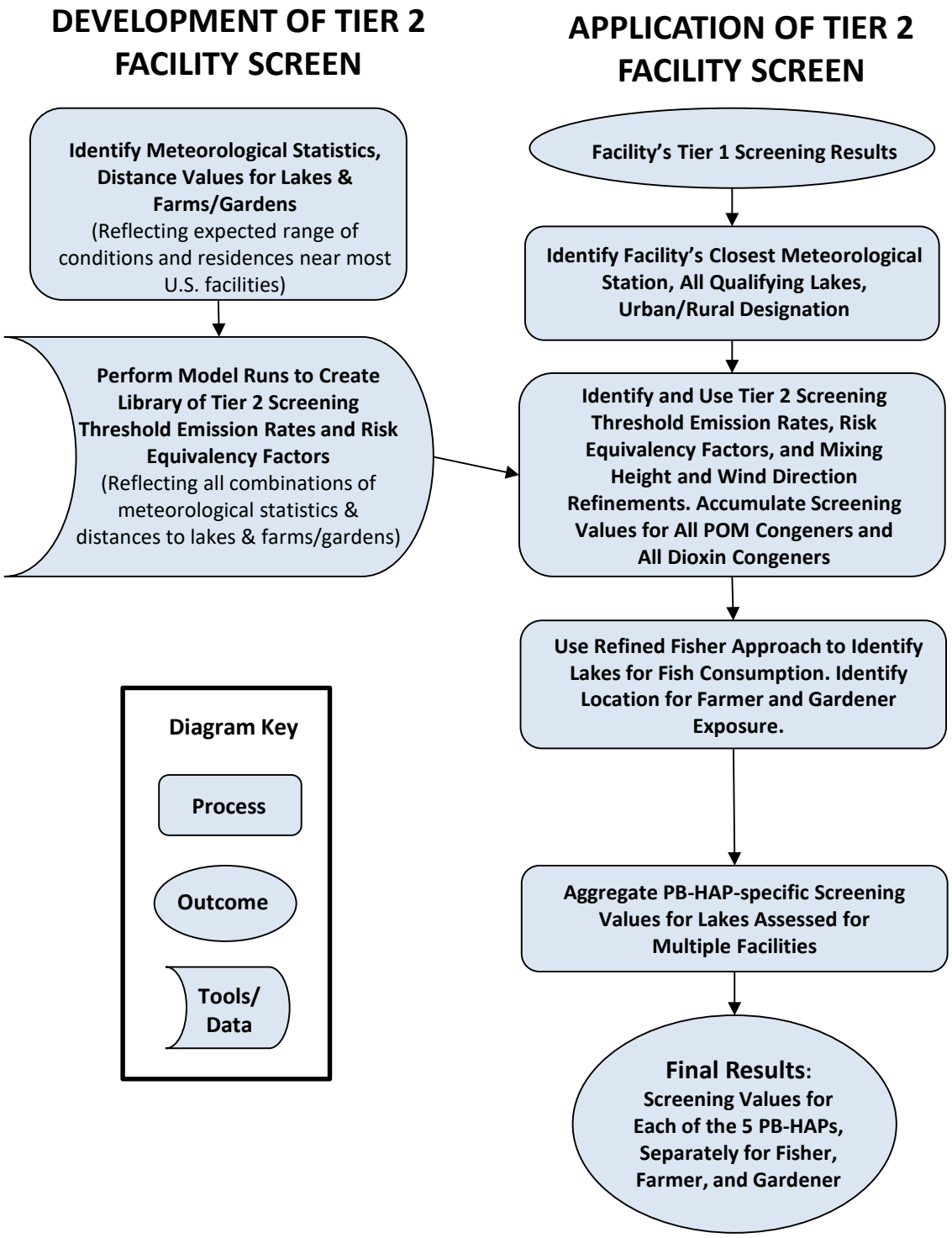
surface area of 25 acres is recommended. Very large lakes and bays (i.e., those larger than 100,000 acres) are not included because their watersheds are too large and their lake dynamics are too complex to realistically model in the TRIM.FaTE system. Lakes and bays larger than 100,000 acres include the Great Lakes, the Great Salt Lake, Lake Okeechobee, Lake Pontchartrain, Lake Champlain, Green Bay, and Galveston Bay.

For the purposes of Tier 2, a "relevant" lake meets the size and designation criteria discussed in the previous paragraph. Second, the lake names are reviewed, and lakes with names suggesting uses related to disposal, evaporation, or treatment are removed from the dataset (sometimes the name indicates one of these uses while the USGS designations do not; for example, the Gavin Fly Ash Impoundment would not be included in the screening process). Third, the lakes around the facility that remain after the first two processing steps are ranked in order of highest to lowest PB-HAP concentrations in fish. These rankings are then used to refine the Tier 2 screen for the fisher.

For the farmer and gardener scenarios, each farm/garden-distance scenario is evaluated from the "one-time" modeling, and the facility's emissions are compared with the Tier 2 screening threshold emission rates for those farm/garden-distance values and site-specific meteorological values. As noted previously, media concentrations developed for the farmer scenario are used for the gardener scenarios.

As with Tier 1, a facility screens out if none of the chemical specific facility emissions exceed the applicable Tier 2 screening threshold emission rates; otherwise, additional evaluation might be needed (i.e., Tier 3; see Section 4).

Exhibit 25. Basic Process for Implementing the Tier 2 Multipathway Screen



3.2 Library of Tier 2 Screening Threshold Emission Rates

This section describes the "one-time" steps presented on the left side of Exhibit 25. Section 3.2.1 discusses use of site-specific meteorological data, and Section 3.2.2 discusses the potential locations of lakes, farms, and gardens. Section 3.2.3 discusses the Gardener exposure

scenario, which is introduced to the screening assessment during the Tier 2 screening. Finally, Section 3.2.4 describes the creation of a library of Tier 2 screening threshold emission rates, REFs, and mixing height refinements.

Attachment D provides information on all the TRIM.FaTE variables that were considered for the Tier 2 screen.[15] Using the criteria above, we ranked variables as high, medium, or low priority. Meteorological parameters and lake location were high priority and feasible to implement with input from public databases.

3.2.1 Meteorological Data

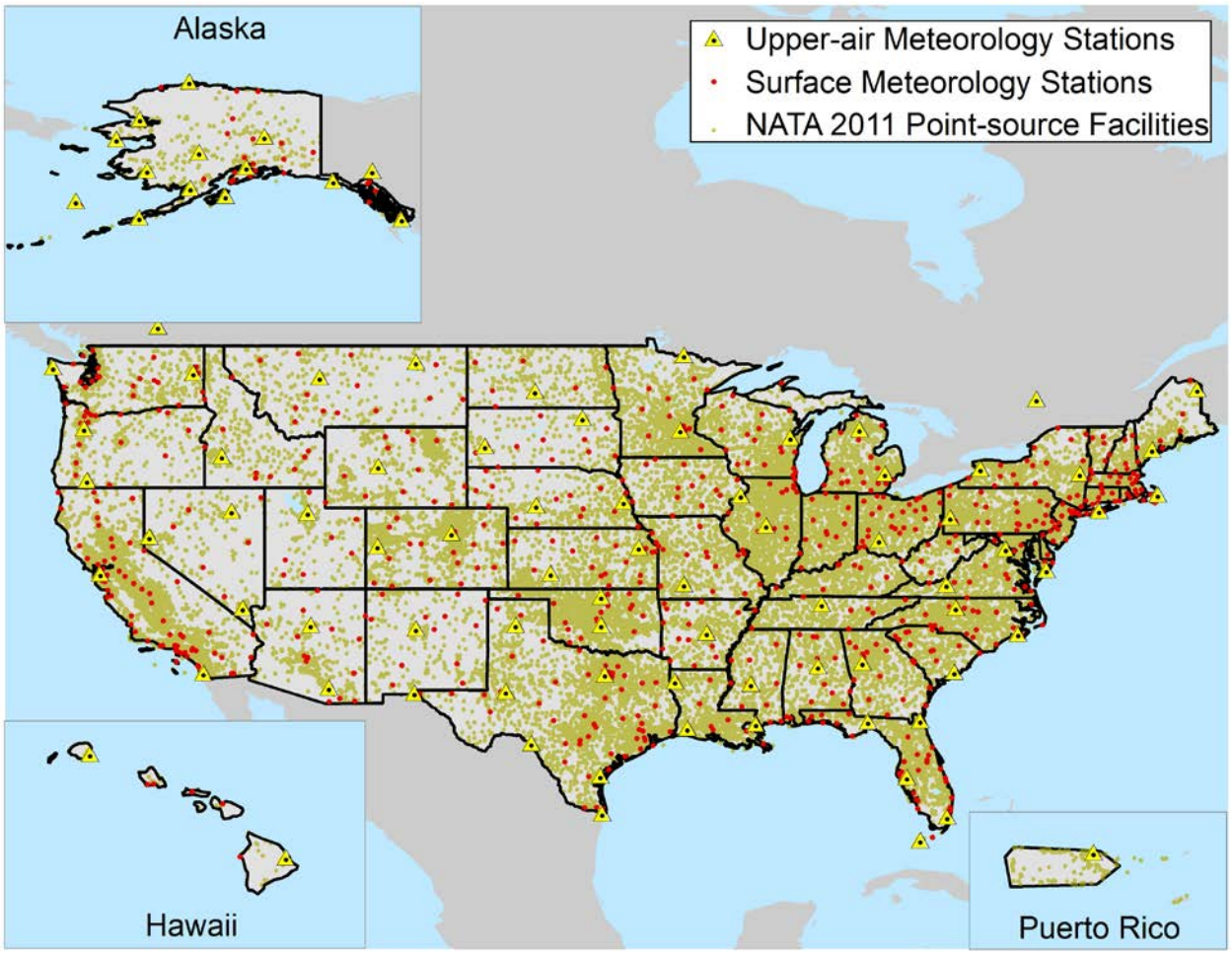
We created a database of the relevant U.S. meteorological data for 824 surface stations paired with their closest upper-air stations located throughout the country. The hourly surface data cover 2016 and are the same AERMOD-ready data used for RTR inhalation modeling. Exhibit 26 shows the surface and upper-air meteorological stations represented in this database along with the locations of U.S. point-source facilities from the 2011 National Air Toxics Assessment (NATA; U.S. EPA 2015). Generally, the spatial density of the surface meteorological stations is similar to the spatial density of the 2011 NATA facilities—i.e., more stations and facilities in areas with more people: in the Great Lakes region, along the East and West Coasts, and in the Southern Plains; and fewer stations and facilities in the Rockies (except Colorado) and Northern Plains.

The meteorological database includes annual summary statistics on wind direction, wind speed, precipitation, and mixing heights. We gathered wind information in directional octants that could be linked to the direction (with respect to the facility location) of the relevant lakes and of the hypothetical locations of farmers/gardeners (facility screening is discussed in Section 3.3). The area around a facility is divided into the eight octants shown below, representing possible wind directions (e.g., N is north, NE is northeast).

N: >337.5 to 360 or >0 to 22.5 degrees	S: >157.5 to 202.5 degrees
NE: >22.5 to 67.5 degrees	SW: >202.5 to 247.5 degrees
E: >67.5 to 112.5 degrees	W: >247.5 to 292.5 degrees
SE: >112.5 to 157.5 degrees	NW: >292.5 to 337.5 degrees

[15]Only TRIM.FaTE parameters were considered for site-specific refinements in Tier 2. Exposure characteristics are considered to be generally consistent across different locations and facilities.

Exhibit 26. The Locations of Meteorological Stations Used in RTR Modeling, and Locations of NATA 2011 Point-Source Facilities for Reference



Note: The 2011 NATA was the most recent, comprehensive, finalized dataset of nationwide point source emitters of hazardous air pollutants, and it is used here only for illustrative purposes. The meteorology locations shown here are those used in the RTR modeling described in this report; NATA used a different meteorology dataset.

From the hourly weather data, we calculated or gathered the annual statistics listed below for each of the 824 surface stations.

- Number of hourly observations
- Number of hours with calm winds or no wind data reported
- Fraction of time the wind blows into each octant (after excluding missing and calm wind hours)
- Median wind speed blowing into each octant (after excluding calm winds)
- Median mixing height (irrespective of wind octant)

- Average annual precipitation (irrespective of wind octant and using 30-year normal data[16] if available, to avoid biasing the screening results in favor of any precipitation anomalies that existed in 2016)

We selected median instead of mean values because medians were usually smaller than mean values and because lower wind speed and mixing height are more health protective (i.e., typically lead to higher chemical deposition in areas near the emission source). We evaluated the distributions of median wind speeds, median mixing heights, and average annual precipitation across all 824 stations. From those distributions, we identified values to represent reasonable low, mid-range low-end, mid-range high-end, and high values across all sites (i.e., roughly 5th-, 35th-, 65th-, and 95th-percentile values). The values shown in Exhibit 27 are those used in TRIM.FaTE model runs as part of developing the library of Tier 2 screening threshold emission rates, REFs, and mixing height refinements (the library is discussed in Section 3.2.4).

Exhibit 27. Values for Meteorological Parameters Used to Develop the Tier 2 Screening Threshold Emission Rates and REFs

Parameter	Value	Risk Direction
Wind Speed (m/s)	**1.6**	As wind speed increases, it carries more airborne chemical out of the modeling domain and decreases risk. Slower wind speeds lead to more chemical deposition closer to the facility.
	2.8	
	3.7	
	5.4	
Precipitation (mm/yr)	240	As precipitation amounts increase, so does wet deposition over the modeled domain.
	706	
	1,069	
	1,474	
Mixing Height (m)	**226**	At higher mixing heights, pollutants released to air mix with larger volumes of air, resulting in lower air concentrations and modeled deposition to surfaces and, consequently, lower ingestion exposures.
	351	
	454	
	674	

Notes: Bold font indicates the value used in Tier 1. Also, we do not show wind direction here because it has a linear effect on exposure and risk modeled in TRIM.FaTE (using the scenario design of the screens). Use of site-specific wind direction data is discussed in Section 3.3.

3.2.2 Locations of Lakes and Farms/Gardens

We model lakes and hypothetical farms/gardens within a 50-km radius around a facility. We believe that a 50-km domain places a reasonable restriction on how far a nearby resident will travel to catch and consume fish from area lakes on a routine basis. Although extending the modeling domain beyond 50 km would increase the amount of deposition "captured" by the modeled watershed, the incremental chemical mass expected to accumulate in the watershed diminishes rapidly with distance. Areas beyond 50 km from the emission source are expected to

[16]We obtained 30-year-average annual precipitation for the period of 1981–2010, from the National Oceanic and Atmospheric Administration. https://www.ncdc.noaa.gov/data-access/land-based-station-data/land-based-datasets/climate-normals/1981-2010-normals-data.

contribute relatively negligible amounts of chemical to the watershed, based on air-to-soil deposition values produced by TRIM.FaTE in the Tier 2 scenarios.[17]

As indicated in Exhibit 28, within the 50-km radius, we evaluate lake and farm/garden impacts at five distances in Tier 2: three distances within a 10-km radius where the majority of chemical deposition occurs (i.e., at 0.5, 5, and 10 km from the facility), and two additional distances beyond 10 km (i.e., at 20 and 40 km from the facility). All farm/garden locations are hypothetical, so potential exposure is evaluated in Tier 2 at each possible distance and octant. Note that if an actual lake distance is in between two of the distances in Exhibit 28, that lake will be assigned the closer location (i.e., the distance expected to yield the greatest risk). For example, if an actual lake is 7 km from the facility being analyzed, that lake will be set 5 km from the facility for screening purposes.

Exhibit 28. Distances Used to Develop the Tier 2 REFs and Screening Threshold Emission Rates

Parameter	Value	Risk Direction
Lake and Farm/Garden Distances, measured from the inside geographic centroid of the feature (km)	**0.5**	With increased distance from the source, chemical deposition typically is reduced and, consequently, exposures are reduced.
	5	
	10	
	20	
	40	

Notes: Bold font indicates the value is equal to the value used in Tier 1.

Exhibit 29 depicts the spatial layouts of each lake- and farm/garden-distance scenario in Tier 2 for a single octant.[18] The 0.5-km-distance scenario is the same layout as the Tier 1 scenario shown in Exhibit 11 and is not repeated in Exhibit 29. The runoff and erosion characteristics are unchanged from the Tier 1 screen.

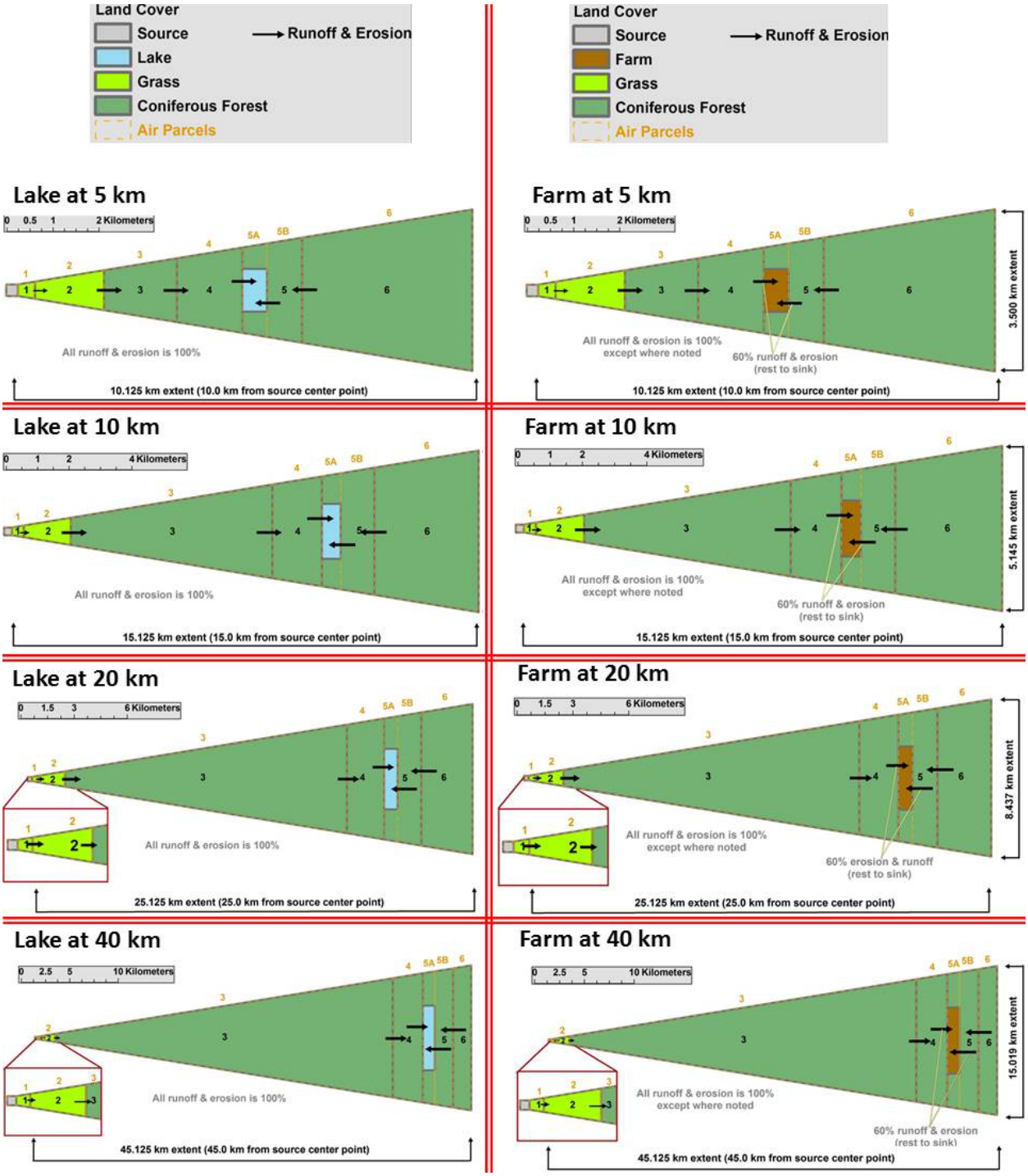
In resituating the lake and farm/garden at these alternative locations, we maintained ratios consistent with those included in the Tier 1 screening scenario for (1) lake or farm area to total land area in the modeled domain, (2) runoff watershed area to lake or farm area, and (3) erosion watershed area to lake or farm area. We used “thin” lake and farm shapes (i.e., downwind width much shorter than the cross-wind length) to minimize distance from the far end of the lake or farm to the facility, resulting in higher and more health protective media concentrations. Situating the lakes or farm/gardens farther from the stack required expansion of the modeled domain. For example, the modeling domain in the top parcel layouts in Exhibit 29 extends 10 km from the middle of the source parcel, while the modeling domain in the bottom

[17]Mass deposited at the outer edge of the watershed (50 km) is expected to result in a negligible increase in estimated exposure via the fisher or farmer scenarios. The TRIM.FaTE runs supporting Tier 2 indicate that chemical deposition rates at 43–45 km from the emission source (at parcel 6 in the Farm at 40 km layout shown in Exhibit 29) are between 1 and 3 orders of magnitude smaller than those at 0.5–2 km from the source (at parcel 2), depending on the chemical and meteorological parameters. Although additional chemical mass could be transported to the lake and farm through erosion and runoff, the amount of chemical deposited beyond 50 km from a facility is less per unit area and would runoff or erode over longer distances, further attenuating the mass that might reach a farm or lake. Wind speeds of 13 m/s (approximately 29 mph) or greater must be sustained for a full hour for the chemical plume to travel farther than 50 km. Wind speeds of that magnitude are unlikely to occur consistently for many days or weeks to substantially affect chronic exposure. In addition, a 50-km limit also puts a reasonable constraint on the domain of lakes for the fisher scenario.

[18]The lake and farm surface areas also were changed for each new distance layout, which allowed for the simulations to maintain a constant ratio between watershed and erosion area compared with lake and farm areas.

parcel layouts extend 45 km from the source (although the diagrams are the same size in Exhibit 29).

Exhibit 29. TRIM.FaTE Surface Layouts for the Tier 2 Multipathway Screen, Using Alternative Distances Between the Facility and the Fishable Lake or Farm/Garden (base layout is the same as Tier 1 shown in Exhibit 11 and not shown here)



Notes: The distance axes are longer for the farther locations of lake and farm/garden from the source (i.e., moving from the top to the bottom diagrams). Heavy, black arrows depict the direction of chemical runoff and erosion.

Maintaining the same overall ratio of land area to lake or farm area in each domain resulted in scenarios with surface areas for lakes and farms increasing with distance from the source. The changes in lake size between these configurations are not expected to have a substantial independent effect on exposure and risk because the effect of increased lake size (i.e., dilution of chemical due to greater volume) is offset by greater watershed area for total deposition and runoff. Furthermore, lake depth was not changed. The changes in farm size between these configurations also are not expected to have a substantial independent effect on exposure and risk because chemical concentrations in soil are estimated as mass per unit area (this also allows us to use the exposure media concentrations developed for the farmer scenario with the gardener scenarios, which have a smaller surface area than the farm). As noted above, we set up the configurations to ensure that the lakes and farms in the different scenarios received runoff and erosion from equivalent watersheds on a per-surface-area basis.

3.2.3 Gardener Exposure Scenario

The gardener scenario comprises the exposure pathways through which individuals might be exposed in an urban or non-farm rural setting. Notably, the gardener exposure scenario is analogous to the HHRAP "Resident" exposure scenario (U.S. EPA 2005a). Similar to the resident scenario in HHRAP, the gardener ingests a subset of the media that the subsistence farmer exposure scenario ingests, namely:

- Soil,
- Exposed fruits and vegetables,
- Protected fruits and vegetables,
- Root vegetables,
- Eggs, and
- Breastmilk (as an infant).

Exhibit 30 compares the ingested media for the gardener exposure scenario to those of the farmer exposure scenario.

Exhibit 30. Ingested Media for Farmer and Gardener Scenarios

Scenario	Soil	Protected Fruit	Exposed Fruit	Protected Vegetable	Exposed Vegetable	Root Vegetable	Dairy	Beef	Pork	Poultry	Eggs	Breast Milk
Farmer	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Gardener	✓	✓	✓	✓	✓	✓					✓	✓

For the RTR inhalation risk assessment, receptor locations are designated as rural or urban, and the gardener in the multipathway screen is designated the same way. For a gardener in a rural environment, we use the same ingestion rates (IRs) as used for the farmer for the produce that the gardener ingests. As discussed above, this is a subset of the media that a farmer would ingest (see Exhibit 30). A reasonable assumption is that a gardener in a rural setting would be more likely to have sufficient land to support a garden large enough to provide for the assumed 90th-percentile IRs and would tend to consume larger amounts of home-produced foods than would gardeners in urban settings. Gardeners in urban settings likely would grow produce on smaller plots and, in general, would likely supplement home grown produce with store-bought

produce, particularly during the non-growing season (ingestion of produce that is not home grown is not assessed in the screens). For gardeners in an urban setting, therefore, the mean instead of 90th-percentile IR for home grown produce from EPA's (2011) *Exposure Factors Handbook* is used. The IRs for the urban gardener generally are between one-third to one-half of those for the farmer and rural gardener, as shown in more detail in Attachment B. Soil ingestion for the urban and rural gardeners is the same high-end rate as for the farmer, and the farmer and gardener (both rural and urban) have higher soil IRs than the general population. A central-tendency soil IR could underestimate soil ingestion for gardeners.

To be health-protective, the gardener scenarios assume concentrations, and thus, the same transfers of chemical from adjacent modeling areas through runoff and erosion as the farming scenario. In a more refined site-specific assessment, an urban garden scenario might assume a raised garden bed or garden boxes that does not receive chemicals through runoff or erosion.

3.2.4 Development of Library of Tier 2 Screening Threshold Emission Rates, REFs, and Mixing Height Refinements

We conducted a large set of modeling runs based on unit emissions of 1 gram per day and taking into account: (1) wind speed, mixing height, and precipitation rate values shown in Exhibit 27; (2) lake and farm distances shown in Exhibit 28; and (3) spatial layouts shown in Exhibit 11 and Exhibit 29. These runs systematically varied each of these parameters so that all possible combinations were evaluated.[19] The resulting matrix of screening-level risk estimates represented each unique combination of PB-HAP and values for wind speed, mixing height, precipitation rate, and distance from the facility to a lake or farm. From these screening-level risk estimates, we calculated Tier 2 REFs and screening threshold emission rates for all the combinations stated above (note that for POMs and dioxins, screening threshold emission rates are only calculated for benzo(a)pyrene and 2,3,7,8-TCDD, respectively).

As in the Tier 1 screen, the Tier 2 screening threshold emission rate is defined as the emission rate that corresponds to a 1-in-one million excess lifetime cancer risk or an HQ of 1 for a given PB-HAP. As in Tier 1, for those chemicals that are not fully parameterized in TRIM.FaTE, the REFs in the Tier 2 screen reflect an individual POM or dioxin chemical's fate, transport, and toxicity relative to the index chemical for each group (BaP for POM and 2,3,7,8-TCDD for dioxin; see Section 2.2.4).

Mixing height has a direct effect on chemical concentrations, and therefore its exposure level. When a chemical is released to air, it mixes with air in the lower atmosphere (i.e., in the mixing layer). With the assumption of instantaneous and complete mixing (which is the assumption used in TRIM.FaTE), there is an inverse linear relationship between changes in mixing height and changes in chemical air concentrations. A lower mixing height (boundary) means that there is a smaller volume of air available for mixing, meaning less dilution and higher chemical concentrations within the mixing layer than when the mixing height is higher. At a given location, higher chemical air concentrations lead to higher deposition and higher chemical concentrations in environmental media, and ultimately higher exposure. Precipitation, on the other hand, affects chemical concentrations in two opposing processes: more precipitation results in more wet chemical deposition, but it also dilutes the deposited chemical in leachate and runoff. Wind speed influences air concentration with distance from an emission source—lower wind speeds result in higher chemical concentrations in air and more deposition closer to the source than

[19]There were 640 independent modeling runs per PB-HAP chemical (including each dioxin/furan congener).

farther away, while higher wind speeds result in relatively lower concentrations and deposition near the source.

Because of the direct and predictable effects of mixing height on exposure, and because the range of median mixing heights across the 824 meteorological stations is substantial (i.e., less than 200 m to more than 2,000 m), we used the matrix of Tier 2 screening threshold emission rates described above to further quantify the relationship between mixing height and exposure. The screening threshold emission rates decrease linearly with decreasing mixing height; put another way, SVs increase linearly with decreasing mixing height. The linear relationships are specific to each combination of PB-HAP, distance from facility to the lake or farm/garden, wind speed, and precipitation rate. Therefore, for each combination of those parameter values, we derived a linear regression to relate changes in mixing height (i.e., the four mixing height values used in the modeling) to changes in screening threshold emission rate. Using the regression coefficients, we are able to estimate the influence of mixing height on Tier 2 SV estimates with a continuous function based on reported mixing heights near a facility.

After developing the mixing height regression coefficients, we condensed the Tier 2 matrix into a Tier 2 library containing REFs, screening threshold emission rates, and mixing height regression coefficients for each unique combination of PB-HAP, wind speed, precipitation rate, and distance from the facility to a lake or farm/garden. With respect to mixing height, the screening threshold emission rates and REFs are derived using the mixing height value of 226 m (i.e., the Tier 1 mixing height) and are then adjusted using the actual mixing height near the facility being screened.[20]

Unlike Tier 1, the Tier 2 screen assesses potential risk from fish ingestion separately from home grown produce, animal products, and soil ingestion; therefore, there is a distinct library of screening threshold emission rates, REFs, and mixing height regressions for each of the exposure scenarios (fisher, farmer, and rural and urban gardeners).

Section 3.3 discusses how the screening threshold emission rates, REFs, and regression coefficients discussed in this section are used to estimate potential multipathway risk.

3.3 Implementing the Tier 2 Multipathway Screen

To implement the Tier 2 multipathway screen, we developed a Microsoft® Access™ tool that is pre-loaded with the (1) U.S. lake location data; (2); U.S. meteorological database; and (3) libraries of Tier 2 screening threshold emission rates, REFs, and mixing height regression coefficients described above.

As noted in Section 3.2.2, the database of lakes used in the Tier 2 screen is available from ESRI® and based on USGS data. The database includes information on the location, surface area, use or type designation, and name (if available) of all lakes (including ponds and reservoirs) in the United States. To focus on lakes that can support harvest of upper-trophic-level fish, we excluded lakes used for disposal, evaporation, or treatment, and we included only lakes greater than 25 acres in area (see Section 3.3.1 below for more detail). We did not include lakes larger than 100,000 acres in area because they are not reliably modeled in TRIM.FaTE

[20]The Tier 2 screening threshold emission rates are all based on a mixing height of 226 m. However, the SVs that are calculated from these screening threshold emission rates will be refined based on a regression equation that takes into account the actual mixing height around the facility. For example, if a facility has an SV of 3 assuming a mixing height of 226, and the mixing height refinement is a factor of 0.8 (based on the regression equation and a site specific median mixing height of 400 m), then the refined Tier 2 SV for this facility would be 2.4 (i.e., 3×0.8). This calculation is further described in Section 3.3.

due to the complexity of near-shore and off-shore hydrodynamics and the size of their watersheds. The database of lakes contains approximately 433,000 fishable lakes for evaluating Tier 2 impacts.

The Tier 2 screening tool identifies all qualifying lakes in the area surrounding a screened facility and determines their distances and directions with respect to the facility; each of these distance values are subsequently matched to the closest lake distance "bin" in the Tier 2 library (fish in lakes closer to the emission source generally accumulate more chemical). The user can vary the radial distance and area limits of qualifying lakes (defaults are set at 50 km and 25 to <100,000 acres, respectively), and the user can also review the nearby lakes and exclude ones that would not be used to harvest fish (e.g., based on names indicating industrial, waste, or treatment purposes). The tool records any excluded lakes to omit them from subsequent screens.

Unlike lake locations, farm and garden locations are not site-specific, so the tool calculates the Tier 2 farmer and gardener SVs at all distances available in the Tier 2 library and in all directions from the facility. SVs are calculated for both the rural and gardener scenarios and the appropriate scenario is selected based on whether the census block nearest to the facility is in an urbanized area based on population density.

Each facility being screened is then matched with the same surface meteorological station used in the RTR inhalation risk assessment (i.e., the closest station). This process currently utilizes over 800 meteorological stations nationwide.[21] From the selected meteorological station, the tool identifies the appropriate precipitation and wind speed bins. The tool matches the meteorological station's annual precipitation amount to the next higher precipitation amount in the Tier 2 library (higher precipitation rates generally lead to greater wet deposition resulting in increased chemical accumulation in fish and surface soil). The tool identifies the annual median wind speed blowing toward each lake or farm/garden location at the facility and matches it to the next lower wind speed in the Tier 2 library (lower wind speeds generally lead to greater near-field chemical accumulation in fish and surface soil).

Given the matching meteorological (wind speed and precipitation) and distance (for farms/gardens and lakes) values, the tool identifies the appropriate Tier 2 screening threshold emission rate and REF from the Tier 2 library for each emitted chemical. The annual median mixing height for the facility's matching meteorological station is then used with the mixing height regression coefficients from the Tier 2 library to account for the impact of mixing height using Equation 6:

$$RefMix_{T2}=(M \times S) + i \qquad \textbf{Eqn. 6}$$

where:

$RefMix_{T2}$ = mixing height multiplier for Tier 2

[21]The process of pairing dozens or hundreds of facilities with meteorological data has precedent. In their report to the Science Advisory Board (SAB) on the 1996 NATA, EPA described pairing each facility with the closest meteorological station in an inventory of over 350 meteorological stations nationwide, creating an average facility-to-station distance of less than 50 km for the 1996 NATA (U.S. EPA 2001b). In a separate 2009 report to the SAB on the RTR program, EPA described using 158 meteorological stations nationwide, with a standard practice of selecting the station closest to a facility unless the facility provides onsite meteorological data (U.S. EPA 2009). Using 156 petroleum refineries as a sample data set, the average facility-to-station distance was 72 km. In both instances, the SAB accepted the approach for modeling large numbers of facilities, although it recommended providing high-level siting maps (e.g., meteorological stations overlaid with terrain gradients or regional climate regimes) to qualify some of the uncertainties related to meteorological data in air dispersion modeling (U.S. EPA 2001a; U.S. EPA 2010b).

i = intercept coefficient of the linear regression

M = median mixing height (in meters) associated with the facility

S = slope coefficient of the linear regression

After the appropriate Tier 2 screening threshold emission rate, REF, and mixing height refinement factor are identified, the final site-specific factor is considered: the frequency that winds blow toward an evaluated lake or hypothetical farm/garden location. In the Tier 2 modeling runs, as in the Tier 1 modeling runs, winds are modeled as blowing toward the lake and farm/garden 43 percent of the time (i.e., three days per week—an unusually consistent, but feasible, long-term wind pattern; e.g., similar to wind direction patterns in Yakima, Washington). The screening threshold emission rates in the Tier 2 library correspond to that direction frequency. Using the Tier 2 database of meteorological data, the Tier 2 screening tool accounts for the percentage of time that the wind actually blows in the direction of the lake or farm/garden being evaluated in the Tier 2 screen using Equation 7:

$$RefWD_{T2} = \frac{FreqWD_{T2}}{FreqWD_{T1}} \qquad \textbf{Eqn. 7}$$

where:

$RefWD_{T2}$ = Tier 2 wind direction multiplier

$FreqWD_{T2}$ = percent of time winds blow toward the Tier 2 lake or farm/garden

$FreqWD_{T1}$ = percent of time winds blow toward the Tier 1 lake and farm/garden (i.e., 43%).

Finally, for each chemical emitted by a facility, the tool calculates the Tier 2 SV for each lake, farm, and garden using Equation 8:

$$SV_{T2} = \left(\frac{ER \times REF_{T2}}{Th_{T2}}\right) \times RefMix_{T2} \times RefWD_{T2} \qquad \textbf{Eqn. 8}$$

where:

SV_{T2} = Facility- and chemical-specific Tier 2 SV (i.e., ratio of facility emissions to threshold for adverse health effects for the chemical)

ER = Chemical-specific facility emission rate

REF_{T2} = Tier 2 REF (for individual dioxins or POM; = 1 for other chemicals)

Th_{T2} = Tier 2 screening threshold emission rate for the PB-HAP (arsenic, cadmium, mercury, 2,3,7,8-TCDD, or BaP) and the lake, farm, and garden.

As with the Tier 1 screen, the Tier 2 SVs for all emitted POM chemicals are summed to a total SV of POM as BaP-equivalents, and the Tier 2 SVs for all emitted dioxin/furan chemicals are summed to a total dioxin SV as 2,3,7,8-TCDD-equivalents.

At this stage of the Tier 2 screen, the Tier 2 fisher-scenario SVs reflect subsistence fishing at each individual lake, regardless of lake size (i.e., whether or not that harvest rate might overfish top trophic level fish in a lake). As discussed in Sections 3.3.1 and 3.3.2 below, we further refine the Tier 2 fisher scenario to better reflect sustainable fish withdrawals. The Tier 2 farmer and gardener exposure scenarios assess exposure at each hypothetical farm and garden location

and the hypothetical farm and garden location with the largest SV for a facility and PB-HAP is identified.

3.3.1 Accounting for Sustainable Fishing

Early in the process of compiling the Tier 2 lake database, we encountered the question: "What size 'lake' is fishable for the purposes of this assessment?" The Tier 2 screen should focus on lakes large enough to support a fish harvest rate that would meet the high-end fish ingestion rates assumed for the exposure scenario (i.e., 373 g-fish/d).

In the TRIM.FaTE model screening scenario, WCCs are modeled at the top of the water-column food chain (e.g., pickerel, largemouth bass), with all of their diet consisting of smaller "prey" or "pan" fish in the water column. In the assumed *linear* water-column food chain, those fish in turn consume smaller fish that are planktivorous (WCH). Thus, the WCCs can be called trophic level 4 (TL4) if the smallest fish are considered trophic level 2 (TL2). The BCs in TRIM.FaTE are modeled to represent an intermediate trophic level between 3 and 4 (i.e., TL3.5) in the benthic food *web*. Benthic carnivores (e.g., catfish) obtain half of their diet from TL2 (benthic invertebrates that feed on detritus at the sediment surface) and half from TL3 fish in the benthic environment, which consume benthic invertebrates only. Together, we refer to the WCC and BC fish compartments as piscivorous fish.

To identify sustainable fish harvest rates by lake size, we made the eight key assumptions listed below. Information and citations to peer-reviewed literature that support these assumptions are provided in Attachment E.

1. Ponds or lakes must exceed a certain size to sustain a population of WCC over the long term (i.e., smaller ponds/lakes might support only two or three trophic levels given limits on total lake productivity per unit area and the 80-to-90-percent loss of food energy per trophic transfer).
2. In lakes with stable fish communities including a reliable WCC fish population, piscivorous fish (i.e., WCC TL4 and BC TL3.5) might comprise approximately 20–22 percent of the total fish biomass (references in Attachment E).
3. Productivity in most lakes of small to moderate size depends substantially on the benthos, with benthic invertebrates consuming detritus derived from both in-lake algae and macrophytes and from plant litter eroded into the lake from terrestrial sources across the watershed. We expect more biomass in the BC than in the WCC compartment. Assuming 21 percent of the standing biomass of fish are piscivorous, BC fish might account for 17.5 percent of the total standing fish biomass, and WCC fish might account for 3.5 percent of the total fish biomass (Attachment E). The remaining 79 percent would include "pan" fish (e.g., sunfish, perch), minnows, young-of-the-year of piscivorous fish. This set of assumptions represents a "point estimate" of fish biomass distribution across different compartments.
4. Humans consume fish from the BC compartment and the WCC compartment, with a 50:50 split, reflecting fishing and consumption preferences rather than relative abundance of fish in the BC and WCC compartments. Depending on the chemical, bioaccumulation over 4.0 trophic transfers might result in higher concentrations in the WCC fish compartment than bioaccumulation over 3.5 trophic transfers in the BC fish compartment. On the other hand, for chemicals that partition primarily to the sediment compartment, benthic invertebrates might accumulate more chemical, resulting in higher concentrations in the BC than the WCC compartment. Because we could not predict, a priori, which fish compartment, the BC or the

WCC, would have higher chemical concentrations for any PB-HAPs, we assumed the 50:50 split in fish harvested from the WCC and BC compartments.

5. The total fish standing biomass is assumed to be 40 g wet weight/m^2, which might be relatively high for natural ponds and lakes across much of the United States; however, it is a mean value for reservoirs. Overestimates of lake productivity would bias results to be more health protective, because more fish could be harvested from contaminated lakes closer to a facility.
6. We assume that the minimum viable population (MVP) size for a single fish species is at least 50 adult fish for a local population to survive over several decades. Interbreeding populations of 500 or more adults (with 50:50 male:female) should be sustainable without signs of inbreeding depression. Actual MVP for a population genome depends on many factors and varies substantially across different species and landscapes. To model MVP for a given species and location, one should specify the timeframe of concern (e.g., 50 years, 100 years) and a target probability of local extirpation (e.g., less than 5 percent). Population modeling for individual species is beyond the scope of RTR screens; we therefore use the estimate of at least 50 breeding individuals to maintain a fish species in a lake.
7. Humans can harvest 10 percent of any single fish compartment without threatening the population due to overharvesting. Although sustainable harvest rates vary with species life history characteristics, for top carnivores, data suggest that 10 percent harvest rates should prevent overfishing (Attachment E).
8. Only 33 percent of the fish caught for consumption is edible fillet muscle. A 0.33 edible fraction is used to estimate total fish biomass that must be harvested for human consumption of fillet only.

Using the above assumptions, we estimated fish-fillet ingestion rates as a function of total standing fish biomass and lake area. Because we assume a 50:50 harvest of BC and WCC fish, and because the standing biomass of WCC fish is approximately one fifth of the standing biomass of BC fish, we focus on lakes that can provide the MVP of 50 breeding individuals for the WCC fish compartment. Attachment E presents the calculations and steps required to estimate which combinations of lake size and productivity could sustain at least 50 individual WCC fish, and the human fish ingestion rates that could be supported for those combinations.

The grey shading in Exhibit 31 indicates combinations of lake size and lake productivity that would not support a MVP of 50 individual adult WCC fish. The white, or unshaded, cells in Exhibit 31 indicate combinations of lake area and productivity that could sustain the listed fish-ingestion rates for WCC plus BC fish over several decades, but might not be sufficient to prevent inbreeding depression. Finally, the yellow shading in Exhibit 31 indicates combinations of lake productivity and lake size that are likely to provide long-term sustainability of WCC fish in the lake.

Once we had established which cells of Exhibit 31 were in the grey, white, and yellow zones, we calculated the fish ingestion rates associated with each cell.

At the assumed standing fish biomass of 40 g ww/m^2, a 25-acre lake is the smallest lake that might sustain a population of 50 or more WCC (smallest lake with unshaded cells). Therefore, we selected 25 acres as the “cutoff” for the minimum size for an actual lake near a facility to be included in the Tier 2 and Tier 3 screens. In addition, we did not consider lakes larger than 100,000 acres because they are not readily modeled in TRIM.FaTE due to the complexity of near-shore and off-shore hydrodynamics and size of their watersheds.

As shown in Exhibit 31, the fish-ingestion rate associated with a 25-acre lake and the assumed fish biomass of 40 g ww/m^2 is 26 g/day, or approximately 1 gram fish/acre/day. Thus, a 25-acre lake cannot by itself support the adult human ingestion rate used in the multipathway screens (i.e., 373 g ww fillet per day) with a 50:50 mix of WCC and BC fish. However, a fisher could fish multiple lakes, totaling 373 acres, to achieve that ingestion rate. In Section 3.3.2 below, we discuss the refined-fisher scenario, whereby a fisher withdraws and consumes fish at an assumed sustainable rate of 1 gram fish/acre/day from as many acres of lake(s) as necessary to harvest 373 grams of fish (wet weight fillet) per day. The refined-fisher scenario also aggregates SVs at lakes impacted by emissions from more than one facility in the source category.

Lakes smaller than 25 acres could be stocked annually to support substantial fish withdrawals. However, we assume that when introduced to the lake, the stocked fish would be uncontaminated by the chemicals of interest. Moreover, the period over which accumulation of chemical from the lake could occur would be approximately three to six months (i.e., the fishing season) for the majority of the fish stocked as large juveniles or adults, instead of several years for fish hatched in or born into the lake. We believe that not taking stocked fish into consideration is a reasonable assumption.

We could have used other assumptions about human fishing behavior. For example, fishers could harvest BC and WCC in proportion to their relative abundance (i.e., 80:20); however, it is not clear which fish compartment might have higher chemical concentrations. Alternatively, fishers could consume “pan” fish like sunfish and small perch to meet their daily fish ingestion rates fishing smaller lakes than predicted in Exhibit 31. Pan fish, however, represent TL3 fish in the water column. Therefore, chemical concentrations in the tissues of pan fish would likely be lower than in the TL3.5 BC or the TL4 WCC fish compartments for mercury, cadmium, dioxins, and POM.

Exhibit 31. Estimated Maximum Fish-ingestion Rate (g/d) Associated with Sustainable Fishing[a]

Total Fish Biomass (g ww/m^2)[b]	Area of Pond or Lake (acres)															
	1	2	3	4	5	7.5	10	15	25	35	50	75	100	150	200	400
2	0	0	0	0	0	0	1	1	1	2	3	4	5	8	10	20
3	0	0	0	0	0	1	1	1	2	3	4	6	8	12	15	31
4	0	0	0	0	1	1	1	2	3	4	5	8	10	15	20	41
5.7	0	0	0	1	1	1	1	2	4	5	7	11	15	22	29	58
10	0	1	1	1	1	2	3	4	6	9	13	19	26	38	51	102
15	0	1	1	2	2	3	4	6	10	13	19	29	38	58	77	154
20	1	1	2	2	3	4	5	8	13	18	26	38	51	77	102	205
30	1	2	2	3	4	6	8	12	19	27	38	58	77	115	154	307
35	1	2	3	4	4	7	9	13	22	31	45	67	90	134	179	359
40	1	2	3	4	5	8	10	15	26	36	51	77	102	154	205	410
50	1	3	4	5	6	10	13	19	32	45	64	96	128	192	256	512
60	2	3	5	6	8	12	15	23	38	54	77	115	154	231	307	615
70	2	4	5	7	9	13	18	27	45	63	90	134	179	269	359	717
80	2	4	6	8	10	15	20	31	51	72	102	154	205	307	410	820
90	2	5	7	9	12	17	23	35	58	81	115	173	231	346	461	922
100	3	5	8	10	13	19	26	38	64	90	128	192	256	384	512	1025
110	3	6	8	11	14	21	28	42	70	99	141	211	282	423	563	1127
120	3	6	9	12	15	23	31	46	77	108	154	231	307	461	615	1229
130	3	7	10	13	17	25	33	50	83	117	166	250	333	499	666	1332

Calculated using a series of basic assumptions and equations discussed in this section and in Attachment E.

[a] Dark gray shading indicates combinations of lake productivity and size that could support 50 or more adult WCC fish over a few decades, the minimum viable population size; yellow-shaded cells indicate a long-term self-sustaining population of WCC with at least 500 adult fish for one (or more) species is likely; no shading (white) indicates medium-term sustainability.

[b] Represents the total fish standing biomass. The biomass of WCC fish is 3.5 percent of the total. Reading from the table, at the assumed fish standing biomass of 40 g ww/m^2, 25 acres could support a water-column WCC fish population, but would provide at most 26 grams of fillet (wet weight) per day for a single fisher over a full year (intersection of the vertical and horizontal red lines). A lake of 100 acres with 40 g ww/m^2 total fish standing biomass could provide as much as 102 g/d of fish fillet. Reading straight across the row at 40 g ww/m^2 total fish biomass, the WCC plus BC fish-fillet-ingestion rate associated with lakes of different sizes turned out to be 1 g ww/acre. Thus, as a rule of thumb, we estimated lake productivity in grams of fish fillet [WCC & BC] per person per day as equal to the lake surface area in acres.

3.3.2 Refined-fisher Scenario

In the Tier 2 screen, the refined-fisher scenario is based on the idea that an adult fisher might fish from multiple lakes if the first lake (i.e., the one with the largest Tier 2 SV from Section 3.3) is unable to provide an adequate catch to satisfy the assumed ingestion rate (i.e., 373 g ww-fish/day for adults). The scenario assumes that lake fish productivity supports a long-term sustainable harvest of no more than 1 gram of fish fillet (wet weight) of top trophic level fish per acre of lake (Attachment E). That means the fisher must harvest fish from 373 acres of lakes to fulfill the assumed ingestion rate.

In the refined fisher scenario, a fisher travels to each relevant lake in order of highest to lowest chemical concentration in fish (of a given PB-HAP) and catches fish up to the lake's assumed sustainable harvest for top trophic level fish. A maximum travel radius of 50 km relative to the facility is used to maintain a realistic scenario. The final Tier 2 SV for the fisher can be expressed as the sum of the SV from each lake that is fished (which is based on the amount of fish ingested from each lake multiplied by the PB-HAP concentration in fish). If the highest-concentration lake is at least 373 acres, the ingestion rate is not altered (i.e., re-mains 373 g/d for adults). If the cumulative size of multiple visited lakes exceeds 373 acres, the fisher catches from the final lake only the amount of fish necessary to satisfy the ingestion rate (i.e., to reach the 373 g/d). If there are not 373 total acres of lakes, the SV reflects a reduced ingestion rate based on the cumulative lake acreage.

Ordering of fished lakes can be different for different PB-HAPs emitted by a facility because fate-and-transport characteristics vary by chemical; thus, it is possible that the most contaminated lake of at least 25 acres for one PB-HAP differs from the most contaminated lake of at least 25 acres for another PB-HAP. In this situation, the order of lakes fished would be different for the two PB-HAPs. In addition, the SVs used to develop the ordering of fished lakes correspond to one facility's emissions, even if another facility in the source category is located nearby. There are three possible lake fishing scenarios, as discussed below: (1) the first lake is 373 acres or larger, (2) multiple lakes total 373 acres or more, and (3) the lakes do not total 373 acres or more.

1. If the first lake fished is 373 acres or larger, the fisher is assumed to catch all fish from that lake. After evaluating the first facility (Facility A), we then identify any other nearby facilities in the source category that also deposit PB-HAP to the lake. If there is another nearby facility (Facility B) that is in the same source category and within 50 km of the lake, we add the fisher SV due to emissions from Facility A to the fisher SV due to emissions from Facility B. This results in the final Tier 2 fisher SV for Facility A incorporating deposition from both facilities.
2. However, if the first lake fished is smaller than 373 acres, multiple lakes must be fished. If *n* lakes are fished, where the total surface area of lakes 1 to *n*-1 is less than 373 acres, the refined-fisher SV of each lake 1 to *n*-1 is calculated using Equation 7 below. For each lake 1 to *n*-1, if the lake is also evaluated for one or more additional facilities in the source category, $SV_{T2RefFish,Lake}$ in Equation 9 (below) is calculated for each contributing facility and all the resulting values are summed into a total value $\left[\sum_{All\ Contributing\ Facilities}\left(SV_{T2RefFish,Lake}\right)\right]$ that incorporates deposition from multiple source category facilities.

   For lakes 1 to *n*-1 which total less than 373 acres:

$$SV_{T2RefFish,\ Lake} = SV_{T2Fish,\ Lake} \times \left(\frac{A_{Lake}}{373\ acres}\right) \qquad \textbf{Eqn. 9}$$

where:

$SV_{T2RefFish,Lake}$ = lake's SV for the Tier 2 refined fisher

$SV_{T2Fish,Lake}$ = lake's SV for the Tier 2 unrefined fisher

A_{Lake} = lake's surface area (acres).

If lakes 1 to *n* total 373 acres or more, the refined fisher SV for lake *n* is calculated using Equation 8 below. As discussed in the preceding paragraphs, this equation is applied for all facilities within 50 miles of lake *n* and the individual SVs are summed into a total $\left[\sum_{All\ Contributing\ Facilities}\left(SV_{T2RefFish,Lake\ n}\right)\right]$ that incorporates deposition from multiple source category facilities.

For lake *n*, where lakes 1 to *n* total 373 acres or more:

$$SV_{T2RefFish,\ Lake\ n} = SV_{T2Fish,\ Lake\ n} \times \left(\frac{373\ acres - \sum\left(A_{Lakes\ 1\ to\ n-1}\right)}{373\ acres}\right) \qquad \textbf{Eqn. 10}$$

Finally, the cumulative Tier 2 SV for the refined fisher is calculated as shown in Equation 11:

$$SV_{T2RefFish,Total} = (Eqn.\ 10) + \sum (Eqn.\ 9) \qquad \textbf{Eqn. 11}$$

or

$$SV_{T2RefFish,Total} = \left[\sum_{All\ Contributing\ Facilities}\left(SV_{T2RefFish,Lake\ 1\ to\ n-1}\right)\right] + \left[\sum_{All\ Contributing\ Facilities}\left(SV_{T2RefFish,Lake\ n}\right)\right]$$

3. If there are *n* total lakes in the modeling domain to assess, and their total surface area is smaller than 373 acres, then we use Equation 9 above to calculate the refined Tier 2 fisher SV for each lake (summing for other contributing source category facilities as necessary, see above), and then Equation 12 below calculates the final fisher SV for all fished lakes combined.

$$SV_{T2RefFish,Total} = \sum (Eq.\ 9) \qquad \textbf{Eqn. 12}$$

or

$$SV_{T2RefFish,Total} = \left[\sum_{All\ Contributing\ Facilities}\left(SV_{T2RefFish,Lake\ 1\ to\ n}\right)\right]$$

3.3.3 Considering Inhalation Risks at Hypothetical Garden Locations

To further prioritize the next steps in risk evaluations and risk management decisions, the screening tool incorporates the RTR total-cancer inhalation risk value at the closest residential

receptor (according to the inhalation modeling) in each of the eight primary directions. Each inhalation-receptor location is matched to the closest hypothetical garden location, and then the garden total-cancer SV (i.e., the sum of the arsenic, POM, and dioxin SVs) is summed with the total-cancer inhalation risk (i.e., the sum of cancer risks from all emitted HAPs, normalized to a "1-in-one million" convention) and the location of the largest total cancer-risk is identified. The combination of inhalation risk and ingestion SV is used to better understand the potential total cancer risk that might exist for individuals living near a facility emitting PB-HAPs. That information guides selection of next steps in the risk assessment of the source category.

3.3.4 Outputs

The screening tool generates several output tables, including an intermediate table that provides information on each lake, farm, and garden, including the amount of fish ingested and SV associated with each lake.

Finally, the tool generates the final screening tables for each facility and PB-HAP group. Summary tables identify the number of facilities exceeding the Tier 2 screening threshold emission rate of each PB-HAP group (separately for the fisher, farmer, and gardener), and which facilities have the largest SVs. All intermediate and final results tables present the Tier 1 and Tier 2 SVs side-by-side for comparison. Exhibit 32 through Exhibit 34 provide screen shots of the primary tool output tables.

Facilities with PB-HAP emissions that do not exceed any Tier 2 screening threshold emission rate are assumed to pose risks below levels of concern and no additional multipathway assessment is required. Facilities having emissions that exceed any of the Tier 2 screening threshold emission rates could be assessed further with Tier 3.

Exhibit 32. Example of Source Category Summary Results Output from Tier 2 Tool

Src Cat Info				Tier 1			Num Facil SV ≥ 2								
							Fisher				Fisher				
Src Cat	PB-HAP Grp	Num Facil in Src Cat (Emitting Any HAP)	Num Facil Emitting this Grp	Num Facil SV ≥ 2	Max SV	Max Facil ID	With Agg Impacts	Without Agg Impacts	Farmer	Inh+Farmer	SV	Agg Impacts?	SV Before Agg Impacts (If Applicable)	Facil ID	SV
Src Cat A	Arsenic	91	90	55	1.E+01	96058	0	0	9	0	2.E-01	Y	3.E-02	12785	3.E+0
Src Cat A	Cadmium	91	90	0	4.E-01	88187	0	0	0	0	7.E-02	N		78624	2.E-02
Src Cat A	Dioxin	91	68	50	7.E+02	40172	45	14	30	0	1.E+02	N		89662	3.E+01
Src Cat A	Methyl Mercury (Hg2)	91	90	68	2.E+02	55584	52	24	0	0	4.E+01	Y	3.E+00	80221	9.E-02
Src Cat A	POM	91	0	0	0.E+00	30537	0	0	0	0	0.E+00	N		37979	0.E+0
Src Cat A	Total Cancer (Arsenic+POM+ Dioxin)	91	90	70	7.E+02	34211	45	19	36	36	1.E+02	N		49159	3.E+01
Src Cat B	Arsenic	91	90	84	3.E+01	75443	0	0	20	0	3.E-01	Y	2.E-01	38743	8.E+0
Src Cat B	Cadmium	91	90	0	1.E+00	66297	0	0	0	0	1.E-01	Y	7.E-02	19292	4.E-02
Src Cat B	Dioxin	91	68	68	7.E+02	51927	72	29	61	0	1.E+02	N		60856	3.E+01
Src Cat B	Methyl Mercury (Hg2)	91	90	70	5.E+02	68523	67	34	0	0	8.E+01	Y	3.E+01	11903	1.E-01
Src Cat B	POM	91	0	0	0.E+00	19060	0	0	0	0	0.E+00	N		48528	0.E+0
Src Cat B	Total Cancer (Arsenic+POM+ Dioxin)	91	90	89	7.E+02	63574	72	35	63	63	1.E+02	N		37307	3.E+01

Note: Only a portion of this table is shown due to space limitations; information not shown here include: numbers of facilities with gardener SVs ≥ 2, numbers of facilities where sums of inhalation risk values with gardener SVs are ≥ 2, largest farmer and gardener SVs and their corresponding facility IDs, and largest sums of inhalation risk values with farmer/gardener SVs and their corresponding facility IDs. The facility IDs shown here are not real (they are only for illustration purposes). Red shading and font indicate where facilities did not screen out (i.e., the SV or sum of SV and inhalation risk rounded to 2 or higher). "Src Cat" = source category. "HAP" = hazardous air pollutant. "PB-HAP Grp" = persistent and bioaccumulative HAP group. "Num" = number. "Facil" = facility. "SV" = screening value (i.e., ratio of facility emission to screening threshold emission rate). "Max" = maximum. "Inh" = inhalation. "ID" = identifier. "Agg" = aggregate (as in emissions from multiple source category facilities impacting the feature of interest).

Exhibit 33. Example of Facility-level Results Output from Tier 2 Tool

Facil Info						Tier 1	Fisher			Farmer			Farmer + Inhalation				
Src Cat	Facil ID	Facil Lat	Facil Long	Facil Met WBAN	PB-HAP Gr	SV	SV	Agg Impacts?	SV Before Agg Impacts (If Applicable)	SV	Oct	Dist from Facil (km)	Total Value	Farmer Value	Inhalation Value	Oct	Inh Di Fa
Src Cat A	67248	34.3134	-70.7310	13876	Arsenic	5.E+00	7.E-02	Y	7.E-02	2.E+00	W	5.E-01					
Src Cat A	67248	34.3134	-70.7310	13876	Cadmium	2.E-01	2.E-02	Y	2.E-02	7.E-03	W	5.E-01					
Src Cat A	67248	34.3134	-70.7310	13876	Dioxin	1.E+02	2.E+01	N		6.E+00	W	5.E-01					
Src Cat A	67248	34.3134	-70.7310	13876	Methyl Mercury (Hg2)	2.E+00	4.E-01	Y	4.E-01	8.E-04	W	5.E-01					
Src Cat A	67248	34.3134	-70.7310	13876	POM	0.E+00	0.E+00	N		0.E+00	NW	4.E+01					
Src Cat A	67248	34.3134	-70.7310	13876	Total Cancer (Arsenic+POM +Dioxin)	2.E+02	2.E+01	Y	2.E+01	8.E+00	W	5.E-01	8.E+00	8.E+00	0.E+00	W	0.E+
Src Cat A	41870	49.9141	-105.7870	93806	Arsenic	7.E-01	1.E-02	N		3.E-01	S	5.E-01					
Src Cat A	41870	49.9141	-105.7870	93806	Cadmium	1.E-02	2.E-03	N		7.E-04	S	5.E-01					
Src Cat A	41870	49.9141	-105.7870	93806	Dioxin	0.E+00	0.E+00	N		0.E+00	NW	4.E+01					
Src Cat A	41870	49.9141	-105.7870	93806	Methyl Mercury (Hg2)	6.E-04	1.E-04	N		3.E-07	S	5.E-01					
Src Cat A	41870	49.9141	-105.7870	93806	POM	0.E+00	0.E+00	N		0.E+00	NW	4.E+01					
Src Cat A	41870	49.9141	-105.7870	93806	Total Cancer	7.E-01	1.E-02	N		3.E-01	S	5.E-01	3.E-01	3.E-01	0.E+00	S	0.E+

Note: Only a portion of this table is shown for space limitations; information not shown completely here include: gardener SVs and sums of inhalation risk values with farmer/gardener SVs. The facility IDs, locations, and meteorology station assignments shown here are not real (they are only for illustration purposes). Red shading and font indicate where facilities did not screen out (i.e., the SV or sum of SV and inhalation risk rounded to 2 or higher). "Src Cat" = source category. "Facil" = facility. "ID" = identifier. "Lat" and "Long" = latitude and longitude. "Met WBAN" = Weather Bureau-Army-Navy identifier for the meteorology station. "HAP" = hazardous air pollutant. "PB-HAP Grp" = persistent and bioaccumulative HAP group. "SV" = screening value (i.e., ratio of facility emission to screening threshold emission rate). "Oct" = one of the eight primary directional octants. "Dist" = distance. "km" = kilometers. "Agg" = aggregate (as in emissions from multiple source category facilities impacting the feature of interest).

Exhibit 34. Example of the Refined-fisher Output for Facility and PB-HAP from Tier 2 Tool

Facil Info			Tier 1	Tier 2													
					Lake Info							Unrefined Fisher SV		Refined Fisher Assessment			
																SV Fisher	
Src Cat	Facil ID	PB-HAP Grp	SV	Oct	Nam	Object ID (USGS)	Area (acres)	Facility-Lake Dist (km)	Lat	Long	Agg Impacts?	Independent Facil	Agg Facil	Fisher Order 0 = fisher did not fish	Fisher Fraction Ingestion (based on lake area)	Independent Facil	Agg Facil
Src Cat A	89341	Arsenic	5.E+00	W	Lake A	278	37	12.8	28.4090	-90.2940	Y	8.E-02	9.E-02	1	0.10	8.E-03	9.E-03
Src Cat A	89341	Arsenic	5.E+00	SW	Lake C	845	974	12.9	28.4380	-90.8240	Y	7.E-02	7.E-02	2	0.90	6.E-02	6.E-02
Src Cat A	89341	Arsenic	5.E+00	SW	Lake W	454	59	10.8	28.6720	-90.6420	Y	7.E-02	7.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake I	260	54	10.0	28.0140	-90.7200	Y	7.E-02	7.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	S	Lake U	789	198	8.1	28.9810	-90.1110	Y	6.E-02	7.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	S	Lake E	660	40	8.9	28.3370	-90.7690	Y	6.E-02	7.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	NW	Lake Q	601	32	6.9	28.4590	-90.9130	Y	6.E-02	7.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SE	Lake B	113	40	9.1	28.1480	-90.3060	Y	4.E-02	5.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	S	Lake D	673	40	19.2	28.3470	-90.3000	Y	4.E-02	5.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	S	Lake F	119	30	11.4	28.7060	-90.4190	Y	4.E-02	5.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	W	Lake G	369	44	39.0	28.8280	-90.6500	N	4.E-02	4.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	E	Lake J	412	37	5.5	28.1330	-90.0920	Y	4.E-02	6.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake V	942	114	20.6	28.9080	-90.4110	Y	3.E-02	4.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake M	894	94	22.7	28.5680	-90.6640	Y	3.E-02	4.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake K	754	86	27.9	28.6740	-90.1760	Y	3.E-02	4.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake S	425	84	32.1	28.1320	-90.8510	Y	3.E-02	4.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake T	838	82	23.9	28.5560	-90.4330	Y	3.E-02	4.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake H	756	74	38.6	28.9480	-90.3070	Y	3.E-02	4.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake N	210	69	30.8	28.8000	-90.4880	Y	3.E-02	4.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake O	515	32	39.1	28.8470	-90.9710	Y	3.E-02	5.E-02	0	0.00	0.E+00	0.E+00
Src Cat A	89341	Arsenic	5.E+00	SW	Lake Z	818	27	26.1	28.2140	-90.8050	Y	3.E-02	4.E-02	0	0.00	0.E+00	0.E+00

Note: The facility IDs and lake information shown here are not real (they are only for illustration purposes). "Src Cat" = source category. Red shading and font indicate where SVs rounded to 2 or higher. "Facil" = facility. "ID" = identifier. "HAP" = hazardous air pollutant. "PB-HAP Grp" = persistent and bioaccumulative HAP group. "SV" = screening value (i.e., ratio of facility emission to screening threshold emission rate). "Oct" = one of the eight primary directional octants. "USGS" = U.S. Geological Survey. "Dist" = distance. "km" = kilometers. "Lat" and "Long" = latitude and longitude. "Agg" = aggregate (as in emissions from multiple source category facilities impacting the feature of interest).

4. Tier 3 Screen

This section describes the methods and assumptions for the Tier 3 screen. We provide an overview of the approach (Section 4.1); description of the lake screen refinement (Section 4.2); evaluations of the farmer and gardener exposure scenarios (Sections 4.3 and 4.4); refinement of the plume rise evaluation (Section 4.5); and a final time-series assessment using hourly instead of annual average meteorological data (Section 4.6).

4.1 Overview of Approach

Tier 3 multipathway screens can be conducted on facilities that do not screen out in Tier 2. Tier 3 consists of five possible individual refinements (described in more detail below) that are based on additional site-specific data. These refinements are applied in sequence because all might not be needed; potential ingestion risk is evaluated at the end of each refinement. Because the Tier 3 screens introduce additional site-specificity to the screening scenario, it can require a higher level of effort than the Tier 2 screen, but still a much lower level of effort than required for a full site-specific assessment. One of the Tier 3 screens (i.e., the lake screen) potentially results in the rescreening of facilities' emissions using the Tier 2 methods described in Section 3 and using a revised lake dataset. The other screens each may result in a refinement of the Tier 2 screening value. The hourly time-series screen, if conducted, supplants the plume-rise screen because it calculates plume rise on an hourly basis.

4.2 Lake Screen

A Tier 3 lake evaluation is conducted if the Tier 2 screen for the fisher scenario indicates a potential for risk. During this evaluation, we examine: (1) whether or not a given lake used in the Tier 2 screen truly exists; (2) the intended purpose of the lake (e.g., recreation, industrial disposal); (3) lake accessibility; and (4) whether or not the lake is likely fishable. This evaluation is conducted because the USGS database of lakes and reservoirs used in the Tier 2 screen does not indicate lake accessibility or which lakes are likely fishable. In addition, USGS occasionally identifies a lake that no longer exists (e.g., has evaporated or been drained) or it uses a classification that might not accurately reflect the lake's purpose or type.

Using aerial imagery and other data sources, non-fishable "lakes" are identified, removed from the Tier 3 screen, and removed from the RTR lake dataset. If one or more lakes are removed from a facility's screen, the facility's emissions are rescreened using the revised lake database and the Tier 2 methods described in Section 3. If removing a lake(s) causes the originally fished lakes to sum to less than 373 total acres, then in the rescreening, the fisher will catch and consume fish from an additional lake(s) if available. In this situation, the Tier 3 lake screen is conducted on the newly added lake(s), and another rescreening is conducted, and so on, until no further lakes are removed or added to the screen.

We use aerial and street-view imagery and internet searches to ascertain if an assessed lake actually exists and whether or not it is likely to be fished. The assessed lakes are those from which the fisher harvests fish according to the Tier 2 methods discussed in Section 3.3.1. Lakes that appear swampy or covered in algae or used for industrial or waste disposal/treatment purposes are not fishable. Lakes adjacent or connected to a river or saltwater body (estuaries and rivers) are likely to have high outflow with limited chemical retention.

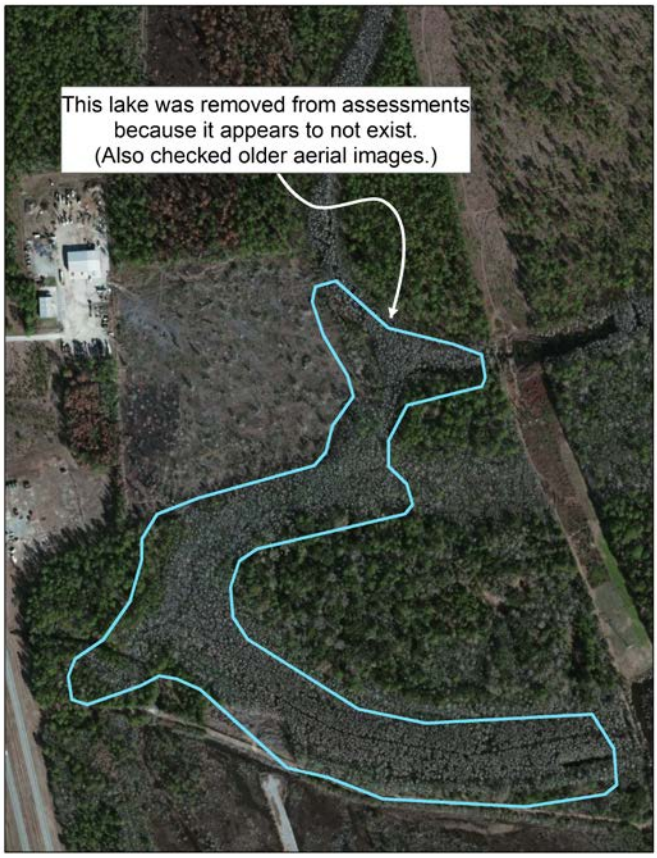
Based on the evaluation described above, we remove from the RTR lake dataset any lakes that are unsuitable for the RTR fisher scenario. For example, the area outlined in blue in Exhibit 35 identifies an area that the USGS dataset originally identified as a lake. However, aerial imagery (current and historical) shows that it is mostly or entirely dried up and not suitable for fishing. The area outlined in blue in Exhibit 36 identifies a lake from the USGS dataset that originally qualified for Tier 2 screening based on that dataset; however, aerial imagery shows that it is directly adjacent to an industrial facility and likely used only for on-site industrial purposes. Both lakes would be permanently removed from the RTR lake dataset and not considered in future.

If we remove a lake during the Tier 3 screen, we often need to include an additional lake for fish harvest to reach 373 fishable acres. We assess the additional lake(s) using the same criteria

and searches discussed in this section. After all lakes fished in the scenario (for the facilities not screening out in Tier 2) have been evaluated, we rerun the Tier 2 screen (using the tool discussed in Section 3) with the revised RTR lake dataset, producing revised screening results.

Lakes removed during this step of Tier 3 could affect screening results for other facilities in the source category. For example, if an assessed facility is within 100 km of another assessed facility, removing a lake might affect the screening results for both facilities. For this reason, we rerun the Tier 2 screen with the revised lake dataset for all facilities in the source category, including lakes contaminated by multiple facilities in the same source category. Screening results for the farmer and gardener scenarios are not affected by the lake screen.

Exhibit 35. Example of Lake Removed from Screening—Likely Evaporated or Drained



Note: Aerial imagery from ESRI World Imagery (2014).

Exhibit 36. Example of Lake Removed from Screening—Likely an Industrial Lake



Note: Aerial imagery from ESRI World Imagery (2014).

4.3 Farmer Scenario Evaluation

In many settings, based on local land use, population density, and other factors, the existence of a full-scale farm capable of providing all of the ingested media that are assumed for the farmer scenario might be unrealistic. If the farmer exposure scenario does not screen out at Tier 2, additional information can be evaluated on the likelihood that full-scale farming operations exist within the modeling domain.

If, after Tier 2, a farmer SV is above a level of concern, EPA will use census data, aerial imagery, and other available data to further assess the likelihood of subsistence farmer operations within 50 km of the facility. If, based on the additional analysis and review, it is determined that no subsistence farming operations are in the area, then the farmer scenario will no longer be used in Tier 3 and only the gardener SVs are reported. That is, EPA will assume that subsistence farming operations are not likely within 50 km of the facility, and only gardener SVs will be evaluated and reported. If information obtained suggests that subsistence farming operations likely exist, then in Tier 3, EPA will identify the farmer SV at the modeled location(s) that best matches the locational data obtained, and EPA will evaluate and report the largest of these SVs. Such location(s) may not be at the location of maximum SV as indicated in the Tier 2 screen. The gardener SVs will continue to be evaluated and reported, even if farmer results are used, because EPA considers the gardener scenario to be possible in all RTR evaluations.

4.4 Gardener Scenario Evaluation

Unlike the farmer exposure scenario, the gardener exposure scenario does not require a large geographic footprint (i.e., relatively small gardens could provide the fruits and vegetables to satisfy the gardener ingestion rates); this is especially true for the urban gardener scenario. Nonetheless, it does require that human receptors be present in the area.

If, after Tier 2, a gardener SV is above a level of concern, then in Tier 3 EPA will evaluate whether there likely are residential areas near the location of the gardener SV. EPA will use

information such as Census data, aerial imagery, and land-use data to determine whether people are likely to live near the SV location. If EPA determines that people likely live there, then the Tier 2 gardener SV is retained. If EPA determines that people likely do not live there, then in Tier 3 EPA will report the next-highest gardener SV at a location where people likely reside.

4.5 Plume-rise Screen

If, after the lake screen, the Tier 3, an SV is still above a level of concern, the risk assessor may choose to conduct a plume-rise screen. Atmospheric conditions coupled with the physical parameters of the chemical release point can cause the chemical plume to rise substantially higher than the physical release height. Plume rise is not explicitly modeled by TRIM.FaTE but can substantially reduce ground-level chemical exposure if the plume frequently rises above the air mixing height. The plume-rise screen varies chemical release height over time to simulate the effect of hourly meteorological conditions and the parameters associated with the chemical release point (i.e., physical release height and diameter, exit velocity, and gas temperature). If the resulting "effective release height" is above the air mixing height for a given hour, then in the TRIM.FaTE modeling system there is no chemical deposition or exposure for that hour.

In TRIM.FaTE modeling, the chemical mass reaching above the mixing layer (i.e., the model's upper-air layer) is unavailable for ground-level exposure (i.e., the upper-air layer functions as a chemical sink). Depending on ambient conditions, the top of the air mixing layer can fall below the top of tall stacks during some hours, and hot exit gas temperatures (i.e., buoyancy) and/or high exit gas velocities (i.e., momentum) can further elevate the chemical plume well above the source height and mixing height. If this occurs across many hours, it will substantially reduce total PB-HAP exposure and reduce the screening value. The plume-rise refinement factor—the number of hours when the effective release height remains below the mixing height, divided by the number of total modeled hours—is multiplied by the Tier 2 screening value, thus lowering the screening value.

The Tier 3 plume-rise screen uses methods summarized by Seinfeld and Pandis (1998) to estimate how often a facility's emissions reach the upper-air sink, which decreases availability for ground-level exposure. The methods use hourly meteorological data (e.g., air temperature and wind speed) from the closest weather station, the mass of the PB-HAP emitted from each source, the physical characteristics of the sources (i.e., release height, inside diameter at the release point, and exit gas temperature and velocity), and an estimate of the size of the facility (needed to estimate the plume height at the estimated edge of the facility).

We use EPA guidance (U.S. EPA 2000) to calculate wind speed at the stack height. We use equations reproduced in Seinfeld and Pandis (1998) to calculate plume rise with the above data and the assumed average vertical gradients of temperature and potential temperature (a calculation that normalizes temperature measurements for differences in height and pressure) corresponding to the stability class (e.g., neutral stability, slight or extreme instability—atmospheric conditions that affect how an air parcel moves vertically).

For each relevant emission source, we compare estimates of the hourly effective release height (i.e., sum of actual release height and plume rise) to the hourly mixing height to determine the mass of chemical remaining in the mixing layer when winds in that layer blow toward the lake or farm/garden of interest. We compare the mass of chemical remaining in the mixing layer, summed across all sources at a given facility, to the total emitted mass of the chemical—this ratio is the plume-rise retention factor. The screening tool described in Section 3 multiplies that

factor by the appropriate farmer, gardener, and fisher SVs following the Tier 3 analyses discussed above (Equation 13):

$$SV_{T3PR} = SV_X \times \left[\frac{Hrs(W \text{ and } E<M)}{Hrs(W)}\right], \qquad \textbf{Eqn. 13}$$

where:

SV_{T3PR} = SVs for the Tier 3 plume-rise screen;

SV_X = SVs either for the Tier 3 fisher scenario from the Tier 3 lake screen or the Tier 2 farmer/gardener scenario;

$Hrs(W \text{ and } E<M)$ = number of hours when winds are blowing toward the lake or farm/garden when the effective release height (physical stack height + plume-rise height) is less than the mixing height;

$Hrs(W)$ = number of hours when winds are blowing toward the lake or farm/garden of interest.

For the fisher scenario, after the SVs at each lake are adjusted based on plume rise, the screening tool reapplies the refined fisher screening calculation discussed in Section 3.3.1, but the order of fished lakes does not change.

For example, suppose two lakes are being assessed, having surface areas of 273 and 100 acres and having Tier 3 lake SVs for mercury of 5.7 and 3.0, respectively. Applying the refined-fisher scenario calculations (Section 3.3.1), the site-wide fisher-scenario SV for mercury is [5.7 × (273 / 373)] + [3.0 × (100 / 373)] = 5.0. Winds blow toward the first lake 1,800 hours per year, and during that time, the effective release height (physical stack height + plume-rise height) is below the mixing layer 1,000 hours; its plume-rise retention factor is 1,000/1,800=0.56. That means that 56 percent of the chemicals emitted remain in the mixing layer. Winds blow toward the second lake 500 hours per year, and during that time, the effective release height is below the mixing layer 400 hours per year; its plume-rise retention factor is 400 / 500 = 0.8. The Tier 3 plume-rise fisher-scenario SVs for mercury are 5.7 × 0.56 = 3.2 and 3.0 × 0.8 = 2.4. With the refined-fisher calculations, the site-wide fisher-scenario SV for mercury is [3.2 × (273 / 373)] + [2.4 × (100 / 373)] = 3.0.

4.6 Screen Using Hourly Time-series Meteorological Data and Effective Release Heights

If a Tier 3 plume-rise screen indicates potential ingestion risks remain, we refine the screen using hourly meteorological data from the closest weather station. Hourly meteorological data and estimates of plume rise are run through TRIM.FaTE and results compiled.[22] The use of time-series meteorological data, which capture hour-by-hour changes in each of the assessed meteorological parameters (instead of using the single annual average values as in the Tiers 1 and 2 screens), increases the accuracy of the assessment by accounting for potential correlations among meteorological parameter values over time.

This screen utilizes hourly effective release heights (computed in the plume-rise screen above) along with the hourly meteorological data associated with the facility instead of the binned

[22]As discussed in Section 3, the Tier 2 multipathway screening results are based on typical meteorological conditions prevailing at the facility being screened. This is in contrast to the modeling with TRIM.FaTE for the Tier 3 screen which incorporates hour-by-hour site-specific meteorological data and plume-rise estimates in new modeling runs.

meteorology statistics used in Tier 2. Using these data in combination with hourly effective release heights is a more complete evaluation of hourly chemical losses due to plume rise compared with the Tier 3 plume-rise screen described above. These time-varying release height and meteorology files are used in a run of TRIM.FaTE that also uses the facility's PB-HAP emissions and the Tier 2 spatial scenario associated with the lake being assessed. The TRIM.FaTE modeling, and subsequent exposure and risk estimation, leads directly to a screening-level cancer risk or HQ (i.e., a revised screening value). For simplicity in the software implementation of the Tiers 2 and 3 screens, the result of this Tier 3 time-series screen is converted to a time-series refinement factor—the revised SV divided by the SV after the Tier 3 lake screen. This ratio can then be multiplied by the SV after the Tier 3 lake screen, yielding the revised SV accounting for the time-series screen.

For a Tier 3 time-series screen, we use the facility's emissions, a time series of hourly effective release heights, a time series of hourly meteorological data (i.e., wind speed and direction, mixing height, temperature, and precipitation), and the Tier 2 spatial scenario that best matches each lake fished by the simulated subsistence fisher (or the relevant farm/garden locations if the farmer or gardener scenario is of concern). The site-specific hourly data and Tier 2 spatial layout are input to TRIM.FaTE, which provides estimated PB-HAP concentrations in environmental media that subsequently are used to estimate exposures and risk. If multiple lakes are fished (to allow for the subsistence fish ingestion rate), the percent of daily-ingested fish caught at each lake is multiplied by the screening level risk or HQ value for that lake. The PB-HAP-specific results are summed across all lakes (i.e., the refined-fisher calculations discussed in Section 3.3.1 are applied to the modeling results, using the screening tool described in Section 3).

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Technical Support Document
for the TRIM-Based Multipathway
Tiered Screening Methodology for RTR

Attachments

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Attachment A. TRIM.FaTE Inputs

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Exhibits, Attachment A

Exhibit A-1. TRIM.FaTE Simulation Parameters for the TRIM.FaTE Screening Scenario A-5
Exhibit A-2. Meteorological Input Values for the TRIM.FaTE Screening Scenario A-6
Exhibit A-3. Air Parameter Values for the TRIM.FaTE Screening Scenario A-7
Exhibit A-4. Soil and Groundwater Parameter Values for the TRIM.FaTE Screening Scenario A-7
Exhibit A-5. Runoff Assumptions for TRIM.FaTE Base Lake (L) Screening Scenario A-10
Exhibit A-6. Runoff Assumptions for TRIM.FaTE Base Farm (F) Screening Scenario A-10
Exhibit A-7. USLE Erosion Parameter Values for the TRIM.FaTE Base Lake (L) Screening Scenario A-11
Exhibit A-8. USLE Erosion Parameter Values for the TRIM.FaTE Base Farm (F) Screening Scenario A-12
Exhibit A-9. Terrestrial Plant Placement for the TRIM.FaTE Base Lake (L) Screening Scenario A-13
Exhibit A-10. Terrestrial Plant Placement for the TRIM.FaTE Base Farm (F) Screening Scenario A-13
Exhibit A-11. Terrestrial Plant Parameter Values for the TRIM.FaTE Screening Scenario A-14
Exhibit A-12. Surface Water Parameters for the TRIM.FaTE Screening Scenario A-17
Exhibit A-13. Sediment Parameter Values for the TRIM.FaTE Screening Scenario A-18
Exhibit A-14. Aquatic Animals Food Chain, Density, and Biomass for the TRIM.FaTE Screening Scenario A-19
Exhibit A-15. Arsenic Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario A-20
Exhibit A-16. Cadmium Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario A-20
Exhibit A-17. Mercury Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario A-21
Exhibit A-18. PAH Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario A-22
Exhibit A-19. Dioxin Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario A-24
Exhibit A-20. Arsenic Chemical-Specific Parameter Values for Abiotic Compartments in the TRIM.FaTE Screening Scenario A-26

Exhibit A-21. Cadmium Chemical-Specific Parameter Values for Abiotic Compartments in the TRIM.FaTE Screening Scenario A-26

Exhibit A-22. Mercury Chemical-Specific Parameter Values for Abiotic Compartments in the TRIM.FaTE Screening Scenario A-27

Exhibit A-23. PAH Chemical-Specific Parameter Values for Abiotic Compartments in the TRIM.FaTE Screening Scenario A-31

Exhibit A-24. Dioxin Chemical-Specific Parameters for Abiotic Compartments in the TRIM.FaTE Screening Scenario A-34

Exhibit A-25. Arsenic Chemical-Specific Parameters for Plant Compartments in the TRIM.FaTE Screening Scenario A-38

Exhibit A-26. Cadmium Chemical-Specific Parameters for Plant Compartments in the TRIM.FaTE Screening Scenario A-38

Exhibit A-27. Mercury Chemical-Specific Parameter Values for Plant Compartments in TRIM.FaTE Screening Scenario A-39

Exhibit A-28. PAH Chemical-Specific Parameter Values for Plant Compartments in TRIM.FaTE Screening Scenario A-40

Exhibit A-29. Dioxin Chemical-Specific Parameter Values for Plant Compartments in the TRIM.FaTE Screening Scenario A-43

Exhibit A-30. Arsenic Chemical-Specific Parameter Values for Aquatic Species in the RTR Screening Scenario A-44

Exhibit A-31. Cadmium Chemical-Specific Parameter Values for Aquatic Species in TRIM.FaTE Screening Scenario A-45

Exhibit A-32. Mercury Chemical-Specific Parameter Values for Aquatic Species in TRIM.FaTE Screening Scenario A-47

Exhibit A-33. PAH Chemical-Specific Parameter Values for Aquatic Species in TRIM.FaTE Screening Scenario A-48

Exhibit A-34. Dioxin Chemical-Specific Parameter Values for Aquatic Species in TRIM.FaTE Screening Scenario A-51

This attachment provides tables of the modeling inputs for the TRIM.FaTE screening scenario. Exhibit A-1 presents runtime settings for TRIM.FaTE. Exhibit A-2 and Exhibit A-3 present the meteorological and air parameter values, respectively, entered into the model.

Exhibit A-4 presents parameter values for soil and groundwater. Exhibit A-5 and Exhibit A-6 present runoff assumptions for the lake and farm scenarios, respectively; while Exhibit A-7 and Exhibit A-8 present parameter values for the universal soil loss equation (USLE) for calculating erosion for the lake and farm scenarios, respectively. Exhibit A-9 and Exhibit A-10 indicate the vegetation type assumed for each terrestrial parcel for the lake and farm scenarios, respectively, while Exhibit A-11 lists parameter values used terrestrial vegetation. Lake-parameter values for abiotic compartments are included in Exhibit A-12 for the surface water column and in Exhibit A-13 for the unconsolidated sediment layer (above the sediment sink). Parameter values for the biotic compartments in the lake (e.g., fish, invertebrates, algae) are included in Exhibit A-14.

Chemical-specific parameter values for the TRIM.FaTE scenarios (e.g., molecular weight, diffusion rate constants, Henry's law constant) are provided in Exhibit A-15 for arsenic, Exhibit A-16 for cadmium, Exhibit A-17 for mercury, Exhibit A-18 for PAHs, and Exhibit A-19 for dioxins. Chemical-specific parameter values for the TRIM.FaTE abiotic compartments (e.g., air, surface water, sediment, surface soil, root-zone soil) are presented in Exhibit A-20 for arsenic, Exhibit A-21 for cadmium, Exhibit A-22 for mercury, Exhibit A-23 for PAHs, and Exhibit A-24 for dioxins. Chemical-specific parameter values for the plant compartments (e.g., leaf, stem, root) are presented in Exhibit A-25 for arsenic, Exhibit A-26 for cadmium, Exhibit A-27 for mercury, Exhibit A-28 for PAHs, Exhibit A-29 for dioxins. Finally, chemical-specific parameter values for aquatic species are presented in Exhibit A-30 for arsenic, Exhibit A-31 for cadmium, Exhibit A-32 for mercury, Exhibit A-33 for PAHs, and Exhibit A-34 for dioxins.

Exhibit A-1. TRIM.FaTE Simulation Parameters for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value Used	Reference
Start of simulation	date/time	1/1/1990, midnight	Consistent with met data.
End of simulation	date/time	1/1/2040, midnight	Consistent with met data set; selected to provide a 50-year modeling period.
Simulation time step	hr	1	Selected value.
Output time step[a]	hr	4	Selected value.

[a]Output time step is set in TRIM.FaTE using the scenario properties "simulationStepsPerOutputStep" and "simulationTimeStep."

Exhibit A-2. Meteorological Input Values for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value Used	Reference
Meteorological Inputs			
Air temperature	degrees K	298	U.S. EPA 2005a.
Horizontal wind speed	m/sec	1.6 or varies	Tiers 1 and 2, and Tier 3 lake screen: The ~5th percentile of median annual wind speeds, partitioned by 8 wind directions, recorded at 824 MET stations across the United States in 2016. Tier 3 plume-rise and time-series screens: Varies by the hour; site-specific hourly meteorological data.
Vertical wind speed	m/sec	0.0	Assumption; vertical wind speed not used by any of the algorithms in the version of the TRIM.FaTE library used for screening.
Wind direction	degrees clockwise from N (blowing from)	3-days-on 4-days-off or varies	Tiers 1 and 2, and Tier 3 lake screen: "On" is defined as time during which wind is blowing across the source into the model domain. The weekly split was determined to be a health-protective setting by evaluating archived meteorology data (NCDC 1995). Tier 3 plume-rise and time-series screens: Varies by the hour; site-specific hourly meteorological data.
Rainfall rate	m^3[rain]/m^2 [surface area]-day	0.0041 or varies	Tiers 1 and 2, and Tier 3 lake screen: The ~95th percentile of the annual average precipitation for 824 MET stations across the United States was approximately 1.5 m/yr or 0.0041 m/day, based on 1981–2010 normals where available (812 MET stations) and based on 2016 values otherwise (11 MET stations). Tier 3 plume-rise and time-series screens: Varies by the hour; site-specific hourly meteorological data.
Mixing height (used to set air VE property named "top")	m	226 or varies	Tiers 1 and 2, and Tier 3 lake screen: The ~5th percentile of annual median mixing heights recorded at 824 MET stations across the United States in 2016. Tier 3 plume-rise and time-series screens: Varies by the hour; site-specific hourly meteorological data.
isDay_SteadyState_forAir	unitless	--	Value not used in current dynamic runs (would need to be reevaluated if steady-state runs are needed).
isDay_SteadyState_forOther	unitless	--	

Acronyms: MET = meteorological.

Exhibit A-3. Air Parameter Values for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value Used	Reference
Atmospheric dust load	kg[dust]/m^3[air]	6.15E-08	Bidleman 1988.
Density of air	g/cm^3	0.0012	U.S. EPA 1997.
Dust density	kg[dust]/m^3[dust]	1,400	Bidleman 1988.
Fraction organic matter on particulates	unitless	0.2	Harner and Bidleman 1998.

Exhibit A-4. Soil and Groundwater Parameter Values for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value Used	Reference
Surface Soil Compartment Type			
Air content	m^3[gas]/m^3[compartment]	0.28[a]	McKone et al. 2001.
Average vertical velocity of water (percolation)	m^3[water]/m^2[surface soil]-day (or m/day)	8.08E-04	Assumed as 0.2 times average precipitation for New England in McKone et al. 2001.
Boundary layer thickness above surface soil	m	0.005	Thibodeaux 1996; McKone et al. 2001 (Table 3).
Density of soil solids (dry weight)	kg[solid]/m^3[solid]	2600	Default in McKone et al. 2001 (Table 3).
Thickness – untilled[b]	m	0.01	McKone et al. 2001 (p. 30).
Thickness – tilled[b]	m	0.20	U.S. EPA 2005a.
Erosion fraction	unitless	varies[c]	See Exhibit A-5 and Exhibit A-6.
Fraction of area available for erosion	m^2[area available]/m^2[total]	1	Area assumed rural.
Fraction of area available for runoff	m^2[area available]/m^2[total]	1	Area assumed rural.
Fraction of area available for vertical diffusion	m^2[area available]/m^2[total]	1	Area assumed rural.
Fraction sand	unitless	0.25	Assumption.
Organic carbon fraction	kg[organic carbon]/kg[solids wet wt]	0.008[a]	U.S. average in McKone et al. 2001 (Tables 16 and A-3).
pH	unitless	6.8	Assumption.
Runoff fraction	unitless	varies[c]	See Exhibit A-5 and Exhibit A-6

Parameter Name	Units	Value Used	Reference
Total erosion rate	kg[soil]/m^2[surface soil]-day	varies[c]	See Exhibit A-7 and Exhibit A-8
Total runoff rate	m^3[water]/m^2[surface soil]-day	1.62E-03[a]	Calculated using scenario-specific precipitation rate and assumptions associated with water balance.
Water content	volume[water]/volume[compartment]	0.19[a]	McKone et al. 2001 (Table 15).
Root Zone Soil Compartment Type			
Air content	m^3[gas]/m^3[compartment]	0.25[a]	McKone et al 2001 (Table 16).
Average vertical velocity of water (percolation)	m^3[water]/m^2[surface soil]-day (or m/day)	8.08E-04	Assumed as 0.2 times average precipitation for New England in McKone et al. 2001.
Density of soil solids (dry weight)	kg[solid]/m^3[solid]	2,600	McKone et al. 2001 (Table 3).
Fraction sand	unitless	0.25	Assumption.
Thickness – untilled[b]	m	0.79	McKone et al. 2001 (Table 16, U.S. average).
Thickness – tilled[b]	m	0.6	Adjusted from McKone et al. 2001 (Table 16).
Organic carbon fraction	kg[organic carbon]/kg[solids wet wt]	0.008[a]	McKone et al. 2001 (Tables 16 and A-3, U.S. average).
pH	unitless	6.8 [a]	Assumption.
Water content	volume[water]/volume[compartment]	0.21[a]	McKone et al. 2001 (Table 16).
Vadose Zone Soil Compartment Type			
Air content	m^3[gas]/m^3[compartment]	0.22[a]	McKone et al. 2001 (Table 17).
Average vertical velocity of water (percolation)	m^3[water]/m^2[surface soil]-day (or m/day)	8.08E-04[a]	Assumed as 0.2 times average precipitation for New England in McKone et al. 2001.
Density of soil solids (dry weight)	kg[solid]/m^3[solid]	2,600	Default in McKone et al. 2001 (Table 3).
Fraction sand	unitless	0.35	Assumption.
Thickness[b]	m	1.4	McKone et al. 2001 (Table 17).
Organic carbon fraction	kg[organic carbon]/kg[solids wet wt]	0.003[a]	McKone et al. 2001 (Tables 16 and A-3, U.S. average).
pH	unitless	6.8	Assumption.
Water content	m^3[liquid]/m^3[compartment]	0.21[a]	McKone et al. 2001 (Table 17, U.S. average).

Parameter Name	Units	Value Used	Reference
Groundwater Compartment Type			
Thickness[b]	m	3	McKone et al. 2001 (Table 3).
Fraction sand	unitless	0.4	Assumption.
Organic carbon fraction	kg[organic carbon]/kg[solids wet wt]	0.004	Assumption.
pH	unitless	6.8	Assumption.
Porosity	L[total pore space]/L [total compartment]	0.2	Default in McKone et al. 2001 (Table 3).
Density of solid material	kg[solid]/m^3[solid]	2,600	Default in McKone et al. 2001 (Table 3).

[a]Scenario-specific parameters.
[b]Set using the volume element properties file.
[c]See Exhibit A-5, Exhibit A-6, Exhibit A-7, and Exhibit A-8 for erosion/runoff fractions and total erosion rates.

Exhibit A-5. Runoff Assumptions for TRIM.FaTE Base Lake (L) Screening Scenario

Originating Compartment	Destination Compartment	Runoff/Erosion Fraction
SurfSoil_Source	Sink	1.0
	SurfSoil_Parcel1	0
SurfSoil_Parcel1	SurfSoil_Source	0
	Lake	1.0
SurfSoil_Parcel2N	Lake	1.0
	SurfSoil_Parcel3	0
SurfSoil_Parcel2S	Lake	1.0
	SurfSoil_Parcel3	0
Lake	SurfSoil_Parcel1	0
	SurfSoil_Parcel2N	0
	SurfSoil_Parcel2S	0
	Lake	1.0
	SurfSoil_Parcel3	0
SurfSoil_Parcel3	SurfSoil_Parcel2N	0
	SurfSoil_Parcel2S	0
	Lake	1.0
	SurfSoil_Parcel4	0
SurfSoil_Parcel4	SurfSoil_Parcel3	1.0

Exhibit A-6. Runoff Assumptions for TRIM.FaTE Base Farm (F) Screening Scenario

Originating Compartment	Destination Compartment	Runoff/Erosion Fraction
SurfSoil_Source	Sink	1.0
	SurfSoil_Parcel1	0
SurfSoil_Parcel1	Sink	0.4
	SurfSoil_Source	0
	SurfSoil_Farm	0.6
	SurfSoil_Parcel2	0
SurfSoil_Farm	Sink	1.0
	SurfSoil_Parcel1	0
	SurfSoil_Parcel2	0
SurfSoil_Parcel2	Sink	0.4
	SurfSoil_Parcel1	0
	SurfSoil_Farm	0.6
	SurfSoil_Parcel3	0
SurfSoil_Parcel3	SurfSoil_Parcel2	1.0
	SurfSoil_Parcel4	0
SurfSoil_Parcel4	SurfSoil_Parcel3	1.0
	SurfSoil_Parcel5	0
SurfSoil_Parcel5	SurfSoil_Parcel4	1.0

Exhibit A-7. USLE Erosion Parameter Values for the TRIM.FaTE Base Lake (L) Screening Scenario

Soil Parcel	Area	Rainfall/ Erosivity Index	Soil Erodibility Index	Length-Slope Factor	Land Use	Cover Mgmt Factor	Supporting Practices Factor	Unit Soil Loss		Sediment Delivery Ratio[a]	Calculated (Adjusted) Erosion Rate
Code:	m^2	R (100 ft-ton/ac)	K (ton/ac/(100 ft-ton/acre))	LS (USCS)	type	C (USCS)	P	A (ton/ac/yr)	A ($kg/m^2/d$)	SDR^a	$kg/m^2/d$
Source	62,500	300	0.39	1.5	untilled soil	0.2	1	35.1	0.02156	0.5281	0.01138
Parcel1	116,891	300	0.39	1.5	grass	0.1	1	17.55	0.01078	0.4884	0.005264
Parcel2N	232,594	300	0.39	1.5	grass	0.1	1	17.55	0.01078	0.4481	0.004830
Parcel2S	232,594	300	0.39	1.5	grass	0.1	1	17.55	0.01078	0.4481	0.0048301
Parcel3	4,082,258	300	0.39	1.5	coniferous forest	0.1	1	17.55	0.01078	0.2088	0.002251
Parcel4	13,386,064	300	0.39	1.5	coniferous forest	0.1	1	17.55	0.01078	0.1800	0.001940

[a]Calculated using $SDR = a * (AL)^{-b}$, where *a* is the empirical intercept coefficient (based on the size of the watershed), AL is the total watershed area receiving deposition (m^2), and *b* is the empirical slope coefficient (always 0.125).

Exhibit A-8. USLE Erosion Parameter Values for the TRIM.FaTE Base Farm (F) Screening Scenario

Soil Parcel	Area	Rainfall/ Erosivity Index	Soil Erodibility Index	Length -Slope Factor	Land Use	Cover Mgmt Factor	Supporting Practices Factor	Unit Soil Loss		Sediment Delivery Ratio[a]	Calculated (Adjusted) Erosion Rate
Code:	m^2	R (100 ft-ton/ac)	K (ton/ac/(100 ft-ton/acre))	LS (USCS)	Type	C (USCS)	P	A (ton/ac/yr)	A ($kg/m^2/d$)	SDR^a	$kg/m^2/d$
Source	62,500	300	0.39	1.5	untilled soil	0.2	1	35.1	0.02156	0.5281	0.01138
Parcel1	116,891	300	0.39	1.5	grass	0.1	1	17.55	0.01078	0.4884	0.005264
Farm	40,633	300	0.39	1.5	tilled soil	0.2	1	35.1	0.02156	0.5573	0.01201
Parcel2	281,012	300	0.39	1.5	grass	0.1	1	17.55	0.01078	0.3960	0.004268
Parcel3	608,730	300	0.39	1.5	grass	0.1	1	17.55	0.01078	0.3595	0.003875
Parcel4	4,082,258	300	0.39	1.5	coniferous forest	0.1	1	17.55	0.01078	0.2088	0.002251
Parcel5	13,386,064	300	0.39	1.5	coniferous forest	0.1	1	17.55	0.01078	0.1800	0.001940

[a]Calculated using $SDR = a * (AL)^{-b}$, where *a* is the empirical intercept coefficient (based on the size of the watershed), AL is the total watershed area receiving deposition (m^2), and *b* is the empirical slope coefficient (always 0.125).

Exhibit A-9. Terrestrial Plant Placement for the TRIM.FaTE Base Lake (L) Screening Scenario

Surface Soil Volume Element	Surface Soil Depth (m)	Coniferous Forest	Grasses/ Herbs	None
Source	0.01 (untilled)			x
Parcel1	0.01		x	
Parcel2N	0.01		x	
Parcel2S	0.01		x	
Parcel3	0.01	x		
Parcel4	0.01	x		

Exhibit A-10. Terrestrial Plant Placement for the TRIM.FaTE Base Farm (F) Screening Scenario

Surface Soil Volume Element	Surface Soil Depth (m)	Coniferous Forest	Grasses/ Herbs	None
Source	0.01 (untilled soil)			x
Parcel1	0.01		x	
Farm	0.2 (tilled soil)			x
Parcel2	0.01		x	
Parcel3	0.01		x	
Parcel4	0.01	x		
Parcel5	0.01	x		

Exhibit A-11. Terrestrial Plant Parameter Values for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Coniferous[a]		Grass/Herb[a]	
		Value Used	Reference	Value Used	Reference
Leaf Compartment Type					
Allow exchange	1=yes, 0=no	1	-	Seasonal	Growing season: for screening scenario begins March 9 (set to 1), ends November 7 (set to 0). Nationwide 80th percentile.
Average leaf area index	m^2[total leaf area]/ m^2[underlying soil area]	5	Representative value for conifers, N. Nikolov, Oak Ridge National Laboratory	5	Mid-range of 4-6 for old fields, R.J. Luxmoore, Oak Ridge National Laboratory.
Calculate wet deposition interception fraction (Boolean)	1=yes, 0=no	1	Selected setting.	1	Selected setting.
Correction exponent, octanol to lipid	unitless	0.76	From roots, Trapp 1995.	0.76	From roots, Trapp 1995.
Degree stomatal opening	unitless	1	Set to 1 for daytime (when stomata are open; stomatal diffusion is turned off at night using a different property, IsDay).	1	Set to 1 for daytime (stomatal diffusion is turned off at night using a different property, IsDay).
Density of wet leaf	kg[leaf wet wt]/m^3[leaf wet]	820	Paterson et al. 1991.	820	Paterson et al. 1991.
Leaf wetting factor	m	3.00E-04	1E-04 to 6E-04 for different crops and elements, Müller and Pröhl 1993.	3.00E-04	1E-04 to 6E-04 for different crops and elements, Müller and Pröhl 1993.
Length of leaf	m	0.01	Professional judgment.	0.05	Professional judgment.
Lipid content	kg[lipid]/kg[leaf wet weight]	0.00224	European beech, Riederer 1995.	0.00224	European beech, Riederer 1995.
Litter fall rate	1/day	0.0021	Value assumes first-order relationship and that 99% of leaves fall in 6 years.	Seasonal	Leaf fall: for screening scenario begins November 7, ends December 6; rate = 0.15/day during this time (value assumes 99 percent of leaves fall in 30 days).

Parameter Name	Units	Coniferous[a]		Grass/Herb[a]	
		Value Used	Reference	Value Used	Reference
Stomatal area normalized effective diffusion path length	1/m	200	Wilmer and Fricker 1996.	200	Wilmer and Fricker 1996.
Vegetation attenuation factor	m^2/kg	2.9	Grass/hay, Baes et al. 1984.	2.9	Grass/hay, Baes et al. 1984.
Water content	kg[water]/kg[leaf wet wt]	0.8	Paterson et al. 1991.	0.8	Paterson et al. 1991.
Wet deposition interception fraction	unitless	0.2	Calculated based on 5 years of local met data, 1987-1991.	0.2	Calculated based on 5 years of local met data, 1987-1991.
Wet mass of leaf per soil area	kg[plant part wet wt]/ m^2[surface soil]	2	Calculated from leaf area index, leaf thickness (Simonich and Hites, 1994), density of wet foliage.	0.6	Calculated from leaf area index and Leith 1975a,b in Leith and Whitaker 1975.
Particle on Leaf Compartment Type					
Allow exchange	1=yes, 0=no	1	-	Seasonal[b]	See leaf compartment.
Volume particle per area leaf	m^3[leaf particles]/m^2[leaf]	1.00E-09	Based on particle density and size distribution for atmospheric particles measured on an adhesive surface, Coe and Lindberg 1987.	1.00E-09	Based on particle density and size distribution for atmospheric particles measured on an adhesive surface, Coe and Lindberg 1987.
Root Compartment Type – Nonwoody Only					
Allow exchange	1=yes, 0=no			Seasonal[b]	See leaf compartment.
Correction exponent, octanol to lipid	unitless			0.76	Trapp 1995.
Lipid content of root	kg[lipid]/kg[root wet wt]			0.011	From bean root, Trapp 1995.
Water content of root	kg[water]/kg[root wet wt]			0.8	Professional judgment.
Wet density of root	kg[root wet wt]/m^3[root wet]			820	Soybean, Paterson et al. 1991.
Wet mass per soil area	kg[plant part wet wt]/m^2[surface soil]			1.4	Temperate grassland, Jackson et al. 1996.

Parameter Name	Units	Coniferous[a]		Grass/Herb[a]	
		Value Used	Reference	Value Used	Reference
Stem Compartment Type – Nonwoody Only					
Allow exchange	1=yes, 0=no			Seasonal[b]	See leaf compartment
Correction exponent, octanol to lipid	unitless			0.76	From roots, Trapp 1995; in Trapp and McFarlane, eds. 1995.
Density of phloem fluid	kg[phloem]/m^3 [phloem]			1,000	Professional judgment.
Density of xylem fluid	kg[xylem fluid]/m^3[xylem fluid]			900	Professional judgment.
Flow rate of transpired water per leaf area	m^3[water]/m^2 [stem]-day			0.0048	Crank et al. 1981, as cited by Paterson et al. 1991.
Fraction of transpiration flow rate that is phloem rate	unitless			0.05	Paterson et al. 1991.
Lipid content of stem	kg[lipid]/kg[stem wet wt]			0.00224	Leaves of European beech, Riederer 1995.
Water content of stem	kg[water]/kg [stem wet wt]			0.8	Paterson et al. 1991.
Wet density of stem	kg[stem wet wt]/ m^3[stem wet]			830	Professional judgement.
Wet mass per soil area	kg[plant part wet wt]/m^2[surface soil]			0.24	Calculated from leaf and root biomass density based on professional judgement.

[a]See for assignment of plant types to surface soil compartments; no parcels are designated as coniferous forest, therefore, no values or references in these two columns can be made to vary seasonally (scenario-specific).

[b]Seasonal values; leaves must be present for exchanges with leaves.

Exhibit A-12. Surface Water Parameters for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value Used	Reference
Algae carbon content (mass fraction; dry wt basis)	unitless	0.465	APHA 1995.
Algae density in water column	g[algae wet wt]/L[water]	0.0025[a]	Millard et al. 1996.
Algae growth rate	1/day	0.7	Hudson et al. 1994, in Watras and Huckabee, eds. 1994. Also cited in Mason et al. 1995b.
Algae radius	um	2.5	Mason et al. 1995b.
Algae water content (mass fraction)	unitless	0.9	APHA 1995.
Average algae cell density (per volume cell, not water)	g[algae]/m^3[algae]	1,000,000	Mason et al. 1995b, Mason et al. 1996.
Boundary layer thickness above sediment	m	0.02	Cal EPA 1993.
Chloride concentration	mg[chloride/L[surface water]	8[a]	Kaushal et al. 2005.
Chlorophyll concentration	mg[chlorophyll]/L[surface water]	0.0029[a]	Nürnberg 1996.
Depth[b]	m	3.12[a]	WI DNR 2007– calculation based on relationship between drainage basin and lake area size[b].
Dimensionless viscous sublayer thickness	unitless	4	Ambrose et al. 1995.
Drag coefficient for water body	unitless	0.0011	Ambrose et al. 1995.
Flush rate	1/year	12.17[a]	Calculated based on pond dimensions and flow calculations.
Fraction Sand	unitless	0.25	Assumption.
Organic carbon fraction in suspended sediments	kg[organic carbon]/kg[solids wet wt]	0.02[a]	Professional judgement.
pH	unitless	7.3[a]	Professional judgement.
Suspended sediment deposition velocity	m/day	2[a]	Assumption (in sediment balance calculations)
Total suspended sediment concentration	kg[sediment]/m^3[water column]	0.05[a]	Assumption (in sediment balance calculations).
Water temperature	degrees K	298[a]	U.S. EPA 2005a.

[a]Scenario-specific parameters, values provided are for RTR screens.
[b]Set using the volume element properties named "top" and "bottom." If not set, depth computed via: d/(A*F), where d is the annual discharge (in m^3/year), A is the lake area (in m^2), and F is the flush rate per year.

Exhibit A-13. Sediment Parameter Values for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value Used	Reference
Depth[a]	m	0.05	McKone et al. 2001 (Table 3).
Fraction sand	unitless	0.25	Assumption.
Organic carbon fraction	kg[organic carbon]/kg[solids wet wt]	0.02[b]	McKone et al. 2001 (Table 3).
Porosity of the sediment zone	volume[total pore space]/volume[sediment compartment]	0.6	Assumption.
Solid material density in sediment	kg[sediment]/m^3[sediment]	2,600	McKone et al. 2001 (Table 3).
pH	unitless	7.3[b]	Assumption.
Sediment resuspension velocity	m/day	7.62E-05[b]	Calculated from sediment balance model.

[a]Unconsolidated sediment layer just below surface water column.
[b]Scenario-specific parameters, values provided are for Tier 1 screenings.

Exhibit A-14. Aquatic Animals Food Chain, Density, and Biomass for the TRIM.FaTE Screening Scenario

Aquatic Biota (Consuming Organism)	Fraction Diet[a]								Biomass (kg/m^2)[a]	Body Weight (kg)[b]	Reference
	Algae	Zooplankton	Benthic Invertebrate	Water Column Herbivore	Benthic Omnivore	Water Column Omnivore	Benthic Carnivore	Water Column Carnivore			
Benthic Invertebrate	0%	0%	0%	0%	0%	0%	0%	0%	0.020	2.55E-04	Assumption.
Water Column Herbivore	0%	100%	0%	0%	0%	0%	0%	0%	0.002	0.025	Assumption.
Benthic Omnivore	0%	0%	100%	0%	0%	0%	0%	0%	0.002	0.25	Assumption.
Water Column Omnivore	0%	0%	0%	100%	0%	0%	0%	0%	0.001	0.25	Assumption.
Benthic Carnivore	0%	0%	50%	0%	50%	0%	0%	0%	0.001	2.0	Assumption.
Water Column Carnivore	0%	0%	0%	0%	0%	100%	0%	0%	0.0002	2.0	Assumption.
Zooplankton	100%	0%	0%	0%	0%	0%	0%	0%	0.0064	5.70E-08	Assumption.

[a]Scenario-specific parameters, values provided are for RTR screening.
[b]Assumption across all scenarios.

Exhibit A-15. Arsenic Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario[a]

Parameter Name[a]	Units	Value	Reference
CAS number[b]	-	7440-38-2	-
Diffusion coefficient in pure air	m^2[air]/day	0.92	U.S. EPA 1996 as cited in U.S. EPA 1999.
Diffusion coefficient in pure water	m^2[water]/day	1.07E-04	U.S. EPA 1996 as cited in U.S. EPA 1999.
Henry's Law constant	Pa-m^3/mol	1.00E-37	U.S. EPA 1999.
Melting point	degrees K	1093	U.S. EPA 2004 as cited in U.S. EPA 2005a.
Molecular weight	g/mol	77.922	NCBI 2017
Octanol-air partition coefficient (Koa)	m^3[air]/m^3[octanol]	-	-
Octanol-water partition coefficient (Kow)	L[water]/kg[octanol]	-	-

[a]All parameters in this table are TRIM.FaTE chemical properties.
[b]This CAS numbers applies to elemental As.

Exhibit A-16. Cadmium Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario

Parameter Name[a]	Units	Value	Reference
CAS number[b]	-	7440-43-9	-
Diffusion coefficient in pure air	m^2[air]/day	0.71	U.S. EPA 1996 as in U.S. EPA 1999, Table A-2-35.
Diffusion coefficient in pure water	m^2[water]/day	8.16E-05	U.S. EPA 1996 as cited in U.S. EPA1999, Table A-2-35).
Henry's Law constant	Pa-m^3/mol	1.00E-37	U.S. EPA 1999 (Table A-2-35; assumed to be zero).
Melting point	degrees K	593.15	U.S. EPA 2004 as cited in U.S. EPA 2005a.
Molecular weight	g/mol	112.41	NCBI 2017 (rounded to five significant digits)
Octanol-air partition coefficient (K_{oa})	m^3[air]/m^3[octanol]	-	-
Octanol-water partition coefficient (Kow)	L[water]/kg[octanol]	-	-

[a]All parameters in this table are TRIM.FaTE chemical properties.
[b]This CAS numbers applies to elemental Cd; however, the cations of cadmium are being modeled.

Exhibit A-17. Mercury Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value			Reference
		Hg(0)[b]	Hg(2)[b]	MHg[b]	
CAS number	unitless	7439-97-6	14302-87-5	22967-92-6	ChemFinder
Diffusion coefficient in pure air	m^2[air]/day	0.478	0.478	0.456	U.S. EPA 1997.
Diffusion coefficient in pure water	m^2[water]/day	5.54E-05	5.54E-05	5.28E-05	U.S. EPA 1997.
Henry's Law constant	Pa-m^3/mol	719	7.19E-05	0.0477	U.S. EPA 1997.
Melting point	degrees K	234[c]	5.50E+02[d]	443[e]	See endnotes.
Molecular weight[f]	g/mol	201	201	216	U.S. EPA 1997.
Octanol-water partition coefficient (Kow)	L[water]/kg[octanol]	4.15	3.33	1.7	Mason et al. 1996.
Vapor washout ratio	m^3[air]/m^3[rain]	1,200	1.6E+06	0	U.S. EPA 1997, based on Petersen et al. 1995.

[a]All parameters in this table are TRIM.FaTE chemical properties.
[b]On this and all following tables, Hg(0) = elemental mercury, Hg(2+) = divalent mercury, and MHg = methyl mercury.
[c]U.S. EPA 2004 as cited in U.S. EPA 2005a.
[d]SRC 2005 as cited in U.S. EPA 2005a.
[e]USDHHS 1992 as cited in CARB 1994.
[f]NCBI 2017 rounded to 3 significant figures.

Exhibit A-18. PAH Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value							
		2Methyl	**712DMB**	**Acenaphthene**	**Acenaphthylene**	**BaA**	**BaP**	**BbF**	**BghiP**
CAS number	unitless	91-57-6	57-97-6	83-32-9	208-96-8	56-55-3	50-32-8	205-99-2	191-24-2
Diffusion coefficient in pure air	m^2/day	0.451	0.691	0.00864	0.388	0.441	0.372	0.00864	0.19
Diffusion coefficient in pure water	m^2/day	6.70E-05	6.91E-05	8.64E-05	6.03E-05	7.78E-05	7.78E-05	8.64E-05	4.54E-05
Henry's Law constant	Pa-m^3/mol	50.56	0.203	18.5	12.7	1.22	0.074	0.0485	0.0278
Melting point	degrees K	307.6	395.5	366.4	364.8	433.5	452.1	441	545.5
Molecular weight	g/mol	142.2	256.34	154.21	152.2	228.29	252.31	252.31	276.33
Octanol-water partition coefficient (Kow)	L[water]/L[octanol]	7.24E+03	6.31E+05	8.32E+03	1.00E+04	6.17E+05	9.33E+05	6.03E+05	4.27E+06

Parameter Name	Units	Value					
		BkF	**Chr**	**DahA**	**Fluoranthene**	**Fluorene**	**IcdP**
CAS number	unitless	207-08-9	218-01-9	53-70-3	206-44-0	86-73-7	193-39-5
Diffusion coefficient in pure air	m^2/day	0.00864	0.00864	0.00864	0.00864	0.00864	0.00864
Diffusion coefficient in pure water	m^2/day	8.64E-05	8.64E-05	8.64E-05	8.64E-05	8.64E-05	8.64E-05
Henry's Law constant	Pa-m^3/mol	0.043	0.53	0.0076	1.96	9.81	0.029
Melting point	degrees K	490	528.5	542.5	383.19	387.77	435
Molecular weight	g/mol	252.31	228.29	278.36	202.25	166.22	276.33
Octanol-water partition coefficient (Kow)	L[water]/L[octanol]	8.71E+05	5.37E+05	3.16E+06	1.45E+05	1.51E+04	5.25E+06

Parameter Name	Units	Reference
CAS number	unitless	Chemfinder database
Diffusion coefficient in pure air	m^2/day	U.S. EPA 2005a. Exceptions include U.S. EPA 1995a (7,12-dimethylbenz(a)anthracene), and U.S. EPA 2004 as cited in U.S. EPA 2005b (2-methylnapthalene, acenaphthylene, and benzo(g,h,i)perylene).
Diffusion coefficient in pure water	m^2/day	U.S. EPA 2005a. Exceptions include U.S. EPA 1995a (7,12-dimethylbenz(a)anthracene), and U.S. EPA 2004 as cited in U.S. EPA 2005b (2-methylnapthalene, acenaphthylene, and benzo(g,h,i)perylene).
Henry's Law constant	Pa-m^3/mol	All values cited in Mackay et al. 2006, with exception of 7,12-dimethylbenz(a)anthracene, which is from ToxNet HSDB, derived from Meylen 1991.[a] [Original studies, cited by Mackay et al. 2006 but not in the reference list for this attachment, include Bamford et al. 1999, Yaws et al. 1991, Staudinger and Roberts 2001, Altschuh et al. 1999, Hulscher et al. 1992, and Eastcott et al. 1988.]
Melting point	degrees K	Lide 2003 as cited in Mackay et al. 2006.
Molecular weight	g/mol	Mackay et al. 2006.
Octanol-water partition coefficient (Kow)	L[water]/L[octanol]	All values cited in Mackay et al. 2006. [Original studies, cited by Mackay et al. 2006 but not in the reference list for this attachment, include Hansch et al. 1995, Passivirta et al. 1999, and Sangster 1993.]

Exhibit A-19. Dioxin Chemical-Specific Parameter Values for the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value						
		1,2,3,4,6,7,8,9-OCDD	1,2,3,4,6,7,8,9-OCDF	1,2,3,4,6,7,8-HpCDD	1,2,3,4,6,7,8-HpCDF	1,2,3,4,7,8,9-HpCDF	1,2,3,4,7,8-HxCDD	1,2,3,4,7,8-HxCDF
CAS number	unitless	3268-87-9	39001-02-0	35822-46-9	67562-39-4	55673-89-7	39227-28-6	70648-26-9
Diffusion coefficient in pure air	m^2/day	0.751	0.168	0.782	0.176	0.176	0.816	0.183
Diffusion coefficient in pure water	m^2/day	6.91E-05	6.91E-05	6.91E-05	6.91E-05	6.91E-05	6.91E-05	6.91E-05
Henry's Law constant	Pa-m^3/mol	0.68	0.19	1.28	1.43	1.42	1.08	1.449
Melting point	degrees K	598.7	532.2	537.7	509.7	495.2	547.2	499.2
Molecular weight	g/mol	460.76	443.76	425.31	409.31	409.31	390.87	374.87
Octanol-water partition coefficient (Kow)	L[water]/L[octanol]	1.58E+08	1.00E+08	1.00E+08	2.51E+07	7.94E+06	6.31E+07	1.00E+07
Parameter Name	**Units**	**Value**						
		1,2,3,6,7,8-HxCDD	1,2,3,6,7,8-HxCDF	1,2,3,7,8,9-HxCDD	1,2,3,7,8,9-HxCDF	1,2,3,7,8-PeCDD	1,2,3,7,8-PeCDF	2,3,4,6,7,8-HxCDF
CAS number	unitless	57653-85-7	57117-44-9	19408-74-3	72918-21-9	40321-76-4	57117-41-6	60851-34-5
Diffusion coefficient in pure air	m^2/day	0.816	0.183	0.816	0.183	0.854	0.192	0.183
Diffusion coefficient in pure water	m^2/day	6.91E-05	6.91E-05	6.91E-05	6.91E-05	6.91E-05	6.91E-05	6.91E-05
Henry's Law constant	Pa-m^3/mol	1.11	0.741	1.11	1.115	0.26	0.507	1.115
Melting point	degrees K	558.7	505.7	516.7	520.7	513.7	499.2	512.7
Molecular weight	g/mol	390.87	374.87	390.87	374.87	356.42	340.42	374.87
Octanol-water partition coefficient (Kow)	L[water]/L[octanol]	1.62E+08	8.31E+07	1.62E+08	3.80E+07	1.86E+07	6.17E+06	8.31E+07

Parameter Name	Units	Value			Reference
		2,3,4,7,8-PeCDF	2,3,7,8-TCDD	2,3,7,8-TCDF	
CAS number	unitless	57117-31-4	1746-01-6	51207-31-9	ChemFinder
Diffusion coefficient in pure air	m^2/day	0.192	0.899	0.203	U.S. EPA 2000b cited in U.S. EPA 2005a. Exception: U.S. EPA 2004 cited in U.S. EPA 2005a (for 2,3,7,8-TCDD and 2,3,7,8-TCDF).
Diffusion coefficient in pure water	m^2/day	6.91E-05	4.84E-05	5.19E-05	U.S. EPA 1995b cited in U.S. EPA 2005a. Exception: U.S. EPA 2004 cited in U.S. EPA 2005a (for 2,3,7,8-TCDD and 2,3,7,8-TCDF).
Henry's Law constant	Pa-m^3/mol	0.505	3.33	1.459	Mackay et al. 1992 cited in U.S. EPA 2000b. Exceptions: Sijm et al. 1989 cited in U.S. EPA 2000b (1,2,3,7,8-PeCDD); and U.S. EPA 2000b cited in U.S. EPA 2005a (for 1,2,3,6,7,8-HxCDD; 1,2,3,7,8,9-HxCDD; 1,2,3,4,7,8-HxCDF; 1,2,3,7,8,9-HxCDF; 2,3,4,6,7,8-HxCDF; OCDF; and 1,2,3,7,8 PeCDF).
Melting point	degrees K	469.4	578.7	500.7	Rordorf 1987 cited in U.S. EPA 2000b. Exception: Friesen et al. 1985 cited in U.S. EPA 2000b (for OCDD).
Molecular weight	g/mol	340.42	321.98	305.98	U.S. EPA 2000b cited in U.S. EPA 2005a.
Octanol-water partition coefficient (Kow)	L[water]/L[octanol]	3.16E+06	6.31E+06	1.26E+06	Mackay et al. 1992 cited in U.S. EPA 2000b. Exceptions: Passivirta et al. 1999 cited in Mackay et al. 2006 (1,2,3,7,8-PeCDD); U.S. EPA 2000a (for 1,2,3,6,7,8-HxCDD, 1,2,3,7,8,9-HxCDD, 1,2,3,6,7,8-HxCDF, 1,2,3,7,8,9-HxCDF and 2,3,4,6,7,8-HxCDF); Sijm et al. 1989 cited in U.S. EPA 2000b (for 1,2,3,7,8-PeCDF); and Broman et al. 1991 cited in Mackay 2006 (for 1,2,3,4,7,8,9-HpCDF).

Exhibit A-20. Arsenic Chemical-Specific Parameter Values for Abiotic Compartments in the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value	Reference
Air Compartment Type			
Particle dry deposition velocity (vdep)	m/day	500	McKone et al. 2001
Washout ratio	m^3[air]/m^3[rain]	200,000	MacKay et al. 1986.
Surface Soil Compartment Type			
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	Set to no.
Root Zone Soil Compartment Type			
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	Set to no.
Vadose Zone Soil Compartment Type			
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	Set to no.
Surface Water Compartment Type			
Ratio of concentration in water to concentration in algae to concentration dissolved in water	L[water]/g[algae wet wt]	0.155	Mean value from Table 5.5 of Crompton 1998.

Exhibit A-21. Cadmium Chemical-Specific Parameter Values for Abiotic Compartments in the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value	Reference
Air Compartment Type			
Particle dry deposition velocity (vdep)	m/day	260	Calculated from Muhlbaier and Tissue 1980.
Washout ratio	m^3[air]/m^3[rain]	200,000	MacKay et al. 1986.
Surface Soil Compartment Type			
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	Set to no.
Root Zone Soil Compartment Type			
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	Set to no.
Vadose Zone Soil Compartment Type			
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	Set to no.
Surface Water Compartment Type			
Ratio of concentration in water to concentration in algae to concentration dissolved in water	L[water]/g[algae wet wt]	1.87	McGeer et al. 2003.

Exhibit A-22. Mercury Chemical-Specific Parameter Values for Abiotic Compartments in the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value			Reference
		Hg(0)	Hg(2)	MHg	
Air Compartment Type					
Particle dry deposition velocity (vdep)	m/day	500	500	500	CalTOX value cited in McKone et al. 2001.
Demethylation rate	1/day	N/A	N/A	0	Assumption.
Methylation rate	1/day	N/A	0	N/A	Assumption.
Oxidation rate	1/day	0.00385	N/A	N/A	Low end of half-life range (6 months to 2 years) in U.S. EPA 1997.
Reduction rate	1/day	N/A	0	N/A	Assumption.
Washout ratio	m^3[air]/m^3[rain]	2E+5	2E+5	2E+5	Mackay et al. 1986.
Surface Soil Compartment Type					
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	0	0	Set to no.
Soil-water partition coefficient	L[water]/kg[soil wet wt]	1,000	58,000	7,000	U.S. EPA 1997.
Vapor dry deposition velocity	m/day	50	2,500	N/A	Hg(0) – from Lindberg et al. 1992; Hg(2) – estimate by EPA using the Industrial Source Complex (ISC) Model - [See Vol. III, App. A of the Mercury Study Report (U.S. EPA 1997)]; MHg not emitted from source.
Demethylation rate	1/day	N/A	N/A	0.06	Range reported in Porvari and Verta 1995 is 3E-2 to 6E-2 /day; value is average maximum potential demethylation rate constant under anaerobic conditions.
Methylation rate	1/day	N/A	0.001	N/A	Range reported in Porvari and Verta 1995 is 2E-4 to 1E-3 /day; value is average maximum potential methylation rate constant under anaerobic conditions.
Oxidation rate	1/day	0	N/A	N/A	Value assumed in U.S. EPA 1997.
Reduction rate	1/day	N/A	1.25E-05	N/A	Value used for untilled surface soil (2 cm), 10% moisture content, in U.S. EPA 1997; general range is 0.0013–0.0001/day x moisture_content for forested region (Lindberg 1996; Carpi and Lindberg 1997).

Parameter Name	Units	Value			Reference
		Hg(0)	Hg(2)	MHg	
Root Zone Soil Compartment Type					
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	0	0	Set to no.
Soil-water partition coefficient (Kd)	L[water]/kg[soil wet wt]	1,000	58,000	7,000	U.S. EPA 1997.
Demethylation rate	1/day	N/A	N/A	0.06	Range reported in Porvari and Verta 1995 is 3E-2 to 6E-2 /day; value is average maximum potential demethylation rate constant under anaerobic conditions.
Methylation rate	1/day	NA	0.001	NA	Range reported in Porvari and Verta 1995 is 2E-4 to 1E-3 /day; value is average maximum potential methylation rate constant under anaerobic conditions.
Oxidation rate	1/day	0	0	0	Value assumed in U.S. EPA 1997.
Reduction rate	1/day	NA	3.25E-06	NA	Value used for tilled surface soil (20cm), 10% moisture content, in U.S. EPA 1997 (Lindberg 1996; Carpi and Lindberg 1997).
Vadose Zone Soil Compartment Type					
Use input characteristic depth (Boolean)	0 = no, Else = yes	0	0	0	Set to no.
Soil-water partition coefficient	L[water]/kg[soil wet wt]	1,000	58,000	7,000	U.S. EPA 1997.
Demethylation rate	1/day	N/A	N/A	0.06	Range reported in Porvari and Verta 1995 is 3E-2 to 6E-2 /day; value is average maximum potential demethylation rate constant under anaerobic conditions.
Methylation rate	1/day	0	0.001	0	Range reported in Porvari and Verta 1995 is 2E-4 to 1E-3 /day; value is average maximum potential methylation rate constant under anaerobic conditions.
Oxidation rate	1/day	0	0	0	Value assumed in U.S. EPA 1997.

Parameter Name	Units	Value			Reference
		Hg(0)	Hg(2)	MHg	
Reduction rate	1/day	0	3.25E-06	0	Value used for tilled surface soil (20cm), 10% moisture content, in U.S. EPA 1997 (Lindberg 1996; Carpi and Lindberg 1997).
Groundwater Compartment Type					
Soil-water partition coefficient	L[water]/kg[soil wet wt]	1,000	58,000	7,000	U.S. EPA 1997.
Demethylation rate	1/day	N/A	N/A	0.06	Range reported in Porvari and Verta 1995 is 3E-2 to 6E-2 /day; value is average maximum potential demethylation rate constant under anaerobic conditions.
Methylation rate	1/day	N/A	0.001	N/A	Range reported in Porvari and Verta 1995 is 2E-4 to 1E-3 /day; value is average maximum potential methylation rate constant under anaerobic conditions.
Oxidation rate	1/day	1.00E-08	N/A	N/A	Small default nonzero value (0 assumed in U.S. EPA 1997).
Reduction rate	1/day	N/A	3.25E-06	N/A	Value used for tilled surface soil (20cm), 10% moisture content, in U.S. EPA 1997 (Lindberg 1996; Carpi and Lindberg 1997).
Surface Water Compartment Type					
Algal surface area-specific uptake rate constant	nmol/[μm^2-day-nmol]	0	2.04E-10	3.60E-10	Assumes radius = 2.5mm, Mason et al. 1995b, Mason et al. 1996; Hg(0) assumed same as Hg(2).
D_{ow} ("overall Kow")	L[water]/kg[octanol]	0	-[a]	-[b]	Mason et al. 1996.
Solids-water partition coefficient	L[water]/kg[solids wet wt]	1E+3	1E+5	1E+5	U.S. EPA 1997.
Vapor dry deposition velocity	m/day	N/A	2,500	N/A	U.S. EPA 1997 (Vol. III, App. A).
Demethylation rate	1/day	N/A	N/A	0.013	Average range of 1E-3 to 2.5E-2/day from Gilmour and Henry 1991.
Methylation rate	1/day	N/A	0.001	N/A	Value used in U.S. EPA 1997; range is 1E-4 to 3E-4/day (Gilmour and Henry 1991).
Oxidation rate	1/day	0	N/A	N/A	Assumption.

Parameter Name	Units	Value			Reference
		Hg(0)	Hg(2)	MHg	
Reduction rate	1/day	N/A	0.0075	N/A	Value used in U.S. EPA 1997; reported values range from less than 5E-3/day for depths greater than 17m, up to 3.5/day (Xiao et al. 1995; Vandal et al. 1995; Mason et al. 1995a; Amyot et al. 1997).
Sediment Compartment Type					
Solids-water partition coefficient (Kd)	L[water]/kg[solids wet wt]	3,000	50,000	3,000	U.S. EPA 1997.
Demethylation rate	1/day	N/A	N/A	0.0501	Average range of 2E-4 to 1E-1/day from Gilmour and Henry 1991.
Methylation rate	1/day	N/A	1.00E-04	N/A	Value used in U.S. EPA 1997; range is 1E-5 to 1E-3/day, Gilmour and Henry 1991.
Oxidation rate	1/day	0	N/A	N/A	Assumption.
Reduction rate	1/day	N/A	1.00E-06	N/A	Inferred value based on presence of Hg(0) in sediment porewater (U.S. EPA 1997; Vandal et al. 1995).

[a]TRIM.FaTE Formula Property, which varies from 0.025 to 1.625 depending on pH and chloride concentration.
[b]TRIM.FaTE Formula Property, which varies from 0.075 to 1.7 depending on pH and chloride concentration.

Exhibit A-23. PAH Chemical-Specific Parameter Values for Abiotic Compartments in the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value								
		2Methyl	712DMB	Acenaph-thene	Acenaph-thylene	BaA	BaP	BbF	BghiP	BkF
Air Compartment Type										
Particle dry deposition velocity	m/day	500	500	500	500	500	500	500	500	500
Half-life	day	0.154	0.092	0.3	0.208	0.125	0.046	0.596	0.215	0.458
Washout ratio	m3[air]/m3[rain]	2E+5	2E+5	2E+5	2E+5	2E+5	2E+5	2E+5	2E+5	2E+5
Surface Soil Compartment Type										
User input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	0	0	0	0
Half-life	day	18	24	56	66.5	680	530	610	415	2140
Root Zone Soil Compartment Type										
User input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	0	0	0	0
Half-life	day	18	24	56	66.5	680	530	610	415	2140
Vadose Zone Soil Compartment Type										
User input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	0	0	0	0
Half-life	day	36	48	112	133	1360	1060	1220	830	4280
Groundwater Compartment Type										
Half-life	day	36	48	112	133	1360	1060	1220	830	4280
Surface Water Compartment Type										
Ratio Of Conc In Algae To Conc Dissolved In Water	(g[chem]/kg[algae]) / (g[chem]/L[water])	2.6	333.4	3	3.7	325	510	317	1539	473
Half-life	day	78	216	25	184	0.375	0.138	90	1670	62.4
Sediment Compartment Type										
Half-life	day	2290	2290	2290	2290	2290	2290	2290	2290	2290

Parameter Name	Units	Value					Reference
		Chr	DahA	Fluor-anthene	Fluor-ene	IcdP	
Air Compartment Type							
Particle dry deposition velocity	m/day	500	500	500	500	500	McKone et al. 2001.
Half-life	day	0.334	0.178	0.46	0.46	0.262	Howard et al. 1991 / upper bound measured or estimated value. Exceptions include ATSDR 2005 (2-Methylnaphthalene), U.S. EPA 1998 (7,12-Dimethylbenz(a)anthracene, Benzo(g,h,i)perylene, and Fluoranthene) / average of range, HSDB 2001d (Acenaphthene), HSDB 2001b (Acenaphthylene), and Spero et al. 2000 (Fluorene).
Washout ratio	m^3[air]/m^3[rain]	2E+5	2E+5	2E+5	2E+5	2E+5	Mackay et al. 1986 (for chemicals that are primarily or entirely in particle form.
Surface Soil Compartment Type							
User input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	Set to no.
Half-life	day	1000	940	275	33	730	MacKay et al. 2000 / average of range. Exceptions include ATSDR 2005 (2-Methylnaphthalene = value recorded for Napthalene), U.S. EPA 1998 (7,12-Dimethylbenz(a) anthracene, Benzo(g,h,i)perylene, and Fluoranthene) / average of range, HSDB 2001d (Acenaphthene), HSDB 2001b (Acenaphthylene), and HSDB 2001e (Fluorene).
Root Zone Soil Compartment Type							
User input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	Set to no.
Half-life	day	1000	940	275	33	730	Howard et al. 1991 / upper bound measured or estimated value. Exceptions include ATSDR 2005 (2-Methylnaphthalene = value recorded for Napthalene), U.S. EPA 1998 (7,12-Dimethylbenz(a)anthracene, Benzo(g,h,i)perylene, and Fluoranthene) / average of range, HSDB 2001d (Acenaphthene), HSDB 2001b (Acenaphthylene), and HSDB 2001e (Fluorene).

Vadose Zone Soil Compartment Type							
User input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	Assumption.
Half-life	day	2000	1880	550	66	1460	Howard et al. 1991 / upper bound measured or estimated value. Exceptions include ATSDR 2005 (2-Methylnaphthalene = value recorded for Napthalene), U.S. EPA 1998 (7,12-Dimethylbenz(a)anthracene, Benzo(g,h,i)perylene, and Fluoranthene) / twice average of range, HSDB 2001d (Acenaphthene) / multiplied by 2, HSDB 2001b (Acenaphthylene) / multiplied by 2, and HSDB 2001e (Fluorene) / multiplied by 2.
Groundwater Compartment Type							
Half-life	day	2000	1880	550	66	1460	Howard et al. 1991 / upper bound measured or estimated value. Exceptions include ATSDR 2005 (2-Methylnaphthalene = value recorded for Napthalene), U.S. EPA 1998 (7,12-Dimethylbenz(a)anthracene, Benzo(g,h,i)perylene, and Fluoranthene) / twice average of range, HSDB 2001d (Acenaphthene) / multiplied by 2, HSDB 2001b (Acenaphthylene) / multiplied by 2, and HSDB 2001e (Fluorene) / multiplied by 2.
Surface Water Compartment Type							
RatioOfConcInAlgaeToConcDissolvedInWater	(g[chem]/kg[algae])/(g[chem]/L[water])	280	1388	67.4	5.8	1653	Calculated from Kow from Del Vento and Dachs 2002.
Half-life	day	1.626	97.8	160	8.5	750	Howard et al. 1991 / upper bound measured or estimated value. Exceptions include HSDB 2005 (2-Methylnaphthalene), HSDB 2001a (7-12 Dimethylbenz(a)anthracene), HSDB 2001d (Acenaphthene), HSDB 2001b (Acenaphthylene), and HSDB 2001c (Benzo(g,h,i)perylene), Montgomery 2000 (Fluoranthene), and Boyle 1985 (Fluorene).
Sediment Compartment Type							
Half-life	day	2290	2290	2290	2290	2290	Mackay et al. 1992 / PAH values are the mean half-life of the log class that Mackay et al. assigned for sediment, except for BbF and IcdP, which were not in Table 2.3 of Mackay et. al.

Exhibit A-24. Dioxin Chemical-Specific Parameters for Abiotic Compartments in the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value					
		1,2,3,4,6,7,8,9-OCDD	1,2,3,4,6,7,8,9-OCDF	1,2,3,4,6,7,8-HpCDD	1,2,3,4,6,7,8-HpCDF	1,2,3,4,7,8,9-HpCDF	1,2,3,4,7,8-HxCDD
Air Compartment Type							
Deposition Velocity	m/day	500	500	500	500	500	500
Half-life	day	162	321	64	137	122	42
Washout Ratio	m^3[air]/m^3[rain]	91000	22000	64000	32000	32000	9000
Surface Soil Compartment Type							
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	0
Halflife	day	3650	3650	3650	3650	3650	3650
Root Zone Soil Compartment Type							
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth	0 = No, Else = Yes	0	0	0	0	0	0
Half-life	day	3650	3650	3650	3650	3650	3650
Vadose Zone Soil Compartment Type							
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	0
Half-life	day	1008	1008	1008	1008	1008	1008
Groundwater Compartment Type							
Half-life	day	1008	1008	1008	1008	1008	1008
Surface Water Compartment Type							
Ratio Of Conc In Algae To Conc Dissolved In Water	(g[chem]/g[algae])/ (g[chem]/L[water])	5.31	4.54	4.54	2.83	1.9	3.88
Half-life	day	0.67	0.58	47	0.58	0.58	6.3
Sediment Compartment Type							
Half-life	day	1095	1095	1095	1095	1095	1095

Parameter Name	Units	Value					
		1,2,3,4,7,8-HxCDF	**1,2,3,6,7,8-HxCDD**	**1,2,3,6,7,8-HxCDF**	**1,2,3,7,8,9-HxCDD**	**1,2,3,7,8,9-HxCDF**	**1,2,3,7,8-PeCDD**
Air Compartment Type							
Deposition velocity	m/day	500	500	500	500	500	500
Half-life	day	78	28	55	28	51	18
Washout ratio	m^3[air]/m^3[rain]	10000	9000	10000	9000	10000	18000
Surface Soil Compartment Type							
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	0
Half-life	day	3650	3650	3650	3650	3650	3650
Root Zone Soil Compartment Type							
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth	0 = No, Else = Yes	0	0	0	0	0	0
Half-life	day	3650	3650	3650	3650	3650	3650
Vadose Zone Soil Compartment Type							
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0	0
Half-life	day	1008	1008	1008	1008	1008	1008
Groundwater Compartment Type							
Half-life	day	1008	1008	1008	1008	1008	1008
Surface Water Compartment Type							
Ratio Of Conc In Algae To Conc Dissolved In Water	(g[chem]/g[algae])/ (g[chem]/L[water])	2.06	5.36	4.25	5.36	3.26	1.55
Half-life	day	0.58	6.3	0.58	6.3	0.58	2.7
Sediment Compartment Type							
Half-life	day	1095	1095	1095	1095	1095	1095

Parameter Name	Units	Value				
		1,2,3,7,8-PeCDF	2,3,4,6,7,8-HxCDF	2,3,4,7,8-PeCDF	2,3,7,8-TCDD	2,3,7,8-TCDF
Air Compartment Type						
Deposition velocity	m/day	500	500	500	500	500
Half-life	day	31	59	33	12	19
Washout ratio	m^3[air]/m^3[rain]	13000	10000	14000	18000	19000
Surface Soil Compartment Type						
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0
Half-life	day	3650	3650	3650	3650	3650
Root Zone Soil Compartment Type						
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth	0 = No, Else = Yes	0	0	0	0	0
Half-life	day	3650	3650	3650	3650	3650
Vadose Zone Soil Compartment Type						
Input characteristic depth	m	0.08	0.08	0.08	0.08	0.08
Use input characteristic depth (Boolean)	0 = No, Else = Yes	0	0	0	0	0
Half-life	day	1008	1008	1008	1008	1008
Groundwater Compartment Type						
Half-life	day	1008	1008	1008	1008	1008
Surface Water Compartment Type						
Ratio Of Conc In Algae To Conc Dissolved In Water	(g[chem]/g[algae])/ (g[chem]/L[water])	1.75	4.26	1.39	1.76	0.71
Half-life	day	0.19	0.58	0.19	2.7	0.18
Sediment Compartment Type						
Half-life	day	1095	1095	1095	1095	1095

Parameter Name	Reference
Air Compartment Type	
Deposition velocity	McKone et al. 2001.
Half-life	Atkinson 1996 as cited in U.S. EPA 2000b; vapor phase reaction with hydroxyl radical.
Washout ratio	Vulykh et al. 2001.
Surface Soil Compartment Type	
Input characteristic depth	Not used (model set to calculate value).
Use input characteristic depth (Boolean)	Set to no.
Half-life	Mackay et al. 2000; the degradation rate was cited by multiple authors, value is for 2,3,7,8-TCDD.
Root Zone Soil Compartment Type	
Input characteristic depth	Not used (model set to calculate value).
Use input characteristic depth	Set to no.
Half-life	Mackay et al. 2000; the degradation rate was cited by multiple authors, value is for 2,3,7,8-TCDD.
Vadose Zone Soil Compartment Type	
Input characteristic depth	Not used (model set to calculate value).
Use input characteristic depth (Boolean)	Set to no.
Half-life	Average value of the range presented in Mackay et al. 2000; based on estimated unacclimated aerobic biodegradation half-life, value is for 2,3,7,8-TCDD.
Groundwater Compartment Type	
Half-life	Average value of the range presented in Mackay et al. 2000; based on estimated unacclimated aerobic biodegradation half-life, value is for 2,3,7,8-TCDD.
Surface Water Compartment Type	
Ratio Of Conc In Algae To Conc Dissolved In Water	Estimated from Kow value using model from DelVento and Dachs 2002
Half-life	Kim and O'Keefe 1998, as cited in U.S. EPA 2000b.
Sediment Compartment Type	
Half-life	Estimation based on Adriaens and Grbic-Galic 1992,1993 and Adriaens et al. 1995 as cited in U.S. EPA 2000b.

Exhibit A-25. Arsenic Chemical-Specific Parameters for Plant Compartments in the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value	Reference
Leaf Compartment Type			
Transfer factor to leaf particle	1/day	0.002	Assumption (assume 1% of transfer factor from leaf particle to leaf).
Particle on Leaf Compartment Type			
Transfer factor to leaf	1/day	0.2	Assumption.
Root Compartment Type - Grasses and Herbs[a]			
Root to Root Soil Partition-Alpha of Steady State	unitless	0.95	Selected value.
Root to Root Soil Partition-Partitioning Coefficient	m^3[bulk root soil]/m^3[root]	0.05	Bergqvist 2013.
Root to Root Soil Partition-Time to Reach Alpha	day	10	Iriel 2015 (time to reach 95% of equilibrium).
Stem Compartment Type - Grasses and Herbs[a]			
Transpiration stream concentration factor (TSCF)	m^3[soil pore water]/m^3[xylem fluid]	0.24	Zhao 2008.

[a]Roots and stems are not modeled for deciduous or coniferous forest in the current version of TRIM.FaTE.

Exhibit A-26. Cadmium Chemical-Specific Parameters for Plant Compartments in the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value	Reference
Leaf Compartment Type			
Transfer factor to leaf particle	1/day	0.002	Assumption (assume 1% of transfer factor from leaf particle to leaf).
Particle on Leaf Compartment Type			
Transfer factor to leaf	1/day	0.2	Assumption.
Root Compartment Type - Grasses and Herbs[a]			
Root to Root Soil Partition-Alpha of Steady State	unitless	0.95	Selected value.
Root to Root Soil Partition-Partitioning Coefficient	m^3[bulk root soil]/m^3[root]	0.23	Nriagu 1980; based on average value calculated from various agricultural plant species.
Root to Root Soil Partition-Time to Reach Alpha	day	28	Henning et al. 2001 (time to reach 95% of equilibrium).
Stem Compartment Type - Grasses and Herbs[a]			
Transpiration stream concentration factor (TSCF)	m^3[soil pore water]/m^3[xylem fluid]	0.45	Tsiros et al. 1999.

[a]Roots and stems are not modeled for deciduous or coniferous forest in the current version of TRIM.FaTE.

Exhibit A-27. Mercury Chemical-Specific Parameter Values for Plant Compartments in TRIM.FaTE Screening Scenario

Parameter Name	Units	Value			Reference
		Hg(0)	Hg(2)	MHg	
Leaf Compartment Type					
Transfer factor to leaf particle	1/day	0.002	0.002	0.002	Assumed based on 1% of transfer factor from leaf particle to leaf.
Demethylation rate	1/day	N/A	N/A	0.03	Calculated from Bache et al. 1973.
Methylation rate	1/day	N/A	0	N/A	Assumed from Gay 1975, Bache et al. 1973.
Oxidation rate	1/day	1.0E+06	N/A	N/A	Assumed to be nearly instantaneous.
Reduction rate	1/day	N/A	0	N/A	Assumption.
Particle on Leaf Compartment Type					
Transfer factor to leaf	1/day	0.2	0.2	0.2	Assumption.
Demethylation rate	1/day	N/A	N/A	0	Assumption.
Methylation rate	1/day	N/A	0	N/A	Assumption.
Oxidation rate	1/day	0	N/A	N/A	Assumption.
Reduction rate	1/day	N/A	0	N/A	Assumption.
Root Compartment Type - Grasses and Herbs[a]					
Alpha for root-root zone bulk soil	unitless	0.95	0.95	0.95	Selected value.
Root/root-zone-soil-water partition coefficient	m^3[bulk root soil] / m^3[root]	0	0.18	1.2	Hg(0) assumption; Hg(2) is geometric mean of values from Leonard et al. 1998, John 1972, and Hogg et al. 1978; MHg is based on Hogg et al. 1978.
t-alpha for root-root zone bulk soil	day	21	21	21	Assumption.
Demethylation rate	1/day	N/A	N/A	0	Assumption.
Methylation rate	1/day	N/A	0	N/A	Assumption.
Oxidation rate	1/day	0	N/A	N/A	Assumption.
Reduction rate	1/day	N/A	0	N/A	Assumption.

Parameter Name	Units	Value			Reference
		Hg(0)	Hg(2)	MHg	
Stem Compartment Type - Grasses and Herbs[a]					
Transpiration stream concentration factor (TSCF)	m^3[soil pore water] / m^3[xylem fluid]	0	0.5	0.2	Calculation from Norway spruce and Scots pine, Bishop et al. 1998.
Demethylation rate	1/day	N/A	N/A	0.03	Calculated from Bache et al. 1973.
Methylation rate	1/day	N/A	0	N/A	Assumption.
Oxidation rate	1/day	0	N/A	N/A	Assumption.
Reduction rate	1/day	N/A	0	N/A	Assumption.

[a]Roots and stems are not modeled for deciduous or coniferous forest in the current version of TRIM.FaTE.

Exhibit A-28. PAH Chemical-Specific Parameter Values for Plant Compartments in TRIM.FaTE Screening Scenario

Parameter Name	Units	Value								
		2Methyl	712DMB	Acenaph-thene	Acenaph-thylene	BaA	BaP	BbF	BghiP	BkF
Leaf Compartment Type										
Transfer factor to leaf particle	1/day	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04
Half-life	day	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5
Particle on Leaf Compartment Type										
Transfer factor to leaf	1/day	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04
Half-life	day	2.31	2.31	2.31	2.31	1.84	2.31	3.56	2.31	17.8
Root Compartment Type - Grasses and Herbs[a]										
Half-life	day	34.6	34.6	34.6	34.6	34.6	34.6	34.6	34.6	34.6
Root soil-water interaction - alpha	unitless	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95
Stem Compartment Type - Grasses and Herbs[a]										
Half-life	day	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5

Parameter Name	Units	Value								
		2Methyl	712DMB	Acenaph-thene	Acenaph-thylene	BaA	BaP	BbF	BghiP	BkF

[a]Roots and stems are not modeled for deciduous or coniferous forest in the current version of TRIM.FaTE.

Parameter Name	Units	Value					Reference
		Chr	DahA	Fluoran-thene	Fluorene	IcdP	
Leaf Compartment Type							
Transfer factor to leaf particle	1/day	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	Assumption.
Half-life	day	3.5	3.5	3.5	3.5	3.5	Approximated from data reported by Edwards 1988 and from unpublished research (McKone 1997).
Particle on Leaf Compartment Type							
Transfer factor to leaf	1/day	1.00E-04	1.00E-04	1.00E-04	1.00E-04	1.00E-04	Assumption.
Half-life	day	4.12	17.8	2.31	2.31	17.8	Calculated as 2 times the measured photolysis half-life from Mackay et al. 1992. Exceptions: value of 2.31 for BaP used for 2-methylnaphthalene, 712DMB, acenaphthene, acenaphthylene, BghiP, fluoranthene, and fluorene.
Root Compartment Type - Grasses and Herbs[a]							
Half-life	day	34.6	34.6	34.6	34.6	34.6	Approximated from data reported by Edwards 1988 (in Cooke and Dennis, eds.,1988); for bush beans in nutrient solution.
Root soil water interaction - alpha	unitless	0.95	0.95	0.95	0.95	0.95	Selected value.
Stem Compartment Type - Grasses and Herbs[a]							
Half-life	day	3.5	3.5	3.5	3.5	3.5	Approximated from data reported by Edwards 1988 (in Cooke and Dennis, eds.,1988); for bush beans in nutrient solution.

[a]Roots and stems are not modeled for deciduous or coniferous forest in the current version of TRIM.FaTE.

Exhibit A-29. Dioxin Chemical-Specific Parameter Values for Plant Compartments in the TRIM.FaTE Screening Scenario

Parameter Name	Units	Value	Reference
		All Dioxins	
Leaf Compartment Type			
Transfer factor to leaf particle	1/day	0.003	Calculated as 1% of transfer factor to leaf; highly uncertain.
Half-life	day	70	Arjmand and Sandermann 1985, as cited in Komoba et al. 1995 (in Trapp and McFarlane, eds., 1995); soybean root cell culture metabolism test data for DDE.
Particle on Leaf Compartment Type			
Transfer factor to leaf	1/day	0.3	Assumption based on U.S. EPA 2000c (an estimate for mercury) and Trapp 1995; highly uncertain.
Half-life	day	4.4	McCrady and Maggard 1993; photodegradation; particles sorbed to grass foliage in sunlight; assumed 10% direct sunlight per day.
Root Compartment Type - Grasses and Herbs[a]			
Half-life	day	70	Arjmand and Sandermann 1985, as cited in Komoba et al. 1995 (in Trapp and McFarlane, eds., 1995); soybean root cell culture metabolism test data for DDE.
Root soil-water interaction - alpha	unitless	0.95	Selected value.
Stem Compartment Type - Grasses and Herbs[a]			
Half-life	day	70	Arjmand and Sandermann 1985, as cited in Komoba et al. 1995 (in Trapp and McFarlane, eds., 1995); soybean root cell culture metabolism test data for DDE.

[a]Roots and stems are not modeled for deciduous or coniferous forest in the current version of TRIM.FaTE.

Exhibit A-30. Arsenic Chemical-Specific Parameter Values for Aquatic Species in the RTR Screening Scenario[a]

Parameter Name	Units	Value	Reference
Benthic Omnivore (BO) Compartment Type			
Biota – Sediment accumulation factor (BSAF)	kg[bulk dry sed] / kg[fish – benthic omnivore wet wt]	1.8E-04	Davis et al. 1996.
Benthic Carnivore (BC) Compartment Type			
Biota – Sediment accumulation factor (BSAF)	kg[bulk dry sed] / kg[fish – benthic carnivore wet wt]	1.8E-04	Davis et al. 1996.
Water-column Herbivore (WCH) Compartment Type			
Bioaccumulation factor (BAF)	L[water] / kg [fish – water-column herbivore wet wt]	71	U.S. EPA 2003b, Table 3.3, highest value for freshwater carp.
Water-column Omnivore (WCO) Compartment Type			
Bioaccumulation factor (BAF)	L[water] / kg [fish – water-column omnivore wet wt]	95	U.S. EPA 2003b, Tables 3.3 and 3.9, highest value for Trophic Level 3 fish, alewife.
Water-column Carnivore (WCC) Compartment Type			
Bioaccumulation factor (BAF)	L[water] / kg [fish – water-column carnivore wet wt]	46	U.S. EPA 2003b,Tables 3.4 and 3.9, highest value for Trophic Level 4, largemouth bass.

[a]Arsenic tends not to bioaccumulate from one trophic level to the next in freshwater ecosystems. Instead, concentrations in top predatory fish tend to be somewhat lower than concentrations in their prey (U.S. EPA 2003). As a result, the biokinetic model of food-web bioaccumulation simulated in TRIM.FaTE was not used for arsenic. Instead, biota-sediment accumulation factors (BSAF) and biota-water bioaccumulation factors (BAF) were sought for freshwater fish. Other investigators have reported different BAF values for arsenic in fish from specific studies than presented by EPA (e.g., Williams et al. 2006 reported that the wet-weight BAF for alewife in the Upper Mystic Lake study by Chen and Folt (2000) was 46, not 95).

Exhibit A-31. Cadmium Chemical-Specific Parameter Values for Aquatic Species in TRIM.FaTE Screening Scenario

Parameter Name	Units	Value	Reference
Zooplankton Compartment Type			
Absorption rate constant	L[water]/kg[plankton wet wt]-day	1500	Goulet 2007.
Assimilation efficiency from algae	unitless	0.5	Goulet 2007.
Elimination rate constant	1/day	0.03	Goulet 2007.
Benthic Invertebrate Compartment Type			
Sediment partitioning - alpha of equilibrium	unitless	0.95	Selected value.
Sediment partitioning - partition coefficient	kg[bulk sed/kg[invertebrate wet wt]	0.27	Assumption.
Sediment partitioning - time to reach alpha of equilibrium	day	21	Hare et al. 2001.
Benthic Omnivore Compartment Type			
Assimilation efficiency from food	unitless	0.1	Assumption based on Yan and Wang 2002.
Absorption rate constant	unitless	1.23[a]	Calculated based on body weight from regression in Hendriks and Heikens 2001.
Elimination rate constant	unitless	1.73E-02	Assumption.
Benthic Carnivore Compartment Type			
Assimilation efficiency from food	unitless	0.1	Assumption based on Yan and Wang 2002.
Absorption rate constant	unitless	0.66[a]	Calculated based on body weight from regression in Hendriks and Heikens 2001.
Elimination rate constant	unitless	1.68E-03[b]	Computed based on empirical equation.
Water-column Herbivore Compartment Type			
Assimilation efficiency from food	unitless	0.1	Assumed value based on Yan and Wang 2002.
Assimilation efficiency from plants	unitless	0.1	Assumed value based on Yan and Wang 2002.
Absorption rate constant	unitless	2.46[a]	Calculated based on body weight from regression in Hendriks and Heikens 2001.
Elimination rate constant	unitless	1.73E-02	Assumption.

Parameter Name	Units	Value	Reference
Water-column Omnivore Compartment Type			
Assimilation efficiency from food	unitless	0.1	Assumption based on Yan and Wang 2002.
Assimilation efficiency from plants	unitless	0.1	Assumption based on Yan and Wang 2002.
Absorption rate constant	unitless	1.23[a]	Calculated based on body weight from regression in Hendriks and Heikens 2001.
Elimination rate constant	unitless	1.73E-02	Assumption.
Water-column Carnivore Compartment Type			
Assimilation efficiency from food	unitless	0.1	Assumption based on Yan and Wang 2002.
Absorption rate constant	unitless	0.66[a]	Calculated based on body weight from regression in Hendriks and Heikens 2001.
Elimination rate constant	unitless	1.73E-02	Assumption

[a]Formula used: 10**(-0.30*log10(compartment.BW)-0.09).
[b]Formula used: 10**(-0.25*log10(compartment.BW)-2.7).

Exhibit A-32. Mercury Chemical-Specific Parameter Values for Aquatic Species in TRIM.FaTE Screening Scenario

Parameter Name	Units	Value			Reference
		Hg(0)	Hg(2)	MHg	
Zooplankton Compartment Type					
Assimilation Efficiency From Algae	unitless	0.015	0.2	0.5	Environment Canada 2002.
Half-life	day	1.0E+09	1.0E+09	1.0E+09	Assumption.
How Much Faster Hg Elimination Is Than For MHg	unitless	3	3	1	Assumption.
Methylation rate	1/day	N/A	0	N/A	Assumption.
Oxidation rate	1/day	1.0E+06	N/A	N/A	Assumption.
Reduction rate	1/day	N/A	0	N/A	Assumption.
Benthic Invertebrate Compartment Type					
Alpha of equilibrium for sediment partitioning	unitless	0.95	0.95	0.95	Selected value (i.e., proportion of equilibrium achieved by time "t").
Benthic invertebrate-bulk sediment partition coefficient	kg[bulk sediment]/kg[in-vertebrate wet wt]	0.0824	0.0824	5.04	Hg(0) value assumed based on Hg(2) value; Hg(2) and MHg from Saouter et al. 1991.
t-alpha for equilibrium for sediment partitioning	day	14	14	14	Experiment duration from Saouter et al. 1991.
All Fish Compartments Types[a]					
Elimination adjustment factor	unitless	3	3	1	Trudel and Rasmussen 1997.
Assimilation efficiency from food	unitless	0.06	0.06	0.5	Williams et al. 2010.
Demethylation rate	1/day	N/A	N/A	0	Assumption.
Methylation rate	1/day	N/A	0	N/A	Assumption.
Oxidation rate	1/day	1.0E+06	N/A	N/A	Assumption.
Reduction rate	1/day	N/A	0	N/A	Assumption.
Water-column Herbivore Compartment Type					
Assimilation efficiency from plankton	unitless	0.06	0.06	0.5	Williams et al. 2010.

[a]Screening scenario includes: Benthic Omnivore, Benthic Carnivore, Water-column Herbivore, Water-column Omnivore, and Water-column Carnivore.

Exhibit A-33. PAH Chemical-Specific Parameter Values for Aquatic Species in TRIM.FaTE Screening Scenario

Parameter Name	Units	Value								
		2Methyl	712DMB	Acenaph-thene	Acenaph-thylene	BaA	BaP	BbF	BghiP	BkF
Zooplankton Compartment Type										
Absorption rate constant	L[water]/kg[plank-ton wet wt]-day	790	42650	42230	42300	42650	42653	42650	42656	42652
Assimilation efficiency from algae	unitless	0.5	0.25	0.5	0.5	0.46	0.25	0.25	0.25	0.25
Elimination rate constant	1/day	170	2.03	148	123	2.07	1.389	2.12	0.33	1.48
Half-life	day	0.00779	17	0.00239	0.00239	1.28	16.5	17	17	17
Benthic Invertebrate Compartment Type										
Clearance constant	unitless	100.6	100.6	100.6	100.6	100.6	100.6	100.6	100.6	100.6
V_d (ratio of concentration in benthic invertebrates to concentration in water)	mL/g	7235	7235	7235	7235	7235	7235	7235	7235	7235
Half-life	day	0.722	17	0.722	0.722	1.284	16.5	17	17	17
All Fish Compartment Types[a]										
Gamma fish	unitless	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Assimilation efficiency from food	unitless	0.5	0.15	0.5	0.32	0.15	0.15	0.15	0.15	0.15
Half-life	day	0.2	2	0.2	0.2	0.408	2	2	2	2

Parameter Name	Units	Value					Reference
		Chr	DahA	Fluoran-thene	Fluorene	IcdP	
Zooplankton Compartment Type							
Absorption rate constant	L[water]/kg[fish wet wt]-day	42650	42656	142000	15000	42656	AQUAWEB-estimated based on Kow (Arnot et al. 2004). Exceptions: 2-methylnaphthalene, fluoranthene, and fluorene from Berrojalbiz et al. 2009.
Assimilation efficiency (AE) from algae	unitless	0.46	0.25	0.49	0.5	0.25	AQUAWEB-estimated based on Kow (Arnot et al. 2004). Exceptions: Value of 0.25, the maximum AE for copepods exposed to BaP (Wang and Wang 2006), is assumed for all higher molecular weight PAHs (i.e., 712DMB, BaA, BaP, BbF, BghiP, DahA, and IcdP).
Elimination rate constant	1/day	2.375	0.4331	8.678	81.87	0.269	AQUAWEB-estimated based on Kow (Arnot et al. 2004).
Half-life	day	0.495	17	0.00239	0.00025	17	McElroy 1990. Exceptions: 2-methyl-naphthalene, fluoranthene, and fluorene from Berrojalbiz et al. 2009; BaA, BaP, and chrysene from Moermond et al. 2007.
Benthic Invertebrate Compartment Type							
Clearance constant	unitless	100.6	100.6	100.6	100.6	100.6	Stehly et al. 1990; estimated for mayfly, 120-day-old nymphs.
V_d (ratio of concentration in benthic invertebrates to concentration in pore water)	mL/g	7235	7235	7235	7235	7235	Stehly et al. 1990; estimated for mayfly, 120-day-old nymphs.
Half-life	day	0.495	17	0.722	0.722	17	Moermond et al. 2007.

Parameter Name	Units	Value					Reference
		Chr	DahA	Fluoran-thene	Fluorene	IcdP	
All Fish Compartment Types[a]							
Gamma fish	unitless	0.2	0.2	0.2	0.2	0.2	Thomann 1989.
Assimilation efficiency from food	unitless	0.15	0.15	0.14	0.14	0.15	Lemair et al. 1992. Exceptions: Barber 2008 (for 2-methyl-naphthalene and acenaphthene); Niimi and Palazzo 1986 (for acenaphthylene, fluoranthene, and fluorene).
Half-life	day	0.533	2	0.165	0.2	2	Moermond et al. 2007. Exceptions see endnote.[b]

[a]Screening scenario includes: Benthic Omnivore, Benthic Carnivore, Water-column Herbivore, Water-column Omnivore, and Water-column Carnivore

[b]Moermond et al. 2007 calculated metabolic degradation rate constants for fluoranthene, chrysene, BaA, BeP, and BaP from experiments conducted on fish. Value of 0.2 days is assumed for the lower molecular weight PAHs based on the value for fluoranthene rounded to one significant digit. Value of 2 days is assumed for the higher molecular weight PAHs based on the value for BaP rounded to one significant digit.

Exhibit A-34. Dioxin Chemical-Specific Parameter Values for Aquatic Species in TRIM.FaTE Screening Scenario

Parameter Name	Units	Value								
		1,2,3,4,6,7,8,9-OCDD	1,2,3,4,6,7,8,9-OCDF	1,2,3,4,6,7,8-HpCDD	1,2,3,4,6,7,8-HpCDF	1,2,3,4,7,8,9-HpCDF	1,2,3,4,7,8-HxCDD	1,2,3,4,7,8-HxCDF	1,2,3,6,7,8-HxCDD	1,2,3,6,7,8-HxCDF
Zooplankton Compartment										
Absorption rate constant	L[water]/kg[fish wet wt]-day	8640	8640	8640	8640	8640	8640	8640	8640	8640
Assimilation efficiency from algae	unitless	0.08	0.05	0.21	0.09	0.2	0.31	0.31	0.31	0.31
Elimination rate constant	1/day	0.0102	0.016	0.016	0.0616	0.1829	0.0252	0.1474	0.0099	0.0194
Half-life	day	7E+06	7E+06	7E+06	7E+06	7E+06	7E+06	7E+06	7E+06	7E+06
Benthic Invertebrate Compartment										
Clearance constant	L[water cleared]/kg[benthic invertebrate wet wt]-hr	0	0	0	0	0	0	0	0	0
Sediment partitioning partition coefficient	kg/kg	0.0013	0.0017	0.0055	0.0012	0.042	0.033	0.0081	0.013	0.02
Sediment partitioning alpha of equilibrium	unitless	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95
Sediment partitioning time to reach alpha of equilibrium	days	120	42	120	42	42	120	42	120	42
V_d (ratio of concentration in benthic invertebrates to concentration in water)	L[water]/kg[benthic invertebrate wet wt]	0	0	0	0	0	0	0	0	0
Half-life	day	5776.2	5776.2	5776.2	5776.2	5776.2	5776.2	5776.2	5776.2	5776.2
All Fish Compartments[a]										
Assimilation efficiency from food	unitless	0.08	0.05	0.21[b]	0.09	0.2	0.31[c]	0.31	0.31	0.31
Fish chemical uptake rate via gill	L[water]/kg[fish wet wt]-day	11	6	56	25	50	102	200	300	200
Half-life	day	70	70	70	70	70	70	70	70	70

Parameter Name	Units	Value							
		1,2,3,7,8,9-HxCDD	1,2,3,7,8,9-HxCDF	1,2,3,7,8-PeCDD	1,2,3,7,8-PeCDF	2,3,4,6,7,8-HxCDF	2,3,4,7,8-PeCDF	2,3,7,8-TCDD	2,3,7,8-TCDF
Zooplankton Compartment									
Absorption Rate Constant	L[water]/kg[fish wet wt]-day	8640	8640	8640	8640	8640	8640	8640	8640
Assimilation Efficiency from Algae	unitless	0.31	0.31	0.42	0.42	0.31	0.42	0.41	0.51
Elimination Rate Constant	1/day	0.0099	0.0413	0.08192	0.2316	0.0192	0.4331	0.2268	1.0375
Half-life	day	7E+06	7E+06	7E+06	7E+06	7E+08	7E+08	7E+06	7E+08
Benthic Invertebrate Compartment									
Clearance Constant	L[water cleared]/kg[benthic invertebrate wet wt]-hr	0	0	0	0	0	0	0	0
Sediment Partitioning Partition Coefficient	kg/kg	0.015	0.067	0.098	0.024	0.072	0.17	0.205	0.056
Sediment Partitioning Alpha of Equilibrium	unitless	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95
Sediment Partitioning Time to Reach Alpha of Equilibrium	days	120	42	120	42	42	42	120	42
V_d (ratio of concentration in benthic invertebrates to concentration in water)	L[water]/kg[benthic invertebrate wet wt]	0	0	0	0	0	0	0	0
Half-life	day	5776.2	5776.2	5776.2	5776.2	5776.2	5776.2	5776.2	5776.2
All Fish Compartments[a]									
Assimilation efficiency from food	unitless	0.31	0.31	0.42	0.42	0.31	0.42	0.41	0.51
Chemical uptake rate via gill	L[water]/kg[fish wet wt]-day	300	200	700	300	200	400	600	400
Half-life	day	70	70	70	70	70	70	70	70

Parameter Name	Units	Reference
Zooplankton Compartment		
Absorption rate constant	L[water]/kg[fish wet wt]-day	Zhang et al. 2011, copepod k_u value.
Assimilation efficiency from algae	unitless	Morrison et al. 1999. Exceptions: Niimi and Oliver 1986 (for 1,2,3,4,6,7,8,9-OCDD, 1,2,3,4,6,7,8,9-OCDF); Berntssen et al. 2007 (for 1,2,3,4,6,7,8-HpCDD, 1,2,3,4,6,7,8-HpCDF); and value for 1,2,3,4,7,8,9-HpCDF set by linear interpolation between values for 1,2,3,4,7,8-HxCDD and 1,2,3,4,6,7,8-HpCDD / 1,2,3,4,6,7,8-HpCDF (i.e., 0.2 interpolated from 0.3 and 0.1).
Elimination rate constant	1/day	AQUQWEB-estimated based on Kow (Arnot and Gobas 2004).
Half-life	day	Morrison et al. 1999, metabolic rates for invertebrates.
Benthic Invertebrate Compartment		
Clearance constant	L[water cleared] /kg[benthic invertebrate wet wt]-hr	Assumption.
Sediment partitioning partition coefficient	kg/kg	Rubinstein et al. 1990 (used TCDD data for sandworm) and U.S. EPA 1999.
Sediment partitioning alpha of equilibrium	unitless	Rubinstein et al. 1990.
Sediment Partitioning Time to Reach Alpha of Equilibrium	days	Rubinstein et al. 1990.
V_d (ratio of concentration in benthic invertebrates to concentration in water)	L[water]/kg[benthic invertebrate wet wt]	Assumption.
Half-life	day	Rubinstein et al. 1990, TCDD value for sandworm; same value assumed for all other congeners.
All Fish Compartments[a]		
Assimilation Efficiency (AE) from Food	unitless	Morrison et al. 1999. Exceptions: Niimi and Oliver 1986 (OCDD, OCDF); value for 1,2,3,4,7,8,9-HpCDF set by linear interpolation between values for 1,2,3,4,7,8-HxCDD and 1,2,3,4,6,7,8-HpCDD / 1,2,3,4,6,7,8-HpCDF (i.e., 0.2 interpolated from 0.3 and 0.1); and two exceptions in endnotes b and c.

Parameter Name	Units	Reference
Chemical Uptake Rate Via Gill	L[water]/kg[fish wet wt]-day	Muir et al. 1985 (explicit or interpolated based on congener-specific differences in relative assimilation efficiencies from food). Exception is Opperhuizen et al. 1986 (1,2,3,7,8,9-HxCDF, 1,2,3,7,8-PeCDD, 1,2,3,7,8-PeCDF, 2,3,4,7,8-PeCDF, 2,3,7,8-TCDD, 2,3,7,8-TCDF).
Half-life	day	Berntssen et al. 2007, representative of 2,3,7,8-substituted dioxins and furans fed to large salmon (calculated half-lives ranged from 36 to 99 days with no trend apparent with degree of chlorination).

[a]Screening scenario includes: Benthic Omnivore, Benthic Carnivore, Water-column Herbivore, Water-column Omnivore, and Water-column Carnivore.

[b]AE value of 0.21 from Berntssen et al. 2007 (for fish smaller than 1 kg body weight) used for Water-column Herbivore, Water-column Omnivore, and Benthic Omnivore. AE values of 0.13 used for the two Carnivore fish compartments (2 kg body weight) based on van den Berg et al. 1984.

[c]AE value of 0.37 from van den Berg et al. 1984 (for smallest fish species) used for Water-column Herbivore. AE value of 0.31 used for remaining fish compartments based on Morrison et al. 1999.

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Attachment B. Multimedia Ingestion Risk Methodology Used for RTR Exposure and Risk Estimates

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CONTENTS, ATTACHMENT B

B.1 Introduction B-1
B.1.1 Purpose and Overview B-1
B.1.2 Organization of This Attachment B-1
B.2 Methodology Overview B-1
B.2.1 Exposure Pathways B-2
B.2.2 Receptor Groups B-3
B.3 Exposure Algorithms B-4
B.3.1 Farm Food Chain Algorithms B-5
B.3.2 Chemical Intake Calculations for Adults and Non-Infant Children B-15
B.3.3 Total Chemical Intake B-24
B.3.4 Chemical Intake Calculations for Nursing Infants B-25
B.4 Dose-Response Values B-33
B.4.1 Arsenic B-38
B.4.2 Cadmium B-38
B.4.3 Mercury B-38
B.4.4 Dioxins (2,3,7,8-TCDD) B-38
B.4.5 Polycyclic Organic Matter B-40
B.5 Risk Estimation B-42
B.5.1 Cancer Risks B-42
B.5.2 Non-cancer Hazard Quotients B-44
B.6 Assessment Data and Parameter Values B-46
B.6.1 Environmental Concentrations B-46
B.6.2 Farm-Food-Chain Parameter Values B-47
B.6.3 Adult and Non-Infant Exposure Parameter Values B-64
B.6.4 Other Exposure Factor Values B-78
B.6.5 Breast-Milk Infant Exposure Pathway Parameter Values B-81
B.7 Summary of Default Exposure Parameter Values B-88
B.7.1 Default Ingestion Rates B-89
B.7.2 Default Screening-Level Population-Specific Parameter Values B-93
B.7.3 Default Chemical-Specific Parameter Values for Screening Analysis B-93
B.7.4 Screening-Level Parameter Values for Nursing Infant Exposure B-96
B.8 References B-97

Exhibits, Attachment B

Exhibit B-1. Transfer Pathways for Farm Food Chain (FFC) Media B-2
Exhibit B-2. Chemical Transfer Pathways for Produce B-5
Exhibit B-3. Estimating Chemical Concentration in Aboveground Produce B-6
Exhibit B-4. Chemical Transfer Pathways for Animal Products B-12
Exhibit B-5. Oral Dose-response Values Used to Calculate RTR Screening Threshold Emission Rates for PB-HAP Chemicals B-34
Exhibit B-6. WHO 2005 Toxic Equivalency Factors (TEFs) for Dioxin Congeners B-39
Exhibit B-7. Oral Dose-response Values for Polycyclic Organic Matter (POM) Groups B-40
Exhibit B-8. Parameters Used to Estimate Chemical Concentrations in Farm Foods B-47
Exhibit B-9. Chemical-Specific Inputs for Produce Parameters for PB-HAP Chemicals Included in RTR Multipathway Assessments B-49
Exhibit B-10. Chemical-Specific Inputs by Plant Type for PB-HAP Chemicals in RTR Multipathway Assessments B-50
Exhibit B-11. Non-Chemical-Specific Produce Inputs B-61
Exhibit B-12. Animal Product Chemical-specific Inputs for PB-HAP Chemicals Included in RTR Multipathway Assessments B-62
Exhibit B-13. Soil and Plant Ingestion Rates for Animals B-63
Exhibit B-14. Mean and Percentile Body Weight Estimates for Adults and Children B-65
Exhibit B-15. Estimated Daily Per Capita Mean and Percentile Water Ingestion Rates for Children and Adults B-66
Exhibit B-16. Summary of Age-Group Specific Food Ingestion Rates for Farm Food Items B-67
Exhibit B-17. Fish Ingestion Rates Used in Screen B-72
Exhibit B-18. Daily Mean and Percentile *Consumer-Only* Fish Ingestion Rates for Children and Adults ($IR_{CO,y}$) B-74
Exhibit B-19. Fraction of Population Consuming Freshwater/Estuarine Fish on a Single Day ($F_{PC,y}$) B-74
Exhibit B-20. Calculated Long-term Mean and Percentile Per Capita Fish Ingestion Rates for Children and Adults ($IR_{PC,y}$) B-75
Exhibit B-21. Calculated Mean and 90th Percentile Per Capita Fish Ingestion Rates for Populations of Recreational Fishers ($IR_{PC,y}$) B-76
Exhibit B-22. Daily Mean and Percentile Soil Ingestion Rates for Children and Adults B-76
Exhibit B-23. Daily Mean and Percentile Per Capita Total Food Intake for Children and Adults B-77
Exhibit B-24. Fraction Weight Losses from Preparation of Various Foods B-79

Exhibit B-25. Scenario- and Receptor-Specific Input Parameter Values Used to Estimate Infant Exposures via Breast Milk B-82
Exhibit B-26. Average Body Weight for Infants B-82
Exhibit B-27. Time-weighted Average Body Weight for Mothers B-83
Exhibit B-28. Infant Breast Milk Intake Rates B-85
Exhibit B-29. Chemical-specific Input Parameter Values for Breast Milk Exposure Pathway B-86
Exhibit B-30. Farm Food Category Ingestion Rates for Health Protective Screening Scenario for Farming Households B-90
Exhibit B-31. Ingestion Rates for Rural Gardeners B-92
Exhibit B-32. Ingestion Rates for Urban Gardeners B-93
Exhibit B-33. Mean Body Weight Estimates for Adults and Children B-93
Exhibit B-34. Chemical-Specific Parameter Values B-95
Exhibit B-35. Chemical and Animal-Type Specific Biotransfer Factor (Ba) Values B-96

Equations, Attachment B

Equation B-1. Chemical Concentration in Aboveground Produce B-6
Equation B-2. Chemical Concentration in Aboveground Produce Due to Root Uptake B-7
Equation B-3. Chemical Concentration in Aboveground Produce Due to Deposition of Particle-phase Chemical B-7
Equation B-4. Chemical Concentration in Aboveground Produce Due to Air-to-Plant Transfer of Vapor-phase Chemical B-8
Equation B-5. Conversion of Aboveground Produce Chemical Concentration from Dry- to Wet-Weight Basis B-9
Equation B-6. Chemical Concentration in Belowground Produce: Nonionic Organic Chemicals B-9
Equation B-7. Chemical Concentration in Belowground Produce: Inorganic Chemicals B-10
Equation B-8. Conversion of Belowground Produce Chemical Concentration from Dry- to Wet-Weight Basis B-11
Equation B-9. Chemical Concentration in Beef, Pork, or Total Dairy B-12
Equation B-10. Chemical Concentration in Poultry or Eggs B-13
Equation B-11. Incidental Ingestion of Chemical in Soil by Livestock B-13
Equation B-12. Ingestion of Chemical in Feed by Livestock B-14
Equation B-13. Chemical Concentration in Livestock Feed (All Aboveground) B-14
Equation B-14. Chemical Concentration in Livestock Feed Due to Root Uptake B-15

Equation B-15. Average Daily Dose for Specified Age Group and Food Type B-15
Equation B-16. Chemical Intake from Soil Ingestion B-17
Equation B-17. Chemical Intake from Fish Ingestion B-17
Equation B-18. Consumption-weighted Chemical Concentration in Fish B-17
Equation B-19. Aquatic Biota Concentrations based on Empirical BAFs B-18
Equation B-20. Aquatic Biota Concentration based on Empirical BSAFs B-19
Equation B-21. Chemical Intake from Consumption of Exposed Fruits B-19
Equation B-22. Chemical Intake from Consumption of Protected Fruits B-19
Equation B-23. Chemical Intake from Exposed Vegetables B-20
Equation B-24. Chemical Intake from Protected Vegetables B-20
Equation B-25. Chemical Intake from Root Vegetables B-20
Equation B-26. Chemical Intake from Ingestion of Beef B-21
Equation B-27. Chemical Intake from Dairy Ingestion B-21
Equation B-28. Chemical Intake from Pork Ingestion B-22
Equation B-29. Chemical Intake from Poultry Ingestion B-22
Equation B-30. Chemical Intake from Egg Ingestion B-23
Equation B-31. Chemical Intake from Drinking Water Ingestion B-23
Equation B-32. ..... B-24
Equation B-33. ..... B-24
Equation B-34. ..... B-24
Equation B-35. ..... B-24
Equation B-36. ..... B-24
Equation B-37. ..... B-24
Equation B-38. Lifetime Average Daily Dose (LADD) B-25
Equation B-39. Average Daily Dose of Chemical to the Nursing Infant B-25
Equation B-40. Chemical Concentration in Breast Milk Fat B-27
Equation B-41. Daily Maternal Absorbed Intake B-28
Equation B-42. Biological Elimination Rate Constant for Chemicals for Non-lactating Women B-29
Equation B-43. Biological Elimination Constant for Lipophilic Chemicals for Lactating Women B-29
Equation B-44. Chemical Concentration in Aqueous Phase of Breast Milk B-30
Equation B-45. Fraction of Total Chemical in Body in the Blood Plasma Compartment B-31
Equation B-46. Biological Elimination Rate Constant for Hydrophilic Chemicals B-32
Equation B-47. Calculation of Infant Average Daily Absorbed Dose of Methyl Mercury B-33

Equation B-48. Calculation of Excess Lifetime Cancer Risk B-42
Equation B-49. ..... B-43
Equation B-50. ..... B-43
Equation B-51. ..... B-43
Equation B-52. ..... B-43
Equation B-53. ..... B-43
Equation B-54. ..... B-43
Equation B-55. ..... B-43
Equation B-56. Hazard Quotient for Chemicals with a Chronic RfD B-44
Equation B-57. Hazard Index Calculation B-45
Equation B-58. Calculation of Age-Group-Specific and Food-Specific Ingestion Rates B-69
Equation B-59. Calculation of Alternative Age-Group-Specific Fish Ingestion Rates B-73

B.1 Introduction

B.1.1 Purpose and Overview

For persistent and bioaccumulative (PB) chemicals, risks from direct inhalation of the chemical can be much less than risks from ingestion of the chemical in water, fish, and food products grown in an area of chemical deposition. For example, households that consume high quantities of self-caught fish or locally grown produce and animal products may be particularly susceptible to ingestion of chemicals transferred from air in the vicinity of an air emissions source. This attachment provides a detailed description of the multimedia ingestion risk estimation methodology developed by EPA's Office of Air Quality Planning and Standards (OAQPS) for use in Risk and Technology Review (RTR) multimedia risk assessments.

The methodology described in this attachment uses equations, assumptions, and default parameter values previously published by EPA and approaches consistent with EPA guidance on human exposure and risk estimation. In particular, the methodology complies with EPA's latest guidelines for exposure and risk assessment, including HHRAP; the Agency's 2005 *Guidelines for Carcinogen Risk Assessment* (Cancer Guidelines)*, Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (Supplemental Guidance), and *Guidance on Selecting Age Groups for Monitoring and Assessing Childhood Exposures to Environmental Contaminants* (U.S. EPA 2005b,c,d); and its Exposure Factors Handbook (U.S. EPA 2008a, 2011a).

B.1.2 Organization of This Attachment

The RTR multimedia ingestion risk estimation methodology (hereafter referred to as "the methodology") is described in Sections B.2 through B.5 of this attachment. Section B.2 identifies the exposure pathways and receptors included in the scope of the methodology. Section B.3 describes the exposure algorithms used in the methodology, including how average daily doses (ADDs) are calculated. Section B.4 presents the toxicity reference values the methodology uses to calculate risks. Section B.5 describes the risk characterization algorithms. Section B.6 describes the data requirements of the methodology, and Section B.7 identifies default parameter assumptions EPA uses for RTR screening assessments. Section B.8 provides references.

Note that EPA used the default parameter values described in Section B.7 to estimate Tier 1 screening threshold emission rates of PB-HAPs from RTR facilities. These emissions levels are assumed to pose negligible risk to subsistence communities in the vicinity of a facility emitting the PB-HAPs to air. For some assessments, it may be appropriate to use values other than the defaults to better represent a specific exposure scenario. This attachment provides tables of alternate values for some parameter values and assumptions (e.g., exposure factors) from previously published EPA sources.

B.2 Methodology Overview

The RTR multimedia ingestion risk methodology provides screening-level estimates of exposures and risks associated with fishing activities and farming and gardening activities in the vicinity of a source of PB chemical emissions to air. The methodology can assess human exposures via ingestion pathways, including drinking water consumption, incidental soil ingestion, fish ingestion, and ingestion of ten types of farm food chain (FFC) products: exposed fruits, protected fruits, exposed vegetables, protected vegetables, root vegetables, beef, total dairy, pork, poultry, and eggs. It also includes breast milk ingestion and risk estimates for

nursing infants. For fruits and vegetables, the terms “exposed” and “protected” refer to whether the edible portion of the plant is exposed to the atmosphere.

Sections B.2.1 and B.2.2 below summarize the ingestion exposure pathways and receptor age categories, respectively, included in the methodology.

B.2.1 Exposure Pathways

The methodology estimates the concentrations of chemicals in farm food chain (FFC) food categories grown in an area of airborne chemical deposition using algorithms and parameter values provided in HHRAP (U.S. EPA 2005a). FFC foods are evaluated in ten categories: exposed fruit, protected fruit, exposed vegetables, protected vegetables, root vegetables, beef, total dairy, pork, poultry, and eggs. Exhibit B-1 summarizes the pathways by which chemicals are transferred to these food media. Note that for a general Tier 1 screening-level assessment, all of the pathways can be estimated, as is the case for EPA’s *Risk and Technology Review* (RTR) calculation of screening threshold emission rates for persistent and bioaccumulative hazardous air pollutants (PB-HAPs) (U.S. EPA 2008b).

Produce types included in the FFC can accumulate a chemical directly from air and/or soil. For exposed produce, chemical mass is assumed to be transferred to plants from the air in two ways. First, particle-bound chemical can deposit directly on the plant surface. Second, the uptake of vapor-phase chemicals by plants through their foliage can occur. For both exposed and protected produce, the concentration in the plant derived from exposure to the chemical in soil is estimated using an empirical bioconcentration factor (BCF) that relates the concentration in the plant to the concentration present in the soil. For belowground root vegetables, a root concentration factor is applied. The algorithms used to estimate produce concentrations are presented in Section B.3.1.1 of this attachment.

Chemical concentrations in animal products are estimated based on the amount of chemical consumed through the diet, including incidental ingestion of soil while grazing. Diet options for farm animals include forage (plants grown on-site for grazing, such as grass), silage (wet forage grasses, fresh-cut hay, or other fresh plant material that has been stored and fermented), and feed grain products grown on the farm (e.g., corn, soybeans). All three animal feed products are assumed to accumulate chemical via root uptake from the soil. Forage and silage also can accumulate chemical via direct deposition of particle-bound chemical and vapor transfer.

Exhibit B-1. Transfer Pathways for Farm Food Chain (FFC) Media

Farm Food Media	Chemical Transfer Pathways
Exposed fruit and vegetables	• Direct deposition from air of particle-bound chemical • Air-to-plant transfer of vapor phase chemical • Root uptake from soil
Protected fruit and vegetables (including root vegetables)	• Root uptake from soil
Beef and total dairy (including milk)	• Ingestion of forage, silage, and grain[a] • Soil ingestion
Pork	• Ingestion of silage and grain[a] • Soil ingestion
Poultry and eggs	• Ingestion of grain[a] • Soil ingestion

[a]Chemical concentrations in forage, silage, and grain are estimated via intermediate calculations analogous to those used for aboveground produce.

The algorithms in the methodology are based on the assumptions that beef and dairy cattle consume all three feed products, while pigs consume only silage and grain and chickens consume only grain. The incidental ingestion of the chemical in soils during grazing or consumption of foods placed on the ground is estimated using empirical soil ingestion values. For secondary animal products (dairy products and eggs), chemical concentrations are estimated by applying a biotransfer factor to the estimated concentration in the "source" animal (cows and chickens, respectively). The algorithms used to estimate animal product concentrations are described in Section B.3.1.2 of this attachment.

B.2.2 Receptor Groups

As noted in EPA risk assessment guidelines (U.S. EPA 2005b,c,d, 2008a), exposures of children are expected to differ from exposures of adults due to differences in body weights, ingestion rates, dietary preferences, and other factors. It is important, therefore, to evaluate the contribution of exposures during childhood to total lifetime risk using appropriate exposure factor values.

EPA's HHRAP (Chapter 4, U.S. EPA 2005a) recommends assessing exposures for children and adults separately, but considers all non-infant children in one category. Specifically, HHRAP recommends eight categories of receptor: farmer, child farmer, resident, child resident, fisher, child fisher, acute receptor, and nursing infant. Over time, different EPA programs have used different child age groupings to evaluate body weights, ingestion rates, and other parameter values needed to estimate chemical exposures and risks to children.

To improve the match between age groups used to estimate values across exposure parameters, in 2005, EPA recommended a standard set of child age categories for exposure and risk assessments (U.S. EPA 2005b). EPA recommended four age groups for infants: birth to <1 month; 1 to <3 months; 3 to <6 months; and 6 to <12 months. For young children, EPA recommended an additional four age groups: 1 to <2 years; 2 to <3 years; 3 to <6 years; and 6 to <11 years. Two age groupings were recommended for teenagers and young adults: 11 to <16 years; and 16 to <21 years. These age groupings correspond to different developmental stages and reflect different food ingestion rates per unit body weight, with the highest ingestion rates occurring for the youngest, most rapidly growing, age groups.

For purposes of RTR assessments using this methodology, the selection of age categories is limited by the categories for which most of the FFC food ingestion rates have been calculated. In Chapter 13 of both its *Exposure Factors Handbook* (EFH; U.S. EPA 2011a) and its *Child-Specific Exposure Factors Handbook* (CSEFH; U.S. EPA 2008a), EPA summarized home grown/raised food ingestion rates for four children's age groups: 1 to <3 years; 3 to <6 years; 6 to <12 years; and 12 to <20 years. Intake rates were not calculated for children younger than 1 year because infants are unlikely to consume those foods. They are more likely to be nursing or to be fed formula and other commercial baby-food products.

Although the age groupings used to estimate FFC ingestion rates do not match precisely the groupings that EPA recommended in 2005 for Agency exposure assessments (U.S. EPA 2005b), they are the only age-groupings for which such data are available. The U.S. Department of Agriculture's (USDA's) 1987-1988 *Nationwide Food Consumption Survey* (USDA 1992, 1993, 1994a) remains the most recent survey of ingestion rates for home grown foods, and EPA's analysis of those data, published in its 2011 EFH, remains the most recently published major analysis of those data. Because ingestion of home grown produce and animal products are the primary exposure pathways for which the multipathway risk methodology was developed, those are the age groupings used for all child parameter values used to estimate exposure and risk.

Thus, values for each exposure parameter were estimated for adults (20 to 70 years) and five children's age groups:

- infants under 1 year (i.e., 0 to <1 year);
- children ages 1 through 2 years (i.e., 1 to <3 years);
- children ages 3 through 5 years (i.e., 3 to <6 years);
- children ages 6 through 11 years (i.e., 6 to <12 years); and
- children ages 12 through 19 years (i.e., 12 to <20 years).

See Sections B.5.1 and B.5.2 for descriptions of the risk characterization algorithms used to calculate cancer and non-cancer effects, respectively, for the above age groupings. Exposure and risks to infants under 1 year of age are estimated only for the breast-milk-ingestion pathway.

For assessment of cancer risks from early-life exposure, EPA recognizes that infants and children may be more sensitive to a carcinogenic chemical than adults, with cancers appearing earlier in life or with lower doses experienced during childhood (U.S. EPA 2005c, d). Thus, the "potency" of a carcinogen might be higher for infants and children than for adults. To date, however, data by which to evaluate the relative sensitivity of children and adults to the same daily dose of a carcinogen remain limited. Based on analyses of radioactive and other carcinogenic chemicals, EPA recommends evaluating two lifestages for children separately from adults for chemicals that cause cancer by a mutagenic mode of action (MOA): from birth to <2 years and from 2 to <16 years (U.S. EPA 2005c,d). EPA also suggests that, as data become available regarding carcinogens with a mutagenic MOA, further refinements of these age groupings may be considered.

For assessing risks from exposures to carcinogenic chemicals that act via a mutagenic MOA, the two early lifestages recommended by EPA (U.S. EPA 2005c,d) also are included in the methodology:

- children under the age of 2 years (i.e., 0 to <2 years); and
- children from 2 through 15 years (i.e., 2 to <16 years).

Different age groupings are needed for the assessment of risks from carcinogenic chemicals with a mutagenic MOA and other carcinogens with other or unknown MOAs. Currently, the only PB-HAPs included in RTR assessments that have a mutagenic mode of carcinogenesis are the carcinogenic PAHs. See Section B.5.1 for a description of the age-dependent adjustment factors (ADAFs) that are used to calculate cancer risks for chemicals with a mutagenic MOA.

B.3 Exposure Algorithms

The exposure algorithms are described below in four sections. Section B.3.1 presents the algorithms used to estimate chemical concentrations in FFC foods from chemical concentrations in soil and air. Pathway-specific algorithms used to estimate chemical intakes by adults and non-infant children are described in Section B.3.2, and total chemical intake calculations are described in Section B.3.3. Finally, the sets of algorithms used to estimate chemical intake via consumption of breast milk by nursing infants are described in Section B.3.4. As noted previously, the exposure algorithms used in in this methodology are based on those presented in HHRAP (U.S. EPA 2005b). Any differences from HHRAP are explained in this section.

B.3.1 Farm Food Chain Algorithms

The algorithms and parameters used to estimate chemical concentrations in produce and animal products are described in Sections B.3.1.1 and B.3.1.2, respectively. Discussions of the parameter value options and the values selected as defaults for RTR risk assessment are provided in Section B.6.2. The use of TRIM.FaTE to model chemical fate and transport in the environment prior to FFC calculations drives the most significant difference between the FFC algorithms included in HHRAP and the equations used for RTR. The approach in HHRAP uses estimated ambient air concentrations and deposition rates from dispersion model simulations that use unit emission rates. Chemical-specific emission rates (adjusted for vapor and particle-bound fractions) are then incorporated into some of the HHRAP FFC algorithms to calculate concentrations in FFC media. Soil concentrations are calculated using a similar approach in HHRAP. For assessment of multipathway exposures for RTR, TRIM.FaTE is used to estimate air concentrations, air-to-surface deposition rates, and soil concentrations, and these outputs are used in the FFC algorithms.

B.3.1.1 Estimating Chemical Concentrations in Produce

Produce (vegetables and fruits) can become contaminated directly by deposition of airborne chemicals to foliage and fruits or indirectly by uptake of chemicals deposited to the soil. Given these two contamination processes, produce is divided into two main groups: aboveground and belowground produce. Aboveground produce is divided into fruits and vegetables. These groups are further subdivided into “exposed” and “protected” depending on whether the edible portion of the plant is exposed to the atmosphere or is protected by a husk, hull, or other outer covering.

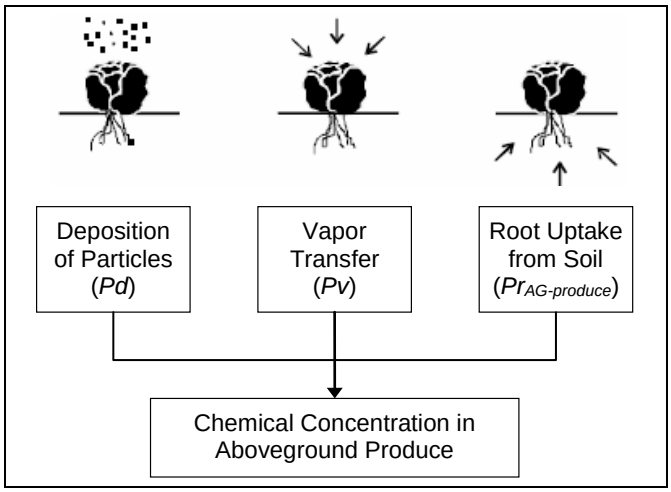
Exhibit B-2 lists the pathways by which chemicals are transferred to the FFC produce categories. Note that for a general screening-level assessment, all of the pathways can be modeled, as was done for EPA’s calculation of Tier 1 screening threshold emission rates for PB-HAPs in its RTR assessments (U.S. EPA 2008b), and as described in the Technical Support Document. The two sections below (Aboveground Produce and Belowground Produce) describe the transfer pathways and algorithms for aboveground and belowground produce, respectively.

Exhibit B-2. Chemical Transfer Pathways for Produce

Farm Food Media		Chemical Transfer Pathways
Aboveground Produce	Exposed fruits and vegetables	Direct deposition from air of particle-bound chemical Air-to-plant transfer of vapor phase chemical Root uptake from soil
	Protected fruits and vegetables	Root uptake from soil
Belowground Produce	Root vegetables	Root uptake from soil

Aboveground Produce

Exhibit B-3. Estimating Chemical Concentration in Aboveground Produce



For aboveground *exposed* produce, chemical mass is assumed to be transferred to plants from the air in three ways, as illustrated in Exhibit B-3. First, particle-bound chemical can deposit directly on the plant surface via deposition (*Pd*). The amount of chemical accumulated is estimated based on the areal fraction of chemical deposition intercepted by the plant surface, minus a loss factor that is intended to account for removal of deposited chemical by wind and rain and changes in concentration due to growth dilution. Second, for chemical present in air in the vapor phase, the concentration of chemical accumulated by the plant's foliage is estimated using an empirical air-to-plant biotransfer factor (*Pv*). Third, the chemical concentration in the plant due to root uptake from the soil ($Pr_{AG\text{-}produce}$) is estimated using an empirical bioconcentration factor ($Br_{AG\text{-}produce}$) that relates the chemical concentration in the plant to the average chemical concentration in the soil at the root-zone depth in the produce-growing area ($Cs_{root\text{-}zone\_produce}$).

The edible portions of aboveground *protected* produce are not subject to contamination via particle deposition (*Pd*) or vapor transfer *(Pv*). Therefore, root uptake of chemicals is the primary mechanism through which aboveground protected produce becomes contaminated. As shown below, the chemical concentration in the aboveground plant due to root uptake from soil ($Pr_{AG\text{-}produce\text{-}DW}$) is estimated using an empirical bioconcentration factor ($Br_{AG\text{-}produce\text{-}DW}$) that relates the chemical concentration in the plant to the average chemical concentration in the soil at the root-zone depth in the produce-growing area ($Cs_{root\text{-}zone\_produce}$). These equations all assume measurements on a dry-weight (DW) basis.

Equation B-1. Chemical Concentration in Aboveground Produce

$$C_{AG-produce-DW(i)} = Pr_{AG-produce-DW(i)} + Pd_{(i)} + Pv_{(i)} \qquad \textit{Eqn. B-1}$$

where:

$C_{AG\text{-}produce\text{-}DW(i)}$ = Concentration of chemical in edible portion of aboveground produce type i, exposed or protected, on a dry-weight (DW) basis (mg/kg produce DW)

$Pd_{(i)}$ = Chemical concentration in edible portion of aboveground produce type i due to deposition of particles (mg/kg produce DW); for protected aboveground produce, Pd equals zero (Equation B-3)

$Pr_{AG\text{-}produce\text{-}DW(i)}$ = Chemical concentration in edible portion of aboveground produce type i, exposed or protected, due to root uptake from soil at the root-zone depth of the produce growing area (mg/kg produce DW) (Equation B-2)

$Pv_{(i)}$ = Chemical concentration in edible portion of aboveground produce type i due to air-to-plant transfer (µg/g [or mg/kg] produce DW); for protected aboveground produce, Pv equals zero (Equation B-4)

Equation B-2. Chemical Concentration in Aboveground Produce Due to Root Uptake

$$Pr_{AG-produce-DW(i)} = Cs_{\text{root-zone\_produce}} \times Br_{AG-produce-DW(i)} \qquad \text{Eqn. B-2}$$

where:

$Pr_{AG\text{-}produce\text{-}DW(i)}$ = Concentration of chemical in edible portion of aboveground produce type *i*, *exposed* or *protected*, due to root uptake from soil at root-zone depth in the produce-growing area, on a dry-weight (DW) basis (mg/kg produce DW)

$Cs_{root\text{-}zone\_produce}$ = Average chemical concentration in soil at root-zone depth in produce-growing area (mg/kg soil DW)

$Br_{AG\text{-}produce\text{-}DW(i)}$ = Chemical-specific plant/soil chemical bioconcentration factor for edible portion of aboveground produce type *i*, *exposed* or *protected* (g soil DW / g produce DW)

Equation B-3. Chemical Concentration in Aboveground Produce Due to Deposition of Particle-phase Chemical

$$Pd_{(i)} = \frac{1{,}000 \times (Drdp + (Fw \times Drwp)) \times Rp_{(i)} \times (1 - e^{(-kp(i) * Tp(i))})}{Yp_{(i)} \times kp_{(i)}} \qquad \textbf{\textit{Eqn. B-3}}$$

where:

$Pd_{(i)}$ = Chemical concentration in aboveground produce type *i* on a dry-weight (DW) basis due to particle deposition (mg/kg produce DW); set equal to zero for *protected* aboveground produce

$Drdp$ = Average annual dry deposition of particle-phase chemical (g/m^2-yr)

Fw = Fraction of wet deposition that adheres to plant surfaces; 0.2 for anions, 0.6 for cations and most organics (unitless)

$Drwp$ = Average annual wet deposition of particle-phase chemical (g/m^2-yr)

$Rp_{(i)}$ = Interception fraction of the edible portion of plant type *i* (unitless)

$kp_{(i)}$ = Plant surface loss coefficient for plant type *i* (yr^{-1})

$Tp_{(i)}$ = Length of exposure to deposition in the field per harvest of the edible portion of plant type *i* (yr)

$Yp_{(i)}$ = Yield or standing crop biomass of the edible portion of plant type i (kg produce DW/m^2)

Note that Equation B-3 differs from Equation 5-14 in HHRAP, from which it is derived. In HHRAP, Equation 5-14 includes the term *Q* x (*1 – Fv*) to indicate the emissions rate, in g/sec, of chemical from the source and the proportion of the chemical that remains in, or partitions to, the particle-phase in the air. Also in HHRAP, the dry and wet particle phase deposition rates, *Dydp* and *Dywp,* respectively, are normalized to the emission rate and are expressed in units of sec/m^2-yr.

The mulitpathway ingestion risk methodology uses both the dry and wet particle-phase deposition rates, *Drdp* and *Drwp*, respectively, in units of g/m^2-yr for a specific location relative to an emissions source. Those deposition rates might be values measured near that location or estimated using a fate and transport model, such as TRIM.FaTE, in conjunction with local

meteorological information and emissions rate data. The chemical emissions term used in HHRAP, *Q*, therefore, is not used in Equation B-3. In addition, *Drdp* and *Drwp*, the average annual dry- and wet-particle-phase deposition rates, respectively, are in units of g/m^2-yr. Users of TRIM.FaTE should note that the dry- and wet-particle-deposition rates output from TRIM.FaTE are in units of g/m^2-day; therefore, it is necessary to adjust the TRIM.FaTE output values to units of g/m^2-yr (i.e., multiply by 365 days/yr) before inputting values for *Drdp* and *Drwp* into Equation B-3.

Equation B-4. Chemical Concentration in Aboveground Produce Due to Air-to-Plant Transfer of Vapor-phase Chemical

$$Pv_{(i)} = \frac{Ca \times Fv \times Bv_{AG(i)} \times VG_{AG(i)}}{\rho_a} \qquad \textit{Eqn. B-4}$$

where:

$Pv_{(i)}$	=	Concentration of chemical in edible portion of aboveground produce type *i* from air-to-plant transfer of vapor-phase chemical on a dry-weight (DW) basis (µg/g produce DW); set equal to zero for *protected* aboveground produce
Ca	=	Average annual *total* chemical concentration in air (g/m^3)
Fv	=	Fraction of airborne chemical in vapor phase (unitless)
$Bv_{AG(i)}$	=	Air-to-plant biotransfer factor for aboveground produce type *i* for vapor-phase chemical in air ([mg/g produce DW] / [mg/g air], i.e., g air/ g produce DW)
$VG_{AG(i)}$	=	Empirical correction factor for aboveground *exposed* produce type *i* to address possible overestimate of the diffusive transfer of chemical from the outside to the inside of bulky produce, such as fruit (unitless)
ρ_a	=	Density of air (g/m^3)

Note that Equation B-4 differs from Equation 5-18 in HHRAP, from which it is derived. In HHRAP, Equation 5-18 includes the term *Q* x *Fv* to indicate the emissions rate, in g/sec, of chemical from the source and the fraction of the chemical in vapor phase in the air. HHRAP also includes the parameter *Cyv*, or the unitized yearly average air concentration of vapor-phase chemical in units of µg-sec/g-m^3. However, the multimedia ingestion risk methodology uses the average annual total air concentration of the chemical, *Ca*, for a specific location relative to the source in units of g/m^3.The air concentration might be a value measured near that location or a value estimated by a fate and transport model such as TRIM.FaTE. Users of TRIM.FaTE should note that the average annual concentration of the total chemical in air (i.e., total of both vapor and particulate phases), *Ca*, output from TRIM.FaTE is in units µg/m^3; therefore, the value must be adjusted to units of g/m^3 (i.e., divide by 1,000 µg/g).

With the publication of HHRAP, EPA provided a companion database that includes default chemical-specific values for *Fv*, as well as certain other parameters.[23]

The calculations of chemical concentration in aboveground produce, ($C_{AG\text{-}produce\text{-}DW}$), shown in Equation B-1 above, are on a dry-weight (DW) basis. The FFC food ingestion rates, on the other

[23] The HHRAP Companion Database is available at https://archive.epa.gov/epawaste/hazard/tsd/td/web/html/riskvol.html

hand, are on a fresh- or wet-weight (WW) basis. Therefore the concentration in aboveground produce must be calculated on a wet-weight basis, $C_{AG\text{-}produce\text{-}WW}$, using Equation B-5 and the moisture adjustment factor (*MAF*) of the FFC food category.

Equation B-5. Conversion of Aboveground Produce Chemical Concentration from Dry- to Wet-Weight Basis

$$C_{AG-produce-WW(i)} = C_{AG-produce-DW(i)} \times \left(\frac{(100 - MAF_{(i)})}{100} \right) \qquad \textit{Eqn. B-5}$$

where:

$C_{AG\text{-}produce\text{-}WW(i)}$ = Chemical concentration in edible portion of aboveground produce type i on a wet-weight (WW) basis (mg/kg produce WW)

$C_{AG\text{-}produce\text{-}DW(i)}$ = Chemical concentration in edible portion of aboveground produce type i on a dry-weight (DW) basis (mg/kg produce DW)

$MAF_{(i)}$ = Moisture adjustment factor for aboveground produce type i to convert the chemical concentration estimated for dry-weight produce to the corresponding chemical concentration for full-weight fresh produce (percent water)

Belowground Produce

The equations by which chemical concentrations are estimated in belowground produce are different for nonionic organic chemicals than for inorganic chemicals and ionic organic chemicals.

(a) Nonionic Organic Chemicals

For belowground produce, the nonionic organic chemical concentration in the tuber or root vegetable is derived from exposure to the chemical in soil and is estimated using an empirical root concentration factor (*RCF*) and the average chemical concentration in the soil at the root-zone depth in the produce-growing area ($Cs_{root\text{-}zone\_produce}$), as shown in Equation B-6. The RCF relates the chemical concentration in the plant on a wet-weight basis to the average chemical concentration in the root-zone soil ($Cs_{root\text{-}zone\_produce}$) on a dry-weight basis. Belowground produce (i.e., tubers or root vegetables) are protected from the deposition and vapor transfer by being covered by soil. Therefore, root uptake of chemicals is the primary mechanism through which belowground produce becomes contaminated.

Equation B-6. Chemical Concentration in Belowground Produce: Nonionic Organic Chemicals

$$C_{BG\text{-}produce\text{-}WW} = \frac{Cs_{root-zone\_produce} \times RCF \times VG_{rootveg}}{Kds \times UCF} \qquad \textit{Eqn. B-6}$$

where:

$C_{BG\text{-}produce\text{-}WW}$	=	Concentration of chemical in belowground (BG) produce (i.e., tuber or root vegetable) on a wet-weight (WW) basis (mg chemical/kg produce WW)*
$Cs_{root\text{-}zone\_produce}$	=	Average chemical concentration in soil at root-zone depth in produce-growing area, on a dry-weight (DW) basis (mg chemical/kg soil DW)
RCF	=	Chemical-specific root concentration factor for tubers and root produce (L soil pore water/kg root WW)*
$VG_{rootveg}$	=	Empirical correction factor for belowground produce (i.e., tuber or root vegetable) to account for possible overestimate of the diffusive transfer of chemicals from the outside to the inside of bulky tubers or roots (based on carrots and potatoes) (unitless)*
Kds	=	Chemical-specific soil/water partition coefficient (L soil pore water/kg soil DW)
UCF	=	Units conversion factor of 1 kg/L

*Note that there is only one type of BG produce; hence there are no plant-type-specific subscripts.

The RCF, as developed by Briggs *et al*. (1982), is the ratio of the chemical concentration in the edible root on a wet-weight basis to its concentration in the soil pore water. RCFs are based on experiments with growth solutions (hydroponic) instead of soils; therefore, it is necessary to divide the soil concentration by the chemical-specific soil/water partition coefficient (*Kds*). There is no conversion of chemical concentrations in belowground produce from DW to WW because the values are already on a WW basis.

For nonionic organic chemicals, it is possible to predict RCF values and Kds values (for a specified soil organic carbon content) from an estimate of the chemical's Kow from empirically derived regression models. Those models are shown in HHRAP Appendix A-2, Equations A-2-14 and A-2-15 (RCF) and in Equations A-29 and A-2-10 (Kds). The RCF and Kds values so calculated for many of the chemicals in HHRAP are included in the HHRAP Companion Database (including the values for PAHs and dioxins).

(b) Inorganic and Ionic Organic Chemicals

For inorganic chemicals and ionized organic chemicals, it is not possible to predict RCF or Kds values from Kow. For inorganic chemicals, chemical-specific empirical values for the root/soil bioconcentration factor must be used. The root/soil bioconcentration factor, now specified as $Br_{BG\text{-}produce\text{-}DW}$, must be obtained from the literature for each inorganic chemical on a DW basis. For inorganic chemicals, therefore, Equation B-7 is used instead of Equation B-6.

Equation B-7. Chemical Concentration in Belowground Produce: Inorganic Chemicals

$$C_{BG\text{-}produce\text{-}DW} = \frac{Cs_{root-zone\_produce} \times Br_{BG-produce-DW} \times VG_{rootveg}}{1} \qquad \textit{Eqn. B-7}$$

where:

$C_{BG\text{-}produce\text{-}DW}$	=	Concentration of chemical in edible portion of aboveground produce, due to root uptake from soil at root-zone depth in the produce-growing area, on a dry-weight (DW) basis (mg/kg produce DW)
$Cs_{root\text{-}zone\_produce}$	=	Average chemical concentration in soil at root-zone depth in produce-growing area (mg/kg soil DW)

$Br_{BG\text{-}produce\text{-}DW}$ = Chemical-specific root/soil chemical bioconcentration factor for edible portion of belowground produce (g soil DW / g produce DW)

$VG_{rootveg}$ = Empirical correction factor for belowground produce (as in Equation B-6) (unitless)

As for the aboveground produce, the DW estimate of concentration of chemical in the root vegetables must be transformed to a WW estimate, as shown in Equation B-8.

Equation B-8. Conversion of Belowground Produce Chemical Concentration from Dry- to Wet-Weight Basis

$$C_{BG-produce-WW} = C_{BG-produce-DW} \times \left(\frac{(100 - MAF_{BG})}{100} \right) \qquad \textit{Eqn. B-8}$$

where:

$C_{BG\text{-}produce\text{-}WW}$ = Chemical concentration in edible portion of belowground produce on a weight-weight (WW) basis (mg/kg produce WW)

$C_{BG\text{-}produce\text{-}DW}$ = Concentration of chemical in edible portion of aboveground produce, due to root uptake from soil at root-zone depth in the produce-growing area, on a dry-weight (DW) basis (mg/kg produce DW)

$MAF_{(BG)}$ = Moisture adjustment factor (as in Equation B-5, but single value for below ground produce) (percent water)

B.3.1.2 Estimating Chemical Concentrations in Animal Products

Chemical concentrations in animal products are estimated based on the amount of chemical consumed by each animal group *m* through each plant feed type *i* ($Plant_{Ch\text{-}Intake(i,m)}$) and incidental ingestion of soil for ground-foraging animals ($Soil_{Ch\text{-}Intake(m)}$). Exhibit B-4 summarizes the pathways by which chemicals are transferred to these home- or farm-raised animal food products. Note that for a general screening-level assessment, all of the pathways can be modeled, as is done for EPA's RTR calculation of screening threshold emission rates for PB-HAPs (U.S. EPA 2008b).

The feed options for farm animals in the mulitpathway ingestion risk methodology include forage (plants grown on-site for animal grazing, such as grass), silage (wet forage grasses, fresh-cut hay, or other fresh plant material that has been stored and fermented), and grain products grown on the farm. As seen in Exhibit B-4, the algorithms for chemical intake with plant feeds ($Plant_{Ch\text{-}Intake(i,m)}$) are based on the assumptions that beef and dairy cattle consume all three plant feed products, while pigs consume only silage and grain, and chickens consume only grain.

Exhibit B-4. Chemical Transfer Pathways for Animal Products

Farm Food Media		Chemical Transfer Pathways
Animal Products	Beef and total dairy (including milk)	• Ingestion of forage, silage, and grain[a] • Incidental soil ingestion
	Pork	• Ingestion of silage and grain[a] • Incidental soil ingestion
	Poultry and eggs	• Ingestion of grain[a] • Incidental soil ingestion

[a]Chemical concentrations in plant feed (i.e., forage, silage, and grain) are estimated via intermediate calculations (see Equation B-13, Equation B-14, Equation B-3, and Equation B-4).

Forage and silage are exposed to the air and can accumulate chemicals via direct deposition of particle-bound chemical and transfer of vapor-phase chemical, while all animal feed grains are assumed to be protected from the air by a husk or pod (e.g., corn, soybeans). All three animal feed products are assumed to accumulate chemical via root uptake.

Chemical concentrations are estimated for animal feeds using algorithms analogous to those for aboveground farm produce described above. The multimedia ingestion risk methodology uses Equation B-9 to calculate the concentration of chemical in beef, pork, or total dairy and Equation B-10 to calculate the concentration of chemical in poultry or eggs. The chemical concentration in mammalian farm animals (i.e., beef and pigs) is adjusted using a metabolism factor (*MF*) that accounts for endogenous degradation of the chemical (see Equation B-9). *MF* is set to 1.0 for chemicals that are not metabolized and for chemicals for which the metabolic degradation rate is unknown. Although other vertebrates, including birds, are likely to have similar metabolic pathways for most chemicals, the health protective assumption is that birds do not metabolize any chemicals; therefore, the *MF* is omitted from Equation B-10 for poultry and eggs.

Equation B-9. Chemical Concentration in Beef, Pork, or Total Dairy

$$C_{mammal(m)} = Ba_{(m)} \times MF \times \left(Soil_{Ch-Intake(m)} + \sum_{i=1}^{n} Plant_{Ch-Intake(i,m)} \right) \qquad \textit{Eqn. B-9}$$

where:

$C_{mammal(m)}$ = Concentration of chemical in mammalian animal product *m*, where *m* = beef, pork, or total dairy (mg chemical/kg animal product WW)

$Ba_{(m)}$ = Chemical-specific biotransfer factor for chemical in diet to chemical in animal food product *m*, where *m* = beef, pork, or total dairy ([mg chemical/kg animal product WW] / [mg chemical intake/day] or day/kg WW)

MF = Chemical-specific mammalian metabolism factor that accounts for endogenous degradation of the chemical (unitless)

$Soil_{Ch-Intake(m)}$ = Incidental ingestion of chemical in surface soils by livestock type *m* during grazing or consumption of foods placed on the ground (mg/day); see Equation B-11 below

$Plant_{Ch-Intake(i,m)}$ = For livestock (animal product) type *m*, ingestion of chemical from plant feed type *i* (mg chemical/kg livestock WW); see Equation B-12 below

(If *m* =beef or total dairy, then *n* = 3 and *i* = forage, silage, and grain; *m* = pork, then *n* = 2 and *i* = silage and grain; *m* = poultry, then *n* = 1 and *I* = grain.)

Equation B-10. Chemical Concentration in Poultry or Eggs

$$C_{poultry(m)} = Ba_{(m)} \times \left(Soil_{Ch-Intake(m)} + Plant_{Ch-Intake(i,m)}\right) \quad \textit{Eqn. B-10}$$

where:

$C_{poultry(m)}$	=	Concentration of chemical in food product *m*, where *m* = poultry or eggs (mg chemical/kg animal product WW)
$Ba_{(m)}$	=	Chemical-specific biotransfer factor for food product *m*, where *m* = poultry or eggs (day/kg animal product WW)
$Soil_{Ch-Intake(m)}$	=	Incidental ingestion of chemical in surface soils by consumption of food on the ground (mg chemical/day) where *m* = poultry; see Equation B-11
$Plant_{Ch-Intake(i,m)}$	=	For poultry (and eggs), animal *m*, ingestion of the chemical in plant feed type *i* (mg chemical/day), which for poultry is limited to grain; see Equation B-12

The incidental ingestion of the chemical in soils by livestock during grazing or consumption of feed placed on the ground ($Soil_{Ch-Intake(m)}$) is estimated using empirical soil ingestion rates (*Qs*) and a soil bioavailability factor for livestock (*Bs*), as shown in Equation B-11. The default value for *Bs* for all chemicals is 1.0 (i.e., the chemical in soil is assumed to be 100 percent bioavailable to the animal). This assumption may be reasonably accurate for the soil surface to which airborne chemical is deposited. The surface soil concentration in Equation B-11, $Cs_{S-livestock}$, is for areas where livestock forage, which may be distinct from the surface soil concentration in areas where produce are grown and where humans might incidentally ingest soils (see Section B.6.1 of this attachment).

Equation B-11. Incidental Ingestion of Chemical in Soil by Livestock

$$Soil_{Ch-Intake(m)} = Qs \times Cs_{s-livestock} \times Bs \quad \textit{Eqn. B-11}$$

where:

$Soil_{Ch-Intake(m)}$	=	Incidental ingestion of the chemical in surface soils by livestock type *m* during grazing or consumption of foods placed on the ground (mg chemical/day)
$Qs_{(m)}$	=	Quantity of soil eaten by animal type *m* each day (kg soil DW/day)
$Cs_{s-livestock}$	=	Chemical concentration in surface soil in contaminated area where livestock feed (mg chemical/kg soil DW)
Bs	=	Soil bioavailability factor for livestock (unitless) (assumed to be the same for birds and mammals)

Animal ingestion of the chemical in feed is calculated for each type of livestock based on their assumed diets. For *m* = beef and dairy cattle, chemical intake is estimated for all three feed types: *i* = forage, silage, and grain. For pork, chemical intake is estimated only for silage and grain. The chemical intake for poultry is based on grain consumption only. The intake of chemical with each feed type, *i*, $Plant_{Ch-Intake(i,m)}$, is calculated separately according to Equation

B-12. Note that the animal feed ingestion rates are on a dry-weight (DW) basis; hence, no DW to wet weight (WW) conversion is needed.

Equation B-12. Ingestion of Chemical in Feed by Livestock

$$Plant_{Ch-Intake(i,m)} = F_{(i,m)} \times Qp_{(i,m)} \times C_{feed(i)} \qquad \textbf{Eqn. B-12}$$

where:

$Plant_{Ch-Intake(i,m)}$ = Ingestion of chemical in plant feed type *i* (mg chemical/day), where *i* = forage, silage, or grain, for livestock type *m*

$F_{(i,m)}$ = Fraction of plant feed type *i* obtained from contaminated area used to grow animal feed, where *I* = forage, silage, or grain (unitless) for livestock type *m*

$Qp_{(i,m)}$ = Quantity of plant feed type *i* consumed per animal per day (kg plant feed DW/day), where *i* = forage, silage, or grain, for livestock type *m*

$C_{feed(i)}$ = Concentration of chemical in ingested plant feed type *i* (mg chemical/kg plant feed DW), where *i* = forage, silage, or grain

The concentrations of chemical in the three different types of plant feeds for livestock are calculated according to Equation B-13. The equation is the same as that for aboveground produce in Equation B-1, with the exception that the concentrations are for plants used as animal feeds (not produce consumed by humans) and all types of plant feed (i.e., forage, silage, and grain) are aboveground.

Equation B-13. Chemical Concentration in Livestock Feed (All Aboveground)

$$C_{feed(i)} = Pr_{feed(i)} + Pd_{(i)} + Pv_{(i)} \qquad \textbf{Eqn. B-13}$$

where:

$C_{feed(i)}$ = Concentration of chemical in plant feed type *i* on a dry-weight (DW) basis (mg chemical/kg plant feed DW), where *i* = forage, silage, or grain

$Pr_{feed(i)}$ = Concentration of chemical in plant feed type *i* due to root uptake from soil (mg/kg DW), where *i* = forage, silage, or grain; see Equation B-14 below

$Pd_{(i)}$ = Concentration of chemical in plant feed type *i* due to wet and dry deposition of particle-phase chemical (mg/kg DW), where *i* = forage, silage, or grain; when *i* = grain, the *Pd* term equals zero

$Pv_{(i)}$ = Concentration of chemical in plant feed type *i* due to air-to-plant transfer of vapor-phase chemical (µg/g [or mg/kg] DW) where *i* = forage, silage, or grain; when *i* = grain, the *Pd* term equals zero

The chemical concentration in animal feed due to root uptake from the soil is calculated with Equation B-14. The equation is the same as Equation B-2, except that a *Br* value appropriate to grasses is used and different soil concentrations could be used for the area used to grow animal feed and the area used to grow produce for human consumption (see Section B.6.1 of this attachment). Note that for feed type *i* = grains, the *Pd* and *Pv* terms do not apply (are set to

zero), because the feed products (i.e., corn kernels, soy beans) are protected from the air (i.e., by husks, pods).

Equation B-14. Chemical Concentration in Livestock Feed Due to Root Uptake

$$Pr_{feed(i)} = Cs_{root-zone\_feed(i)} \times Br_{feed(i)} \qquad \textit{Eqn. B-14}$$

where:

$Pr_{feed(i)}$ = Concentration of chemical in plant feed type *i* due to root uptake from soil on a dry-weight (DW) basis (mg chemical/kg plant feed DW), where *i* = forage, silage, or grain

$Cs_{root\text{-}zone\_feed(i)}$ = Average chemical concentration in soil at root-zone depth in area used to grow plant feed type *i* (mg chemical/kg soil DW), where *i* = forage, silage, or grain

$Br_{feed(i)}$ = Chemical-specific plant-soil bioconcentration factor for plant feed type *i* (kg soil DW/kg plant feed DW), where *i* = forage, silage, or grain

The algorithms used to calculate $Pd_{(i)}$ and $Pv_{(i)}$ when plant feed type *i* = forage and silage are identical to those used to calculate $Pd_{(i)}$ and $Pv_{(i)}$ for aboveground exposed produce (i.e., Equation B-3 and Equation B-4, respectively).

There are no conversions of DW feed to WW feed, because all feed ingestion rates for livestock are based on DW feed.

B.3.2 Chemical Intake Calculations for Adults and Non-Infant Children

The multimedia ingestion risk methodology calculates human chemical intake rates from the ingestion of home grown foods as average daily doses (ADDs) normalized to body weight for each age group, chemical, and food type separately. ADDs, calculated using Equation B-15, are expressed in milligrams of chemical per kilogram of receptor body weight per day (mg/kg-day).

Equation B-15. Average Daily Dose for Specified Age Group and Food Type

$$ADD_{(y,i)} = \left(\frac{C_{(i)} \times IR_{(y,i)} \times FC_{(i)} \times ED_{(y)}}{BW_{(y)} \times AT_{(y)}}\right)\left(\frac{EF_{(y)}}{365 \text{ days}}\right) \qquad \textit{Eqn. B-15}$$

where:

$ADD_{(y,i)}$ = Average daily dose for age group *y* from food type or ingestion medium *i* (mg chemical/kg body weight-day)

$C_{(i)}$ = Concentration of chemical in food type *i* harvested from the contaminated area (mg chemical/kg food or mg food/L water)

$IR_{(y,i)}$ = Ingestion rate for age group *y* of food type *i* (kg/day or L/day)

$FC_{(i)}$ = Fraction of food type *i* that was harvested from contaminated area (unitless)

$ED_{(y)}$ = Exposure duration for age group *y* (years)

$BW_{(y)}$ = Body weight for age group *y* (kg)

$AT_{(y)}$ = Averaging time for calculation of daily dose (years) for age group *y*, set equal to ED

$EF_{(y)}$ = Annual exposure frequency for age group *y* (days)

Equation B-15 takes into account the chemical concentration in each food type *i* (or in water), the quantity of food brought into the home for consumption, the loss of some of the mass of the foods due to preparation and cooking, how much of the food is consumed per year, the amount of the food obtained from contaminated areas, and the consumer's body weight (U.S. EPA 2011a, 2003a). ADDs are calculated separately for each chemical, home grown food type, and consumer age group.

ADD values, expressed as intakes, not absorbed doses, are appropriate for comparison with RfDs and for use with cancer slope factors (CSFs) to estimate risk, as discussed in Section B.5 of this attachment. An exception is for the breast-milk exposure pathway, where calculating the dose available to and absorbed by the nursing infant is related to the dose absorbed by the mother as discussed in Section B.3.4 of this attachment.

For screening-level assessments, all components of Equation B-15 are assumed to remain constant for consumers in a given age group over time (e.g., seasonal and annual variations in diet are not explicitly taken into account). To calculate an $ADD_{(y,i)}$ from the contaminated area for food group *i* over an entire lifetime of exposure, age-group-specific ingestion rates and body weights are used for the age groups described in Section B.2.2 of this attachment. The averaging time used to calculate the daily dose for an age group (AT_y) is equal to the exposure duration for that group (ED_y); therefore these variables drop out of Equation B-15.

For each chemical included in a screening scenario, total average daily exposure for age group *y* ($ADD_{(y)}$) is estimated as the sum of chemical intake from all ingestion pathways combined: Note that the last exposure pathway is limited to infants.

- Incidental soil ingestion;
- Ingestion of fish;
- Ingestion of homegrown fruits (exposed and protected);
- Ingestion of homegrown vegetables (exposed, protected, and root);
- Ingestion of animal products from home-raised animals:
  - Milk and other dairy products from cows,
  - Beef products,
  - Pork products, and
  - Poultry and eggs;
- Ingestion of drinking water from specified source; and
- Ingestion of breast milk by infants.

The algorithms for the first six exposure pathways listed above are described in Sections B.3.2.1 through B.3.2.6 of this attachment. The algorithms for the breast milk ingestion pathway are described in Section B.3.4.

B.3.2.1 Chemical Intake from Soil Ingestion

Equation B-16 shows the equation used to estimate chemical intake through incidental ingestion of soil.

Equation B-16. Chemical Intake from Soil Ingestion

$$ADD_{Soil(y)} = \left(\frac{C_{Soil} \times IR_{Soil(y)} \times FC_{Soil} \times 0.001 \ \frac{mg}{\mu g}}{BW_{(y)}} \right) \left(\frac{EF}{365 \text{ days}} \right) \qquad \textit{Eqn. B-16}$$

where:

$ADD_{Soil(y)}$ = Average daily chemical intake from incidental ingestion of soil or ingestion by child in age group *y* (mg chemical/kg body weight-day)

C_{Soil} = Concentration of chemical in soil from contaminated area on a dry-weight (DW) basis (µg/g soil DW)

$IR_{Soil(y)}$ = Soil ingestion rate for age group *y* (g DW/day)

FC_{Soil} = Fraction of soil ingested that is from contaminated area (unitless)

$BW_{(y)}$ = Body weight for age group *y* (kg)

EF = Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario (≤ 365 days)

B.3.2.2 Chemical Intake from Fish Ingestion

The multimedia ingestion risk methodology includes ingestion of locally caught fish as a possible exposure pathway (Equation B-17). Two types of fish are included in the exposure algorithm: trophic level 3.5 (abbreviated as TL3) fish, equivalent to benthic carnivores such as catfish and trophic level 4 (TL4) fish in the water column, equivalent to game fish such as lake trout and walleye. The chemical concentration in fish in Equation B-17 is estimated as the consumption-weighted chemical concentration using Equation B-18.

Equation B-17. Chemical Intake from Fish Ingestion

$$ADD_{Fish(y)} = \left(1 - L1_{Fish}\right) \times \left(1 - L2_{Fish}\right) \times \frac{\left(C_{Fish} \times IR_{Fish(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{Fish} \right)}{BW_{(y)}} \times \left(\frac{EF}{365 \text{ days}} \right) \qquad \textit{Eqn. B-17}$$

Equation B-18. Consumption-weighted Chemical Concentration in Fish

$$C_{Fish} = (C_{FishTL3} \times F_{TL3}) + (C_{FishTL4} \times F_{TL4}) \qquad \textit{Eqn. B-18}$$

where:

$ADD_{Fish(y)}$ = Average daily chemical intake from ingestion of local fish for age group *y* (mg/kg-day)

$L1_{Fish}$* = Weight of fish brought into home that is discarded during preparation (e.g., head, bones, liver, other viscera, belly fat, skin with fat) (unitless)

$L2_{Fish}$* = Loss of weight during cooking, such as evaporation and loss of fluids into pan (unitless)

$C_{FishTL3}$ = Chemical concentration in whole fish for trophic level 3.5 (TL3) fish on a wet-weight (WW) basis (mg/kg WW)

$C_{FishTL4}$ = Chemical concentration in whole fish for trophic level 4 (TL4) fish on a wet-weight (WW) basis (mg/kg WW)

F_{TL3} = Fraction of fish intake that is from TL3 (unitless)

F_{TL4} = Fraction of fish intake that is from TL4 (unitless)

C_{Fish} = Consumption-weighted mean chemical concentration in total fish (i.e., as specified by Equation B-18) (mg/kg WW)

FC_{Fish} = Fraction of local fish consumed derived from contaminated area (unitless)

$BW_{(y)}$ = Body weight for age *y* (kg)

$IR_{Fish(y)}$* = Local fish ingestion rate for age *y* (g WW/day)

EF = Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario ($\leq$ 365 days)

*Parameter values must be internally consistent. In contrast to the ingestion rates for homegrown food products, which are based on the products as brought into the home from the field (see Section B.6.3.3), the fish ingestion rates are on an "as consumed" basis (i.e., after preparation and cooking losses), and L1 and L2 therefore are set equal to zero. If an assessment will include local fish ingestion rates on an "as harvested" basis, L1 and L2 values also should be included as specified in Section B.6.4.3.

When whole fish are prepared for cooking, it is usual for the viscera, head, and fins to be removed, particularly for larger fish. Many persons also remove (or do not eat) the skin, bones, and belly fat. EPA has, therefore, estimated the proportion of the weight of whole fish that tends to be lost during preparation and cooking across a variety of fish species (EFH; U.S. EPA 2011a) and included those losses in its HHRAP algorithms for chemical intake from fish ($L1_{Fish}$ and $L2_{Fish}$ in Equation B-17).

For arsenic, TRIM-FaTE-calculated water and sediment concentrations are multiplied by empirical bioaccumulation factors (BAFs) and biota-sediment accumulation factors (BSAFs) to estimate fish tissue concentrations in the WCC and BC, respectively. (Fish tissue concentrations for other PB-HAPs are calculated in TRIM.FaTE's biokinetic food web model.) The BAF/BSAF approach to aquatic bioaccumulation could easily be extended to other inorganic chemicals included in an assessment. Equation B-19 presents the algorithm for estimating aquatic biota concentrations based on BAFs. Equation B-20 presents the algorithm for estimating aquatic biota concentrations based on BSAFs and is appropriate for sediment-dwelling fish and organisms.

Equation B-19. Aquatic Biota Concentrations based on Empirical BAFs

$$C_{Fish} = BAF \times C_{SurfaceWater} \qquad \textbf{Eqn. B-19}$$

where:

C_{Fish} = Chemical concentration in whole fish on a wet-weight (WW) basis (mg/kg WW)

$C_{Surface\ Water}$ = Chemical concentration in surface water phase (mg/L WW)

BAF = Bioaccumulation Factor (L/kg WW fish)

Equation B-20. Aquatic Biota Concentration based on Empirical BSAFs

$$C_{Fish}=BSAF \times C_{Sediment} \qquad \textit{Eqn. B-20}$$

where:

C_{Fish} = Chemical concentration in whole fish on a wet-weight (WW) basis (mg/kg WW)

$C_{Sediment}$ = Chemical concentration in sediment (mg/kg DW)

$BSAF$ = Biota Sediment Factor (kg DW sediment / kg WW fish)

B.3.2.3 Chemical Intake from Fruit Ingestion

Average daily doses of a chemical from homegrown exposed fruits are calculated separately for exposed and protected fruits (Equation B-21 and Equation B-22, respectively).

Equation B-21. Chemical Intake from Consumption of Exposed Fruits

$$ADD_{ExpFruit(y)} = \left(1 - L1_{ExpFruit}\right) \times \left(1 - L2_{ExpFruit}\right) \times \left(C_{ExpFruit} \times IR_{ExpFruit(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{ExpFruit}\right) \times \left(\frac{EF}{365 \text{ days}}\right)$$

Eqn. B-21

Equation B-22. Chemical Intake from Consumption of Protected Fruits

$$ADD_{ProFruit(y)} = \left(1 - L1_{ProFruit}\right) \times \left(C_{ProFruit} \times IR_{ProFruit(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{ProFruit}\right) \times \left(\frac{EF}{365 \text{ days}}\right)$$

Eqn. B-22

where:

$ADD_{ExpFruit(y)}$ / $ADD_{ProFruit(y)}$ = Average daily chemical intake from ingestion of exposed fruit or protected fruit (depending on subscript) (mg chemical/kg body weight-day)

$L1_{ExpFruit}$ = Mean reduction in fruit weight resulting from removal of skin or peel, core or pit, stems or caps, seeds and defects, and from draining liquids from canned or frozen forms (unitless)

$L1_{ProFruit}$ = Mean reduction in fruit weight that results from paring or other preparation techniques for protected fruits (unitless)

$L2_{ExpFruit}$ = Mean reduction in fruit weight that results from draining liquids from cooked forms of the fruit (unitless)

$C_{ExpFruit}$ / $C_{ProFruit}$ = Chemical concentration in whole exposed fruits or whole protected fruits (depending on subscript) on a wet-weight (WW) basis (mg chemical/kg exposed fruit WW)

EF = Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario ($\leq$ 365 days)

$FC_{ExpFruit}$ / $FC_{ProFruit}$ = Fraction of exposed fruits or protected fruits (depending on subscript) obtained from contaminated area (unitless)

$IR_{ExpFruit(y)}$ / $IR_{ProFruit(y)}$ = Ingestion rate of home grown exposed fruits or protected fruits (depending on subscript) for age *y* (g WW/kg body weight-day)

Fruit ingestion rates in the survey were based on weights of unprepared fruits (e.g., one apple; one pear) or the weight of a can of fruit (e.g., 8 oz. can). The weight of the fruit ingested is less than the initial weight owing to common preparation actions ($L1_{ExpFruit}$ and $L1_{ProFruit}$; e.g., coring apples and pears; peeling apples; pitting cherries). Cooking of exposed fruit (e.g., berries, apples, peaches) often results in further weight loss that results from liquids lost during cooking and drained from the cooking vessel ($L2_{ExpFruit}$). EPA has assumed that cooking of protected fruit results in no loss of weight for the fruit.

B.3.2.4 Chemical Intake from Vegetable Ingestion

The methodology includes three separate algorithms for homegrown vegetables adapted from EPA's HHRAP Modeling System (U.S. EPA 2005a): one for exposed vegetables such as asparagus, broccoli, lettuce, and tomatoes (although they are actually a fruit); one for protected vegetables such as corn, cabbage, soybeans, and peas; and one for root vegetables such as carrots, beets, and potatoes (see Equation B-23, Equation B-24, and Equation B-25, respectively).

Equation B-23. Chemical Intake from Exposed Vegetables

$$ADD_{ExpVeg(y)} = \left(1 - L1_{ExpVeg}\right) \times \left(C_{ExpVeg} \times IR_{ExpVeg(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{ExpVeg}\right) \times \left(\frac{EF}{365 \text{ days}}\right) \quad \textit{Eqn. B-23}$$

Equation B-24. Chemical Intake from Protected Vegetables

$$ADD_{ProVeg(y)} = \left(1 - L1_{ProVeg}\right) \times \left(C_{ProVeg} \times IR_{ProVeg(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{ProVeg}\right) \times \left(\frac{EF}{365 \text{ days}}\right) \quad \textit{Eqn. B-24}$$

Equation B-25. Chemical Intake from Root Vegetables

$$ADD_{RootVeg(y)} = \left(1 - L1_{RootVeg}\right) \times \left(1 - L2_{RootVeg}\right) \times \left(C_{RootVeg} \times IR_{RootVeg(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{RootVeg}\right) \times \left(\frac{EF}{365 \text{ days}}\right) \quad \textit{Eqn. B-25}$$

where:

$ADD_{ExpVeg(y)}$ $ADD_{ProVeg(y)}$ $ADD_{RootVeg(y)}$	=	Average chemical intake from ingestion of exposed vegetables, protected vegetables, or root vegetables (depending on subscript) for age group *y* (mg chemical/kg body weight-day)
$L1_{ExpVeg}$	=	Mean net preparation and cooking weight loss for exposed vegetables (unitless); includes removing stalks, paring skins, discarding damaged leaves
$L1_{ProVeg}$	=	Mean net cooking weight loss for protected vegetables (unitless); includes removing husks, discarding pods of beans and peas, removal of outer leaves
$L1_{RootVeg}$	=	Mean net cooking weight loss for root vegetables (unitless); includes losses from removal of tops and paring skins

$L2_{RootVeg}$ = Mean net post cooking weight loss for root vegetables from draining cooking liquids and removal of skin after cooking (unitless)

C_{ExpVeg}, C_{ProVeg}, $C_{RootVeg}$ = Chemical concentration in exposed vegetables, protected vegetables, or root vegetables (depending on subscript) on a wet-weight (WW) basis (mg chemical/kg vegetable WW)

EF = Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario (≤365 days)

FC_{ExpVeg}, FC_{ProVeg}, $FC_{RootVeg}$ = Fraction of exposed vegetables, protected vegetables, or root vegetables (depending on subscript) obtained from contaminated area (unitless)

$IR_{ExpVeg(y)}$, $IR_{ProVeg(y)}$, $IR_{RootVeg(y)}$ = Ingestion rate of exposed vegetables, protected vegetables, or root vegetables (depending on subscript) for age group *y* (g vegetable WW/kg body weight-day)

B.3.2.5 Chemical Intake from Animal Product Ingestion

Calculations of chemical intake from the consumption of farm animals and related food products are provided below in Equation B-26 through Equation B-30 for homegrown beef, dairy (milk), pork, poultry, and eggs, respectively.

Equation B-26. Chemical Intake from Ingestion of Beef

$$ADD_{Beef(y)} = \left(1 - L1_{Beef}\right) \times \left(1 - L2_{Beef}\right) \times \left(C_{Beef} \times IR_{Beef(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{Beef} \right) \times \left(\frac{EF}{365 \text{ days}} \right) \quad \textbf{Eqn. B-26}$$

where:

$ADD_{Beef(y)}$ = Average daily chemical intake from ingestion of beef for age group y (mg/kg-day)

$L1_{Beef}$ = Mean net cooking loss for beef (unitless)

$L2_{Beef}$ = Mean net post cooking loss for beef (unitless)

C_{Beef} = Concentration of contaminant in beef (mg/kg WW))

EF = Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario (≤ 365 days)

$IR_{Beef(y)}$ = Ingestion rate of contaminated beef for age group y (g WW/kg-day)

$FCBeef$ = Fraction of beef consumed raised on contaminated area or fed contaminated silage and grains (unitless)

Equation B-27. Chemical Intake from Dairy Ingestion

$$ADD_{Dairy(y)} = \left(C_{Dairy} \times IR_{Dairy(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{Dairy} \right) \times \left(\frac{EF}{365 \text{ days}} \right) \quad \textbf{Eqn. B-27}$$

where:

$ADD_{Dairy(y)}$ = Average daily chemical intake from ingestion of total dairy for age group *y* (mg/kg-day)

C_{Dairy} = Average concentration of contaminant in total dairy (mg/kg WW)

EF = Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario (≤ 365 days)

$IR_{Dairy(y)}$ = Ingestion rate of contaminated total dairy for age group *y* (g WW/kg-day)

FC_{Dairy} = Fraction of total dairy products from contaminated area (unitless)

Equation B-28. Chemical Intake from Pork Ingestion

$$ADD_{Pork(y)} = (1 - L1_{Pork}) \times (1 - L2_{Pork}) \times \left(C_{Pork} \times IR_{Pork(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{Pork} \right) \times \left(\frac{EF}{365 \text{ days}} \right)$$

Eqn. B-28

where:

$ADD_{Pork(y)}$ = Average daily chemical intake from ingestion of pork for age group *y* (mg/kg-day)

$L1_{Pork}$ = Mean net cooking loss for pork (unitless); includes dripping and volatile losses during cooking; averaged over various cuts and preparation methods

$L2_{Pork}$ = Mean net post cooking loss for pork (unitless); includes losses from cutting, shrinkage, excess fat, bones, scraps, and juices; averaged over various cuts and preparation methods

C_{Pork} = Concentration of contaminant in pork (mg/kg WW)

EF = Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario (≤365 days)

$IR_{Pork(y)}$ = Ingestion rate of contaminated pork for age *y* (g WW/kg-day)

FC_{Pork} = Fraction of pork obtained from contaminated area (unitless)

Equation B-29. Chemical Intake from Poultry Ingestion

$$ADD_{Poultry(y)} = (1 - L1_{Poultry}) \times (1 - L2_{Poultry}) \times \left(C_{Poultry} \times IR_{Poultry(y)} \times 0.001 \frac{\text{kg}}{\text{g}} \times FC_{Poultry} \right) \times \left(\frac{EF}{365 \text{ days}} \right)$$

Eqn. B-29

where:

ADDPoultry(y) = Average daily dose (chemical intake) from ingestion of poultry (mg/kg-day)

L1Poultry = Mean net cooking loss for poultry (unitless)

L2Poultry = Mean net post cooking loss for poultry (unitless)

CPoultry	=	Concentration of chemical in poultry (mg/kg WW)
EF	=	Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario (≤365 days)
IRPoultry(y)	=	Ingestion rate of poultry for age group *y* (g WW/kg-day)
$FC_{Poultry}$	=	Fraction of poultry from contaminated area or fed contaminated grains (unitless)

The reduction in the weight of beef, pork, and poultry during and after cooking may correlate with an increase or decrease in the concentration of the chemical in the food as consumed depending on the chemical and depending on the cooking method.

Equation B-30. Chemical Intake from Egg Ingestion

$$ADD_{Egg(y)} = \left(C_{Egg} \times IR_{Egg(y)} \times 0.001\frac{\text{kg}}{\text{g}} \times FC_{Egg} \right) \times \left(\frac{EF}{365\ \text{days}} \right) \qquad \textit{Eqn. B-30}$$

where:

$ADD_{Egg(y)}$	=	Average daily chemical intake from ingestion of eggs for age group *y* (mg/kg-day)
C_{Egg}	=	Concentration of contaminant in eggs (mg/kg WW)
EF	=	Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario (≤ 365 days)
$IR_{Egg(y)}$	=	Ingestion rate of contaminated eggs for age group *y* (g WW/kg-day)
FC_{Egg}	=	Fraction of eggs obtained from contaminated area (unitless)

B.3.2.6 Chemical Intake from Drinking Water Ingestion

Assessments that evaluate chemical ingestion via drinking water use chemical concentration in g/L (equivalent to mg/mL), which could represent water from groundwater wells, community water, nearby surface waters, or other source. For this exposure pathway, ingestion rates are in units of milliliters of water per day (mL/day) (see Equation B-31).

Equation B-31. Chemical Intake from Drinking Water Ingestion

$$ADD_{DW(y)} = \left(\frac{C_{DW} \times IR_{DW(y)} \times FC_{DW}}{BW_{(y)}} \right) \times \left(\frac{EF}{365\ \text{days}} \right) \qquad \textit{Eqn. B-31}$$

where:

$ADD_{DW(y)}$	=	Average daily chemical intake from ingestion of drinking water from local residential water source for age group *y* (mg/kg-day)
C_{DW}	=	Concentration of contaminant in drinking water (g/L)

$IR_{DW(y)}$ = Drinking water ingestion rate for age group *y* (mL/day)

FC_{DW} = Fraction of drinking water obtained from contaminated area (unitless)

$BW_{(y)}$ = Body weight of age group *y* (kg)

EF = Exposure frequency; number of days per year of exposure for family(ies) as specified for scenario (≤365 days)

B.3.3 Total Chemical Intake

To estimate the total ADD, or intake of a chemical from all of the exposure media that a single individual in each age group is expected to contact (e.g., soil, local fish, five types of home grown produce, and five types of home-raised animals or animal products), the media-specific chemical intakes are summed for each age group. Total average daily exposure for a particular age group *y* ($ADD_{(y)}$) is estimated as the sum of chemical intake from all ingestion pathways combined, as illustrated in Equation B-32 through Equation B-37 below.

Equation B-32. - Equation B-37. Total Average Daily Dose of a Chemical for Different Age Groups

Equation B-32. $$ADD_{(<1)} = ADD_{breastmilk}$$

Equation B-33. $$ADD_{(1-2)} = \sum_{i=1}^{n} ADD_{(1-2,i)}$$

Equation B-34. $$ADD_{(3-5)} = \sum_{i=1}^{n} ADD_{(3-5,i)}$$

Equation B-35. $$ADD_{(6-11)} = \sum_{i=1}^{n} ADD_{(6-11,i)}$$

Equation B-36. $$ADD_{(12-19)} = \sum_{i=1}^{n} ADD_{(12-19,i)}$$

Equation B-37. $$ADD_{(adult)} = \sum_{i=1}^{n} ADD_{(adult,i)}$$

where *i* represents the i^{th} food type or ingestion medium and *n* equals the total number of food types or ingestion media, and *ADD* parameters are defined below:

$ADD_{(<1)}$ = Total average daily dose of chemical for infants less than one year from ingestion of breast milk (mg/kg-day)

$ADD_{(1-2)}$ = Total average daily dose of chemical from all ingestion sources for children ages 1 through 2 years (mg/kg-day)

$ADD_{(3-5)}$ = Total average daily dose for children ages 3 through 5 years (mg/kg-day)

$ADD_{(6-11)}$ = Total average daily dose for children ages 6 through 11 years (mg/kg-day)

$ADD_{(12-19)}$ = Total average daily dose for children ages 12 through 19 years (mg/kg-day)

$ADD_{(adult)}$ = Total average daily dose for adult age 20 up to 70 years (mg/kg-day)

The lifetime average daily dose (*LADD*) is calculated as the time-weight average of the *ADD* values for each age group (Equation B-38).

Equation B-38. Lifetime Average Daily Dose (LADD)

$$LADD = ADD_{(<1)}\left(\frac{1}{70}\right) + ADD_{(1\text{-}2)}\left(\frac{2}{70}\right) + ADD_{(3\text{-}5)}\left(\frac{3}{70}\right) + ADD_{(6\text{-}11)}\left(\frac{6}{70}\right) + ADD_{(12\text{-}19)}\left(\frac{8}{70}\right) + ADD_{(adult)}\left(\frac{50}{70}\right)$$

Eqn. B-38

The time-weighting factors simply equal the duration of exposure for the specified age category in years divided by the total lifespan, assumed to be 70 years.

B.3.4 Chemical Intake Calculations for Nursing Infants

The scientific literature indicates that infants can be exposed to some chemicals via their mothers' breast milk. The magnitude of the exposure can be estimated from information on the mother's exposure, data on the partitioning of the chemical into various compartments of the mother's body and into breast milk, and information on the infant's consumption of breast milk and absorption of the chemical. This exposure pathway is included in the multimedia ingestion risk methodology with adapted exposure algorithms and default assumptions from EPA's *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions* (U.S. EPA 1998), hereafter referred to as MPE, as explained below.

Note that this pathway generally is of most concern for *lipophilic* bioaccumulative chemicals (e.g., dioxins) that can cause developmental effects. The period of concern for the more *hydrophilic* chemicals that cause developmental effects generally is earlier, that is, from conception to birth. Hydrophilic chemicals generally exchange well between the maternal and fetal blood supplies at the placenta.

B.3.4.1 Infant Average Daily Absorbed Dose

The average daily dose of chemical *absorbed* by the infant (DAI_{inf}) is estimated with Equation B-39. This basic exposure equation relies on the concentration of the chemical in the breast milk, the infant's breast-milk ingestion rate (IR_{milk}), the absorption efficiency of the chemical by the oral route of exposure (AE_{inf}), the bodyweight of the infant (BW_{inf}), and the duration of breast feeding (ED). Equation B-39 is EPA's (U.S. EPA 1998) modification of an average daily dose for the infant model first published by Smith (1987) and includes variables for both the concentration of the chemical in the breast milk fat ($C_{milkfat}$) and the concentration of the chemical in the aqueous phase of breast milk ($C_{aqueous}$). The remainder of the DAI_{inf}-associated equations assume that most chemicals of concern will partition *either* to the lipid phase *or* to the aqueous phase of breast milk, although some chemicals may partition significantly to both phases of milk. Thus, the remaining equations assume that either $C_{milkfat}$ or $C_{aqueous}$ is equal to zero and hence drops out of the equation.

For the parameters in Equation B-39 (and the equations that follow) that are not calculated from another equation, an EPA default value and options for other values for the infant breast-milk-exposure pathway are described in Section B.6.4 of this attachment.

Equation B-39. Average Daily Dose of Chemical to the Nursing Infant

$$DAI_{inf} = \frac{\left[\left(C_{milkfat} \times f_{mbm}\right) + \left(C_{aqueous} \times \left(1 - f_{mbm}\right)\right)\right] \times IR_{milk} \times AE_{inf} \times ED}{BW_{inf} \times AT}$$

Eqn. B-39

where:

DAI_{inf} = Average daily dose of chemical absorbed by infant (mg chemical/kg body weight-day)

$C_{milkfat}$ = Concentration of chemical in lipid phase of maternal milk (mg chemical/kg milk lipid; calculated using Equation B-40)

f_{mbm} = Fraction of fat in breast milk (unitless)

$C_{aqueous}$ = Concentration of chemical in aqueous phase of maternal milk (mg chemical/kg aqueous phase milk; calculated using Equation B-44)

IR_{milk} = Infant milk ingestion rate over the duration of nursing (kg milk/day)

AE_{inf} = Absorption efficiency of the chemical by the oral route of exposure (i.e., chemical-specific fraction of ingested chemical that is absorbed by the infant) (unitless)

ED = Exposure duration, i.e., duration of breast feeding (days)

BW_{inf} = Body weight of infant averaged over the duration of nursing (kg)

AT = Averaging time associated with exposure of interest; equal to ED (days)

As mentioned above, Equation B-39 includes terms for the chemical in both the lipid- and non-lipid phases of milk. The remaining equations, however, assume that a chemical of concern will partition to the lipid or aqueous phase of breast milk, but not to both. Different models are used to estimate $C_{milkfat}$ (described in Section B.3.4.2) and $C_{aqueous}$ (described in Section B.3.4.3).

B.3.4.2 Chemical Concentration in Breast Milk Fat

When developing the Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions (MPE) (U.S. EPA 1998), EPA reviewed three first-order kinetics models for estimating chemical concentration in breast milk fat. The model selected for use with the multimedia ingestion risk methodology is the model selected for MPE. The other two models were not selected for use in MPE because one used a biotransfer factor (BTF) approach considered more of a screening model than a predictive tool (Travis *et al.* 1988) and the other assumed that the contaminant concentration in the maternal fat compartment is at steady state and that the concentration in breast milk fat is the same as in maternal body fat (Smith 1987). The model used in the multimedia ingestion risk methodology is a changing-concentration model that EPA adapted from a model by Sullivan *et al*. (1991). The model, shown in Equation B-40, estimates the average chemical concentration in the breast milk over the entire period of breast feeding by reference to a maximum theoretical steady-state concentration. Studies of lipophilic chemicals such as dioxins suggest that concentrations in the maternal milk are highest during the first few weeks of breast feeding and then decrease over time (ATSDR 1998). Equation B-40 accounts for the changing concentration in breast milk fat, but estimates one average value to represent the concentration over the entire duration of breast feeding. The model is dependent on the maternal body burden of the chemical and assumes that the chemical concentration in breast milk fat is the same as the concentration in general maternal body fat. According to reviewers of the model, this assumption warrants further investigation because milk fat appears to be synthesized in the mammary glands and may have lower chemical concentrations than general body fat stores (U.S. EPA 2001a).

Equation B-40. Chemical Concentration in Breast Milk Fat

$$C_{milkfat} = \frac{DAI_{mat} \times f_f}{k_{elim} \times f_{fm}} \times \left[\frac{k_{elim}}{k_{fat\_elac}} + \frac{1}{k_{fat\_elac} \times t_{bf}} \left(1 - e^{-k_{elim} t_{pn}} - \frac{k_{elim}}{k_{fat\_elac}} \right) \left(1 - e^{-k_{fat\_elac} t_{bf}} \right) \right] \quad \textit{Eqn. B-40}$$

where:

$C_{milkfat}$ = Concentration of chemical in lipid phase of maternal milk (mg chemical/kg lipid)

DAI_{mat} = Daily absorbed maternal chemical dose (mg chemical/kg maternal body weight-day; calculated using Equation B-41)

f_f = Fraction of total maternal body burden of chemical that is stored in maternal fat (mg chemical in body fat / mg total chemical in whole body; value from literature or EPA default - see Section B.6.5 of this attachment)

k_{elim} = Chemical-specific total elimination rate constant for elimination of the chemical by non-lactating women (per day; e.g., via urine, bile to feces, exhalation; value from literature or calculated using Equation B-42)

f_{fm} = Fraction of maternal body weight that is fat stores (unitless)

k_{fat_elac} = Chemical-specific rate constant for total elimination of chemical in the lipid phase of milk during nursing (per day; value from literature or calculated using Equation B-43)

t_{bf} = Duration of breast feeding (days)

t_{pn} = Duration of mother's exposure prior to parturition and initiation of breast feeding (days)

Equation B-40 relies on the daily maternal absorbed intake (DAI_{mat}) to determine the concentration of the chemical in the breast milk fat. DAI_{mat} is multiplied by the fraction of the chemical that is stored in maternal fat (f_f) to determine the amount (i.e., mass) of chemical in the fat. This product, divided by the chemical-specific elimination rate constant (k_{elim}) for non-lactating adult women and the fraction of the mother's weight that is fat (f_{fm}), represents the maximum theoretical steady-state concentration of the chemical in an adult woman. If used alone to estimate the chemical concentration in breast milk fat, the equation as explained thus far is likely to overestimate the chemical concentration in milk fat because it does not account for losses due to breast feeding. Alone, this term ($DAI_{mat} f_f / k_{elim} f_{fm}$) also assumes that the biological half-life of the chemical in the mother's breast milk fat is small relative to the duration of the mother's exposure. However, for chemicals with half-lives that are longer than the exposure duration, which are the chemicals of concern in the applications of this methodology to date, an additional term is needed to determine the average concentration in the milk fat over the duration of her exposure.

To account for breast feeding losses and longer chemical half-lives in the mother than the exposure duration, an additional term is included in Equation B-40. This term includes a fraction dependent on two rate constants, k_{elim} and the elimination constant for a lipophilic chemical in lactating women via the lipid phase of breast milk (k_{fat_elac}), the duration of the mother's chemical exposure prior to nursing (t_{pn}), and the duration of breast feeding (t_{bf}). The whole body concentration ($DAI_{mat} f_f / k_{elim} f_{fm}$), the maximum theoretical steady-state concentration, is multiplied by the rate of elimination averaged over the duration of the mother's exposure,

including her exposure prior to and during lactation. To review the derivation of Equation B-40, see Appendix B of MPE (U.S. EPA 1998).

To estimate an average daily dose *absorbed* by an infant's mother, or DAI_{mat}, the average daily dose (*ADD*) (in mg/kg-day) for the chemical from all sources calculated for adults (ADD_{adult}), described in Section B.3.3 of this attachment, Equation B-37), is multiplied by an absorption efficiency (AE_{mat}) or fraction of the chemical absorbed by the oral route of exposure, as shown in Equation B-41. The value for AE_{mat} can be estimated from absorption efficiencies for adults in general. Available data for some chemicals, in particular some inorganic compounds, indicate AE values for ingestion exposures of substantially less than 100 percent. For a few of these chemicals, data also indicate lower AEs for the chemical when ingested in food or in soil than when ingested in water (e.g., cadmium). For a screening level assessment, however, it is reasonable to either assume 100 percent for the AE_{mat} or to use the higher AE_{mat} of the food and water AE_{mat} values if available; hence, a single AE_{mat} parameter is included in Equation B-41.

Equation B-41. Daily Maternal Absorbed Intake

$$DAI_{mat} = ADD_{(adult)} \times AE_{mat} \qquad \textbf{Eqn. B-41}$$

where:

DAI_{mat} = Daily maternal dose of chemical absorbed from medium *i* (mg/kg-day)

$ADD_{(adult)}$ = Average daily dose to the mother (mg/kg-day) (see Section B.3.3 of this attachment, Equation B-37)

AE_{mat} = Absorption efficiency of the chemical by the oral exposure route (i.e., chemical-specific fraction of ingested chemical that is absorbed) by the mother (unitless) (value from literature or EPA default – see Section B.6.4 of this attachment)

Equation B-37, used to calculate $ADD_{(adult)}$, is based on many medium-specific ingestion rates that are normalized to body weight. The adult body weights to which the homegrown food ingestion rates are normalized are the body weights of the consumers in the original USDA survey (see Section B.6.3.3 of this attachment), which included both males and females. An assumption in the breast-milk exposure pathway is that those ingestion rates also are applicable to nursing mothers. The original data for ingestion rates for soil, drinking water, and fish are on a per person basis for males and females combined. This methodology divides those chemical intakes by an adult body weight for males and females combined (i.e., 71.4 kg mean value) to estimate the ADD normalized to body weight from those sources. If the assessor finds that those exposure media contribute the majority of the chemical intake for the risk scenario under consideration, they may use alternative ingestion rates for those media and alternative body weights for nursing women, as described in Section B.6.5 of this attachment.

Elimination rates for chemicals often are reported as the half-life of the chemical in the body following a known dose of chemical. Many chemicals exhibit a two-phase elimination process, the first being more rapid than the second. For screening risks for persistent and bioaccumulative chemicals, the half-life of the slower phase of elimination, presumably from non-blood compartments of the body, is the more important of the two. Assuming first-order kinetics, Equation B-42 is used to convert a measured half-life for elimination of a chemical for adults or non-lactating women to an elimination rate constant (U.S. EPA 1998). The equation

can be used to estimate any kind of chemical loss rate constant from a measured chemical half-life.

Equation B-42. Biological Elimination Rate Constant for Chemicals for Non-lactating Women

$$k_{elim} = \frac{\ln 2}{h} \qquad \textit{Eqn. B-42}$$

where:

k_{elim} = Chemical-specific elimination rate constant for elimination of the chemical for non-lactating women (per day; e.g., via urine, bile to feces, exhalation)

$ln2$ = Natural log of 2 (unitless constant)

h = Chemical-specific biological half-life of chemical for non-lactating women (days)

For chemicals transferred from the body of lactating women to breast milk, the rate of chemical elimination is augmented by the rate of chemical loss via the milk through breast feeding. The total elimination rate for lactating women sometimes is measured directly and reported in the literature. Where direct measurements are not available, and for chemicals that partition predominantly to the lipid-phase of milk, EPA has used Equation B-43 to estimate the total chemical elimination rate for lactating women, k_{fat_elac} (U.S. EPA 1998).

Equation B-43. Biological Elimination Constant for Lipophilic Chemicals for Lactating Women

$$k_{fat\_elac} = k_{elim} + \frac{IR_{milk} \times f_f \times f_{mbm}}{f_{fm} \times BW_{mat}} \qquad \textit{Eqn. B-43}$$

where:

k_{fat_elac} = Rate constant for total elimination of chemical during nursing (per day); accounts for both elimination by adults in general and the additional chemical elimination via the lipid phase of milk in nursing women

k_{elim} = Elimination rate constant for chemical from adults, including non-lactating women (per day; e.g., via urine, bile to feces, exhalation; chemical-specific; value from literature or calculated from half-life using Equation B-42)

IR_{milk} = Infant milk ingestion rate over the duration of nursing (kg/d)

f_f = Fraction of total maternal body burden of chemical that is stored in maternal fat (mg chemical in body fat / mg chemical total in body; value from literature or EPA default)

f_{mbm} = Fraction of fat in breast milk (unitless)

f_{fm} = Fraction of maternal body weight that is fat stores (unitless)

BW_{mat} = Maternal body weight over the entire duration of the mother's exposure to the chemical including during pregnancy and lactation (kg)

Equation B-43 is based on a model from Smith (1987) and accounts for the additional elimination pathway for lipophilic chemicals via the breast milk fat. The term K_{fat_elac} is estimated by adding an estimate of the first-order elimination constant for breast feeding losses to k_{elim}, which is the chemical-specific total elimination rate constant for non-lactating women. The breast feeding losses are estimated from the infant's intake rate of breast milk (IR_{milk}), the fraction of the total maternal body burden of the chemical that is stored in maternal body fat (f_f), the fraction of the mother's breast milk that consists of fat (lipids) (f_{mbm}), the mother's body weight (BW_{mat}), and the fraction of the mother's weight that is body fat (f_{fm}). In Equation B-43, the value for the mother's body weight should be specific to women of child-bearing age, as opposed to a body weight value for both males and females that is used to estimate an adult average daily dose and the mother's absorbed daily intake in Equation B-41. Body weight values for the mother are described in Section B.6.5 of this attachment. Smith's (1987) model assumes that the chemical partitions to the lipid-phase of breast milk to the same degree that it partitions into the mother's body fat. For highly lipophilic compounds, losses from breast feeding can be larger than losses by all other pathways (U.S. EPA 1998).

B.3.4.3 Chemical Concentration in Aqueous Phase of Breast Milk

When developing MPE (U.S. EPA 1998), EPA also considered models to estimate chemical concentrations in the aqueous phase of breast milk ($C_{aqueous}$). EPA adapted Smith's (1987) steady state concentration model for estimating $C_{milkfat}$ and developed the $C_{aqueous}$ model shown in Equation B-44 (U.S. EPA 1998). Chemicals that would partition to the aqueous phase of human milk include water-soluble chemicals, such as salts of metals, and other hydrophilic chemicals that may be in equilibrium with bound forms of the chemical in different tissues. The $C_{aqueous}$ equation assumes that the chemical concentration in the aqueous phase of milk is directly proportional to the chemical concentration in the mother's blood plasma. The portion of chemical sequestered in red blood cells (e.g., bound to RBC proteins) is assumed to be unavailable for direct transfer to breast milk.

Equation B-44. Chemical Concentration in Aqueous Phase of Breast Milk

$$C_{aqueous} = \frac{DAI_{mat} \times f_{pl} \times Pc_{bm}}{k_{aq\_elac} \times f_{pm}} \qquad \textit{Eqn. B-44}$$

where:

$C_{aqueous}$ = Concentration of chemical in aqueous phase of maternal milk (mg/kg)

DAI_{mat} = Daily absorbed maternal chemical dose (mg/kg-day; calculated by Equation B-41)

f_{pl} = Fraction of chemical in the body (based on absorbed intake) that is in the blood plasma compartment (unitless; value from literature or calculated by Equation B-45)

Pc_{bm} = Partition coefficient for chemical between the plasma and breast milk in the aqueous phase (unitless); assumed to equal 1.0

k_{aq_elac} = Chemical-specific rate constant for total elimination of chemical in the aqueous phase of milk during nursing (per day; value from literature or calculated in Equation B-46)

f_{pm} = Fraction of maternal weight that is blood plasma (unitless)

Equation B-44 is a steady-state concentration model that, like the Equation B-40 for $C_{milkfat}$, is dependent on the maternal absorbed daily intake (DAI_{mat}). In Equation B-44, DAI_{mat} is multiplied by the fraction of the absorbed chemical that is circulating in the blood plasma compartment (f_{pl}) and a partitioning coefficient for the chemical between plasma and the aqueous phase of breast milk (Pc_{bm}). For highly water-soluble chemicals that are not transported via special carrier molecules, the chemical is assumed to diffuse passively from the mother's blood serum to the aqueous phase of her milk, in which case Pc_{bm} would equal 1.0. The denominator includes the biological elimination constant for the chemical in the aqueous phase of breast milk in lactating women (k_{aq_elac}) and the fraction of the mother's weight that is plasma (f_{pl}). Because the model assumes steady-state, it does not account for chemical species with long half-lives in the body or for body burden losses due to lactation. These factors are important for highly lipophilic chemicals and for non-lipophilic chemicals such as methyl mercury, lead, and cadmium that partition into body compartments such as red blood cells and bone. While these latter chemicals or forms of these chemicals are water-soluble when free, they have relatively long half-lives because they are in equilibrium with the chemical bound to macromolecules in some tissue compartments. Lead is of particular concern because it can be released from the bone into the blood during lactation, and thus into the breast milk (U.S. EPA 2001a). Due to this limitation, the model may over- or underestimate exposure to the infant.

Because Equation B-44 is based on the relationship between the chemical concentrations in the aqueous phase of breast milk and the blood plasma, a value for the fraction of the chemical in the mother's blood plasma (f_{pl}) is required. Ideally, an empirical value for f_{pl} should be used. If empirical values are not available, f_{pl} can be estimated from Equation B-45, provided that an empirical value can be found for the fraction of the chemical in the body that is in the mother's whole blood compartment (f_{bl}; U.S. EPA 1998).

Equation B-45. Fraction of Total Chemical in Body in the Blood Plasma Compartment

$$f_{pl} = \frac{f_{bl} \times f_{bp}}{f_{bp} + Pc_{RBC}\left(1 - f_{bp}\right)} \qquad \textit{Eqn. B-45}$$

where:

f_{pl} = Fraction of chemical in body (based on absorbed intake) that is in the blood plasma compartment (unitless); chemical-specific

f_{bl} = Fraction of chemical in body (based on absorbed intake) in the whole blood compartment (unitless); chemical-specific

f_{bp} = Fraction of whole blood that is plasma (unitless)

$PcRBC$ = Partition coefficient for chemical between red blood cells and plasma (unitless); chemical-specific

If the fraction of the total chemical in the body that is in the whole blood compartment (f_{bl}) is known for a given chemical, then the fraction of that chemical that is in blood plasma depends only on the partition coefficient for the chemical between the red blood cells and the plasma (Pc_{RBC}) and the fraction of whole blood that is plasma (f_{bp}).

Another parameter for which a value is needed to solve Equation B-44 is the total chemical elimination rate for lactating women for hydrophilic chemicals, k_{aq_elac}. As for k_{fat_elac} for lipophilic

chemicals, k_{aq_elac} for hydrophilic chemicals would be equal to k_{elim} plus the loss rate for the chemical in the aqueous phase of breast-milk during lactation. In the case of hydrophilic chemicals, EPA has yet to propose a term for the additional elimination of a chemical in the aqueous phase of milk from breast feeding. Given basic physiological mechanisms, we assume that chemical loss rates via urine are likely to be significantly higher than loss rates from nursing, however. This is because the counter-current anatomy of kidney tubules allows substantial concentration of chemicals in the tubules for elimination in urine compared with the concentration in circulating blood and because of active secretion of some chemicals into urine. Therefore, the best estimation of elimination of hydrophilic chemicals by lactating women is simply k_{elim}, the elimination of the chemical from a non-lactating woman, as shown in Equation B-42. The extent to which k_{elim} is an underestimate of k_{aq_elac} for a given chemical will determine the extent of health protective bias in k_{aq_elac}.

Equation B-46. Biological Elimination Rate Constant for Hydrophilic Chemicals

$$k_{aq\_elac} = k_{elim} \qquad \textit{Eqn. B-46}$$

where:

k_{aq_elac} = Chemical-specific rate constant for total elimination of chemical by lactating women for hydrophilic chemicals (per day)

k_{elim} = Chemical-specific rate constant for total elimination of chemical by non-lactating women (per day; e.g., via urine, bile to feces, exhalation; value from literature or calculated from half-life using Equation B-42)

B.3.4.4 Alternative Model for Infant Intake of Methyl Mercury

EPA has not fully parameterized the aqueous model for mercury. In particular, no empirical value could be found for the steady-state fraction of total hydrophilic chemical body burden in the mother that is in the blood plasma (f_{pl}). This parameter could be estimated using Equation B-45 if a suitable chemical-specific fraction of chemical in the body that is in the whole blood (f_{bl}) could be found. However, the value found for f_{bl} is based on a single-dose study and is not considered reliable for use in chronic exposure calculations.

A literature search was conducted to identify existing physiologically based toxicokinetic (PBTK) models of lactational transfer of methylmercury (MeHg) in humans. Most PBTK models identified focused on gestational transfer of mercury between mother and fetus, including a PBTK dynamic compartmental model for gestational transfer of MeHg in humans developed by Gearhart *et al*. (1995, 1996), and reparameterized by Clewell *et al*. (1999).

Byczkowski and Lipscomb (2001) added a lactational transfer module to the Clewell *et al*. (1999) model. Byczkowski and Lipscomb compared their model's predictions to epidemiological data from mother-nursing-infant pairs obtained following an accidental high-dose poisoning in Iraq (Amin-Zaki *et al*. 1976) and from 34 mother-nursing-infant pairs examined in a low-dose, chronic exposure environment (Fujita and Takabatake 1977). Using data from the Iraq incident, Byczkowski and Lipscomb (2001) found good agreement between their model's predictions and the clinical data relating MeHg concentrations in breast milk to MeHg concentrations in infant's blood with time following the poisoning. To compare their model's predictions to data from chronic exposure to low doses of MeHg, Byczkowski and Lipscomb (2001) simulated MeHg intake for 500 days prior to conception, continued through gestation, and 6.5 months (200 days)

of lactation. Their model's predictions were consistent with Fujita and Takabatake's (1977) study, although use of hair/blood partition coefficients based on the results of the 1977 study precluded use of this comparison as model validation. Both the model predictions and the mean values from the 1977 data indicated that the concentration of MeHg in the blood of nursing infants was close to the MeHg concentration in their mothers' blood (approximately 0.025 to 0.027 mg/L, Figure 4 of report). At those blood concentrations, the PBTK model estimated the average maternal intake of MeHg to be 0.68 ± 0.33 (SD) µg/kg-day and the average infant intake of MeHg to be 0.80 ± 0.38 µg/kg-day. Therefore, for purposes of this methodology, the DAI_{inf} of MeHg is estimated to be the same as the maternal intake per unit body weight (Equation B-47).

Equation B-47. Calculation of Infant Average Daily Absorbed Dose of Methyl Mercury

$$DAI_{inf\_MeHg} = DAI_{mat\_MeHg} \qquad \textit{Eqn. B-47}$$

where:

DAI_{inf_MeHg} = Average daily dose of MeHg absorbed by infant from breast milk (mg/kg-day)

DAI_{mat_MeHg} = Average daily dose of methyl mercury absorbed by the mother, predominantly from fish (mg/kg-day)

B.4 Dose-Response Values

The chemical dose-response values used with the multimedia ingestion risk methodology include carcinogenic potency slope factors for ingestion and non-cancer oral reference doses (RfDs) for chronic exposures. The dose-response values currently used for RTR assessments are shown in Exhibit B-5. OAQPS identified dose-response values for use in RTR based on the following hierarchy of sources: EPA's Integrated Risk Information System (IRIS); the Centers for Disease Control's Agency for Toxic Substances and Disease Registry (ATSDR); and the California Environmental Protection Agency's (CalEPA's) Office of Environmental Health Hazard Assessment (OEHHA) Toxicity Criteria Database. For PB-HAPs without OAQPS-identified dose-response values, alternative methods for deriving values were used (see Sections B.4.4 and B.4.5).

As provided in Exhibit B-5, TEFs from van den Berg et al. (2006) are used except for two congeners for which EPA's IRIS program has developed a CSF—1,2,3,6,7,8-hexachlorodibenzo-p-dioxin and 1,2,3,7,8,9-hexachlorodibenzo-p-dioxin. Collectively across RTR assessments that EPA has conducted in recent years, these two congeners together constitute roughly 4 percent of total dioxin emissions from point sources. When the dioxin emissions are weighted by TEFs (to calculate TEQs), the two congeners constitute about 4 percent of the total dioxin TEQ emissions from point sources using TEF=0.1 from van den Berg et al. (2006) and about 2 percent using TEF=0.04 derived from the IRIS-based CSF. Therefore, the impact of changing the TEFs of the two congeners is small. Moreover, as noted above, IRIS is the most preferred source for toxicity data.

Exhibit B-5. Oral Dose-response Values Used to Calculate RTR Screening Threshold Emission Rates for PB-HAP Chemicals[a]

Chemical	CAS No.	Cancer Slope Factor		Reference Dose	
		Value $(mg/kg\text{-}day)^{-1}$	Source	Value (mg/kg-day)	Source
Inorganics					
Arsenic	7440382	1.5	IRIS	0.0003	IRIS
Cadmium compounds in food[b]	7440439	not available		1.0E-03	IRIS
Mercury (elemental)	7439976	not available		not available	
Mercuric chloride	7487947	not available		3.0E-04	IRIS
Methyl mercury (MeHg)	22967926	not available		1.0E-04	IRIS
Dioxins					
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	35822469	1.50E+03	Derived from WHO 2005 TEF[c]	7.0E-08	c
1,2,3,4,6,7,8-Heptachlorodibenzofuran	67562394	1.50E+03	Derived from WHO 2005 TEF[c]	7.0E-08	c
1,2,3,4,7,8,9-Heptachlorodibenzofuran	55673897	1.50E+03	Derived from WHO 2005 TEF[c]	7.0E-08	c
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	39227286	1.5E+04	Derived from WHO 2005 TEF[c]	7.0E-09	c
1,2,3,4,7,8-Hexachlorodibenzofuran	70648269	1.5E+04	Derived from WHO 2005 TEF[c]	7.0E-09	c
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	57653857	6.2E+03	IRIS	1.8E-08	c
1,2,3,6,7,8-Hexachlorodibenzofuran	57117449	1.5E+04	Derived from WHO 2005 TEF[c]	7.0E-09	c
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin	19408743	6.2E+03	IRIS	1.8E-08	c
1,2,3,7,8,9-Hexachlorodibenzofuran	72918219	1.5E+04	Derived from WHO 2005 TEF[c]	7.0E-09	c
2,3,4,6,7,8-Hexachlorodibenzofuran	60851345	1.5E+04	Derived from WHO 2005 TEF[c]	7.0E-09	c
1,2,3,4,6,7,8,9-Octachlorodibenzo-p-dioxin	3268879	4.5E+01	Derived from WHO 2005 TEF[c]	2.3E-06	c

Chemical	CAS No.	Cancer Slope Factor		Reference Dose	
		Value (mg/kg-day)$^{-1}$	Source	Value (mg/kg-day)	Source
1,2,3,4,6,7,8,9-Octachlorodibenzofuran	39001020	4.5E+01	Derived from WHO 2005 TEF[c]	2.3E-06	c
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	40321764	1.5E+05	Derived from WHO 2005 TEF[c]	7.0E-10	c
1,2,3,7,8-Pentachlorodibenzofuran	57117416	4.5E+03	Derived from WHO 2005 TEF[c]	2.3E-08	c
2,3,4,7,8-Pentachlorodibenzofuran	57117314	4.5E+04	Derived from WHO 2005 TEF[c]	2.3E-09	c
2,3,7,8-Tetrachlorodibenzo-p-dioxin	1746016	1.5E+05	EPA ORD	7.0E-10	IRIS
2,3,7,8-Tetrachlorodibenzofuran	51207319	1.5E+04	Derived from WHO 2005 TEF[c]	7.0E-09	c
Polycyclic Organic Matter					
1-Methylnaphthalene	90120	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	7.0E-02	ATSDR
2-Acetylaminofluorene	53963	1.0E+00	U.S. EPA 1999, POM Group 75002[d]	not available	
2-Chloronaphthalene	91587	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	8.0E-02	IRIS
2-Methylnaphthalene	91576	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	4.0E-3	IRIS
3-Methylcholanthrene	56495	2.2E+01	CalEPA	not available	
7,12-Dimethylbenz(a)anthracene	57976	2.5E+02	CalEPA	not available	
Acenaphthene	83329	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	6.0E-02	IRIS
Acenaphthylene	208968	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	
Anthracene	120127	0[e]	IRIS	3.0E-01	IRIS
Benz(a)anthracene	56553	1.0E-01	IRIS CSF for BaP with EPA Chemical Potency Factor	not available	

Chemical	CAS No.	Cancer Slope Factor		Reference Dose	
		Value (mg/kg-day)$^{-1}$	Source	Value (mg/kg-day)	Source
Benz(a)anthracene/chrysene (7-PAH)	103	5.0E-02	U.S. EPA 1999, POM Group 71002[d]	not available	
Benzo(a)pyrene	50328	1.0E+00	IRIS	not available	
Benzo(a)fluoranthene	203338	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	
Benzo(b)fluoranthene	205992	1.0E-01	IRIS CSF for BaP with EPA Chemical Potency Factor	not available	
Benzo(b+k)fluoranthene	207089	1.0E-01	U.S. EPA 1999, POM Group 76002[d]	not available	
Benzo(c)phenanthrene	195197	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	
Benzo(e)pyrene	192972	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	
Benzo(g,h,i)fluoranthene	203123	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	
Benzo(g,h,i)perylene	191242	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	
Benzo(j)fluoranthene	205823	1.0E-01	IRIS CSF for BaP with CAL Chemical Potency Factor	not available	
Benzo(k)fluoranthene	207089	1.0E-02	IRIS CSF for BaP with EPA Chemical Potency Factor	not available	
Benzofluoranthenes	56832736	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	
Carbazole	86748	2.0E-02	HEAST	not available	
Chrysene	218019	1.0E-03	IRIS CSF for BaP with EPA Chemical Potency Factor	not available	

Chemical	CAS No.	Cancer Slope Factor		Reference Dose	
		Value (mg/kg-day)$^{-1}$	Source	Value (mg/kg-day)	Source
Dibenzo(a,h)anthracene	53703	1.0E+00	IRIS CSF for BaP with EPA Chemical Potency Factor	not available	
Dibenzo(a,i)pyrene	189559	1.0E+01	IRIS CSF for BaP with CAL Chemical Potency Factor	not available	
Dibenzo(a,j)acridine	224420	1.0E-01	IRIS CSF for BaP with CAL Chemical Potency Factor	not available	
Fluoranthene	206440	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	4.0E-02	IRIS
Fluorene	86737	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	4.0E-02	IRIS
Indeno(1,2,3-c,d)pyrene	193395	1.0E-01	IRIS CSF for BaP with EPA Chemical Potency Factor	not available	
PAH, total	234	5.0E-02	U.S. EPA 1999, POM Group 71002[d]	not available	
Perylene	198550	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	
Phenanthrene	85018	0[e]	IRIS	not available	
Polycyclic Organic Matter	246	5.0E-02	U.S. EPA 1999, POM Group 71002[d]	not available	
Pyrene	129000	0[e]	IRIS	3.0E-02	IRIS
Retene	483658	5.0E-02	U.S. EPA 1999, POM Group 72002[d]	not available	

Chemical	CAS No.	Cancer Slope Factor		Reference Dose	
		Value $(mg/kg\text{-}day)^{-1}$	Source	Value (mg/kg-day)	Source

ATSDR = Agency for Toxic Substances and Disease Registry, IRIS = Integrated Risk Information System, EPA OAQPS = EPA's Office of Air Quality Planning and Standards, CalEPA = California EPA, EPA ORD = EPA's Office of Research and Development, HEAST = EPA Health Effects Assessment Tables, TEF = toxic equivalency factor

[a]Values as of July 2018; these values may be updated as newer ones become available.

[b]There are RfDs for both water ingestion and food ingestion for cadmium – the RfD for food is used.

[c]Dose-response values for these dioxin congeners are not available from EPA sources. CSFs and/or RfDs for these congeners were derived as discussed in Section B.4.4 of this attachment.

[d]The method to assign oral cancer slope factors to polycyclic organic matter (POM) without CSFs available from other EPA sources is the same as that used in the 1999 National Air Toxics Assessment (U.S. EPA 1999). A complete description of the methodology is available at: https://archive.epa.gov/nata2002/web/pdf/pom_approach.pdf and is summarized in Section B.4.5 of this attachment.

[e]Weight of evidence evaluations indicated that the available data were adequate to determine that this chemical was not carcinogenic (U.S. EPA 2010a).

B.4.1 Arsenic

EPA has developed a CSF of 1.5 per mg/kg-day for arsenic compounds based on data and analysis reported in IRIS. The data derived from 40,000 persons exposed to arsenic in drinking water and 7,500 relatively unexposed controls. A multistage model with time was used to predict dose-specific and age-specific skin cancer prevalence rates associated with ingestion of inorganic arsenic. IRIS also reports an RfD for arsenic; the RfD, however, was not considered because the use of the CSF is more health protective regardless of emission scenario.

B.4.2 Cadmium

EPA has developed two chronic RfDs for cadmium (Cd), one for food and one for water, based on data in IRIS indicating a lower absorption efficiency of cadmium from food than from water. The default RfD for RTR assessments is the higher RfD value for Cd compounds in food (as described in Section B.3.2.6, the drinking water exposure pathway is not modeled in the screening scenario because the likelihood that humans would use a lake as a drinking water source is assumed to be low). Users of this methodology who assess exposures via drinking water would need to use the RfD for Cd compounds in water (i.e., 5.0E-04 mg/kg-day).

B.4.3 Mercury

The RfD applies to the pregnant mother as well as young children. EPA has not specified the minimum exposure duration at the RfD level of exposure that is appropriate to use in characterizing risk. For this methodology, EPA assumes 10 years for women of childbearing age and 1 year for infants. EPA notes that human exposures to MeHg are primarily through the consumption of fish and shellfish (U.S. EPA 2001b). EPA found that, on average, approximately 76 percent of the exposure to MeHg for women of childbearing age could be attributed to ingestion of mercury in freshwater and estuarine fish and shellfish, with the remaining 24 percent derived from marine fish and shellfish. Other sources accounted for less than 0.06 percent of total exposures (U.S. EPA 2001b).

B.4.4 Dioxins (2,3,7,8-TCDD)

For chemicals for which the critical health effect is developmental, either *in utero* and/or during the first months or years of life, the exposure duration and timing of exposure for comparison with the RfD (or comparable values) require special consideration. The most sensitive health endpoints for both mercury and 2,3,7,8-TCDD are neurological effects during development that have long-lasting effects on learning and social behaviors. To ensure a protective risk characterization for these chemicals, it is important to use the shortest exposure duration appropriate, at the appropriate life stage, for comparison with the toxicity reference values. This

approach avoids "dilution" of an estimated average ADD that would result from averaging the lower daily chemical intake rates normalized to body weight for older children and adults with the potentially higher daily intake rates of infants over a longer exposure averaging period.

The convention for assessing risk from mixtures of dioxins is by application of toxic equivalency factors (TEFs) to dioxin concentrations, which are then expressed as toxic equivalents (TEQs). Of the dioxin congeners, 2,3,7,8-TCDD is the most widely studied and considered to be one of the most toxic congeners. It is therefore assigned a TEF of 1, with the other dioxin congener TEQ concentrations scaled relative to 2,3,7,8-TCDD concentrations on the basis of toxicity. For risk assessment of dioxins for RTR with the exception of the two 0.04 TEF values), the World Health Organization (WHO) 2005 TEFs presented in Exhibit B-6 were used to derive both the CSFs and RfDs (shown in Exhibit B-5) for dioxin congeners without available EPA dose-response values. The two exceptions are the CSFs for 1,2,3,7,8,9-hexachlorodibenzo-p-dioxin and 1,2,3,6,7,8-hexachlorodibenzo-p-dioxin, which are from IRIS.

The TEF values from the WHO 2005 reevaluation are based on effects mediated by dioxins binding to the aryl hydrocarbon receptor (AhR) (van den Berg et al. 2006). Blocking AhR receptors contributes to several health effects in mammals, including impaired immune response, reproduction, development (e.g., cleft palate), and liver function and a variety of neoplastic lesions. Some in vitro studies of dioxin congeners compared with the same type of study with 2,3,7,8-TCDD (e.g., specific enzyme induction in mammalian tissue cultures) contribute to the weight of evidence used to estimate TEFs. The TEFs can therefore be multiplied by the CSF for 2,3,7,8-TCDD to estimate the CSF for other dioxins or the RfD for 2,3,7,8-TCDD can be divided by the TEF to estimate RfDs for the other dioxins.

As provided in Exhibit B-6, WHO TEFs from van den Berg et al. (2006) are used except for two congeners for which EPA's IRIS program has developed a CSF—1,2,3,6,7,8-hexachlorodibenzo-p-dioxin and 1,2,3,7,8,9-hexachlorodibenzo-p-dioxin. Collectively across RTR assessments that EPA has conducted in recent years, these two congeners together constitute roughly 4 percent of total dioxin emissions from point sources. When the dioxin emissions are weighted by TEFs (to calculate TEQs), the two congeners constitute about 4 percent of the total dioxin TEQ emissions from point sources using TEF=0.1 from van den Berg et al. (2006) and about 2 percent using TEF=0.04 derived from the IRIS-based CSF. Therefore, the impact of changing the TEFs of the two congeners is small.

Exhibit B-6. WHO 2005 Toxic Equivalency Factors (TEFs) for Dioxin Congeners

Dioxin Congener	CAS No.	WHO 2005 Toxic Equivalency Factor[a]
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	35822469	0.01
1,2,3,4,6,7,8-Heptachlorodibenzofuran	67562394	0.01
1,2,3,4,7,8,9-Heptachlorodibenzofuran	55673897	0.01
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	39227286	0.1
1,2,3,4,7,8-Hexachlorodibenzofuran	70648269	0.1
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin[b]	57653857	0.04
1,2,3,6,7,8-Hexachlorodibenzofuran	57117449	0.1
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin[b]	19408743	0.04
1,2,3,7,8,9-Hexachlorodibenzofuran	72918219	0.1

Dioxin Congener	CAS No.	WHO 2005 Toxic Equivalency Factor[a]
2,3,4,6,7,8-Hexachlorodibenzofuran	60851345	0.1
1,2,3,4,6,7,8,9-Octachlorodibenzo-p-dioxin	3268879	3E-04
1,2,3,4,6,7,8,9-Octachlorodibenzofuran	39001020	3E-04
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	40321764	1
1,2,3,7,8-Pentachlorodibenzofuran	57117416	0.03
2,3,4,7,8-Pentachlorodibenzofuran	57117314	0.3
2,3,7,8-Tetrachlorodibenzo-p-dioxin	1746016	1
2,3,7,8-Tetrachlorodibenzofuran	51207319	0.1

[a]Source: van den Berg et al. 2006, with the one exception in the next footnote.
[b]For 1,2,3,7,8,9-HexCDD and 1,2,3,6,7,8-HexCDD, OAQPS identified an oral CSF from IRIS. For RTR multipathway assessments, EPA uses the TEF derived from this IRIS oral CSF (6200 per mg/kg-d, equaling a TEF of 0.04) rather than the WHO 2005 TEF of 0.1.

B.4.5 Polycyclic Organic Matter

Dose-response values for some of the POM species that are included in the screens were not identified by OAQPS; for these POM species, an alternative methodology for identifying CSFs was needed. Previously, for risk assessment of inhalation exposures to POM for EPA's National-Scale Air Toxics Assessments (NATA) and for RTR, OAQPS developed an approach for characterizing risks associated with the individual POM species and POM groups reported in the National Emissions Inventory (NEI). Individual POMs were assigned to one of eight POM groups according to cancer potencies derived by EPA for IRIS and by CalEPA, and based on assumptions regarding relative carcinogenicity. OAQPS then estimated an inhalation CSF for each POM group. The same approach was used to derive oral CSFs for POMs without available CSFs. Exhibit B-7 presents each POM group (with all its member POM species reported in NEI, not just those currently evaluated in this assessment) and the corresponding CSFs using this methodology. These group CSFs are used only when OAQPS has not, for the purposes of RTR, identified a CSF specific to the individual chemical.

Exhibit B-7. Oral Dose-response Values for Polycyclic Organic Matter (POM) Groups[a]

Individual POM or POM Group	CAS No.	Cancer Slope Factor[b] $(mg/kg\text{-}day)^{-1}$
POM Group 71002		
Benz(a)anthracene/chrysene (7-PAH)	103	0.5
Total PAH	234	
Polycyclic organic matter	246	
16-PAH	40	
16-PAH–7-PAH	75040	
POM Group 72002		
Anthracene	120127	0.5
Pyrene	129000	
Benzo(g,h,i) perylene	191242	

Individual POM or POM Group	CAS No.	Cancer Slope Factor[b] (mg/kg-day)$^{-1}$
Benzo(e)pyrene	192972	
Benzo(c)phenanthrene	195197	
Perylene	198550	
Benzo(a)fluoranthene	203338	
Fluoranthene	206440	
Acenaphthylene	208968	
1-Methylpyrene	2381217	
12-Methylbenz(a)anthracene	2422794	
Methylbenzopyrenes	247	
Methylanthracene	26914181	
Retene	483658	
Benzofluoranthenes	56832736	
9-Methylbenz(a)anthracene	779022	
1-Methylphenanthrene	832699	
Acenaphthene	83329	
Phenanthrene	85018	
Fluorene	86737	
1-Methylnaphthalene	90120	
2-Methylnaphthalene	91576	
2-Chloronaphthalene	91587	
POM Group 73002		
7,12-Dimethylbenz(a)anthracene	57976	1000
POM Group 74002		
Dibenzo(a,i)pyrene	189559	100
Dibenzo(a,h)pyrene	189640	
3-Methylcholanthrene	56495	
POM Group 75002		
Dibenzo(a,e)pyrene	192654	10
Methylchrysene	41637905	
5-Methylchrysene	3697243	
Benzo(a)pyrene	**50328**	
Dibenzo(a,h)anthracene	53703	
2-Acetylaminofluorene	53963	
POM Group 76002		
Benzo(b+k)fluoranthene	102	1
Indeno(1,2,3-c,d)pyrene	193395	

Individual POM or POM Group	CAS No.	Cancer Slope Factor[b] (mg/kg-day)$^{-1}$
Benzo(g,h,i)fluoranthene	203123	
Benzo(j)fluoranthene	205823	
Benzo(b)fluoranthene	205992	
Benzo(k)fluoranthene	207089	
Dibenzo(a,j)acridine	224420	
Benz(a)anthracene	56553	
POM Group 77002		
Chrysene	218019	0.1
Carbazole	86748	
POM Group 78002		
7-PAH	75	0.5

[a]These group CSFs are used only when OAQPS has not identified a CSF specific to the individual chemical.
[b]The method to assign oral cancer slope factors to POM groups was the same as that used in the 1999 National Air Toxics Assessment (U.S. EPA 1999). A complete description of the methodology is available at: http://archive.epa.gov/nata2002/web/pdf/pom_approach.pdf.

B.5 Risk Estimation

For persistent and bioaccumulative hazardous air pollutants (PB-HAPs), risks from inhalation of a chemical directly from air generally will be negligible compared with risks from ingestion of the chemical from foodstuffs grown in an area subject to air deposition of the chemical. Risk characterization for carcinogens with a linear mode of action at low doses is described in Section B.5.1 of this attachment. Risk characterization for chemicals likely to exhibit a threshold for response (e.g., non-cancer hazards) is described in Section B.5.2.

B.5.1 Cancer Risks

The estimated risk of developing cancer from exposure to a chemical from a specified source is characterized as the excess lifetime cancer risk (ELCR). The ELCR represents the incremental probability of an individual developing cancer over a lifetime as a result of lifetime exposure to the chemical. For a known or suspected carcinogen with a low-dose linear mode of action, the estimated ELCR is calculated as the product of the lifetime average daily dose (LADD) and the cancer slope factor (CSF):

Equation B-48. Calculation of Excess Lifetime Cancer Risk

$$ELCR = LADD \times CSF \qquad \textbf{Eqn. B-48}$$

where:

ELCR = Estimated excess lifetime cancer risk from a chemical summed across all exposure pathways and media (unitless)

LADD = Lifetime average total daily dose from all exposure pathways and media (mg/kg-day)

CSF = Oral carcinogenic potency slope factor for chemical (per mg/kg-day)

As described in Section B.3.3, the *LADD* (in mg/kg-day) for a chemical is calculated to reflect age-related differences in exposure rates that are experienced by a hypothetical individual throughout his or her lifetime of exposure. The total chemical intake is normalized to a lifetime, which for the purposes of this assessment is assumed to be 70 years.

EPA considers the possibility that children might be more sensitive than adults to toxic chemicals, including chemical carcinogens (U.S. EPA 2005b,c). Where data allow, EPA recommends development of lifestage-specific cancer potency CSFs. To date, EPA has developed a separate slope factor for early lifestage exposure for only one chemical (i.e., 1,1,1-trichloroethane; U.S. EPA 2007a), and current data availability for most chemicals preclude this approach. EPA has, therefore, examined options for default adjustments of the CSF to protect children. To date, the only mode of action (MOA) for carcinogenesis for which EPA has adequate data to develop a reasonable quantitative default approach is mutagenesis (U.S. EPA 2005b,c). For carcinogens with a mutagenic MOA for cancer, EPA concluded that the carcinogenic potency of a chemical may be approximately tenfold greater for the first 2 years of life (i.e., birth up to second birthday) and threefold greater for the next 14 years of life (i.e., ages 2 through 15) than for adults (U.S. EPA 2005c). These conclusions are represented by age-dependent adjustment factors (ADAFs) of 10, 3, and 1 for the first two lifestages and for adults, respectively.

These three lifestages do not match the age categories for the home grown food ingestion rates in the multimedia ingestion risk methodology. As a consequence, ADAFs for the age groups are adapted as time-weighted average values as follows:

$$ADAF_{(<1)} = 10 \qquad ADAF_{(6\text{-}11)} = 3$$

$$ADAF_{(1\text{-}2)} = \frac{(10 \times 1\,\text{yr}) + (3 \times 1\,\text{yr})}{2} = 6 \qquad ADAF_{(12\text{-}19)} = \frac{(3 \times 4\ \text{yrs}) + (1 \times 4\ \text{yrs})}{8} =$$

$$ADAF_{(3\text{-}5)} = 3 \qquad ADAF_{(adult)} = 1$$

To estimate total lifetime risk from a lifetime of exposure to such a chemical, EPA recommends estimating the cancer risk for each of the three lifestages separately and then adding the risks for i = 1 to 6 age groups.

Equation B-49. − Equation B-55. Lifetime Cancer Risk: Chemicals with a Mutagenic MOA for Cancer

Equation B-49. $$Risk_{(<1)} = ADD_{(0-<1)} \times 10 \times CSF \times (1\ yr/70\ yr)$$

Equation B-50. $$Risk_{(1-2)} = ADD_{(1-2)} \times 6.5 \times CSF \times (2\ yr/70\ yr)$$

Equation B-51. $$Risk_{(3-5)} = ADD_{(3-5)} \times 3 \times CSF \times (3\ yr/70\ yr)$$

Equation B-52. $$Risk_{(6-11)} = ADD_{(6-11)} \times 3 \times CSF \times (6\ yr/70\ yr)$$

Equation B-53. $$Risk_{(12-19)} = ADD_{(12-19)} \times 2 \times CSF \times (8\ yr/70\ yr)$$

Equation B-54. $$Risk_{(adult)} = ADD_{(adult)} \times 1 \times CSF \times (50\ yr/70\ yr)$$

Equation B-55. $$\boldsymbol{ELCR} = \sum_{i=1}^{n} Risk_{(i)}$$

In other words, Equation B-55 indicates that the total excess lifetime cancer risk (ELCR) equals the sum of the age-group-specific risks estimated by Equation B-49 through Equation B-54,

where:

$Risk_{(<1)}$ = Risk from chemical ingestion in first year of life

$Risk_{(1-2)}$ = Risk from chemical ingestion from first birthday through age 2 years

$Risk_{(3-5)}$ = Risk from chemical ingestion from age 3 through 5 years of age

$Risk_{(6-11)}$ = Risk from chemical ingestion from age 6 through 11 years of age

$Risk_{(12-19)}$ = Risk from chemical ingestion from age 12 through 19 years of age

$Risk_{(adult)}$ = Risk from chemical ingestion from age 20 to 70 years age

$ADD_{(<1)}$ = Average daily dose for infants under one year of age (mg/kg-day)

$ADD_{(1-2)}$ = Average daily dose from first birthday through age 2 years of age (mg/kg-day)

$ADD_{(3-5)}$ = Average daily dose from age 3 through 5 years of age (mg/kg-day)

$ADD_{(6-11)}$ = Average daily dose from age 6 through 11 years of age (mg/kg-day)

$ADD_{(12-19)}$ = Average daily dose from age 12 through 19 years of age (mg/kg-day)

$ADD_{(adult)}$ = Average daily dose for adults age 20 to 70 years of age (mg/kg-day)

CSF = Oral carcinogenic potency slope factor for chemical (per mg/kg-day)

$Risk_{(i)}$ = Risk from chemical ingestion for the i^{th} age group

$ELCR$ = Total extra lifetime cancer risk (incremental or extra risk)

n = Number of age groups (i.e., 6)

B.5.2 Non-cancer Hazard Quotients

Non-cancer risks are presented as hazard quotients (HQs), that is, the ratio of the estimated daily intake (i.e., ADD) to the reference dose (e.g., chronic RfD). If the HQ for a chemical is equal to or less than 1, EPA believes there is no appreciable risk that non-cancer health effects will occur. If the HQ is greater than 1 then there is at least some possibility for an adverse health effect. The larger the HQ value, the more likely an adverse health effect may occur.

B.5.2.1 Hazard Quotients for Chemicals with a Chronic RfD

For chemicals with a chronic RfD, HQs are calculated for each age group separately using Equation B-56 to indicate the potential for adverse health effects associated with chronic exposure via ingestion pathways. The HQ is the ratio of a long-term, daily average exposure normalized to the receptor's body weight (i.e., ADD) to the RfD for that chemical. HQs are threshold effects and are not additive across age groups.

Equation B-56. Hazard Quotient for Chemicals with a Chronic RfD

$$HQ = \frac{ADD}{RfD} \qquad \textbf{\textit{Eqn. B-56}}$$

where:

HQ = Hazard quotient for chemical (unitless)

ADD = Average daily ingested dose of chemical (mg/kg-day) from all food types and ingested media for the age group

RfD = Chronic oral reference dose for chemical (mg/kg-day)

B.5.2.2 Hazard Quotients for Chemicals with RfD Based on Developmental Effects

For chemicals for which the toxicity reference value is an RfD based on developmental effects, a shorter exposure duration (ED) and averaging time (AT) may be required. For this type of chemical (e.g., methylmercury, 2,3,7,8-TCDD), the appropriate ED/AT and sensitive lifestage for exposure may need to be estimated from the information provided in the critical developmental study(ies) from which the RfD was derived (e.g., in consultation with the RfD documentation in EPA's IRIS or in a toxicological profile developed for the chemical). For screening-level risk assessments, however, a health protective approach is to compare the highest ADD from among the child age categories to the RfD, as is done for all PB-HAPs. This approach ensures that the highest exposure from among the various age groups evaluated is taken into consideration, regardless of which age group might be most relevant to the health effect of interest (i.e., the age group on which the RfD is based).

B.5.2.3 Hazard Index for Chemicals with RfDs

When conducting screening-level assessments for multiple chemicals, it can be informative to calculate a hazard index (HI) for toxicologically similar chemicals (U.S. EPA 2000). The HI is the sum of HQs across chemicals (not age groups) as shown in Equation B-57. As with the HQ, if the HI value is less than 1, adverse health effects are not expected for that suite of chemicals. If the screening level HI exceeds 1, however, the risk assessor may in some instances, evaluate the assumptions of the screening-level assessment to determine if more realistic local values are available for parameters that drive risk. In addition, the risk assessor may need to examine the mode of action (MOA) and target organ(s) for the chemicals with the highest HQs to develop an appropriate approach to assessing their potential joint action.

Equation B-57. Hazard Index Calculation

$$HI = HQ_1 + HQ_2 \ldots HQ_n \qquad \textbf{Eqn. B-57}$$

where:

HI = Hazard index (unitless)

HQ_1 = Hazard Quotient for chemical 1 (unitless)

HQ_2 = Hazard Quotient for chemical 2 (unitless)

HQ_n = Hazard Quotient for chemical *n* (unitless)

The HI approach can be appropriate for chemicals with the same MOA and same target organ; however, MOA often is difficult to determine. An HI usually is "developed for each exposure route of interest, and for a single toxic effect or for toxicity to a single target organ" (U.S. EPA 2000; p 79). If a receptor is exposed to multiple chemicals that affect different target organs or

that operate by different MOAs, and if more than one HQ is close to 1, the risk assessor in some circumstances, may consider whether chemical interactions play a role in chemical toxicity (U.S. EPA 2000). Exposures to more than one chemical can result in a greater or lesser toxic response than might be predicted on the basis of one or the other chemical acting alone (toxicologically independent) or acting in concert (toxicologically similar chemicals). Users are referred to EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* for approaches to assessing the potential for adverse health effects from exposure to multiple chemicals (U.S. EPA 2000).

Users of the multimedia ingestion risk methodology are responsible for determining how to interpret HQs for multiple chemicals for each assessment.

B.6 Assessment Data and Parameter Values

This section describes the types of data and parameter values required for the multimedia ingestion risk methodology. Where applicable, default parameter values recommended by EPA are presented and discussed. In general, parameter values recommended by EPA were identified from HHRAP and EPA publications. For HHRAP parameters, including chemical-specific parameters, values were originally provided in the HHRAP Companion Database (U.S. EPA 2005a).

Required data for environmental media concentrations and air deposition rates, such as those predicted by (output of) TRIM.FaTE, are described in Section B.6.1 of this attachment. Values for farm-food-chain (FFC) parameters for specific types of produce and animal products are discussed in Section B.6.2. Receptor characterization parameters are described in Section B.6.3, including age-group-specific parameter values for body weight, water ingestion, and food ingestion by food type. Other exposure parameter values, such as exposure frequency and loss of chemical during food preparation and cooking, are discussed in Section B.6.4.

Where values for chemical-specific parameters are presented, values are presented only for PB-HAP chemicals currently evaluated using the TRIM-based RTR screening scenario. The data presented in this section were reviewed and used by EPA to develop the set of modeling defaults used to calculate screening threshold emission rates for RTR. Note that the default values used to estimate RTR screening thresholds, and the justification for selecting a specific value from the data sets described in this section, are discussed in Section B.7.

B.6.1 Environmental Concentrations

As noted in Section B.2, the multimedia ingestion risk methodology is intended to estimate exposures and risks to farming, gardening, and fishing families from ingestion of contaminated media in an area of airborne chemical deposition.

Acccordingly, the following values, specific to the air pollutant of concern, are required:

- a single air concentration (in g/m^3);
- the fraction of chemical in the air that is in the vapor phase;
- air-to-surface deposition rates for both vapor- and particle-phase chemical in the air (in g/m^2-yr);
- two fish tissue concentrations, one each for forage and game fish (i.e., fish in TL 3 and TL 4) (in mg/kg wet weight);
- concentrations in drinking water (in g/L) (only if drinking water exposure is assessed); and

- four chemical concentrations in soil (in µg/g dry weight), one each for:
  - surface soil in produce growing area,
  - surface soil where livestock feed,
  - root-zone soil in produce growing area, and
  - root-zone soil in livestock feed growing area.

The methodology as described in this attachment includes algorithms to estimate ingestion exposures via drinking water for a specified chemical concentration in the drinking water source (e.g., groundwater well). However, no exposure via drinking water is assumed to occur when calculating the Tier 1 screening thresholds. As discussed in Section B.3.2.6, drinking water exposure is not estimated for the scenario developed for the Tier 1 assessment because the likelihood that humans would use a lake as a drinking water source is assumed to be low.

For RTR assessements, EPA uses media concentrations output by TRIM.FaTE: EPA provides no default values for the data requriements listed above.

B.6.2 Farm-Food-Chain Parameter Values

Using the chemical information specified in Section B.6.1, above, chemical concentrations are calculated for foods that are commonly grown or raised on family farms: exposed and protected fruits; exposed and protected vegetables; root vegetables; beef; total dairy products; pork; and poultry and eggs.

B.6.2.1 List of Farm-Food-Chain (FFC) Parameters

The multimedia ingestion risk methodology uses algorithms from HHRAP (U.S. EPA 2005a) to estimate chemical concentrations in the produce identified above, as described in Section B.3.2. Parameters required for these HHRAP algorithms, including chemical-specific media transfer factors (e.g., soil-to-plant transfer coefficients) and plant- and animal-specific properties (e.g., plant interception fraction, quantity of forage consumed by cattle) are described in Exhibit B-8. As described in Section B.7, the default values recommended by EPA for RTR assessments are HHRAP-recommended parameter values, where available.

Exhibit B-8. Parameters Used to Estimate Chemical Concentrations in Farm Foods

Parameter	Description	Units
Plants/Produce		
$Br_{AG\text{-}produce\text{-}DW(i)}$	Chemical-specific plant/soil chemical bioconcentration factor for edible portion of aboveground produce type i, exposed or protected	Unitless (g soil DW / g produce DW)
$Bv_{AG(i)}$	Chemical-specific air-to-plant biotransfer factor for aboveground produce type i for vapor-phase chemical in air	Unitless ([mg chemical / g DW plant] / [mg chemical / g air])
Fw	Fraction of wet deposition that adheres to plant surfaces; 0.2 for anions, 0.6 for cations and most organics	Unitless
Kds	Chemical-specific soil/water partition coefficient	L soil pore water / kg soil DW
$kp_{(i)}$	Plant-specific surface loss coefficient for aboveground exposed produce and animal forage and silage	yr^{-1}

Parameter	Description	Units
$MAF_{(i)}$	Moisture adjustment factor for aboveground produce type *i* to convert the chemical concentration estimated for dry-weight produce to the corresponding chemical concentration for full-weight fresh produce	Percent water
RCF	Chemical-specific root concentration factor for tubers and root produce on a wet-weight (WW) basis	L soil pore water/ kg root WW
$Rp_{(i)}$	Plant-specific interception fraction for the edible portion of aboveground exposed produce or animal forage and silage	Unitless
$Tp_{(i)}$	Length of plant exposure to deposition per harvest of the edible portion of aboveground exposed produce or animal forage and silage	Year
$VG_{AG(i)}$	Empirical correction factor for aboveground exposed produce type i to address possible overestimate of the diffusive transfer of chemical from the outside to the inside of bulky produce, such as fruit	Unitless
$VG_{rootveg}$	Empirical correction factor for belowground produce (i.e., tuber or root vegetable) to account for possible overestimate of the diffusive transfer of chemicals from the outside to the inside of bulky tubers or roots (based on carrots and potatoes)	Unitless
$Yp_{(i)}$	Plant-specific yield or standing crop biomass of the edible portion of produce or animal feed	kg produce DW/m^2
Animal Products		
Bs	Soil bioavailability factor for livestock	Unitless
MF	Chemical-specific mammalian metabolism factor that accounts for endogenous degradation of the chemical	Unitless
$Ba_{(beef)}$	Chemical-specific biotransfer factor for chemical in diet of cow to chemical in beef on a fresh-wet (FW; equivalent to WW) basis	mg chemical/kg FW tissue/mg chemical/day or day/kg FW tissue
$Ba_{(dairy)}$	Biotransfer factor in dairy	day/kg FW tissue
$Ba_{(pork)}$	Biotransfer factor in pork	day/kg FW tissue
$Ba_{(poultry)}$	Biotransfer factor in poultry	day/kg FW tissue
$Ba_{(eggs)}$	Biotransfer factor in eggs	day/kg FW tissue
$Qs_{(m)}$	Quantity of soil eaten by animal type m each day	kg/day
$Qp_{(i,m)}$	Quantity of plant feed type i consumed per animal type m each day	kg/day

Source: EPA Source: U.S. EPA 2005a
DW = dry weight; FW = fresh weight; WW = wet weight

B.6.2.2 Produce Parameter Values

Exhibit B-9 and Exhibit B-10 provide the default chemical-specific input values that EPA uses for RTR assessments. Exhibit B-11 presents additional non-chemical-specific input values for parameters used in the algorithms that calculate chemical concentrations in produce. Unless otherwise noted, the default parameter values were obtained from HHRAP. Refer to HHRAP (U.S. EPA 2005a, Chapter 5 and associated appendices) for detailed descriptions of these parameters and documentation of input values.

Exhibit B-9. Chemical-Specific Inputs for Produce Parameters for PB-HAP Chemicals Included in RTR Multipathway Assessments

Chemical	Fraction of Wet Deposition (*Fw*) (unitless)[a]	Root Concentration Factor (*RCF*) (belowground) (L/kg)[b]	Soil-Water Partition Coefficient (*Kds*) (L/kg)[c]	Chemical Air-to-Plant Biotransfer Factor ($Bv_{AG(i)}$) (unitless)[d]
Inorganics				
Arsenic compounds	0.6	NA	2.5E+03	NA[e]
Cadmium compounds	0.6	NA	7.5E+01	NA[e]
Mercury (elemental)	0.6	NA	1.0E+03	0[f]
Mercuric chloride	0.6	NA	5.8E+04	1.8E+03
Methyl mercury	0.6	NA	7.0E+03	0[f]
Dioxins				
OctaCDD, 1,2,3,4,6,7,8,9-	0.6	4.8E+05	7.8E+05	2.4E+06
OctaCDF, 1,2,3,4,6,7,8,9-	0.6	3.4E+05	4.9E+05	2.3E+06
HeptaCDD, 1,2,3,4,6,7,8-	0.6	3.4E+05	4.9E+05	9.1E+05
HeptaCDF, 1,2,3,4,6,7,8-	0.6	1.2E+05	1.2E+05	8.3E+05
HeptaCDF, 1,2,3,4,7,8,9-	0.6	4.8E+04	3.9E+04	8.3E+05
HexaCDD, 1,2,3,4,7,8-	0.6	2.4E+05	3.1E+05	5.2E+05
HexaCDF, 1,2,3,4,7,8-	0.6	5.7E+04	4.9E+04	1.6E+05
HexaCDD, 1,2,3,6,7,8-	0.6	4.9E+05	8.0E+05	5.2E+05
HexaCDF, 1,2,3,6,7,8-	0.6	2.9E+05	4.1E+05	1.6E+05
HexaCDD, 1,2,3,7,8,9 -	0.6	4.9E+05	8.0E+05	5.2E+05
HexaCDF, 1,2,3,7,8,9-	0.6	1.6E+05	1.9E+05	1.6E+05
HexaCDF, 2,3,4,6,7,8-	0.6	2.9E+05	4.1E+05	1.6E+05
PentaCDD, 1,2,3,7,8-	0.6	9.2E+04	9.2E+04	2.4E+05
PentaCDF, 1,2,3,7,8-	0.6	3.9E+04	3.0E+04	9.8E+04
PentaCDF, 2,3,4,7,8-	0.6	2.3E+04	1.6E+04	9.8E+04
TetraCDD, 2,3,7,8-	0.6	4.0E+04	3.1E+04	6.6E+04
TetraCDF, 2,3,7,8-	0.6	1.2E+04	6.2E+03	4.6E+04
POMs				
2-Methylnaphthalene	0.6	2.2E+02	5.0E+01	1.4E+00
7,12-Dimethylbenz(a)anthracene	0.6	6.8E+03	4.0E+03	4.2E+04
Acenaphthene	0.6	2.4E+02	3.9E+01	4.6E+00
Acenaphthylene	0.6	2.8E+02	6.8E+01	8.1E+00
Benz(a)anthracene	0.6	6.7E+03	2.9E+03	6.8E+03
Benzo(a)pyrene	0.6	9.2E+03	7.8E+03	1.7E+05
Benzo(b)fluoranthene	0.6	6.6E+03	3.8E+03	1.7E+05
Benzo(g,h,i)perylene	0.6	3.0E+04	2.6E+04	2.3E+06
Benzo(k)fluoranthene	0.6	8.7E+03	5.5E+03	2.8E+05
Chrysene	0.6	6.0E+03	3.4E+03	1.4E+04
Dibenz(a,h)anthracene	0.6	2.3E+04	1.4E+04	6.2E+06
Fluoranthene	0.6	2.2E+03	3.9E+02	9.0E+02

Chemical	Fraction of Wet Deposition (*Fw*) (unitless)[a]	Root Concentration Factor (*RCF*) (belowground) (L/kg)[b]	Soil-Water Partition Coefficient (*Kds*) (L/kg)[c]	Chemical Air-to-Plant Biotransfer Factor ($Bv_{AG(i)}$) (unitless)[d]
Fluorene	0.6	3.8E+02	6.2E+01	1.6E+01
Indeno(1,2,3-cd)pyrene	0.6	3.5E+04	3.2E+04	2.8E+06

Source: U.S. EPA 2005a. NA = not applicable.

[a]6E-01 is the value for cations and most organic chemicals. As described in HHRAP (U.S. EPA 2005a), Appendix B (available at https://archive.epa.gov/epawaste/hazard/tsd/td/web/pdf/05hhrapapb.pdf), EPA estimated this value (U.S. EPA 1994a, 1995a) from a study by Hoffman et al. (1992) in which soluble gamma-emitting radionuclides and insoluble particles tagged with gamma-emitting radionuclides were deposited onto pasture grass via simulated rain. Note that the values developed experimentally for pasture grass may not accurately represent all aboveground produce-specific values. Also note that values based on the behavior of insoluble particles tagged with radionuclides may not accurately represent the behavior of organic compounds under site-specific conditions.

[b]For nonionic organic chemicals, as described in HHRAP (U.S. EPA 2005a, Section 5, available at https://archive.epa.gov/epawaste/hazard/tsd/td/web/html/riskvol.html#volume2), RCF is used to calculate the below-ground transfer of contaminants from soil to a root vegetable on a wet-weight basis as shown in Equation B-6. EPA estimated chemical-specific values for RCF from empirical regression equations developed by Briggs et al. (1982) based on their experiments measuring uptake of compounds into barley roots from growth solution. Briggs' regression equations allow calculation of RCF values from log Kow. For metals and mercuric compounds, empirical values for soil to root vegetable transfer on a dry-weight basis are available in the literature, thus the RCF was not needed.

[c]As discussed in HHRAP (EPA 2005a), Appendix A, Kds describes the partitioning of a compound between soil pore-water and soil particles and strongly influences the release and movement of a compound into the subsurface soils and underlying aquifer. Kds values for mercuric compounds were obtained from EPA (1997b). Kds for cadmium compounds were obtained from EPA 1996. For all PAHs and dioxins, Kds was calculated by multiplying Koc times the screening scenario's fraction organic carbon content (0.008). Empirical information for Koc was available for acenaphthene, benz(a)anthracene, benzo(a)pyrene, dibenz(a,h)anthracene, fluoranthene, and fluorene in USEAP 1996. For all other organic compounds, the Koc was calculated using the correlation equations presented in USEAP 2005a.

[d]As discussed in HHRAP (U.S. EPA 2005a), Appendix A, the value for mercuric chloride was obtained from U.S. EPA 1997b. BvAG(i) values for PAHs were calculated using the correlation equation derived for azalea leaves as cited in Bacci et al. (1992), then reducing this value by a factor of 100, as suggested by Lorber (1995), who concluded that the Bacci factor reduced by a factor of 100 was similar to his own observations. The values for dioxins were obtained from Lorber and Pinsky (2000).

[e]It is assumed that metals, with the exception of vapor-phase elemental mercury, do not transfer significantly from air into leaves.

[f]Speciation and fate and transport of mercury from emissions suggest that BvAG(i) values for elemental and methyl mercury are likely to be zero (U.S. EPA 2005a).

Exhibit B-10. Chemical-Specific Inputs by Plant Type for PB-HAP Chemicals in RTR Multipathway Assessments

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
Inorganics				
Arsenic compounds	Exp. Fruit	6.3E-03	-	1.0E+00
	Exp. Veg.	6.3E-03	-	1.0E+00
	Forage	3.6E-02	-	1.0E+00
	Grain	4.0E-03	-	-
	Prot. Fruit	6.3E-03	-	-
	Prot. Veg.	6.3E-03	-	-
	Root	8.0E-03	1.0E+00	-
	Silage	3.6E-02	-	5.0E-01
Cadmium compounds	Exp. Fruit	1.3E-01	-	1.0E+00

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Exp. Veg.	1.3E-01	-	1.0E+00
	Forage	3.6E-01	-	1.0E+00
	Grain	6.2E-02	-	-
	Prot. Fruit	1.3E-01	-	-
	Prot. Veg.	1.3E-01	-	-
	Root	6.4E-02	1.0E+00	-
	Silage	3.6E-01	-	5.0E-01
Mercury (elemental)	Exp. Fruit	-	-	1.0E+00
	Exp. Veg.	-	-	1.0E+00
	Forage	-	-	1.0E+00
	Grain	-	-	-
	Prot. Fruit	-	-	-
	Prot. Veg.	-	-	-
	Root	-	1.0E+00	-
	Silage	-	-	5.0E-01
Mercuric chloride	Exp. Fruit	1.5E-02	-	1.0E+00
	Exp. Veg.	1.5E-02	-	1.0E+00
	Forage	0.0E+00	-	1.0E+00
	Grain	9.3E-03	-	-
	Prot. Fruit	1.5E-02	-	-
	Prot. Veg.	1.5E-02	-	-
	Root	3.6E-02	1.0E+00	-
	Silage	0.0E+00	-	5.0E-01
Methyl mercury	Exp. Fruit	2.9E-02	-	1.0E-02
	Exp. Veg.	2.9E-02	-	1.0E-02
	Forage	0.0E+00	-	1.0E+00
	Grain	1.9E-02	-	-
	Prot. Fruit	2.9E-02	-	-
	Prot. Veg.	2.9E-02	-	-
	Root	9.9E-02	1.0E-02	-
	Silage	0.0E+00	-	5.0E-01
PAHs				
2-Methylnaphthalene	Exp. Fruit	2.3E-01	-	1.0E+00

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Exp. Veg.	2.3E-01	-	1.0E+00
	Forage	2.3E-01	-	1.0E+00
	Grain	2.3E-01	-	-
	Prot. Fruit	2.3E-01	-	-
	Prot. Veg.	2.3E-01	-	-
	Root	4.4E+00	1.0E+00	-
	Silage	2.3E-01	-	5.0E-01
7,12-Dimethylbenz(a)anthracene	Exp. Fruit	1.7E-02	-	1.0E-02
	Exp. Veg.	1.7E-02	-	1.0E-02
	Forage	1.7E-02	-	1.0E+00
	Grain	1.7E-02	-	-
	Prot. Fruit	1.7E-02	-	-
	Prot. Veg.	1.7E-02	-	-
	Root	1.7E+00	1.0E-02	-
	Silage	1.7E-02	-	5.0E-01
Acenaphthene	Exp. Fruit	2.1E-01	-	1.0E+00
	Exp. Veg.	2.1E-01	-	1.0E+00
	Forage	2.1E-01	-	1.0E+00
	Grain	2.1E-01	-	-
	Prot. Fruit	2.1E-01	-	-
	Prot. Veg.	2.1E-01	-	-
	Root	6.2E+00	1.0E+00	-
	Silage	2.1E-01	-	5.0E-01
Acenaphthylene	Exp. Fruit	1.9E-01	-	1.0E-02
	Exp. Veg.	1.9E-01	-	1.0E-02
	Forage	1.9E-01	-	1.0E+00
	Grain	1.9E-01	-	-
	Prot. Fruit	1.9E-01	-	-
	Prot. Veg.	1.9E-01	-	-
	Root	4.1E+00	1.0E-02	-
	Silage	1.9E-01	-	5.0E-01
Benz(a)anthracene	Exp. Fruit	1.7E-02	-	1.0E-02
	Exp. Veg.	1.7E-02	-	1.0E-02

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Forage	1.7E-02	-	1.0E+00
	Grain	1.7E-02	-	-
	Prot. Fruit	1.7E-02	-	-
	Prot. Veg.	1.7E-02	-	-
	Root	2.3E+00	1.0E-02	-
	Silage	1.7E-02	-	5.0E-01
Benzo(a)pyrene	Exp. Fruit	1.4E-02	-	1.0E-02
	Exp. Veg.	1.4E-02	-	1.0E-02
	Forage	1.4E-02	-	1.0E+00
	Grain	1.4E-02	-	-
	Prot. Fruit	1.4E-02	-	-
	Prot. Veg.	1.4E-02	-	-
	Root	1.2E+00	1.0E-02	-
	Silage	1.4E-02	-	5.0E-01
Benzo(b)fluoranthene	Exp. Fruit	1.8E-02	-	1.0E-02
	Exp. Veg.	1.8E-02	-	1.0E-02
	Forage	1.8E-02	-	1.0E+00
	Grain	1.8E-02	-	-
	Prot. Fruit	1.8E-02	-	-
	Prot. Veg.	1.8E-02	-	-
	Root	1.7E+00	1.0E-02	-
	Silage	1.8E-02	-	5.0E-01
Benzo(g,h,i)perylene	Exp. Fruit	5.7E-03	-	1.0E-02
	Exp. Veg.	5.7E-03	-	1.0E-02
	Forage	5.7E-03	-	1.0E+00
	Grain	5.7E-03	-	-
	Prot. Fruit	5.7E-03	-	-
	Prot. Veg.	5.7E-03	-	-
	Root	1.1E+00	1.0E-02	-
	Silage	5.7E-03	-	5.0E-01
Benzo(k)fluoranthene	Exp. Fruit	1.4E-02	-	1.0E-02
	Exp. Veg.	1.4E-02	-	1.0E-02
	Forage	1.4E-02	-	1.0E+00

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Grain	1.4E-02	-	-
	Prot. Fruit	1.4E-02	-	-
	Prot. Veg.	1.4E-02	-	-
	Root	1.6E+00	1.0E-02	-
	Silage	1.4E-02	-	5.0E-01
Chrysene	Exp. Fruit	1.9E-02	-	1.0E-02
	Exp. Veg.	1.9E-02	-	1.0E-02
	Forage	1.9E-02	-	1.0E+00
	Grain	1.9E-02	-	-
	Prot. Fruit	1.9E-02	-	-
	Prot. Veg.	1.9E-02	-	-
	Root	1.7E+00	1.0E-02	-
	Silage	1.9E-02	-	5.0E-01
Dibenz(a,h)anthracene	Exp. Fruit	6.8E-03	-	1.0E-02
	Exp. Veg.	6.8E-03	-	1.0E-02
	Forage	6.8E-03	-	1.0E+00
	Grain	6.8E-03	-	-
	Prot. Fruit	6.8E-03	-	-
	Prot. Veg.	6.8E-03	-	-
	Root	1.6E+00	1.0E-02	-
	Silage	6.8E-03	-	5.0E-01
Fluoranthene	Exp. Fruit	4.0E-02	-	1.0E-02
	Exp. Veg.	4.0E-02	-	1.0E-02
	Forage	4.0E-02	-	1.0E+00
	Grain	4.0E-02	-	-
	Prot. Fruit	4.0E-02	-	-
	Prot. Veg.	4.0E-02	-	-
	Root	5.6E+00	1.0E-02	-
	Silage	4.0E-02	-	5.0E-01
Fluorene	Exp. Fruit	1.5E-01	-	1.0E-02
	Exp. Veg.	1.5E-01	-	1.0E-02
	Forage	1.5E-01	-	1.0E+00
	Grain	1.5E-01	-	-

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Prot. Fruit	1.5E-01	-	-
	Prot. Veg.	1.5E-01	-	-
	Root	6.2E+00	1.0E-02	-
	Silage	1.5E-01	-	5.0E-01
Indeno(1,2,3-cd)pyrene	Exp. Fruit	5.1E-03	-	1.0E-02
	Exp. Veg.	5.1E-03	-	1.0E-02
	Forage	5.1E-03	-	1.0E+00
	Grain	5.1E-03	-	-
	Prot. Fruit	5.1E-03	-	-
	Prot. Veg.	5.1E-03	-	-
	Root	1.1E+00	1.0E-02	-
	Silage	5.1E-03	-	5.0E-01
Dioxins				
OctaCDD, 1,2,3,4,6,7,8,9-	Exp. Fruit	7.1E-04	-	1.0E-02
	Exp. Veg.	7.1E-04	-	1.0E-02
	Forage	7.1E-04	-	1.0E+00
	Grain	7.1E-04	-	-
	Prot. Fruit	7.1E-04	-	-
	Prot. Veg.	7.1E-04	-	-
	Root	6.1E-01	1.0E-02	-
	Silage	7.1E-04	-	5.0E-01
OctaCDF, 1,2,3,4,6,7,8,9-	Exp. Fruit	9.2E-04	-	1.0E-02
	Exp. Veg.	9.2E-04	-	1.0E-02
	Forage	9.2E-04	-	1.0E+00
	Grain	9.2E-04	-	-
	Prot. Fruit	9.2E-04	-	-
	Prot. Veg.	9.2E-04	-	-
	Root	6.8E-01	1.0E-02	-
	Silage	9.2E-04	-	5.0E-01
HeptaCDD, 1,2,3,4,6,7,8-	Exp. Fruit	9.2E-04	-	1.0E-02
	Exp. Veg.	9.2E-04	-	1.0E-02
	Forage	9.2E-04	-	1.0E+00
	Grain	9.2E-04	-	-

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Prot. Fruit	9.2E-04	-	-
	Prot. Veg.	9.2E-04	-	-
	Root	6.8E-01	1.0E-02	-
	Silage	9.2E-04	-	5.0E-01
HeptaCDF, 1,2,3,4,6,7,8-	Exp. Fruit	2.0E-03	-	1.0E-02
	Exp. Veg.	2.0E-03	-	1.0E-02
	Forage	2.0E-03	-	1.0E+00
	Grain	2.0E-03	-	-
	Prot. Fruit	2.0E-03	-	-
	Prot. Veg.	2.0E-03	-	-
	Root	9.4E-01	1.0E-02	-
	Silage	2.0E-03	-	5.0E-01
HeptaCDF, 1,2,3,4,7,8,9-	Exp. Fruit	4.0E-03	-	1.0E-02
	Exp. Veg.	4.0E-03	-	1.0E-02
	Forage	4.0E-03	-	1.0E+00
	Grain	4.0E-03	-	-
	Prot. Fruit	4.0E-03	-	-
	Prot. Veg.	4.0E-03	-	-
	Root	1.2E+00	1.0E-02	-
	Silage	4.0E-03	-	5.0E-01
HexaCDD, 1,2,3,4,7,8-	Exp. Fruit	1.2E-03	-	1.0E-02
	Exp. Veg.	1.2E-03	-	1.0E-02
	Forage	1.2E-03	-	1.0E+00
	Grain	1.2E-03	-	-
	Prot. Fruit	1.2E-03	-	-
	Prot. Veg.	1.2E-03	-	-
	Root	7.6E-01	1.0E-02	-
	Silage	1.2E-03	-	5.0E-01
HexaCDF, 1,2,3,4,7,8-	Exp. Fruit	3.5E-03	-	1.0E-02
	Exp. Veg.	3.5E-03	-	1.0E-02
	Forage	3.5E-03	-	1.0E+00
	Grain	3.5E-03	-	-
	Prot. Fruit	3.5E-03	-	-

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Prot. Veg.	3.5E-03	-	-
	Root	1.2E+00	1.0E-02	-
	Silage	3.5E-03	-	5.0E-01
HexaCDD, 1,2,3,6,7,8-	Exp. Fruit	7.0E-04	-	1.0E-02
	Exp. Veg.	7.0E-04	-	1.0E-02
	Forage	7.0E-04	-	1.0E+00
	Grain	7.0E-04	-	-
	Prot. Fruit	7.0E-04	-	-
	Prot. Veg.	7.0E-04	-	-
	Root	6.1E-01	1.0E-02	-
	Silage	7.0E-04	-	5.0E-01
HexaCDF, 1,2,3,6,7,8-	Exp. Fruit	1.0E-03	-	1.0E-02
	Exp. Veg.	1.0E-03	-	1.0E-02
	Forage	1.0E-03	-	1.0E+00
	Grain	1.0E-03	-	-
	Prot. Fruit	1.0E-03	-	-
	Prot. Veg.	1.0E-03	-	-
	Root	7.1E-01	1.0E-02	-
	Silage	1.0E-03	-	5.0E-01
HexaCDD, 1,2,3,7,8,9-	Exp. Fruit	7.0E-04	-	1.0E-02
	Exp. Veg.	7.0E-04	-	1.0E-02
	Forage	7.0E-04	-	1.0E+00
	Grain	7.0E-04	-	-
	Prot. Fruit	7.0E-04	-	-
	Prot. Veg.	7.0E-04	-	-
	Root	6.1E-01	1.0E-02	-
	Silage	7.0E-04	-	5.0E-01
HexaCDF, 1,2,3,7,8,9-	Exp. Fruit	1.6E-03	-	1.0E-02
	Exp. Veg.	1.6E-03	-	1.0E-02
	Forage	1.6E-03	-	1.0E+00
	Grain	1.6E-03	-	-
	Prot. Fruit	1.6E-03	-	-
	Prot. Veg.	1.6E-03	-	-

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Root	8.5E-01	1.0E-02	-
	Silage	1.6E-03	-	5.0E-01
HexaCDF, 2,3,4,6,7,8-	Exp. Fruit	1.0E-03	-	1.0E-02
	Exp. Veg.	1.0E-03	-	1.0E-02
	Forage	1.0E-03	-	1.0E+00
	Grain	1.0E-03	-	-
	Prot. Fruit	1.0E-03	-	-
	Prot. Veg.	1.0E-03	-	-
	Root	7.1E-01	1.0E-02	-
	Silage	1.0E-03	-	5.0E-01
PentaCDD, 1,2,3,7,8-	Exp. Fruit	2.4E-03	-	1.0E-02
	Exp. Veg.	2.4E-03	-	1.0E-02
	Forage	2.4E-03	-	1.0E+00
	Grain	2.4E-03	-	-
	Prot. Fruit	2.4E-03	-	-
	Prot. Veg.	2.4E-03	-	-
	Root	1.0E+00	1.0E-02	-
	Silage	2.4E-03	-	5.0E-01
PentaCDF, 1,2,3,7,8-	Exp. Fruit	4.6E-03	-	1.0E-02
	Exp. Veg.	4.6E-03	-	1.0E-02
	Forage	4.6E-03	-	1.0E+00
	Grain	4.6E-03	-	-
	Prot. Fruit	4.6E-03	-	-
	Prot. Veg.	4.6E-03	-	-
	Root	1.3E+00	1.0E-02	-
	Silage	4.6E-03	-	5.0E-01
PentaCDF, 2,3,4,7,8-	Exp. Fruit	6.8E-03	-	1.0E-02
	Exp. Veg.	6.8E-03	-	1.0E-02
	Forage	6.8E-03	-	1.0E+00
	Grain	6.8E-03	-	-
	Prot. Fruit	6.8E-03	-	-
	Prot. Veg.	6.8E-03	-	-
	Root	1.5E+00	1.0E-02	-

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]
	Silage	6.8E-03	-	5.0E-01
TetraCDD, 2,3,7,8-	Exp. Fruit	4.5E-03	-	1.0E-02
	Exp. Veg.	4.5E-03	-	1.0E-02
	Forage	4.5E-03	-	1.0E+00
	Grain	4.5E-03	-	-
	Prot. Fruit	4.5E-03	-	-
	Prot. Veg.	4.5E-03	-	-
	Root	1.3E+00	1.0E-02	-
	Silage	4.5E-03	-	5.0E-01
TetraCDF, 2,3,7,8-	Exp. Fruit	1.2E-02	-	1.0E-02
	Exp. Veg.	1.2E-02	-	1.0E-02
	Forage	1.2E-02	-	1.0E+00
	Grain	1.2E-02	-	-
	Prot. Fruit	1.2E-02	-	-
	Prot. Veg.	1.2E-02	-	-
	Root	1.9E+00	1.0E-02	-
	Silage	1.2E-02	-	5.0E-01

[a]As discussed in HHRAP (U.S. EPA 2005a), the $Br_{AG\text{-}produce\text{-}DW(i)}$ for aboveground produce and forage accounts for the uptake from soil and the subsequent transport of contaminants through the roots to the aboveground plant parts. For organics, correlation equations to calculate values for *Br* on a dry weight basis were obtained from Travis and Arms (1988). For cadmium, *Br* values were derived from uptake slope factors provided in U.S. EPA 1992. Uptake slope is the ratio of contaminant concentration in dry weight plant tissue to the mass of contaminant applied per hectare soil. *Br* aboveground values for mercuric chloride and methyl mercury were calculated using methodology and data from Baes, et al. (1984). *Br* forage values for mercuric chloride and methyl mercury (on a dry weight basis) were obtained from U.S. EPA 1997b. The HHRAP methodology assumes that elemental mercury doesn't deposit onto soils. Therefore, it's assumed that there is no plant uptake through the soil.

[b]As discussed in HHRAP (U.S. EPA 2005a), Appendix B, $VG_{rootveg}$ represents an empirical correction factor that reduces produce concentration. Because of the protective outer skin, size, and shape of bulky produce, transfer of lipophilic chemicals (i.e., log Kow greater than 4) to the center of the produce is not likely. In addition, typical preparation techniques, such as washing, peeling, and cooking, further reduce the concentration of the chemical in the vegetable as consumed by removing the high concentration of chemical on and in the outer skin, leaving the flesh with a lower concentration than would be the case if the entire vegetable were pureed without washing. For belowground produce, HHRAP (U.S. EPA 2005a) recommends using a $VG_{rootveg}$ value of 0.01 for PB-HAP with a log Kow greater than 4 and a value of 1.0 for PB-HAP with a log Kow less than 4 based on information provided in U.S. EPA 1994b. In developing these values, U.S. EPA (1994b) assumed that the density of the skin and the whole vegetable are equal (potentially overestimating the concentration of PB-HAP in belowground produce due to root uptake).

[c]As discussed in HHRAP (U.S. EPA 2005a), Appendix B, VG_{ag} represents an empirical correction factor that reduces aboveground produce concentration and was developed to estimate the transfer of PB-HAP into leafy vegetation versus bulkier aboveground produce (e.g., apples). Because of the protective outer skin, size, and shape of bulky produce, transfer of lipophilic PB-HAP (log Kow greater than 4) to the center of the produce is not likely. In addition, typical preparation techniques, such as washing, peeling, and cooking, further reduce residues. For aboveground produce, HHRAP (U.S. EPA 2005a) recommends using a VG_{ag} value of 0.01 for PB-HAP with a log Kow greater than 4 and a value of 1.0 for PB-HAP with a log Kow less than 4 based on information provided in U.S. EPA 1994b. In developing these values, U.S. EPA (1994b) assumed the following: (1) translocation of compounds deposited on the surface of aboveground vegetation to inner parts of aboveground produce would be insignificant (potentially underestimating the concentration of PB-HAP in aboveground produce due to air-to-plant transfer); (2) the density of the skin and the whole vegetable are equal (potentially overestimating the concentration of PB-HAP in aboveground produce due to air-to-plant transfer); and (3) the thickness of vegetable skin and broadleaf tree skin are equal (effects on the concentration of PB-HAP in aboveground produce due to air-to-plant transfer unknown). For forage, HHRAP recommends a VG_{ag} value of 1.0, also based on information provided in U.S. EPA 1994b. A VG_{ag} value for silage is not provided in U.S. EPA 1994b; the VGag value for

Compound Name	Plant Part	Plant-Soil Bio-Concentration Factor ($Br_{AG\text{-}produce\text{-}DW(i)}$) (unitless)[a]	Empirical Correction Factor- Belowground Produce ($VG_{rootveg}$) (unitless)[b]	Empirical Correction Factor- Aboveground Produce ($VG_{AG(i)}$) (unitless)[c]

silage of 0.5 was obtained from NC DEHNR (1997); however, NC DEHNR does not present a specific rationale for this recommendation. Depending on the composition of the site-specific silage, this value may under- or overestimate the actual value.

Exhibit B-11. Non-Chemical-Specific Produce Inputs

Plant Part	Interception Fraction ($Rp_{(i)}$) (unitless)[a]	Plant Surface Loss Coefficient ($kp_{(i)}$) (1/year)[b]	Length of Plant Exposure to Deposition ($Tp_{(i)}$) (year)[c]	Yield or Standing Crop Biomass ($Yp_{(i)}$) (kg/m^2)[d]	Plant Tissue-Specific Moisture Adjustment Factor ($MAF_{(i)}$) (percent)[e]
Exposed Vegetable	0.982	18	0.16	5.66	92
Protected Fruit	NA	NA	NA	NA	90
Protected Vegetable	NA	NA	NA	NA	80
Forage (animal feed)	0.5	18	0.12	0.24	NA[f]
Exposed Fruit	0.053	18	0.16	0.25	85
Root Vegetables	NA	NA	NA	NA	87
Silage (animal feed)	0.46	18	0.16	0.8	NA[f]
Grain (animal feed)	NA	NA	NA	NA	NA[f]

Source: U.S. EPA 2005a. NA = not applicable.

[a]Baes *et al*. (1984) used an empirical relationship developed by Chamberlain (1970) to identify a correlation between initial Rp values and pasture grass productivity (standing crop biomass [*Yp*]) to calculate *Rp* values for exposed vegetables, exposed fruits, forage, and silage. Two key uncertainties are associated with using these values for *Rp*: (1) Chamberlain's (1970) empirical relationship developed for pasture grass may not accurately represent aboveground produce. (2) The empirical constants developed by Baes *et al*. (1984) for use in the empirical relationship developed by Chamberlain (1970) may not accurately represent the site-specific mixes of aboveground produce consumed by humans or the site-specific mixes of forage or silage consumed by livestock.

[b]The term *kp* is a measure of the amount of chemical that is lost to natural physical processes (e.g., wind, water) over time. The HHRAP-recommended value of 18 yr^{-1} (also recommended by U.S. EPA 1994a and 1998) represents the midpoint of a range of values reported by Miller and Hoffman (1983). There are two key uncertainties associated with using these values for *kp*: (1) The recommended equation for calculating *kp* includes a health protective bias in that it does not consider chemical degradation processes. (2) Given the reported range of *kp* values from 7.44 to 90.36 yr^{-1}, plant concentrations could range from about 1.8 times higher to about 5 times lower than the plant concentrations estimated in FFC media using the midpoint *kp* value of 18.

[c]HHRAP (U.S. EPA 2005a) recommends using a *Tp* value of 0.16 years for aboveground produce and cattle silage. This is consistent with earlier reports by U.S. EPA (1994a, 1998) and NC DEHNR (1997), which recommended treating *Tp* as a constant based on the average period between successive hay harvests. Belcher and Travis (1989) estimated this period at 60 days. *Tp* is calculated as 60 days ÷ 365 days/year = 0.16 years. For forage, the average of the average period between successive hay harvests (60 days) and the average period between successive grazing (30 days) is used (that is, 45 days), and *Tp* is calculated as (60 days + 30 days)/ 2 ÷ 365 days/yr = 0.12 yr. Two key uncertainties are associated with use of these values for *Tp*: (1) The average period between successive hay harvests (60 days) may not reflect the length of the growing season or the length between successive harvests for site-specific aboveground produce crops. The concentration of chemical in aboveground produce due to direct (wet and dry) deposition (*Pd*) will be underestimated if the site-specific value of *Tp* is less than 60 days, or overestimated if the site-specific value of *Tp* is more than 60 days.

[d]*Yp* values for aboveground produce and forage were calculated using an equation presented in Baes *et al*. (1984) and Shor *et al*. (1982): $Yp = Y_{hi} / A_{hi}$, where Y_{hi} = Harvest yield of i^{th} crop (kg DW) and A_{hi} = Area planted to i^{th} crop (m2), and using values for Y_h and A_h from USDA (1994b and 1994c). A production-weighted U.S. average *Yp* of 0.8 kg DW/m2 for silage was obtained from Shor *et al*. 1982.

[e]*MAF* represents the plant tissue-specific moisture adjustment factor to convert dry-weight concentrations into wet-weight concentrations (which are lower owing to the dilution by water compared with dry-weight concentrations). Values obtained from Chapter 10 of EPA's 2003 SAB Review materials for 3MRA Modeling System, Volume II, "Farm Food Chain and Terrestrial Food Web Data" (U.S. EPA 2003a), which references U.S. EPA 1997c (available at https://archive.epa.gov/epawaste/hazard/web/pdf/2-chap10.pdf).

[f]MAFs were not implemented for animal feed groups as the calculations for chemical concentration are based on dry weight not wet weight. Previous values used for these groups were 92, 92, and 90 respective; however, note that the value for grain used as animal feed is based on corn and soybeans, not seed grains such as barley, oats, or wheat.

B.6.2.3 Animal Product Parameter Values

The multimedia ingestion risk methodology requires chemical-specific inputs for many of the animal product algorithms. The relevant values are shown in Exhibit B-12 for the PB-HAP chemicals included in RTR multipathway assessments to date. The HHRAP algorithms require additional inputs for the animal products calculations that are not specific to PB-HAPs, but are

specific to the animal and animal product type. The soil and plant ingestion rates recommended in HHRAP for beef cattle, dairy cattle, swine, and chicken are provided in Exhibit B-13.

Exhibit B-12. Animal Product Chemical-specific Inputs for PB-HAP Chemicals Included in RTR Multipathway Assessments

Compound Name	Biotransfer Factors (Ba_m) (day/kg FW tissue)[a] and Metabolism Factors (MF) (unitless)[b]						
	Mammal				Non-mammal		
	Beef (Ba_{beef})	Dairy (Ba_{dairy})	Pork (Ba_{pork})	*MF*	Eggs (Ba_{eggs})	Poultry ($Ba_{poultry}$)	*MF*
Inorganics							
Arsenic compounds	2.0E-03	6.0E-04	1.0E-02	1	7.0E-02	3.0E-02	NA
Cadmium compounds	1.2E-04	6.5E-06	1.9E-04	1	2.5E-03	1.1E-01	NA
Mercury (elemental)	0	0	0	1	0	0	NA
Mercuric chloride	1.1E-04	1.4E-06	3.4E-05	1	2.4E-02	2.4E-02	NA
Methyl mercury	1.2E-03	1.7E-05	5.1E-06	1	3.6E-03	3.6E-03	NA
Dioxins							
OctaCDD, 1,2,3,4,6,7,8,9-	6.9E-03	1.4E-03	8.3E-03	1	2.9E-03	5.1E-03	NA
OctaCDF, 1,2,3,4,6,7,8,9-	8.8E-03	1.8E-03	1.1E-02	1	3.7E-03	6.5E-03	NA
HeptaCDD, 1,2,3,4,6,7,8-	8.8E-03	1.8E-03	1.1E-02	1	3.7E-03	6.5E-03	NA
HeptaCDF, 1,2,3,4,6,7,8-	1.6E-02	3.5E-03	2.0E-02	1	6.9E-03	1.2E-02	NA
HeptaCDF, 1,2,3,4,7,8,9-	2.4E-02	5.1E-03	3.0E-02	1	1.0E-02	1.8E-02	NA
HexaCDD, 1,2,3,4,7,8-	1.1E-02	2.3E-03	1.3E-02	1	4.6E-03	8.1E-03	NA
HexaCDF, 1,2,3,4,7,8-	2.3E-02	4.8E-03	2.8E-02	1	9.6E-03	1.7E-02	NA
HexaCDD, 1,2,3,6,7,8-	6.8E-03	1.4E-03	8.2E-03	1	2.9E-03	5.0E-03	NA
HexaCDF, 1,2,3,6,7,8-	9.7E-03	2.0E-03	1.2E-02	1	4.1E-03	7.1E-03	NA
HexaCDD, 1,2,3,7,8,9 -	6.8E-03	1.4E-03	8.2E-03	1	2.9E-03	5.0E-03	NA
HexaCDF, 1,2,3,7,8,9-	1.4E-02	2.9E-03	1.7E-02	1	5.8E-03	1.0E-02	NA
HexaCDF, 2,3,4,6,7,8-	9.6E-03	2.0E-03	1.2E-02	1	4.1E-03	7.1E-03	NA
PentaCDD, 1,2,3,7,8-	1.8E-02	3.9E-03	2.2E-02	1	7.8E-03	1.4E-02	NA
PentaCDF, 1,2,3,7,8-	2.6E-02	5.5E-03	3.2E-02	1	1.1E-02	1.9E-02	NA
PentaCDF, 2,3,4,7,8-	3.1E-02	6.5E-03	3.8E-02	1	1.3E-02	2.3E-02	NA
TetraCDD, 2,3,7,8-	2.6E-02	5.5E-03	3.2E-02	1	1.1E-02	1.9E-02	NA
TetraCDF, 2,3,7,8-	3.6E-02	7.7E-03	4.4E-02	1	1.5E-02	2.7E-02	NA
POMs							
2-Methylnaphthalene	2.4E-02	5.0E-03	2.9E-02	0.01	1.0E-02	1.7E-02	NA
7,12-Dimethylbenz(a)anthracene	3.9E-02	8.3E-03	4.8E-02	0.01	1.7E-02	2.9E-02	NA
Acenaphthene	2.5E-02	5.2E-03	3.0E-02	0.01	1.0E-02	1.8E-02	NA
Acenaphthylene	2.6E-02	5.5E-03	3.1E-02	0.01	1.1E-02	1.9E-02	NA
Benz(a)anthracene	3.9E-02	8.3E-03	4.8E-02	0.01	1.7E-02	2.9E-02	NA
Benzo(a)pyrene	3.8E-02	8.0E-03	4.6E-02	0.01	1.6E-02	2.8E-02	NA
Benzo(b)fluoranthene	3.9E-02	8.3E-03	4.8E-02	0.01	1.7E-02	2.9E-02	NA
Benzo(g,h,i)perylene	2.9E-02	6.1E-03	3.5E-02	0.01	1.2E-02	2.1E-02	NA

Compound Name	Biotransfer Factors (Ba_m) (day/kg FW tissue)[a] and Metabolism Factors (MF) (unitless)[b]						
	Mammal				Non-mammal		
	Beef (Ba_{beef})	Dairy (Ba_{dairy})	Pork (Ba_{pork})	*MF*	Eggs (Ba_{eggs})	Poultry ($Ba_{poultry}$)	*MF*
Benzo(k)fluoranthene	3.8E-02	8.0E-03	4.6E-02	0.01	1.6E-02	2.8E-02	NA
Chrysene	4.0E-02	8.4E-03	4.8E-02	0.01	1.7E-02	2.9E-02	NA
Dibenz(a,h)anthracene	3.1E-02	6.5E-03	3.8E-02	0.01	1.3E-02	2.3E-02	NA
Fluoranthene	4.0E-02	8.5E-03	4.9E-02	0.01	1.7E-02	3.0E-02	NA
Fluorene	2.9E-02	6.1E-03	3.5E-02	0.01	1.2E-02	2.1E-02	NA
Indeno(1,2,3-cd) pyrene	2.7E-02	5.8E-03	3.3E-02	0.01	1.2E-02	2.0E-02	NA

Source: U.S. EPA 2005a, unless otherwise indicated. NA = not applicable.

[a]As discussed in HHRAP (U.S. EPA 2005a), Appendix A, biotransfer factors for mercury compounds were obtained from U.S. EPA 1997b. Considering speciation, fate, and transport of mercury from emission sources, elemental mercury is assumed to be vapor-phase and hence is assumed not to deposit to soil or transfer into aboveground plant parts. As a consequence, there is no transfer of elemental mercury into animal tissues. Biotransfer factors for cadmium compounds were obtained from U.S. EPA 1995b. Biotransfer factors for dioxins and PAHs were calculated from chemical octanol-water partitioning coefficients (Kow values) using the correlation equation from RTI (2005) and assuming the following fat contents: milk - 4%; beef - 19%; pork - 23%; poultry -14%; and eggs - 8%.

[b]As discussed in HHRAP (U.S. EPA 2005a), U.S. EPA (1995c) recommends using a metabolism factor (MF) to account for metabolism of PAHs by mammals to offset the amount of bioaccumulation suggested by biotransfer factors. EPA has recommended an MF of 0.01 for bis(2-ethylhexyl)phthalate (BEHP) and 1.0 for all other chemicals (U.S. EPA 1995d). An MF of 0.01 is also used to calculate concentrations of PAHs in food products from mammalian species based on the work of Hofelt et al. (2001). This factor takes into account the P450-mediated metabolism of PAHs in mammals; applying this factor in our approach reduced the concentrations of chemicals in beef, pork, and dairy by two orders of magnitude.

Exhibit B-13. Soil and Plant Ingestion Rates for Animals

Animal	Soil Ingestion Rate – $Qs_{(m)}$ (kg/day)[a]	Plant Part Consumed by Animal	Plant Ingestion Rate – $Qp_{(l,m)}$ (kg/day)
Beef cattle[b]	0.5	Silage	2.5
		Forage	8.8
		Grain	0.47
Dairy cattle[c]	0.4	Silage	4.1
		Forage	13.2
		Grain	3.0
Swine[d]	0.37	Silage	1.4
		Grain	3.3
Chicken (eggs)[e]	0.022	Grain	0.2

Source: U.S. EPA 2005a HHRAP (Chapter 5).

[a]Beef cattle: NC DEHNR (1997) and U.S. EPA (1994b) recommended a soil ingestion rate for subsistence beef cattle of 0.5 kg/day based on Fries (1994) and NAS (1987). As discussed in HHRAP, Fries (1994) reported soil ingestion to be 4 percent of the total dry matter intake. NAS (1987) cited an average beef cattle weight of 590 kg, and a daily dry matter intake rate (non-lactating cows) of 2 percent of body weight. This results in a dry matter intake rate of 11.8 kg DW/day and a daily soil ingestion rate of about 0.5 kg/day.

Dairy cattle: NC DEHNR (1997) and U.S. EPA (1994b) recommended a soil ingestion rate for dairy cattle of 0.4 kg/day based on Fries (1994) and NAS (1987). As discussed in HHRAP, Fries (1994) reported soil ingestion to be 2 percent of the total dry matter intake. NAS (1987) cited an average beef cattle weight of 630 kg and a daily dry matter intake rate (non-lactating 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. Uncertainties associated with Qs include the lack of current empirical data to support soil ingestion rates for dairy cattle and the assumption of uniform contamination of soil ingested by cattle.

Swine: NC DEHNR (1997) recommended a soil ingestion rate for swine of 0.37, estimated by assuming a soil intake that is 8% of the plant ingestion rate of 4.3 kg DW/day. Uncertainties include the lack of current empirical data to support soil ingestion rates and the assumption of uniform contamination of the soil ingested by swine.

Chicken: HHRAP (U.S. EPA 2005a) assumes that chickens consume 10 percent of their total diet (which is approximately 0.2 kg/day grain) as soil, a percentage that is consistent with the study from Stephens *et al*. (1995). Uncertainties 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.

[b]The beef cattle ingestion rates of forage, silage, and grain are based on the total daily intake rate of about 12 kg DW/day (based on NAS [1987] reporting a daily dry matter intake that is 2 percent of an average beef cattle body weight of 590 kg) and are supported by NC DEHNR (1997), U.S. EPA (1994b and 1990), and Boone *et al*. (1981). The principal uncertainty associated with these *Qp* values is the variability between forage, silage, and grain ingestion rates for cattle.

[c]The dairy cattle ingestion rates of forage, silage, and grain are based on the total daily intake rate of about 20 kg DW/day (NAS 1987; U.S. EPA 1992) as recommended by NC DEHNR (1997). Uncertainties include the proportion of each food type in the diet, which varies from location to location. Assuming uniform contamination of plant materials consumed by cattle also introduces uncertainty.

[d]Swine are not grazing animals and are assumed not to eat forage (U.S. EPA 1998). U.S. EPA (1994b and 1998) and NC DEHNR (1997) recommended including only silage and grains in the diet of swine. EPA (1995c) recommended an ingestion rate of 4.7 kg DW/day for a swine, referencing NAS (1987). Assuming a diet of 70 percent grain and 30 percent silage (U.S. EPA 1990), HHRAP estimated ingestion rates of 3.3 kg DW/day (grain) and 1.4 kg DW/day (silage). Uncertainties associated with *Qp* include variability of the proportion of grain and silage in the diet, which varies from location to location.

[e]Chickens consume grain provided by the farmer. The daily quantity of grain feed consumed by chicken is assumed to be 0.2 kg/day (Ensminger (1980), Fries (1982), and NAS (1987). Uncertainties associated with this variable include the variability of actual grain ingestion rates from site to site. In addition, assuming uniform contamination of plant materials consumed by chicken introduces some uncertainty.

B.6.3 Adult and Non-Infant Exposure Parameter Values

The exposure parameters included in the multimedia ingestion risk methodology and their default values are summarized in the following subsections. EPA selected the default values to result in a highly health protective screening scenario. Also presented are alternatives to the default values (e.g., typically based on other percentiles from the distribution), which may be appropriate on a case-by-case basis. These parameter value options were primarily obtained or estimated from EPA's *Exposure Factors Handbook* (EFH; U.S. EPA 2011a) and *Child-Specific Exposure Factors Handbook* (CSEFH; U.S. EPA 2008a). Where values were reported for age groupings other than those used in the methodology (see Section B.2.2), time-weighted average values were estimated for the methodology's age groups from the available data.

Ingestion rates for home-produced farm food items were identified for exposed fruit, protected fruit, exposed vegetables, protected vegetables, root vegetables, beef, total dairy, pork, poultry, and eggs. Those ingestion rates are already normalized to body weight (i.e., $g_{wet\ weight}$/kg-day) (U.S. EPA 2011a). The body weight parameter values presented in Exhibit B-14, therefore, are not applied in the chemical intake (ADD) equations for these food types.

Ingestion rates also are identified for drinking water (mL/day), soil (mg/day), and fish (g/day). These ingestion rates, however, are on a per person basis (i.e., not normalized for body weight). The body weight parameter values presented in Exhibit B-14, therefore, are applied in the chemical intake (ADD) equations for these media.

B.6.3.1 Body Weights

Body weight (BW) options include mean, 5th, 10th, 50th, 90th, and 95th percentile values for adults and the five children's age groups: <1 year; 1–2 years; 3–5 years; 6–11 years; and 12–19 years. For its default screen, EPA uses the mean BW for each age group. The BW values are listed in Exhibit B-14.

In general, BW values for the five children's age groups were calculated from the summary data provided in Table 8-3 of EPA's 2008 CSEFH. For purposes of comparison, alternative BW values for children ages 12 through 19 years also were estimated using data from Portier *et al*. (2007). These values are listed in the last row of Exhibit B-14, but are not used in RTR screening assessments. The means calculated using the two methods for children ages 12 through 19 years were essentially identical at 64 kg. The other percentile values for this age group using the two methods differed by approximately 10 percent or less.

Exhibit B-14. Mean and Percentile Body Weight Estimates for Adults and Children

Lifestage (years)	Duration (years)	Body Weight (kg)					
		Mean	5th	10th	50th	90th	95th
Adult[a] (20-70)	50	80.0[a]	52.9	56	69.3	89.7	97.6
Child <1[b]	1	7.83	6.03	6.38	7.76	9.24	9.66
Child 1-2[c]	2	12.6	9.9	10.4	12.5	14.9	15.6
Child 3-5[d]	3	18.6	13.5	14.4	17.8	23.6	26.2
Child 6-11[e]	6	36.0	22.1	24.0	33.5	51.2	58.6
Child 12-19[f]	8	64.2	39.5	45	64.2	83.5	89
[Child 12-19[g]	*8*	*64.3*	*41.1*	*44.6*	*60.9*	*88.5*	*98.4]*

[a]BW represents the recommended body weight from EPAs 2011 EFH. Although the 18 to 74 year age category in EPA's EFH does not match exactly the age 20 to 70 year categorization of adults for the methodology, the magnitude of error in the mean and percentile body weights is likely to be very small (i.e., less than 1%).
[b]Each BW represents a time-weighted average of body weights for age groups birth to <1 month, 1 to <3 months, 3 to <6 months, and 6 to <12 months from Table 8-3 of the 2008 CSEFH. Original sample sizes for each of these age groups can also be found in Table 8-3.
[c]Each BW represents a time-weighted average of body weights for age groups 1 to <2 years and 2 to <3 years from Table 8-3 of the 2008 CSEFH. Original sample sizes for each of these age groups can also be found in Table 8-3.
[d]BWs obtained directly from Table 8-3 of the 2008 CSEFH (age group 3 to <6 years).
[e]Each BW represents a time-weighted average of body weights for age groups 6 to <11 years and 11 to <16 years from Table 8-3 of the 2008 CSEFH. Original sample sizes for each of these age groups can also be found in Table 8-3.
[f]Mean BW estimated using Table 8-22 of the 2008 CSEFH, which is based on NHANES IV data as presented in Portier et al. (2007). This estimate was calculated as the average of the 8 single-year age groups from 12 to 13 years through 19 to 20 years. Values for the other percentiles were estimated using Portier et al., 2007.
[g]Each BW represents a time-weighted average of body weights for age groups 11 to <16 years and 16 to <21 years from Table 8-3 of the 2008 CSEFH. Estimated values include 11-year-olds and individuals through age 20, which contributes to uncertainty in the estimates for 12 to 19 years. Those values are provided for comparison purposes only and are not used in RTR assessments.

B.6.3.2 Water Ingestion Rates

The methodology can be used to calculate chemical ingestion via drinking water obtained from surface-water sources or from wells (i.e., from groundwater) in the contaminated area. The 2008 CSEFH recommends values for drinking water ingestion rates for children are based on a study reported by Kahn and Stralka (2008). Table 3-4 of the CSEFH provides per capita estimates of community water ingestion rates by age categories. Community water ingestion includes both direct and indirect ingestion of water from the tap. *Direct ingestion* is defined as direct consumption of water as a beverage, while *indirect ingestion* includes water added during food or beverage preparation. The source of these data is the 1994–1996 and 1998 U.S. Department of Agriculture's (USDA's) *Continuing Survey of Food Intakes by Individuals* (CSFII) (USDA 2000). Exhibit B-15 includes the recommended median and other drinking water ingestion rates for children.

Mean and percentile adult drinking water ingestion rates were obtained from EPA (2004a), which presents estimated per capita water ingestion rates for various age categories based on data collected by the USDA's 1994–1996 and 1998 CSFII (USDA 2000). Adult ingestion rates, presented in Exhibit B-15, represent community water ingestion, both direct and indirect as defined above, for males and females combined, ages 20 years and older.

Exhibit B-15. Estimated Daily Per Capita Mean and Percentile Water Ingestion Rates for Children and Adults[a]

Lifestage (years)	Ingestion Rates, Community Water (mL/day)				
	Mean	50th	90th	95th	99th
Child <1[b]	504	482	969	1113	1440
Child 1-2[c]	332	255	687	903	1318
Child 3-5[d]	382	316	778	999	1592
Child 6-11[e]	532	417	1149	1499	2274
Child 12-19[f]	698	473	1641	2163	3467
Adult[g]	1219	981	2534	3087	4567

Sources: U.S. EPA 2004a, 2008a

*The sample size does not meet minimum reporting requirements as described in U.S. EPA 2008a. For some of the age groupings presented, the values are based on the time-weighted average value for 2 or more age ranges from CSEFH Table 3-4. One or more age ranges within the group may not meet the minimum reporting requirements, but not necessarily all of them fall within this category.

[a]Source is Kahn and Stralka 2008, also presented in the CSEFH (U.S. EPA 2008a).

[b]Each IR represents a time-weighted average of ingestion rates for age groups birth to <1 month, 1 to <3 months, 3 to <6 months, and 6 to <12 months from Table 3-4 of the 2008 CSEFH.

[c]Each IR represents a time-weighted average of ingestion rates for age groups 1 to <2 years and 2 to <3 years from Table 3-4 of the 2008 CSEFH.

[d]Each IR represents the ingestion rate for age group 3 to <6 years from Table 3-4 of the 2008 CSEFH.

[e]Each IR represents the ingestion rate for age group 6 to <11 years from Table 3-4 of the 2008 CSEFH. This value represents a health protective (i.e., slightly low) estimate of IR for ages 6 through 11 years since 11-year olds are not included in this CSEFH age group.

[f]Each IR represents a time-weighted average of ingestion rates for age groups 11 to <16 years, 16 to <18, and 18 to <21 years from Table 3-4 of the 2008 CSEFH. Note that estimated values include 11-year-olds and individuals through age 20, which contributes to uncertainty in the estimates for 12 to 19 years.

[g]Adult drinking water ingestion rates were obtained from EPA (2004b), Appendix E, Part I, Table A1 for community water, both sexes (ages 20+), direct plus indirect water ingestion.

B.6.3.3 Local Food Ingestion Rates

Exhibit B-16 presents mean, median, 90th, 95th, and 99th percentile food-specific ingestion rates (IRs) for consumers-only of farm food chain (FFC) media for adults and children. The mean and percentile values are from EPA's analysis of data from the USDA's 1987 to 1988 *Nationwide Food Consumption Survey* (NFCS) (USDA 1993), as presented in Chapter 13 of the Agency's *Exposure Factors Handbook* (i.e., Intake Rates for Various Home Produced Food Items) (U.S. EPA 2011a). Consumers-only means that individuals who did not report eating a specified type of food during the three-day period covered by the food ingestion part of the survey were not included in the analysis of ingestion rates for that food type. The questionnaire included the options for a household to self-identify in one or more of five categories: as a household that gardens, raises animals, hunts, fishes, or farms. As of September 2008, that survey was the most recent NFCS available (U.S. EPA 2008a, CSEFH), and we are not aware of any that might be more recent.[24]

For the adult age group, data were compiled on food-specific IRs separately for two types of households as indicated in the "Response to Questionnaire" (U.S. EPA 2011a, Chapter 13): (1) households that farm (F) and (2) households that garden or raise animals (HG for homegrown). This division reflects EPA's data analysis. EPA tabulated IRs for fruits and vegetables

[24]Note that EPA's 2008 CSEFH does not distinguish between exposed and protected fruits and vegetables when recommending food ingestion rates based on the same data set for the same age categories. EPA's 1997 analysis for its EFH therefore remains the most appropriate data source for use with this methodology.

separately for households that farm and households that garden. Similarly, EPA tabulated IRs for animals and animal products for households that farm and for households that raise animals. Thus, the first type of household, F, represents farmers who may both grow crops and raise animals and who are likely to consume more homegrown/raised foods than the second type of household. The second type of household, HG, represents the non-farming households that may consume lower amounts of home grown or raised foods (i.e., HG encompasses both households that garden and households that raise animals).

The food-specific ingestion rates are based on the amount of each food type that households that farm (F) or households that garden and raise animals (HG) produced and brought into their homes for consumption and the number of persons consuming the food. EPA averaged the actual consumption rate for home grown foods over the 1-week survey period.

Exhibit B-16. Summary of Age-Group Specific Food Ingestion Rates for Farm Food Items

Product	Child (age in yr)					Adult (20–70 yrs)
	<1	1–2	3–5	6–11	12–19	
Mean ingestion rates (g/kg-day)						
Beef[a]	N/A	4.14	4.00	3.77	1.72	1.93
Dairy[b]	N/A	91.6	50.9	27.4	13.6	2.96
Eggs[a]	N/A	2.46	1.42	0.86	0.588	0.606
Exposed Fruit[a]	N/A	6.14	2.60	2.52	1.33	1.19
Exposed Vegetable[a]	N/A	3.48	1.74	1.39	1.07	1.38
Pork[a]	N/A	2.23	2.15	1.50	1.28	1.10
Poultry[a]	N/A	3.57	3.35	2.14	1.50	1.37
Protected Fruit[a]	N/A	16.6	12.4	8.50	2.96	5.19
Protected Vegetable[a]	N/A	2.46	1.30	1.10	0.78	0.862
Root Vegetable[a]	N/A	2.52	1.28	1.32	0.94	1.03
Water (mL/day)[c]	N/A	332	382	532	698	1218
Median ingestion rates (g/kg-day)						
Beef[a]	N/A	2.51	2.49	2.11	1.51	1.55
Dairy[b]	N/A	125	66.0	34.4	15.5	2.58
Eggs[a]	N/A	1.51	0.83	0.561	0.435	0.474
Exposed Fruit[a]	N/A	N/A	1.82	1.11	0.61	0.68
Exposed Vegetable[a]	N/A	1.89	1.16	0.64	0.66	0.812
Pork[a]	N/A	1.80	1.49	1.04	0.89	0.802
Poultry[a]	N/A	3.01	2.90	1.48	1.30	0.922
Protected Fruit[a]	N/A	7.59	5.94	3.63	1.23	2.08
Protected Vegetable[a]	N/A	1.94	1.04	0.79	0.58	0.564
Root Vegetable[a]	N/A	0.92	0.46	0.52	0.57	0.63
Water (mL/day)[c]	N/A	255	316	417	473	981

Product	Child (age in yr)					Adult (20–70 yrs)
	<1	1–2	3–5	6–11	12–19	
90th percentile ingestion rates (g/kg-day)[d]						
Beef[a]	N/A	9.49	8.83	11.4	3.53	4.41
Dairy[b]	N/A	185	92.5	57.4	30.9	6.16
Eggs[a]	N/A	4.90	3.06	1.90	1.30	1.31
Exposed Fruit[a]	N/A	12.7	5.41	6.98	3.41	2.37
Exposed Vegetable[a]	N/A	10.7	3.47	3.22	2.35	3.09
Pork[a]	N/A	4.90	4.83	3.72	3.69	2.23
Poultry[a]	N/A	7.17	6.52	4.51	3.13	2.69
Protected Fruit[a]	N/A	44.8	32.0	23.3	7.44	15.1
Protected Vegetable[a]	N/A	3.88	2.51	2.14	1.85	1.81
Root Vegetable[a]	N/A	7.25	4.26	3.83	2.26	2.49
Water (mL/day)[c]	N/A	687	778	1149	1640	2534
95th percentile ingestion rates (g/kg-day)						
Beef[a]	N/A	12.9	12.5	12.5	3.57	5.83
Dairy[b]	N/A	167	89.9	56.0	32.3	7.80
Eggs[a]	N/A	5.38	3.62	2.37	1.43	1.59
Exposed Fruit[a]	N/A	14.6	6.07	11.7	4.78	3.38
Exposed Vegetable[a]	N/A	11.9	6.29	5.47	3.78	4.46
Pork[a]	N/A	6.52	6.12	4.73	6.39	2.60
Poultry[a]	N/A	8.10	7.06	5.07	3.51	3.93
Protected Fruit[a]	N/A	48.3	35.1	26.9	11.4	19.2
Protected Vegetable[a]	N/A	9.42	5.10	3.12	2.20	2.83
Root Vegetable[a]	N/A	10.4	4.73	5.59	3.32	3.37
Water (mL/day)[c]	N/A	903	999	1499	2163	3087
99th percentile ingestion rates (g/kg-day)						
Beef[a]	N/A	20.9	19.8	13.3	4.28	6.84
Dairy[b]	N/A	180	87.2	54.8	34.7	9.20
Eggs[a]	N/A	16.2	11.2	8.19	4.77	1.83
Exposed Fruit[a]	N/A	25.2	32.5	15.7	5.9	13.0
Exposed Vegetable[a]	N/A	12.1	7.36	13.3	5.67	8.42
Pork[a]	N/A	8.71	9.74	6.61	4.29	3.87
Poultry[a]	N/A	9.63	10.24	6.12	4.60	4.93
Protected Fruit[a]	N/A	109	71.2	58.2	19.1	34.4
Protected Vegetable[a]	N/A	9.42	5.31	5.40	2.69	5.56

Product	Child (age in yr)					Adult (20–70 yrs)
	<1	1–2	3–5	6–11	12–19	
Root Vegetable[a]	N/A	10.4	4.73	7.47	5.13	7.57
Water (mL/day)[c]	N/A	1318	1592	2274	3467	4567

[a]Primary source for values was the 1987–1988 NFCS survey; compiled results are presented in Chapter 13 of 2011 Exposure Factors Handbook (U.S. EPA 2011a). When data were unavailable for a particular age group, intake rate for all age groups was used multiplied by the age-specific ratio of intake based on national population intake rates from CSFII.

[b]Primary source for values was the 1987–1988 NFCS survey; compiled results are presented in Chapter 13 of 2011 Exposure Factors Handbook (U.S. EPA 2011a). When data were unavailable for a particular age group, intake rate for all age groups was used multiplied by the age-specific ratio of intake based on national population intake rates from an NHANES 2003–2006 analysis in Chapter 11 of the Exposure Factors Handbook.

[c]Primary source for children less than 3 years of age was a Kahn and Stralka (2008) analysis of CSFII data, and from EPA's analysis of NHANES 2003–2006 data for children and adults greater than three. All data tables that were used and justifications for data sources are presented in Chapter 3 of the 2011 Exposure Factors' Handbook.

[d]Default ingestion rate percentile for Tier 1 assessments and chemical threshold calculations.

For children, EPA estimated food-specific IRs for four age categories (U.S. EPA 2011a): 1–2 years, 3–5 years, 6–11 years, and 12–19 years. Sample sizes were insufficient to distinguish IRs for children in different types of households; hence, for children, a single IR value represents both F and HG households for a given food type and age category (Exhibit B-16). For some food types and age categories, there were insufficient data for EPA to provide consumer-only intake rates (i.e., data set for the subpopulation consisted of fewer than 20 observations). The HHRAP methodology, Section 6.2.2.2, recommends a method by which to calculate the "missing" age-specific consumer-only ingestion rates, as explained below. Food-specific intake rates (IRs) for those child age groups and food items not included in Chapter 13 of the 2011 EFH, that is $IR_{age\_group\_x}$, were derived using the following information:

- Mean or percentile-specific consumer-only intake of the farm food item, as brought into the home, for the total NFCS survey population (from EFH Chapter 13) – IRCO_total;
- Mean or percentile-specific per capita intake of the food type from all sources, as consumed, for the specific child age group, from Chapter 3 of the CSFII Analysis of Food Intake Distributions (U.S. EPA 2003c) – IRPC, age_group_x; and
- Mean or percentile-specific per capita intake of the farm food item for the total CSFII survey population (from Chapter 3 of U.S. EPA 2003c) – IRPC_total.

The ratio of $IR_{PC,\ age\_group\_x}$ to IR_{PC_total} from the CSFII data shows the consumption rate of a particular food type by a specific age group relative to the consumption rate for that food type for the population as a whole. The ratio of $IR_{CO,\ age\_group\_x}$ to IR_{CO_total}, that is the consumption rate of a particular food type by a specific age group (consumers only) relative to the consumption rate for that food type for the NFCS survey population as a whole (consumers only), should be approximately the same. Given the assumption that the two ratios are equal, Equation B-58 was used to calculate the "missing" age-specific consumer-only *IRs*:

Equation B-58. Calculation of Age-Group-Specific and Food-Specific Ingestion Rates

$$IR_{CO,\ age\_group\_x} = \frac{IR_{CO\_total} \times IR_{PC,\ age\_group\_x}}{IR_{PC\_total}}$$

where:

$IR_{CO,\ age\_group\_x}$ = Mean or percentile-specific *consumer-only* intake of the food type from all sources, as consumed, for the specific child age group X

IR_{CO_total} = Mean or percentile-specific *consumer-only* intake of the farm food item, as brought into the home, for the total NFCS survey population

$IR_{PC,\ age\_group\_x}$ = Mean or percentile-specific *per capita* intake of the food type from all sources, as consumed, for the specific child age group X from the CSFII

IR_{PC_total} = Mean or percentile-specific per capita intake of the farm food item for the total CSFII survey population

In this discussion, *per capita* (as opposed to *consumer-only*) indicates the intake rates are based on the entire population rather than the subset of the population that ingests the particular food category (i.e., consumers). Here, the use of per capita ingestion rates is recommended by the HHRAP methodology because no consumer-only percentile-specific intakes are provided for the different age groups.

The above calculation implicitly assumes that the distribution of the consumption rate for a food type for a specific age group (consumers only) has the same shape as the distribution of the consumption rate for a food type for a specific age group in the general population (per capita). Otherwise, the separate calculation of each percentile might yield intake estimates that decrease as the percentile increases. This calculation artifact could occur if the shapes of the two distributions differ in the upper percentiles (or "tails") of the distributions.

In the instances where the above calculations were used to fill data gaps in the above exhibit, only the dairy child-specific age group intake estimates are not strictly increasing with increasing percentile. The distributions likely track better (and thus the above assumption of equal ratios is more reasonable) for lower percentiles, with deviations occurring due to outlier ingestion rates based on only a few respondents in the tails of the distributions. The default ingestion rates for this methodology are the 90th percentiles, which are likely more reliable than the 95th or 99th percentile estimates in this particular calculation.

B.6.3.4 Local Fish Ingestion Rates

Screening Scenario

The USDA's 1987 to 1988 Nationwide Food Consumption Survey (NFCS) (USDA 1993, 1994a), as presented in Chapter 13 of the Agency's Exposure Factors Handbook (i.e., Intake Rates for Various Home Produced Food Items) (U.S. EPA 2011a), includes family-caught fish ingestion rates by age category. There are several disadvantages, however, to using that data source to estimate fish ingestion rates. First, due to inadequate sample sizes, EPA did not report fish ingestion rates for children less than 6 years of age. Second, the NFCS data were collected more than two decades ago. Third, the reported fish ingestion rates are for ages 6 to 11 and 12 to 19 and are based on 29 and 21 individuals in each age category, respectively (U.S. EPA 2011a, Table 13-20). Finally, the ingestion rates from NFCS data are based on total weight of fish as brought into the home, and do not include losses from preparation of the fish (i.e., removal of inedible parts and, possibly, the skin). Estimates of preparation losses for fish intended to apply to the NFCS fish ingestion rate data are very uncertain and are based on a wide variety of freshwater, estuarine, and marine fish, and squid (U.S. EPA 2011a, Table 13-69). Additionally, when considering the multipathway screening methodology, it is important that potential health effects to those individuals who are most likely to have the greatest PB-HAP

exposure are not underestimated and, therefore, ingestion rates that are reflective of subsistence fisher ingestion rates are desired. Therefore, a more recent survey was sought that included larger sample sizes, data for children younger than six years, ingestion rates for the parts of fish actually consumed, and ingestion rates reflective of subsistence fisher ingestion rates.

Taking all of these issues into consideration, the default adult fish ingestion rate selected for use is 373 g/day, which is the estimated 99th percentile of fish ingestion rates for woman fishers as reported by Burger (2002). This rate is based upon ingestion of "wild-caught" fish, which includes freshwater, estuarine, and marine species, while our screening scenarios focus only on freshwater fish from lakes. This is notable because a number of studies indicate that fish ingestion rates are limited by species and habitat (i.e., lake, river, estuary, and ocean) and that the majority of the fish consumed in the United States are from river, marine and estuarine habitats versus lakes. Thus, although the fish ingestion rate for this group of subsistence fishers is not the highest fish ingestion rate available for use by EPA, it strikes the appropriate balance between being health protective and having screening scenarios so conservative that they are of limited use in the decision making process. This high-end fish ingestion rate is appropriate in the context of the conservative screening scenario used in the RTR process. This methodology is particularly applicable for national rulemakings given that it is very likely that subsistence woman fishers of child bearing age are located throughout the United States. Finally, using a high-end subsistence fish ingestion rate is consistent with section 112 of the CAA, which focuses on risks associated with maximally exposed individuals.

Because Burger (2002) did not estimate fish ingestion rates for children, another data source was needed to develop ingestion rates for the child age categories. The child ingestion rates need to be consistent with the Burger adult ingestion rate, reflective of subsistence fisher ingestion rates, and based on adequate sample sizes. To satisfy these requirements, data on child ingestion rates from EPA's *Estimated Per Capita Fish Consumption in the United States* (U.S. EPA 2002) were selected for use. Specifically, the estimated 99th percentile of *as-prepared*, *consumer-only* ingestion rates for finfish and shellfish were selected (see Section 4.2.1.1 Table 5 of U.S. EPA 2002).The original data were collected as part of the 1994–96 and 1998 USDA Continuing Survey of Intakes by Individuals (CSFII; USDA 2000). Values reflect "as prepared" ingestion rates for which cooking and preparation losses (*L1* and *L2*) did not need to be considered. "Total" fish as reported in this source represents consumption of finfish plus shellfish.

Because the child age categories used in the methodology differ from the CSFII age categories presented in U.S. EPA 2002, the CSFII data were adjusted. The CSFII data did not provide ingestion rates for the 1–2 year age category. To estimate ingestion rates for this age group, EPA used the ingestion rate for the 3–5 year age category scaled downward by the ratio of the mean body weight of the 1–2 year age group to the mean body weight of the 3–5 year age group. Because the methodology uses a 3–5 year age category, no adjustment was needed for CSFII data from that age category. For the 6–11 and 12–19 age categories, time-weighted average ingestion rates were calculated based on the CSFII ingestion rates. Exhibit B-17 provides the fish ingestion rates used in the screen.

Exhibit B-17. Fish Ingestion Rates Used in Screen

Ingestion Rates (g/day)					
Infants <1 yr	**Child 1–2 yrs**	**Child 3–5 yrs**	**Child 6–11 yrs**	**Child 12–19 yrs**	**Adult 20–70 yrs**
NA	107.7[a]	159.0[b]	268.2[c]	331.0[c]	373[d]

[a]A fish ingestion rate for ages 1-2 years was not available. The value represents the consumer-only fish ingestion rate for ages 3 to 5 from U.S. EPA (2002) (Section 4.2.1.1 Table 5 [freshwater/estuarine habitat]), scaled down by the ratio of the mean Child 1-2 body weight to the mean Child 3-5 body weight.
[b]This value represents the consumer-only fish ingestion rate for ages 3 to 5 from U.S. EPA (2002), Section 4.2.1.1 Table 5 (freshwater/estuarine habitat) rounded to the nearest full number.
[c]These values represent time-weighted average consumer-only fish ingestion rates based on ingestion rates from U.S. EPA (2002), Section 4.2.1.1 Table 5 (freshwater/estuarine habitat).
[d]This value represents the 99th percentile ingestion rate of wild caught fish for women as reported by Burger (2002).

Alternative Values

EPA's (2002) analysis of freshwater and estuarine fish consumption from the USDA's Continuing Survey of Food Intake by Individuals (CSFII) for 1994-96 and 1998 was chosen to provide fish ingestion rate options by age category for the methodology. Although the fish consumption rates reported in the CSFII include all sources, commercial and self-caught, for purposes of screening level risk assessments, it was assumed that all freshwater and estuarine fish consumed are self-caught. The inclusion of commercially obtained and estuarine fish will overestimate locally caught freshwater fish ingestion rates for most populations in the United States; however, it also might underestimate locally caught fish ingestion rates for some populations (e.g., Native Americans, Asian and Pacific Island communities, rural African American communities). Because consumption of locally caught fish varies substantially from region to region in the United States and from one population or ethnic group to the next, assessors are encouraged to use more locally relevant data whenever available.

For children, EPA identified values for the mean and the 90th, 95th, and 99th percentile fish *per-capita* ingestion rates (freshwater and estuarine fish only) based on EPA's analysis of 1994-96 and 1998 CSFII data (U.S. EPA 2002, 2008a). Those rates include individuals who eat fish and those who do not eat fish. As shown in EPA's 2008 CSEFH, Table 10-7, the 90th percentile per capita ingestion rates estimated from the two-day CSFII recall period are zero for some child age groups. Although not presented in CSEFH Table 10-7, median ingestion rates for all child age groups would be zero (considering the "consumer only" sample sizes [CSEFH Table 10-9] relative to the "*per-capita*" sample sizes in Table 10-7).

The high percentile fish ingestion rates that are zero result from the short duration of the CSFII recall period (two days) compared with the averaging time of interest (a year) and the relatively infrequent consumption of fish (e.g., on the order of once a week to once a month or less) compared with the near daily ingestion of other types of food products (e.g., dairy, produce, meat). Use of zero for fish ingestion rates, however, is not useful. As a result, an alternative method was used to estimate fish ingestion rates for children and adults that could provide reasonable, non-zero values for all age groups and percentiles.

The alternative, age-group-specific fish ingestion rates were derived using values for each age group, *y*:

- Mean or other appropriate percentile consumer-only fish ingestion rates for age group *y*, IRCO,*y*, from EPA's Estimated Per Capita Fish Consumption in the United States (U.S. EPA 2002), Section 5.2.1.1, Table 5, for freshwater/estuarine habitat.[25]
- Fraction of the population consuming freshwater/estuarine fish, FPC,*y*, calculated as consumer-only sample size / U.S. population sample for age group *y*. The data to calculate those fractions are available in the 2008 CSEFH and U.S. EPA 2002.

Equation B-59 was used to calculate the alternative, per capita fish ingestion rates by age group ($IR_{PC,y}$):

Equation B-59. Calculation of Alternative Age-Group-Specific Fish Ingestion Rates

$$IR_{PC,y} = IR_{CO,y} \times F_{PC,y}$$

where:

$IR_{PC,y}$ = *Per capita* fish ingestion rate for age group *y* (g/day)

$IR_{CO,y}$ = *Consumer-only* fish ingestion rates for age group *y* (g/day) (U.S. EPA 2002, Section 5.2.1.1, Table 5, for freshwater/estuarine habitat)

$F_{PC,y}$ = Fraction of the population consuming freshwater/estuarine fish, calculated as consumer-only sample size / total U.S. population sample size for age group *y* (unitless) (2008 CSEFH, U.S. EPA 2002)

In the above, *per capita* (as opposed to *consumer-only*) indicates the intake rates are based on the entire population rather than the subset of the population that ingests the particular food category. Here, per capita ingestions are recommended by the HHRAP methodology because no consumer-only percentile-specific intakes are provided for the different age groups.

The mean and percentile consumer-only fish ingestion rates for children and adults and the fraction of the population consuming freshwater/estuarine fish used in calculating long-term per capita fish ingestion rates by age group are presented in Exhibit B-18 and Exhibit B-19. The mean and percentile per capita fish ingestion rates estimated using the methodology are summarized in Exhibit B-20. The fish ingestion rates provided in Exhibit B-20 are intended to represent the harvest and consumption of fish in surface waters in a hypothetical depositional area. For site-specific assessments, more localized survey data may be more appropriate to estimate fish ingestion rates. The fishing season varies substantially across the United States by latitude, and fish consumption patterns also vary by type of water body (e.g., ponds, lakes, rivers, streams, estuaries, coastal marine), cultural heritage, and general geographic area. Therefore, use of more localized information is encouraged. Note that as indicated in Exhibit B-17, in developing the screening threshold emission rates, health protective fish ingestion rates for child and adult fish consumers that more closely represent exposures of a high-end recreational fisher were used.

As noted in Section B.6.4.3, if the fish IRs shown in Exhibit B-20 are replaced with fresh-weight as caught values (e.g., values obtained from a local creel survey), the assessor is advised to set

[25]Most of these data also are provided in Table 10-9 of the CSEFH; the median values, however, are not presented in the CSEFH, and values for the mean and all other percentiles are slightly different due to rounding.

non-zero values for the preparation and cooking loss factors L1 and L2 in Equation B-15. Suggested values are presented in Section B.6.4.3.

Exhibit B-18. Daily Mean and Percentile *Consumer-Only* Fish Ingestion Rates for Children and Adults ($IR_{CO,y}$)[a]

Lifestage (years)	Ingestion Rates, All Fish (g/day)				
	Mean	50th	90th	95th	99th
Child <1	NA	NA	NA	NA	NA
Child 1-2[b]	27.31	15.61	64.46	87.60	138.76*
Child 3-5[c]	40.31	23.04	95.16	129.31	204.84*
Child 6-11[d]	61.49	28.46	156.86*	247.69*	385.64*
Child 12-19[e]	79.07	43.18	181.40*	211.15*	423.38*
Adult[f]	81.08	47.39	199.62*	278.91	505.65*

Sources: U.S. EPA 2002, 2008a

NA = not applicable; it is assumed that children <1 year of age do not consume fish.

*Indicates that the sample size does not meet minimum reporting requirements as described in U.S. EPA 2002. Owing to the small sample sizes, these upper percentiles values are highly uncertain.

[a]Per capita fish ingestion (FI) rates for children by age group are available from Chapter 10 of the CSEFH (U.S. EPA 2008a); however, all 50th and some 90th percentile ingestion rates are zero. Per capita FI rates were therefore estimated as described in Equation B-59 to provide reasonable, non-zero values for all age groups and percentiles.

[b]A fish IR for ages 1-2 years was not available. The value represents the consumer-only fish ingestion rate for ages 3 to 5 from U.S. EPA (2002) (Section 5.2.1.1 Table 5 [freshwater/estuarine habitat]), scaled down by the ratio of the mean Child 1-2 body weight to the mean Child 3-5 body weight.

[c]These values represent the consumer-only fish ingestion rate for ages 3 to 5 from U.S. EPA (2002), Section 5.2.1.1 Table 5 (freshwater/estuarine habitat). Sample size = 442.

[d]These values represent the consumer-only fish ingestion rate for ages 6 to 10 from U.S. EPA (2002), Section 5.2.1.1 Table 5 (freshwater/estuarine habitat). Sample size = 147.

[e]These values represent the time-weighted average per capita fish ingestion rate for ages 11 to 15 and 16 to 17 years from U.S. EPA (2002), Section 5.1.1.1 Table 5 (freshwater/estuarine habitat); the value may underestimate ingestion rate for ages 12 to 19 years. Sample size = 135.

[f]These values represent the consumer-only fish ingestion rate for individuals 18 years and older from U.S. EPA (2002), Section 5.2.1.1 Table 4 (freshwater/estuarine habitat). Sample size = 1,633.

Exhibit B-19. Fraction of Population Consuming Freshwater/Estuarine Fish on a Single Day ($F_{PC,y}$)

Lifestage (years)	Fraction Consuming Fish
Child 3-5	0.0503[a]
Child 6-11	0.0440[b]
Child 12-19	0.0493[c]
Adult	0.08509[d]

Sources: U.S. EPA 2002, 2008a

[a]This value was calculated using the ages 3 to 5 sample size for consumers only divided by the sample size for the U.S. population divided by 2 to represent the proportion consuming fish on a single day (the consumers-only group includes individuals who consumed fish on at least one of two survey days) to match the one-day ingestion rate.

[b]As in footnote a, the value was calculated using the ages 6 to 10 sample size for consumers only divided by the sample size for U.S. population divided by 2.

[c]The value was calculated by summing the ages 11 to 15 and 16 to 17 sample sizes for consumers only and dividing by both by the sum of the sample sizes for U.S. population and by a factor of 2.

[d]The value was calculated using the ages 18 and older sample size for consumers only divided by the sample size for U.S. population from Section 5.1.1.1 Table 4. The result was divided by 2 to represent a one-day sampling period in order to match the one-day ingestion rate.

Exhibit B-20. Calculated Long-term Mean and Percentile Per Capita Fish Ingestion Rates for Children and Adults ($IR_{PC,y}$)

Lifestage (years)	Ingestion Rates, All Fish (g/day)				
	Mean	50th	90th	95th	99th
Child <1	NA	NA	NA	NA	NA
Child 1-2[a]	1.37	0.79	3.24	4.41	6.98
Child 3-5[b]	2.03	1.16	4.79	6.51	10.3
Child 6-11[c]	2.71	1.25	6.90	10.9	17.0
Child 12-19[d]	3.90	2.13	8.95	10.4	20.9
Adult[e]	6.90	4.03	16.99	23.73	43.02

Sources: U.S. EPA 2002, 2008a

NA = not applicable; it is assumed that children <1 year of age do not consume fish.

[a]Values were calculated as (consumer-only IR for Child 1-2) x (fraction of population consuming fish for Child 3-5).

[b]Values were calculated as (consumer-only IR for Child 3-5) x (fraction of population consuming fish for Child 3-5).

[c]Values were calculated as (consumer-only IR for Child 6-11) x (fraction of population consuming fish for Child 6–11).

[d]Values were calculated as (consumer-only IR estimated for Child 12-19) x (fraction of population estimated to consume fish for Child 12-19).

[e]Values were calculated as (consumer-only IR for Adults) x (fraction of population consuming fish for Adults).

Exhibit B-21 provides mean and the 90th percentile fish ingestion rates for recreational fishers, black and female recreational fishers, and fishers of Hispanic, Laotian, and Vietnamese descent. These latter three populations are culturally or economically disposed to higher rates of fish ingestion than the general population. Recreational fisher values are from the EFH (EPA, 2011a). Black and female recreational fishers ingestion rates are presented in Burger (2002). The fish ingestion rates for Hispanic, Laotian, and Vietnamese populations were derived from a study by Shilling *et al*. (2010) of contaminated fish consumption in California's Central Valley Delta. Shilling *et al*. (2010) reported mean and 95th percentile ingestion rates for each subpopulation. In part due to the low sample size in the Shilling study (n of 30 to 45), 95th percentile values were believed to be unrealistically high. The 90th percentile ingestion rate estimates presented in Exhibit B-21 for Hispanic, Laotian, and Vietnamese fishers were derived by EPA using information from Shilling *et al*. (2010; EPA, 2010).

RTR multipathway assessments to date have used whole fish concentrations estimated by TRIM.FaTE (or by application of BAF and BSAF values for arsenic). The proportion of lipid in TL3 and TL4 fish in TRIM.FaTE is assumed to be 5.7 percent (by weight) for the whole fish, based on information provided by Thomann (1989). The lipid content of the part(s) of the fish normally consumed is likely to be less than 5.7 percent. For example, EPA estimated a consumption-weighted mean lipid value for fillets of fish from TL3 to be 2.6 percent and from TL4 to be 3.0 percent (Table 6-9 in U.S. EPA 2003b). If an assessor wishes to account for reduced chemical concentration in fillet compared with whole fish for lipophilic chemicals, they can use a "preparation" loss of chemical (see Section B.6.4).

For lipophilic chemicals (e.g., log Kow greater than 4), which partition primarily into the fatty tissues of fish, much of the higher concentration tissues might be stripped from the fish during preparation (e.g., belly fat, viscera which includes fat in liver, etc., fat under skin). The degree to which the concentration of chemical in a fillet is less than the average total concentration in the whole fish is chemical specific. Assuming the chemical concentration in the fillet is the same as in the whole fish may result in a health protective bias for highly lipophilic chemicals. For persons who prefer to consume fillets with the skin on and do not discard belly fat, assuming the same concentration of chemical in the fish consumed as in the whole fish also is protective.

Exhibit B-21. Calculated Mean and 90th Percentile Per Capita Fish Ingestion Rates for Populations of Recreational Fishers ($IR_{PC,y}$)

Subpopulation							
Percentile	**Units**	**Recreational Fisher[a]**	**Female Recreational Fisher[b]**	**Black Recreational Fisher[b]**	**Hispanic Recreational Fisher[c]**	**Laotian Recreational Fisher[c]**	**Vietnamese Recreational Fisher[c]**
Ingestion of Fish							
Mean	g/day	8	39.1	171	25.8	47.2	27.1
90th	g/day	11	123	446	98	144.8	99.1

[a]1997 Exposure Factors Handbook (U.S. EPA, 1997a)
[b]Burger (2002) weights are "as consumed" for locally caught fish.
[c]Shilling, F., A. White, L. Lippert, and M. Lubell. 2010. Contaminated fish consumption in California's Central Valley Delta. Environmental Research 110:334-344.

B.6.3.5 Soil Ingestion Rates

Adult gardeners and farmers may incidentally ingest soils from gardening activities, and gardening and farming families might ingest soil particles that adhere to exposed fruits and exposed and belowground vegetables. Soils that are re-suspended in the air by wind can resettle on exposed fruits and vegetables. Children may incidentally ingest soils in those ways, but in addition, children playing outdoors may ingest soils directly or by hand-to-mouth activities during play. Exhibit B-23 includes soil ingestion rate options by age group for these types of exposures. It does not include options for children who may exhibit pica, or the recurrent ingestion of unusually high amounts of soil (i.e., on the order of 1,000 - 5,000 mg/day or more) (U.S. EPA 2008a).

Data on soil ingestion rates are sparse; the soil ingestion rates listed in Exhibit B-22 are based on limited data. The studies evaluated by EPA for children generally focused on children between the ages of 1 and 3 to 6 years and were not specific to families that garden or farm. The default ingestion rates recommended by EPA for RTR multipathway assessments are the 90th percentile values, as for other FFC ingestion rate parameters.

Exhibit B-22. Daily Mean and Percentile Soil Ingestion Rates for Children and Adults

Age Group (years)	Soil Ingestion Rate (mg/day)				
	Mean[a]	**50th[a]**	**90th**	**95th**	**99th**
Child <1	NA				
Child 1-2	50	50	200[b]	200[b]	200[b]
Child 3-5	50	50	200[b]	200[b]	200[b]
Child 6-11	50	50	201[c]	331[d]	331[d]
Child 12-19	50	50	201[c]	331[d]	331[d]
Adult 20-70	20	20	201[c]	331[d]	331[d]

Sources: U.S. EPA 2008a, U.S. EPA 2011a
[a]For mean and 50th percentile soil ingestion rates for children, value represents a "central tendency" estimate from EPA's 2008 CSEFH, Table 5-1. For adults, value is the recommended mean value for adults from EPA's 2011 EFH, Chapter 5, Table 5-1.
[b]Values are the recommended "upper percentile" value for children from EPA's 2011 EFH, Chapter 5, Table 5-1. The 2008 CSEFH and 2011 EFH included a high-end value associated with pica only, but this value has not been used.
[c]Values are 90th percentile adult ingestion rates calculated in Stanek et al. 1997; used to represent older children and adults.
[d]Values are 95th percentile adult ingestion rates calculated in Stanek et al. 1997; used to represent older children and adults.

B.6.3.6 Total Food Ingestion Rates

Although the multimedia ingestion risk methodology was developed to perform deterministic screening-level exposure and risk assessments, total food ingestion rates could be included if the methodology is adapted for a pobablilistic assessment. In particular, the total food ingestion rates presented in Exhibit B-23 could be used to normalize or to truncate the sum of food-specific ingestion rates to ensure reasonable values. This procedure is particularly important when chemical intake from multiple upper-percentile food ingestion rates for different types of food are added together. Individuals representing the upper percentile ingestion rate for one food category might not be the same individuals who reported high percentile ingestion rates for one or any of the other food categories.

Exhibit B-23. Daily Mean and Percentile Per Capita Total Food Intake for Children and Adults

Lifestage (years)	Percent of Group Consuming Food	Mean	50th	90th	95th	99th
Total Food Intake (g/day, as consumed)						
Child <1[a]	67.0% - 99.7% [h]	322	270	599	779	1152
Child 1-2[b]	100%	1,032	996	1537	1703	2143
Child 3-5[c]	100%	1,066	1,020	1,548	1,746	2,168
Child 6-11[d]	100%	1,118	1,052	1,642	1,825	2,218
Child 12-19[e]	100%	1,197	1,093	1,872	2,231	2,975
Adult[f]	100%	1,100	1,034	1,738	2,002	2,736
Total Food Intake (g/kg-day, as consumed)						
Child <1[a]	67.0% - 99.7% [h]	39	34	72	95	147
Child 1-2[b]	100%	82	79	125	144	177
Child 3-5[c]	100%	61	57	91	102	132
Child 6-11[d]	100%	40	38	61	70	88
Child 12-19[e]	100%	21	19	34	40	51
Adult[g]	100%	14.8	13.9	23.7	27.6	35.5

Sources: U.S. EPA 2005e, 2008a

[a]These values represent a time-weighted average for age groups birth to <1 month (N=88), 1 to <3 months (N=245), 3 to <6 months (N=411), and 6 to <12 months (N=678) from Table 14-3 of the 2008 CSEFH.

These values represent a time-weighted average for age groups 1 to <2 years (N=1,002) and 2 to <3 years (N=994) from Table 14-3 of the 2008 CSEFH.

[c]These values were obtained from Table 14-3 of the 2008 CSEFH (age group 3 to <6 years, N=4,112).

[d]These values were obtained from Table 14-3 of the 2008 CSEFH (age group 6 to <11 years, N=1,553). These values represents a health protective (i.e., slightly low) estimate for ages 6 through 11 years since 11-year olds are not included in this CSEFH age group.

[e]These values represent a time-weighted average for age groups 11 to <16 years (N=975) and 16 to <21 (N=743) years from Table 14-3 of the 2008 CSEFH. Note that estimated values include 11-year-olds and individuals through age 20, which contributes to uncertainty in the estimates.

[f]These values represent a time-weighted average for age groups 20 to 39 years (N=2,950) and 40 to 69 years (N=4,818) from Table 5B of the 2005 EPA analysis of CSFII.

[g]These values represent a time-weighted average for age groups 20 to 39 years (N=2,950) and 40 to 69 years (N=4,818) from Table 5A of the 2005 EPA analysis of CSFII.

[h]Percents consuming foods from Table 14-3 of the 2008 CSEFH include: 67.0% (birth to <1 month); 74.7% (1 to <3 months); 93.7% (3 to <6 months); and 99.7% (6 to <12 months). Infants under the age of 1 that consume breast milk are classified as "non-consumers" of food.

B.6.4 Other Exposure Factor Values

The other exposure parameters included in the algorithms are exposure frequency (Section B.6.4.1), fraction of the food type obtained from the contaminated area (Section B.6.4.2), and reduction in the weight of the food types during preparation and cooking (Section B.6.4.3). For the breast milk ingestion pathway, additional exposure parameters are included in the FFC algorithms (Section B.6.5).

B.6.4.1 Exposure Frequency

The exposure frequency (EF) represents the number of days per year that an individual consumes home-produced food items that are contaminated with the chemical being evaluated. The default value for EF is 350 days/year for all exposure sources and all potential receptors. This assumption is consistent with the food ingestion rates (i.e., daily intake rates equivalent to annual totals divided by 365 days) and does not imply that residents necessarily consume home-produced food products every day of the year.

If an assessor wishes to evaluate daily intake rates based on shorter averaging times, they can replace both the food-specific ingestion rates and the EF for each home grown food product. For example, they may want to specify a lower EF values for various food types where residents obtain some of their diet from commercial sources and where consumption of home grown produce is seasonal.

B.6.4.2 Fraction Contaminated

The fraction contaminated (FC) represents the fraction of each food product consumed that is contaminated by the chemical at a level consistent with environmental concentrations in the area of concern (e.g., area with maximum deposition rates). Obviously, the most health protective assumption is that all food products consumed (i.e., 100 percent) are from the location represented by the assessment scenario.

For RTR screening assessments, the default FC is 1, assuming that households that farm, garden, or raise animals produce 100 percent of the food product consumed, and 100% of the fish consumed is home caught. The assessor can vary this default FC value for individual food products to tailor the assessment to a particular exposure scenario.

B.6.4.3 Preparation and Cooking Losses

Food preparation and cooking losses are included in the FFC exposure calculations to account for the amount of a food product as brought into the home that is not ingested due to loss during preparation, cooking, or post-cooking. These losses need to be accounted for in the ADD equations because the food ingestion rates calculated from the USDA 1987 to 1988 NFCS are based on the weight of home grown produce and animal products brought from the field into the house prior to any type of preparation. Not all of the produce or products were eventually ingested. In general, some parts of the produce and products are discarded during preparation while other parts might not be consumed even after cooking (e.g., bones). Thus, the actual food ingested is generally less than the amount brought into the home.

Three distinct types of preparation and cooking losses are included in the ingestion exposure algorithms: (1) loss of parts of the food type from paring (i.e., removing the skin from vegetables and fruits) or other types of preparation (e.g., removing pits, coring, deboning), (2) additional loss of weight for the food type during cooking (e.g., evaporation of water), and (3) post-cooking losses (e.g., non-consumption of bones, draining cooking liquid [e.g., spinach]). The methodology uses mean values for these three types of preparation and cooking losses for exposed fruit, protected fruit, exposed vegetables, protected vegetables, root vegetables, beef,

pork, poultry, and fish. Different types of losses apply to different types of foods. Therefore, the losses can be represented by only two parameters, *L1* and *L2*, the definitions of which vary according to the food type as explained in the endnotes in Exhibit B-24. All preparation and cooking loss parameter values were estimated as specified in the exhibit's endnotes from data presented in Chapter 13 of the EPA's 1997 and 2011 EFH (U.S. EPA 1997a and 2011a).

Exhibit B-24. Fraction Weight Losses from Preparation of Various Foods

Product	Mean Cooking, Paring, or Preparation Loss (Cooking Loss Type 1 [L1]) (unitless)[a]	Mean Net Post Cooking (Cooking Loss Type 2 [L2]) (unitless)[b]
Exposed Fruit[c]	0.244	0.305
Exposed Vegetable	0.162[d]	NA
Protected Fruit	0.29[e]	NA
Protected Vegetable	0.088[f]	NA
Root Vegetable[g]	0.075	0.22
Beef	0.27	0.24
Pork	0.28	0.36
Poultry	0.32	0.295[h]
Fish[i]	0.0	0.0

NA = Not Available

[a]For *fruits*, includes losses from draining cooked forms. For *vegetables*, includes losses due to paring, trimming, flowering the stalk, thawing, draining, scraping, shelling, slicing, husking, chopping, and dicing and gains from the addition of water, fat, or other ingredients. For *meats*, includes dripping and volatile losses during cooking.

[b]For *fruits*, includes losses from removal of skin or peel, core or pit, stems or caps, seeds and defects; may also include losses from removal of drained liquids from canned or frozen forms. For *vegetables*, includes losses from draining or removal of skin. For *meats*, includes losses from cutting, shrinkage, excess fat, bones, scraps, and juices.

[c]These values represent averages of means for all fruits with available data (except oranges) (U.S. EPA 1997a,Table 13-6).

[d]This value represents an average of means for all exposed vegetables with available data (U.S. EPA 1997a,Table 13-7). Exposed vegetables include asparagus, broccoli, cabbage, cucumber, lettuce, okra, peppers, snap beans, and tomatoes.

[e]This value was set equal to the value for oranges (U.S. EPA 1997a,Table 13-6).

[f]This value represents an average of means for all protected vegetables with available data (U.S. EPA 1997a,Table 13-7). Protected vegetables include pumpkin, corn, peas, and lima beans.

[g]These values represent averages of means for all root vegetables with available data (U.S. EPA 1997a,Table 13-7). Root vegetables include beets, carrots, onions, and potatoes.

[h]This value represents an average of means for chicken and turkey (U.S. EPA 1997a,Table 13-5).

[i]If the assessor changes fish ingestion rates to match a survey of the whole weight of fish brought into the home from the field (divided by the consumers of the fish), an appropriate value for L1 would be 0.31 and an appropriate L2 would be 0.11 (U.S. EPA 2011a, Table 13-69).

There are substantial uncertainties associated with the L1 and L2 parameters, including the wide variation in values across produce types that were averaged together to recommend a central tendency value for each. For example, the L2 factor does not distinguish between weight loss during cooking by water evaporation, which might leave the chemical in the food (chemical not lost), and pouring the cooking liquid down the drain (chemical lost) or using the liquid to create a sauce (chemical not lost). In addition, the concentration of chemical might be highest in the skin, which often is discarded, and lower in the consumed portion of many bulky fruits and vegetables. Finally, the data EPA used to evaluate L1 included negative losses (i.e., weight gains) due to hydration of dried vegetables (e.g., peas and lima beans), which increases the range of L1 values across different vegetables.

Note that the default L1 and L2 values for fish are set to zero. That is because the data source for the fish ingestion rates is not the USDA's 1987 to 1988 NFCS (USDA 1993, 1994a) as reported in EPA's EFH, which reported food as brought into the home, as is the case for the other food categories. Instead, the fish IR data are based on parts actually consumed, and so no loss processes for preparation are needed.

If the assessor uses fish ingestion rates to match a local survey of the whole weight of fish brought into the home from the field (divided by number of persons consuming the fish), they should also use non-zero values for the L1 and L2 parameters.

B.6.4.4 Food Preparation/Cooking Adjustment Factor (FPCAF) for Fish

In addition to estimating the weight of the food that is lost to preparation and cooking, there also can be changes in the chemical concentrations due to cooking. Because the fish consumption rates are "as consumed" and the fish concentration is based on uncooked fish, adjustments should be made to reflect the chemical concentrations in fish after cooking. In order to account for this phenomenon, an FPCAF can be applied to the uncooked fish concentration to estimate a concentration in cooked fish. The following sections discuss FPCAFs for each of the 4 PB-HAPs.

Mercury

In the U.S. EPA Revised Technical Support Document: National-Scale Assessment of Mercury Risk to Populations with High Consumption of Self-caught Freshwater Fish (U.S. EPA 2011b), a food preparation/cooking adjustment factor (FPCAF) of 1.5 was used to adjust methyl-mercury (MeHg) concentrations in consumed fish (i.e., a 50% increase in MeHg concentration due to cooking). Cooking fish typically increases MeHg levels per unit fish (as consumed) because MeHg concentrates in the muscle, while preparation involves removal primarily of non-muscle elements of the fish. The value is based on a study by Morgan et al. (1997).

Arsenic

Similar to mercury, arsenic will bind to muscle and will be retained during the cooking process. As such, the same FPCAF of 1.5 that is used for mercury is assumed for cadmium.

Cadmium

Similar to mercury and arsenic, cadmium will bind to muscle and will be retained during the cooking process. As such, the same FPCAF of 1.5 that is used for mercury is assumed for cadmium.

Dioxin

Dioxins are lipophilic and have been demonstrated to be lost during cooking. Based on a literature review, an FPCAF of 0.7 to is applied to account for these losses during the cooking process. A brief summary of supporting literature follows.

- Schecter et al. (1998) found that the mass of PCDD and PCDF in fresh catfish fillet (skin on) decreased by about 50 percent per serving portion during cooking. Given the simultaneous losses of moisture/fats during broiling of the catfish, the PCDDs and PCDFs concentrations decreased by 33 percent (i.e., multiply uncooked concentration in fresh fish by a factor of 0.66 = 0.70 to one significant digit).
- Reinert et al. (1972) reported higher losses of another highly lipophilic chemical, DDT, from cooking fish fillets of bloaters, yellow perch, lake trout, and coho salmon. Concentrations of DDT in fish fillet portions for lake trout and coho salmon, top

predators, were reduced by 64 to 72 percent by frying or broiling, primarily through preferential loss of fat (and lipophilic DDT) during cooking. The investigators did not report skin on or off; however, they used steak cuts instead of flat fillets, which provide a smaller ratio of skin to muscle than is the case for fillets that constitute one side of the fish.

- Zabik and Zabik (1995) quantified the reduction in TCDD concentration of cooked, with the skin off, fillets compared with uncooked fillet with skin for fish harvested from the Great Lakes. Concentrations in the cooked fish with the skin off were reduced relative to the raw fillet with the skin on by approximately 44 percent for walleye, 80 percent for white bass, and 61 percent for lake trout. Comparing losses of TCDD for fillets cooked with the skin on versus fillets that were both skinned and cooked, Zabik and Zabik (1995) found reductions in TCDD concentrations of approximately 43 percent for Chinook Salmon cooked with the skin on and 57 percent for chinook salmon cooked with the skin off. They found a 37 percent reduction of TCDD concentration for carp fillets cooked with the skin on and 54 percent reduction if the skin was removed.

The three studies listed above indicate that the 0.7 factor is not likely to overestimate loss of PCDD/PCDFs from fish during cooking (pan frying, broiling, grilling). Reductions in TCDD concentrations could be much higher with skin removal and trimming of fat.

POMs

While it is reasonable to assume that there might be losses of lipophilic PAHs during the cooking process, there is insufficient information to distinguish what the net loss (or gain) during cooking might be because cooking can create PAHs from proteins in the tissue. The literature acknowledges these competing forces, but does not provide information to disentangle the gain and loss mechanisms. As such, a neutral approach was taken, which is to assume an adjustment factor of 1.0 (i.e., no adjustment) for PAHs.

B.6.5 Breast-Milk Infant Exposure Pathway Parameter Values

Values used for parameters in the breast-milk exposure pathway algorithms (see Section B.3.4 of this attachment) can be scenario-specific, receptor-specific, and/or chemical-specific and might be empirically derived or estimated by an appropriate model. For parameters that are scenario-specific or for which empirical values are required, the default values are listed. For parameters for which algorithms calculate values, the appropriate equation is listed. Scenario- and receptor-specific parameters are discussed in Section B.6.5.1 and chemical-specific parameters are discussed in Section B.6.5.2.

B.6.5.1 Receptor-specific Parameters

Receptor-specific values are needed for parameters that describe the characteristics or activities of the exposed individual. In this context, there are two relevant receptors: the mother and the infant. Exhibit B-25 lists the parameters and their default values. The text that follows describes the recommended value or alternative values for each exposure parameter needed to calculate the infant absorbed chemical intake rate, or DAI_{inf}. For parameter values that can be estimated when empirical values are not available, see the equation description in Section B.3.4 of this attachment.

Exhibit B-25. Scenario- and Receptor-Specific Input Parameter Values Used to Estimate Infant Exposures via Breast Milk

Parameter	Description	Default Value
AT	Averaging time for infant's exposure via breast milk, i.e., duration of nursing (days)	=ED
BW_{inf}	Body weight of infant (kg) averaged over duration of nursing exposure	7.8
BW_{mat}	Body weight of mother (kg) averaged over duration of mother's exposure	66
DAI_{mat}	Daily absorbed intake of chemical by mother (mg/kg-day)	Equation B-38
ED	Exposure duration for infant, i.e., duration of breast feeding (days)	=AT
AT/ED	Averaging time divided by exposure duration	1.0
f_{bp}	Fraction of mother's whole blood that is plasma (unitless)	0.65
f_{fm}	Fraction of mother's body weight that is fat (unitless)	0.30
f_{mbm}	Fraction of fat in mother's breast milk (unitless)	0.04
f_{pm}	Fraction of mother's body weight that is plasma (unitless)	0.046
IR_{milk}	Mean infant milk ingestion rate over duration of nursing (kg/day)	0.709
t_{bf}	Duration of breast feeding (days)	365
t_{pn}	Duration of maternal chemical exposure prior to nursing (days)	3285

Averaging time (*AT*) and exposure duration (*ED*). AT refers to the time over which the infant's exposure to the chemical of concern is averaged. *ED* refers to the duration of the infant's exposure. For the exposure scenario considered for this age group, both *AT* and *ED* equal the duration of the nursing period, and they therefore cancel each other out in the infant average daily dose equation.

Infant body weight (BW_{inf}). The assessor selects a value for BW_{inf}, the time-weighted average body weight of the infant over the entire duration of breast feeding, based on the age at which the infant stops breast feeding. For example, if the infant breast feeds for one year, the assessor should select the body weight for an infant that is averaged from birth to the first birthday. Similarly, if an infant breast feeds for 6 months, the assessor should select the body weight for an infant that is averaged from birth to six months. Because the default breast feeding duration (t_{bf}) is one year (i.e., 365 days), the default infant body weight is 7.8 kg, which is the time-weighted average for the mean infant body weight between birth and the first birthday from EPA's 2008 *Child Specific Exposure Factors Handbook* (CSEFH; U.S. EPA 2008a). Exhibit B-26 presents additional percentile values for the infant body weight parameter that may be appropriate for some assessments.

Exhibit B-26. Average Body Weight for Infants

Statistic	0 to <6 months (kg)	0 to <12 months (kg)	0 to <18 months (kg)	0 to <24 months (kg)
Mean	6.5	7.8[a]	9.0	9.6
5th percentile	5.0	6.0	7.0	7.5
10th percentile	5.3	6.4	7.4	7.8
15th percentile	5.5	6.7	7.7	8.2
25th percentile	5.8	7.0	8.1	8.7
50th percentile	6.4	7.8	8.9	9.5
75th percentile	7.1	8.6	9.9	10.5

Statistic	0 to <6 months (kg)	0 to <12 months (kg)	0 to <18 months (kg)	0 to <24 months (kg)
85th percentile	7.4	9.0	10.3	11.0
90th percentile	7.7	9.2	10.6	11.3
95th percentile	8.0	9.7	11.1	11.8

Source: EPA (2008a); each value is the time-weighted average from the data summaries presented in the CSEFH, Table 8-3.
[a]Default value used for RTR assessments

Maternal body weight (BW_{mat}). This parameter represents the body weight of the mother averaged over the entire duration of the mother's exposure to the chemical of concern. The maternal body weight is needed to calculate the biological elimination constant for the lipophilic chemical in lactating women (k_{fat_elac}). The methodology assumes that the mother will be pregnant for 9 months (i.e., 0.75 year) and will be lactating for 1 year. The recommended default maternal body weight also assumes that the mother has been exposed for 10 years total. For 8.25 years, she is not pregnant or lactating, for 0.75 year she is pregnant, and for 1 year she is lactating. The default BW_{mat} of 66 kg is based on CSFII data compiled by EPA for non-lactating and non-pregnant women between the ages of 15 and 44 (i.e., women of child-bearing age), lactating women, and pregnant women (U.S. EPA 2004a). Exhibit B-27 presents additional values for the maternal body weight parameter which might be appropriate for some assessments. The BW_{mat} value is *not* the value that the methodology uses to estimate the mother's absorbed daily intake (DAI_{mat}). The daily ingestion rates for home grown/raised food products are for men and women combined, with the rates normalized to body weight. The ingestion rates for soil, water, and fish are not normalized to body weight but are based on both men and women. For those ingestion rates, the methodology uses an average body weight value for males and females to estimate the average daily dose (intake) of the chemical in mg/kg-day. These values are subject to the assumption that the body-weight normalized ingestion rates and resulting ADD values are applicable to nursing mothers.

Exhibit B-27. Time-weighted Average Body Weight for Mothers

Statistic	Weight (kg)
Mean	66.0[a]
5th	47.1
10th	50.2
25th	54.3
50th	62.0
75th	72.0
90th	85.7
95th	97.0

Source: U.S. EPA 2004a
[a]Default value

Exposure duration (*ED*). See discussion of *AT* and *ED* above.

Fraction of mother's whole blood that is plasma (f_{bp}). Steinbeck (1954) reported that plasma volume accounts for approximately 60 percent of the total blood volume in non-lactating human females (U.S. EPA 1998). Harrison (1967) and Ueland (1976) reported plasma volumes

between 63 to 70 percent in postpartum women (U.S. EPA 1998). The default value of 65 percent (0.65) is the value recommended by EPA in its *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions* (MPE, U.S. EPA 1998).

Fraction of mother's body weight that is fat (f_{fm}). A limitation of using a steady-state, instead of a dynamic partitioning, model for lactational transfer of chemicals is that several key parameters change over the course of exposure. For example, Equation B-40, used to estimate the concentration of a lipophilic chemical in breast milk fat, assumes that the mother's body fat will remain constant over the entire duration of breast feeding (t_{bf}), which is unlikely to be true (U.S. EPA 2001a). Another limitation of the single analytic model is that chemical transfer rates from blood to milk are unlikely to be the same as the rate of mobilization of the chemical from fat stores to the blood (U.S. EPA 2001a). Studies cited in ATSDR's toxicological profile for chlorinated dibenzo-p-dioxins show a correlation between percent body fat and the elimination rate of dioxins, with longer half-lives for dioxins in individuals with a higher proportion of fat in their bodies (ATSDR 1998). In the context of a screening model, however, EPA recommends a default value for the fraction of a mother's body comprised of fat of 0.3 based on data and discussions presented by Smith (1987) and Sullivan *et al*. (1991) (U.S. EPA 1998). A fraction of 0.3 indicates that 30 percent of the mother's body weight is fat, which is a health protective value (U.S. EPA 2001a). To establish a health protective screening scenario, a default value for f_{fm} of 0.30 is used**.**

Fraction of fat in mother's breast milk (f_{mbm}). The $C_{milkfat}$ model (Equation B-40) assumes that a constant fraction of breast milk is fat, even though there is evidence that indicates variation in the fat content of breast milk throughout lactation (Sim and McNeil 1992). Different studies suggest a fat content of breast milk in humans of between 1 and 5 percent (Jensen 1987, Schecter *et al*. 1994, Hong *et al*. 1994, McLachlan 1993, Bates *et al*. 1994, NAS 1991, Butte *et al*. 1984, Maxwell and Burmaster 1993, U.S. EPA 2011a, Smith 1987, Sullivan *et al*. 1991). The default value for f_{mbm} of 0.04 (i.e., 4 percent) is the value EPA recommended for MPE (U.S. EPA 1998).

Fraction of maternal weight that is plasma (f_{pm}). Altmann and Dittmer (1964) estimated that plasma volume for adult women ranged from 37 to 60 mL/kg of body weight and averaged about 45 mL/kg. Ueland (1976) observed that the average plasma volume of women 6 weeks postpartum was 45 mL/kg of body weight. Using a value of 1.026 for the specific gravity of plasma from Conley (1974), EPA estimated a value of 0.046 for the fraction of maternal weight that is plasma (U.S. EPA 1998). The default value for f_{pm} therefore is 0.046.

Infant breast milk ingestion rate (IR_{milk}). Milk ingestion rates vary with several factors, including the age and size of the infant and use of other foods such as formula. Based on its review of a several studies, EPA recommended time-weighted average and upper percentile milk ingestion rates for infants that nurse for six and for twelve months (U.S. EPA 2011a, Table 15-3). To estimate an "average" value, EPA first estimated study-sample-size weighted average values for 1 through 12 months of age and then developed time-weighted average milk ingestion rates from those (U.S. EPA 2011a). EPA estimated an upper percentile (upper bound) value as the mean plus two standard deviations. The ingestion rates, measured volumetrically (mL/day), are converted to mass-based estimates (kg/day) assuming the density of human milk to be 1.03 g/mL (reported by NAS 1991 and recommended by U.S. EPA 2011a). The resulting values are shown in the first two rows of Exhibit B-28. The screening-level default value of 980 mL/day is an upper-bound estimate based on a one-year nursing period.

Exhibit B-28. Infant Breast Milk Intake Rates

Age Category	Average (mL/d)	Average (kg/d)	"Upper Bound" (mL/d)	"Upper Bound" (kg/d)	Reference
1 to 6 months	742	0.764	1,033	1.064	U.S. EPA 2011a†
0 to <12 months	688	0.709	980[a]	1.01[a]	U.S. EPA 2011a†
0 to <1 month	510	0.525	950	0.979	U.S. EPA 2008a††
1 to <3 months	690	0.711	980	1.01	U.S. EPA 2008a†
3 to <6 months	770	0.793	1,000	1.03	U.S. EPA 2008a†
6 to <12 months	620	0.639	1,000	1.03	U.S. EPA 2008a†

[a]Default; † Based on review of multiple studies; †† Based on a single study

Exhibit B-28 also includes the recommended values for four non-overlapping age categories from the CSEFH (U.S. EPA 2008a, Table 15-1). The values demonstrate that although infants grow substantially from birth to one year of age, the "upper bound" estimates of their milk ingestion rates are very close to 1 liter per day at all stages of development in the first year.

Duration of breast feeding (t_{bf}). This parameter is equal to the infant's exposure duration (*ED*) and the infant's averaging time (*AT*). In its MPE Methodology, EPA asserts a health protective value for the duration of breast feeding of 1 year (i.e., 365 days) and a central tendency estimate of 6 months (180 days) (U.S. EPA 1998). Reviewers of MPE noted that 365 days may be overly health protective, given that only 20 percent of infants are breast fed for 6 months, at which point alternative foods are introduced, at least in addition to breast milk (U.S. EPA 2001a). Nonetheless, to establish a health protective screening scenario, the default value for t_{bf} is 365 days.

Duration of the mother's exposure to the chemical of concern prior to nursing (t_{pn}). The model shown as Equation B-40 includes this parameter to reduce the over-estimate of chemical concentration in milk fat that occurs if the model is applied to a chemical with a long biological half-life (e.g., many years). The factor is needed for applications of the model to scenarios with a brief exposure duration (e.g., beginning a few months prior to the start of nursing) relative to the chemical half-life. As the duration of an exposure scenario increases to meet and exceed the chemical half-life, however, the overestimate that occurs without this parameter is reduced. For example, assume a chemical biological half-life of 8 years and a nursing period of 1 year. If exposure of the mother starts at the beginning of nursing, using Equation B-40 without the t_{pn} term results in an over-estimate of the concentration of the chemical in breast milk by a factor of 28.1 compared with the prediction using Equation B-40 with the t_{pn} term (U.S. EPA 1998, Table 9-6). However, at longer pre-natal exposures of the mother, the magnitude of the over-estimate is reduced: for a 10-year exposure, the magnitude of the overestimate without the t_{pn} term is 2.28, and for a 30-year exposure, the overestimate is reduced to 1.39.

For purposes of the screening-level assessment, the methodology uses an exposure duration equal to the default half-life for dioxins, or 10 years. Only 3,285 days of that period are pre-natal (i.e., 3650 minus 365 days, assuming 1 year lactation period). Although longer exposure periods are possible for the screening scenario, there is sufficient uncertainty in the model to merit accepting a health protective bias for this parameter value.

B.6.5.2 Chemical-Specific Parameter Values

The chemical-specific parameters in the breast-milk pathway are listed in Exhibit B-29. Note that the parameters for which values are needed are different for the lipophilic chemicals (i.e.,

dioxins), for which lactational transfer is assumed to occur via milk fat, and inorganic chemicals, for which the transfer is assumed to occur via the aqueous phase of breast milk (i.e., mercury). All dioxin congeners were assumed to manifest identical values as TCDD in regard to breast milk-related parameters.

Exhibit B-29. Chemical-specific Input Parameter Values for Breast Milk Exposure Pathway

Parameter and Description		2,3,7,8-TCDD	MeHg
AE_{inf}	Infant absorption efficiency of the chemical by the oral route of exposure (i.e., fraction of ingested chemical that is absorbed by the infant; unitless)	1.0 (default)	1.0 (default)
AE_{mat}	Maternal absorption efficiency of the chemical by the oral route of exposure (i.e., fraction of ingested chemical that is absorbed by the mother; unitless)	1.0 (default)	1.0 (default)
f_{bl}	Fraction of steady-state total body burden of hydrophilic chemical in mother that is in the mother's whole blood compartment (unitless)	NA	0.059 (Kershaw *et al*. 1980)[a]
f_f	Fraction of steady-state lipophilic chemical body burden in mother that is stored in body fat (unitless)	≥0.90 (ATSDR 1992)	NA
f_{pl}	Fraction of steady-state total hydrophilic chemical body burden in mother that is in the blood plasma compartment (unitless)	NA	Not yet identified[b]
h	Biological half-life for chemical in non-lactating women (days)	3650 (U.S. EPA 1994c)	50 (Sherlock *et al*. 1984)
k_{aq_elac}	Rate constant for total elimination of hydrophilic chemicals by lactating women (per day)	NA	= k_{elim}
k_{elim}	Rate constant for elimination of chemical for non-lactating women (per day; related to chemical half-life)	1.9E-04[b]	1.4E-02 [c]
k_{fat_elac}	Rate constant for total elimination of lipophilic chemicals by lactating women (per day)	Est. using Equation B-43	NA
Pc_{bm}	Partition coefficient for hydrophilic chemical between maternal *blood plasma* and aqueous phase of breast milk (g milk/g plasma; model assumption)	NA	1.0 (model assumption)
Pc_{RBC}	Partition coefficient for hydrophilic or protein-bound chemical between *red blood cells* (RBC) and *plasma* in maternal blood (mL whole blood/mL RBC)	NA	40 (Hollins *et al*. 1975)

NA = not applicable. ND = not yet determined from literature.
[a]This value is based on a single-dose study and may not be appropriate for a chronic exposure model.
[b]An empirical value for this variable is currently missing for application of model.
[c]This value was calculated from biological half-life (*h*) using Equation B-42.

Absorption efficiency of the chemical by the oral route of exposure for the infant (AE_{inf}). The models included in the methodology assume that the AE_{inf} from the lipid phase of breast milk is equal to the AE_{inf} from the aqueous phase of the milk. Reviewers of the model stated that this assumption may not be valid and that ideally, the equation DAI_{inf} would include variables for the AE_{inf} from the breast milk fat and the AE_{inf} from the aqueous phase of breast milk (U.S. EPA 2001a). However, since the methodology assumes that chemicals will partition to either the lipid or aqueous phase of milk, it is not necessary at this time to have multiple AE_{inf} values for a given chemical. If data on the AE from the mother or an adult but not for the infant are available, data for the adult may be used for AE_{inf}. Reviewers also recommended that chemical-specific values

come from studies that account for absorption of the chemical from milk, because absorption from other matrices (e.g., solid foods) may not be relevant (U.S. EPA 2001a). If chemical-specific data are not available for adults or infants, a health protective default value for AE_{inf} for a screening level assessment is 1.0, which assumes 100 percent absorption (U.S. EPA 1998).

The default value for AE_{inf} for both MeHg and dioxin is 1.0. For ingested lipophilic chemicals, it is reasonable to assume that absorption will be high (U.S. EPA 2004b). ATSDR (1998) reported that dioxins are well absorbed by the oral route of exposure, with one human experiment indicating more than 86 percent absorption. It is EPA policy to assume 100 percent absorption for chemicals with reported *AEs* of 50 percent or higher (U.S. EPA 2004b). MeHg also is well absorbed, with measured values as high as 95 percent, and so a value of 100 percent is used (U.S. EPA 2001b).

Absorption efficiency of the chemical by the oral route of exposure for the mother (AE_{mat}). The default value for both dioxins and MeHg is 1.0, as described in the previous paragraph.

Fraction of total maternal chemical body burden that is in the whole blood (f_{bl}). The default value for MeHg, 0.059, is from Kershaw *et al*. (1980), which reported kinetics of blood deposition and clearance of MeHg in humans. Individuals consumed one meal of fish that contained between 18 and 22 µg Hg/kg body weight. The fraction of the dose deposited in the blood volume after mercury was fully distributed in tissues was 5.9 percent or 0.059. This study used a single-dose and thus may not be appropriate for a chronic exposure analysis.

Fraction of total maternal chemical body burden that is in body fat (f_f). Based on ATSDR's *Toxicological Profile for Selected PCBs* (ATSDR 1992) and Sullivan *et al*. (1991), EPA concluded that the "fraction of ingested contaminant stored in fat may be >90%" for lipophilic chemicals such as PCBs and dioxins (U.S. EPA 1998). This statement was interpreted to mean that 90 percent of the maternal body burden of chemical at "steady state" is located in body fat for dioxins at steady state.

Fraction of total maternal chemical body burden that is in blood plasma (f_{pl}). For hydrophilic chemicals, this parameter represents the steady-state fraction of the total chemical in the body that is circulating in the blood plasma. Values for f_{pl} may be available for some chemicals in the scientific literature. No value for this parameter for methyl mercury has been identified from the literature at this time. A value can be calculated using Equation B-45. However, this equation requires a reliable value for f_{bl}, and the value found for mercury may not be appropriate for a chronic exposure analysis (see above).

Chemical half-life in non-lactating women (h). In general, highly lipophilic chemicals tend to have relatively long biological half-lives. EPA estimates that the half-life for dioxins is between 7 and 10 years (U.S. EPA 1994a). ATSDR estimates that the half-life for 2,3,7,8-TCDD in particular may be as long as 12 years (ATSDR 1998). To establish a health protective screening scenario, the default half-life for dioxins is set to 10 years or 3650 days.

The half-life for methylmercury is on the order of weeks, not years. Greenwood *et al*. (1978) measured blood clearance rates for MeHg in lactating Iraqi women exposed accidentally to MeHg via bread prepared from wheat treated with a fungicide that contained MeHg. The data indicated a mean half-life for MeHg of approximately 42 days. Sherlock *et al*. (1984) reported an average measured half-life for MeHg of 50 days with a range of 42-70 days. The default for MeHg is set to the longer average half-life of 50 days.

Chemical elimination rate constant for lactating women – aqueous (k_{aq_elac}). The parameter k_{aq_elac} is equal to k_{elim} plus the loss rate for the chemical in the aqueous phase of breast-milk

during lactation. EPA has yet to propose a term for the additional elimination of a chemical in the aqueous phase of milk from breast feeding. In the absence of empirical values, a reasonable assumption for water soluble chemicals is that k_{aq_elac} is equal to k_{elim} as discussed for Equation B-45. The extent to which k_{elim} is an underestimate of k_{aq_elac} for a given chemical will determine the extent of health protective bias in k_{aq_elac}.

Chemical elimination rate constant for non-lactating women (k_{elim}). Although values for this parameter often are reported directly in the literature, the methodology estimates k_{elim} from chemical half-life assuming first-order kinetics as shown in Equation B-42. For example, for a biological half-life of 3,650 days for dioxins, k_{elim} is estimated to be 1.9E-04 per day. Assuming a biological half-life of 50 days for MeHg, the value for k_{elim} is estimated to be 0.014 per day.

Rate constant for total elimination of lipophilic chemicals by lactating women (k_{fat_elac}). Although values for this parameter might be found in the scientific literature for some chemicals, k_{fat_elac} for dioxins is calculated from Equation B-43. When the parameters in that equation use the default values for dioxins, the estimated value of k_{fat_elac}. is 0.0015 per day

Partition coefficient for chemical between maternal blood plasma and aqueous phase of breast milk (Pc_{bm}). The aqueous model, presented in Equation B-44, assumes that the concentrations in the plasma and aqueous phase of breast milk are directly proportional (U.S. EPA 1998). Therefore, the default value for this parameter for MeHg is 1.0.

Partition coefficient for chemical between red blood cells and plasma in maternal blood (Pc_{RBC}). Chemical-specific values for this parameter should be located in the scientific literature. If chemical-specific values are unavailable and it is assumed that there is equal distribution of the chemical in the plasma and red blood cells, EPA suggests a default value of 1.0 (U.S. EPA 1998). For MeHg, the methodology uses a value of 40 based on Hollins *et al*. (1975) study of cats exposed to MeHg, which reported a ratio of radio-labeled mercury in red blood cells to plasma of 97.7 to 2.3 (i.e., ratio of 42.5).

B.7 Summary of Default Exposure Parameter Values

The default parameter values used in the multimedia ingestion risk methodology are intended to be characteristic of a health protective (but plausible) exposure scenario that results in a negligible or extremely low chance of underestimating risk. EPA used these default parameter values to derive the screening threshold emission rates used for screening emissions of PB-HAPs from sources included in RTR risk assessments. These values are the default for parameter values as described in Section B.6 of this attachment. This section summarizes the default parameter values used to calculate screening thresholds.

This section is organized to present the chemical- and scenario-specific parameters by data type. The screening-level analysis uses the following ingestion rates for each ingestion scenario of interest and population-specific characteristic assumptions (presented in Section B.7.1), that are generally health protective in nature:

- Fisher Scenario: 99th percentile ingestion rates for fish (presented in Section B.7.1.1)
- Farmer Scenario: 90th percentile ingestion rates for soil, breast milk, and farm food items (presented in Section B.7.1.2)
- Gardener Scenario: Urban gardener uses mean ingestion rates for fruits and vegetables and eggs and 90th percentile ingestion rates for soil and breast milk. Rural gardener

uses 90th percentile rates for fruits and vegetables, eggs, soil, and breast milk (presented in Section B.7.1.3).

Screening threshold emission rates were derived for five RTR chemical groups: arsenic compounds, cadmium compounds, methyl mercury, 2,3,7,8-TCDD, and benzo(a)pyrene. Section B.7.3 presents chemical-specific parameter inputs for these five chemicals. Finally, Section B.7.4 presents default parameter values for the nursing infant exposure scenario, which applied only to dioxin and methyl mercury as discussed in Section B.3.4.

B.7.1 Default Ingestion Rates

The screening-level (or default) values for ingestion rates for soil, breast milk, and for each farm food item are set to the 90th percentile or mean of the distribution of national data for that medium based on the exposure scenario of interest. In general, these values were obtained from the 2011 Exposure Factors Handbook or the 2008 Child-Specific Exposure Factors Handbook (see Exhibit B-16). Fish ingestion rates also are available from these sources; however, as described in Section B.6.3.4, other sources were used to obtain fish ingestion rates.

B.7.1.1 Fisher Ingestion Scenario

The adult fish ingestion rate was obtained from Burger (2002), a study that examined daily consumption of wild-caught fish for high-end recreationalists (white, black, female) in South Carolina. For female high-end consumers of wild-caught fish, Burger identified average and higher-percentile consumption rates as follows: 39.1 g/day (mean), 123 g/day (90th percentile), 172 g/day (95th percentile), and 373 g/day (99th percentile). As shown in Exhibit B-17 and discussed in Section B.6.3.4, for adults, the rate of fish ingestion assumed in the screening scenario is 373 g/day, which corresponds to the 99th percentile value estimated by Burger for adult females. This value was selected to be representative of subsistence fishers.

For the child age groups, as discussed in Section B.6.3.4, the baseline fish ingestion rates for the screening scenario are based on “as prepared” total freshwater/estuarine fish ingestion rates at the 99th percentile of the distribution for the *consumer-only* population (i.e., inclusive only of people who consume fish, rather than per-capita rates, which include both consumers and non-consumers), as estimated in U.S. EPA (2002), Section 4.2.1.1. Some adjustments were necessary because the age groups evaluated for RTR (which correspond to the age groups for which farm food ingestion rates are available) do not all directly correspond to the age groups in the U.S. EPA (2002) report. As described in Section B.6.3.4, these adjustments convert the available age-specific data on fish ingestion rates to the age-specific values needed for the methodology.

For the screening-level fish ingestion exposure scenario, the consumer evaluated is an individual who regularly consumes a large amount of fish that he or she has caught locally over the course of a 70-year lifetime. Estimated exposures are intended to encompass those of a subsistence fisher whose diet comprises a substantial proportion of fish. The scenario is not, however, intended to represent the maximum possible exposure an individual subsistence fisher might experience.

Although the fish ingestion rates presented here are representative of the 99th percentile of the evaluated data set, the use of these values (compared with 90th percentile values used for other food types) is not considered to be inconsistent. This is due to the idiosyncrasies of the survey data on fish consumption, the fact that the data sets for homegrown foods and fish are not parallel, and the consideration of rates appropriate for subsistence fishers, as described above.

As discussed above, EPA believes that use of these fish ingestion rates strikes the appropriate balance between being health protective and having screening scenarios so conservative that they are of limited use in the decision making process. This high-end fish ingestion rate is appropriate in the context of the conservative screening scenario used in the RTR process and is applicable for national rulemakings given that it is very likely that subsistence woman fishers of child bearing age are located throughout the United States. Using a high-end subsistence fish ingestion rate also is consistent with section 112 of the CAA, which focuses on risks associated with maximally exposed individuals.

B.7.1.2 Farmer Ingestion Scenario

The default parameter values assume that all food types are obtained from the area of chemical deposition specified by the assessment scenario (i.e., fraction of food from contaminated area = 1.0).

For estimates of screening threshold emission rates for PB-HAPS, environmental concentrations and air deposition rates were estimated using TRIM.FaTE for the area of maximal deposition in the vicinity of a hypothetical facility, and thus represent risks estimated for a maximally exposed individual/farm/family.

Exhibit B-30 also includes a sum of the 90th percentile ingestion rates for homegrown food categories and 99th percentile fish ingestion to show the implied total food ingestion rate associated with setting multiple food-type-specific ingestion rates at upper percentiles. Because these upper percentile values for each farm food category are likely to reflect different individuals, it is likely that addition of multiple upper percentile intake values will exceed the total food ingestion rates expected for the general population. This sum is shown on the third row from the bottom (Total Food: Homegrown Only).

The second row from the bottom presents the 90th percentile of the distribution of *individual total food ingestion rates* from the USDA's 1994-96 and 1998 *Continuing Survey of Food Intakes by Individuals* (CSFII) (USDA 2000) data sets, as analyzed by EPA (U.S. EPA 2005e). The total ingestion rate for the farming households (third row from bottom) takes into account the cooking losses typical of each food category to provide a better comparison with the 90th percentile individual total food ingestion rates from CSFII (which are based on consumption of prepared foods). The final row of Exhibit B-30 shows the likely magnitude of the overestimates by age category by presenting the ratio of the two preceding rows. The values in this row demonstrate the potential for overestimating intake by using upper percentile values for all food groups. This bias may be considered when evaluating the results estimated with the methodology.

Exhibit B-30. Farm Food Category Ingestion Rates for Health Protective Screening Scenario for Farming Households

Product	Screening-Level Consumer Ingestion Rate						Units
	Infants <1 yr	Child 1–2 yrs	Child 3–5 yrs	Child 6–11 yrs	Child 12–19 yrs	Adult 20–70 yrs	
Farm Food Item							
Beef[a]	NA	9.49	8.83	11.4	3.53	4.41	g/kg-day
Dairy[b]	NA	185	92.5	57.4	30.9	6.16	g/kg-day
Eggs[a]	NA	4.90	3.06	1.90	1.30	1.31	g/kg-day

Product	Screening-Level Consumer Ingestion Rate						Units
	Infants <1 yr	Child 1–2 yrs	Child 3–5 yrs	Child 6–11 yrs	Child 12–19 yrs	Adult 20–70 yrs	
Exposed Fruit[a]	NA	12.7	5.41	6.98	3.41	2.37	g/kg-day
Exposed Vegetable[a]	NA	10.7	3.47	3.22	2.35	3.09	g/kg-day
Pork[a]	NA	4.90	4.83	3.72	3.69	2.23	g/kg-day
Poultry[a]	NA	7.17	6.52	4.51	3.13	2.69	g/kg-day
Protected Fruit[a]	NA	44.8	32.0	23.3	7.44	15.1	g/kg-day
Protected Vegetable[a]	NA	3.88	2.51	2.14	1.85	1.81	g/kg-day
Root Vegetable[a]	NA	7.25	4.26	3.83	2.26	2.49	g/kg-day
Other							
Breast milk[c]	1.01	NA	NA	NA	NA	NA	kg/day
Soil (dry)	NA	200[d]	200[d]	201[e]	201[e]	201[e]	mg/day
Fish (per individual)[f]	NA	107.7[g]	159.0[g]	268.2[h]	331.0[h]	373	g/day
Total Food Ingestion Rates (for comparison only not, used in RTR screening; excludes soil and water)							
Total Food: Homegrown only[i]	NA	259	142	99	51	35.5	g/kg-day
Total Food: All Sources[j]	NA	125	91	61	34	23.7	g/kg-day
Overestimate (ratio of Homegrown/Total)	NA	2.1	1.6	1.6	1.5	1.3	(unitless)

Sources: U.S. EPA 2011a, U.S. EPA 2008a, unless otherwise noted.

NA = not applicable

[a]Primary source for values was the 1987–1988 NFCS survey; compiled results are presented in Chapter 13 of the 2011 Exposure Factors Handbook (U.S. EPA 2011a). When data were unavailable for a particular age group, the intake rate for all age groups was multiplied by the age-specific ratio of intake based on national population intake rates from CSFII.

[b]Primary source for values was the 1987–1988 NFCS survey, compiled results are presented in Chapter 13 of the 2011 Exposure Factors Handbook (U.S. EPA 2011a). When data were unavailable for a particular age group, the intake rate for all age groups was multiplied by the age-specific ratio of intake based on national population intake rates from an NHANES 2003–2006 analysis in Chapter 11 of the Exposure Factors Handbook.

[c]Infants are assumed to consume only breast milk for one year.

[d]These values are the recommended "upper percentile" value for children from EPA's 2011 EFH, Chapter 4, Table 4-23. The 2008 CSEFH and 2011 EFH included a high-end value associated with pica only, but this value has not been used.

[e]These values are 90th percentile adult ingestion rates calculated in Stanek et al. 1997, and they are used to represent older children and adults.

[f]The ingestion rate for adults was obtained from Burger (2002) and is the 99th percentile value for adult females considered high-end recreationists; this value is believed to be representative of subsistence fishers. The 99th percentile values for children were derived based on EPA's Estimated Per Capita Fish Consumption in the United States (2002)—Section 4.2.1.1 Table 5 (for child age categories) adjusted and scaled. Values reflect "as prepared" ingestion rates.

[g]The fish ingestion rate for children aged 3–5 years was obtained directly from Section 4.2.1.1, Table 5 in the U.S. EPA (2002) report (value presented is rounded); for these children, the RTR age-group range matches the U.S. EPA (2002) age category. Fish ingestion rates for children less than 3 years old, however, were not provided. Therefore, for children aged 1–2 years, the fish ingestion rate was calculated using the ingestion rate for children aged 3–5 years scaled downward by the ratio of the mean body weight of children aged 1–2 years to the mean body weight of children aged 3–5-years.

[h]Time-weighted average ingestion rates were calculated using the U.S. EPA 2002 fish ingestion estimates in order to adjust for the differences between the age group ranges used for the RTR screening and those presented in the 2002 EPA report.

[i]Sum of post-cooking food ingestion rates. This estimate is calculated by multiplying the food ingestion rates on previous rows (excluding soil and water) by $(1-L_1)x(1-L_2)$, where L_1 and L_2 are the loss rates from Exhibit B-24. The rows are then summed to get the total post-cooking ingestion rate.

[j]90th percentile total food intake rates from U.S. EPA 2008a and 2005e based on CSFII data 1994-96 and 1998; see Section B.6.3.6 of this document.

B.7.1.3 Gardener Ingestion Scenario

As discussed in Section 3.2.3, two potential gardener scenarios can be evaluated depending on the surrounding land use: a rural gardener and an urban gardener. Similar to the farmer ingestion scenario, the default settings assume that all food types are obtained from the area of chemical deposition specified by the user (i.e., fraction of food from contaminated area = 1.0); however, an urban gardener is expected to have a lower ingestion rate for home produced goods than a rural gardener. The rural gardener is assumed to have the same 90th percentile ingestion rate as the farmer for the produce consumed. Exhibit B-31 and Exhibit B-32 provide ingestion rates for the rural gardener and urban gardener, respectively.

For estimates of screening threshold emission rates for PB-HAPS, environmental concentrations and air deposition rates were estimated using TRIM.FaTE for the area of maximal deposition in the vicinity of a hypothetical facility, and thus represent risks estimated for a maximally exposed resident in an urban or rural setting.

Exhibit B-31. Ingestion Rates for Rural Gardeners

Product	Screening-Level Consumer Ingestion Rate						Units
	Infants <1 yr	Child 1–2 yrs	Child 3–5 yrs	Child 6–11 yrs	Child 12–19 yrs	Adult 20–70 yrs	
Exposed Fruit[a]	NA	12.7	5.41	6.98	3.41	2.37	g/kg-day
Exposed Vegetable[a]	NA	10.7	3.47	3.22	2.35	3.09	g/kg-day
Protected Fruit[a]	NA	44.8	32.0	23.3	7.44	15.1	g/kg-day
Protected Vegetable[a]	NA	3.88	2.51	2.14	1.85	1.81	g/kg-day
Root Vegetable[a]	NA	7.25	4.26	3.83	2.26	2.49	g/kg-day
Eggs[a]	N/A	4.90	3.06	1.90	1.30	1.31	g/kg-day
Other							
Soil	NA	200[b]	200[b]	201[c]	201[c]	201[c]	mg/day

NA = not applicable

[a]Primary source for values was the 1987–1988 NFCS survey; compiled results are presented in Chapter 13 of the 2011 Exposure Factors Handbook (U.S. EPA 2011a). When data were unavailable for a particular age group, the intake rate for all age groups was multiplied by the age-specific ratio of intake based on national population intake rates from CSFII. Ingestion rates presented are the 90th percentile values.

[b]These values are the recommended "upper percentile" value for children from EPA's 2011 EFH, Chapter 4, Table 4-23. The 2008 CSEFH and 2011 EFH included a high-end value associated with pica only, but this value has not been used.

[c]These values are 90th percentile adult ingestion rates calculated in Stanek et al. 1997, and they are used to represent older children and adults.

Exhibit B-32. Ingestion Rates for Urban Gardeners

Product	Screening-Level Consumer Ingestion Rate						Units
	Infants <1 yr	Child 1–2 yrs	Child 3–5 yrs	Child 6–11 yrs	Child 12–19 yrs	Adult 20–70 yrs	
Exposed Fruit[a]	NA	6.14	2.60	2.52	1.33	1.19	g/kg-day
Exposed Vegetable[a]	NA	3.48	1.74	1.39	1.07	1.38	g/kg-day
Protected Fruit[a]	NA	16.6	12.4	8.50	2.96	5.19	g/kg-day
Protected Vegetable[a]	NA	2.46	1.30	1.10	0.78	0.86	g/kg-day
Root Vegetable[a]	NA	2.52	1.28	1.32	0.94	1.03	g/kg-day
Eggs[a]	N/A	2.46	1.42	0.86	0.58	0.606	g/kg-day
Other							
Soil	NA	200[b]	200[b]	201[c]	201[c]	201[c]	mg/day

NA = not applicable

[a]Primary source for values was the 1987–1988 NFCS survey; compiled results are presented in Chapter 13, Table 13-58 to Table 13-62, of the 2011 Exposure Factors Handbook (U.S. EPA 2011a). When data were unavailable for a particular age group, the intake rate for all age groups was multiplied by the age-specific ratio of intake based on national population intake rates from CSFII. Ingestion rates presented are the mean values.

[b]These values are the recommended "upper percentile" value for children from EPA's 2011 EFH, Chapter 4, Table 4-23. The 2008 CSEFH and 2011 EFH included a high-end value associated with pica only, but this value has not been used.

[c]These values are 90th percentile adult ingestion rates calculated in Stanek et al. 1997, and they are used to represent older children and adults.

B.7.2 Default Screening-Level Population-Specific Parameter Values

The screening-level values for body weights (BWs) for the RTR screening threshold analysis, which serve as the default values, are mean values and are presented in Exhibit B-33. As stated in Section B.6.3.1 of this attachment, EPA recommends using the mean BW for each age group when using upper percentile values for ingestion rates. Use of the mean body weights introduces no bias toward over- or underestimating risk. The default exposure duration for each age group also is presented in Exhibit B-33.

Exhibit B-33. Mean Body Weight Estimates for Adults and Children[a]

Lifestage (years)	Duration (years)	Mean Body Weight (kg)
Adult[b] (20-70)	50	80.0
Child <1[c]	1	7.83
Child 1-2[c]	2	12.6
Child 3-5[d]	3	18.6
Child 6-11[e]	6	36.0
Child 12-19[f]	8	64.2

[a]Sources: U.S. EPA 1997, 2008a

[b]EPA-recommended value (U.S. EPA 2011a).

[d]These values were obtained directly from Table 8-3 of the 2008 CSEFH.

[e]Each BW represents a time-weighted average of BWs for age groups 6 to <11 years and 11 to <16 years from Table 8-3 of the 2008 CSEFH. Original sample sizes for each of these age groups can also be found in Table 8-3.

[f]These values were calculated as time-weighted average BW for age groups 11 to <16 years and 16 to <21 years from Table 8-3 of the 2008 CSEFH. The direction of the possible bias is unknown. The values match the estimate based on Table 8-22 of the NHANES IV data as presented by Portier et al. (2007).

B.7.3 Default Chemical-Specific Parameter Values for Screening Analysis

Exhibit B-34 presents chemical-specific parameter values for the screening-level analysis. Values for bioavailability when ingested in soil (*Bs*), mammalian metabolism factors (*MF*), correction factors for belowground produce ($VG_{rootveg}$), wet deposition fractions (*Fw*), air to plant

transfer factors (Bv_{AG}), root concentration factors (*RCF*), and soil-water partition coefficient (*Kds*) are presented.

Only single estimates were developed for each of these parameters for HHRAP (U.S. EPA 2005a), and the potential direction and magnitude of bias toward over- or underestimating risks were not investigated in this assessment. The inputs that are both chemical-specific and plant-type-specific, as presented in Exhibit B-10, are *not* repeated here. Finally, Exhibit B-35 presents biotransfer factors for each of the chemicals and animal types for which screening threshold emissions were calculated.

Exhibit B-34. Chemical-Specific Parameter Values[a]

Parameter	Description	Benzo(a)-pyrene	Cadmium	Mercuric chloride	Methyl mercury	2,3,7,8-TCDD	Arsenic	Units
Bs	Soil bioavailability factor for livestock	1	1	1	1	1	1	unitless
SoilAdjFactor	Soil bioavailability factor	1	1	1	1	1	0.6[b]	unitless
MF	Mammalian metabolism factor	0.01	1	1	1	1	1	unitless
VGrootveg	Empirical correction factor for belowground produce, i.e., tuber or root vegetable, to account for possible overestimate of the transfer of chemicals from the outside to the inside of bulky tubers or roots (based on carrots and potatoes)	0.01	1	1	0.01	0.01	1	unitless
Fw	Fraction of wet deposition that adheres to plant surfaces; 0.2 for anions, 0.6 for cations and most organics	0.6	0.6	0.6	0.6	0.6	0.6	unitless
BvAG	Air-to-plant biotransfer factor for aboveground produce for vapor-phase chemical in air	174,523	0	1,800	0	65,500	0	[mg/g produce DW]/[mg/g air]
RCF	Chemical-specific root concentration factor for tubers and root produce	9,180	0	0	0	40,002	0	L soil pore water/kg root WW
Kds	Chemical-specific soil/water partition coefficient	7,750	332	58,000	7,000	31,126	2,512	L soil pore water/kg soil DW

[a]Values presented in this exhibit are also presented in previous exhibits; however exact values used in the assessment are presented here, rather than values restricted by significant figures. In addition, only values for those chemicals that are specifically used in the screen are provided here.
[b]From U.S. EPA (2012). Compilation and Review of Data on Relative Bioavailability of Arsenic in Soil. Relative bioavailability (RBA) of arsenic in soils compared with arsenate dissolved in water. Fewer than 5 percent of 103 estimates of RBA of arsenic exceeded 0.60 (in vivo studies of juvenile swine, n=64; monkeys, n=24; and mice, n=15).

Exhibit B-35. Chemical and Animal-Type Specific Biotransfer Factor (Ba) Values
([mg chemical/kg WW tissue or dairy] / [mg chemical intake/day] = day/kg WW tissue or dairy)

Chemical	Beef	Dairy	Pork	Eggs	Poultry
Benzo(a)pyrene	3.8E-02	8.0E-03	4.6E-02	1.6E-02	2.8E-02
Cadmium	1.2E-04	6.5E-06	1.9E-04	2.5E-03	1.1E-01
Mercuric chloride	1.1E-04	1.4E-06	3.4E-05	2.4E-02	2.4E-02
Methyl mercury	1.2E-03	1.7E-05	5.1E-06	3.6E-03	3.6E-03
2,3,7,8-TCDD	2.6E-02	5.5E-03	3.2E-02	1.1E-02	1.9E-02
Arsenic	2.0E-03	6.0E-05	1.0E-02	7.0E-02	3.0E-02

B.7.4 Screening-Level Parameter Values for Nursing Infant Exposure

For dioxins, chemical intake via breast milk by nursing infants was estimated using the model presented in EPA's *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions* (MPE, U.S. EPA 1998). The assumption that lactational transfer of dioxins to the infant occurs via the lipid-phase of milk appears reasonable. The following screening-level assumptions used in that model should bias the results toward health-protective estimates of risks.

- Duration of nursing is a full year and no other foods or liquids are consumed by the infant; a more "typical" value would be six months.
- Absorption efficiency of dioxin in food or milk by mother and infant are 100 percent; this assumption might overestimate absorption but probably by no more than 15 percent (see Section B.6.5.2).
- The fat content of human milk is assumed to be 4 percent, a value toward the high end of the reported range of values (1 to 5 percent).
- The maternal chemical intake is estimated using upper percentile ingestion rates for the different homegrown foods (see discussion for Exhibit B-30); this assumption might overestimate total ingestion of homegrown foods by a factor of more than 2 (see Exhibit B-30).
- If the fraction of the maternal body burden of dioxin that is in the body fat compartment is greater than 90 percent, as suggested by ATSDR (1998), then actual exposures of the infant may be less than estimated.

There also are parameter values and assumptions for the lipid-phase breast-milk pathway for which possible bias is unknown.

- The accuracy of the model is unknown; it has not been verified or validated with empirical data.
- Using a half-life of 10 years for dioxins may over- or under-estimate risks.

Finally, there is one assumption that might possibly introduce some bias toward underestimating risks. The results are sensitive to the biological half-life of the chemical in the mother relative to the length of her exposure prior to the lactation period. Using an exposure duration for the mother equal to the assumed half-life for dioxins, 10 years, may underestimate the duration of exposure of the mother.

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Attachment C. Dermal Risk Screening

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CONTENTS, ATTACHMENT C

C.1 Hazard Identification and Dose Response Assessment....C-5

C.2 Dermal Exposure Estimation....C-6
C.2.1 Equations for Estimating Dermal Exposure....C-6
C.2.2 Exposure Factors and Assumptions....C-7
C.2.3 Receptor-Specific Parameters....C-8
C.2.4 Scenario-Specific Parameters....C-8
C.2.5 Chemical-Specific Parameters....C-9

C.3 Screening-Level Cancer Risks and Non-Cancer Hazard Quotients....C-10
C.3.1 Dermal Cancer Risk....C-10
C.3.2 Dermal Hazard Quotient....C-11

C.4 Dermal Screening Results....C-11

C.5 References....C-12

Exhibits, Attachment C

Exhibit C-1. Cancer Slope Factors and Reference Doses Based on Absorbed Dose....C-6

Exhibit C-2. Receptor-Specific Body Surface Area Assumed to be Exposed to Chemicals....C-8

Exhibit C-3. Scenario-Specific Exposure Values for Water and Soil Contact....C-9

Exhibit C-4. Chemical-Specific Dermal Exposure Values for Water and Soil Contact....C-10

Exhibit C-5. Dermal Noncancer HQs – Summed for Water and Soil Exposures....C-11

Exhibit C-6. Dermal Cancer Risks – Summed for Water and Soil Exposures....C-11

Exhibit C-7. Comparison of Ingestion Risk/HQ to Dermal Risk/HQ ……C-12

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Non-inhalation exposure to PB-HAPs can occur by dermal contact with PB-HAP-contaminated soil and water. Although dermal absorption of chemicals that are originally airborne generally is considered a relatively minor pathway of exposure compared to other exposure pathways, in certain settings it can be significant (U.S. EPA 2006, Cal/EPA 2012). This section demonstrates that for the conservative tiered screening scenario developed for RTR multipathway evaluation, the dermal exposure route is not a significant risk pathway compared with the ingestion pathway. In general, the RTR dermal assessment follows the protocol for evaluating a reasonable maximum exposure as described in EPA's *Risk Assessment Guidance for Superfund (RAGS), Volume I: Human Health Evaluation Model, Part E, Supplemental Guidance for Dermal Risk Assessment* (U.S. EPA 2004)

C.1 Hazard Identification and Dose Response Assessment

To assess the potential contribution of dermal exposure to non-inhalation exposure, we evaluated the potential for cancer and chronic non-cancer effects for the five PB-HAPs currently assessed in the multipathway screening evaluation for RTR: arsenic, cadmium, divalent mercury, 2,3,7,8-TCDD, and benzo(a)pyrene. EPA has not developed carcinogenic potency slope factors (CSFs) and non-cancer reference doses (RfDs) specifically for evaluating potential human health concerns associated with dermal exposure to PB-HAPs. Instead, dermal toxicity values can be derived from oral toxicity values via route-to-route extrapolation by adjusting for gastrointestinal (GI) absorption. EPA recommends making this adjustment only when GI absorption of the chemical is significantly less than 100% (i.e., less than 50 percent). Otherwise, a default value of complete (100 percent) oral absorption is assumed, and no adjustment is made (U.S. EPA 2004).

The absorbed cancer slope factor (CSF_{ABS}) is based on the oral cancer slope factor (CSF_O) and the fraction of the contaminant absorbed in the gastrointestinal track (ABS_{GI}), as follows:

$$CSF_{ABS} = \frac{CSF_o}{ABS_{GI}}$$

where:

CSF_{ABS} = Absorbed slope factor $(mg/kg\text{-}day)^{-1}$

CSF_O = Oral slope factor $(mg/kg\text{-}day)^{-1}$

ABS_{GI} = Fraction of chemical absorbed in gastrointestinal tract (unitless)

The absorbed reference dose (RfD_{ABS}) is based on the oral reference dose (RFD_O) and the fraction of the contaminant absorbed in the gastrointestinal tract (ABS_{GI}), as shown below.

$$RfD_{ABS} = RfD_o \times ABS_{GI}$$

where:

RfD_{ABS} = Absorbed reference dose (mg/kg-day)

RfD_o = Oral reference dose (mg/kg-day)

ABS_{GI} = Fraction of chemical absorbed in gastrointestinal tract (unitless)

The GI absorptions for arsenic, 2,3,7,8-TCDD, and all polycyclic aromatic hydrocarbons (PAHs) (which includes benzo[a]pyrene) are estimated to be greater than 50 percent based on data provided in RAGS Part E, Exhibit 4-1. Therefore, as shown in Exhibit C-1, no adjustments to the available oral toxicity values (RfD or CSF) were required for these chemicals. For cadmium and divalent mercury, adjustments were made based on absorption data provided in RAGS Part E, Exhibit 4-1. The absorbed RfDs for cadmium and divalent mercury, adjusted to account for GI absorption, also are provided in Exhibit C-1.

Exhibit C-1. Cancer Slope Factors and Reference Doses Based on Absorbed Dose

PB-HAP	Fraction of Contaminant Absorbed in GI Tract (ABS_{GI}) (unitless)	Absorbed Cancer Slope Factor (CSF_{ABS}) $(mg/kg\text{-}day)^{-1}$	Absorbed Reference Dose (RfD_{ABS}) (mg/kg-day)
Arsenic	No adjustment required[a]	1.5E+00	3.0E-04
Cadmium Compounds	0.05	NA	2.5E-05[b]
Divalent Mercury	0.07	NA	2.1E-05[c]
2,3,7,8-TCDD	No adjustment Required[a]	1.5E+05	7.0E-10
Benzo[a]pyrene	No adjustment Required[a]	1.0E+00	3.0E-04

NA = Not applicable
[a]According to RAGS Part E, Exhibit 4-1, GI absorption is expected to be greater than 50%.
[b]Cadmium RfD for water = 5.0E-4.
[c]Divalent mercury RfD for = 3.0E-4.

C.2 Dermal Exposure Estimation

Dermal exposures and risks resulting from absorption of the chemical through the skin from contact with contaminated water and soil were evaluated for the RTR screening scenario. Individuals were assumed to be exposed on a fraction of their bodies (i.e., their head, forearms, hands, lower legs, and feet) to contaminated soil from the TRIM.FaTE surface soil parcel with the highest concentration (Farm) on a daily basis. For the water evaluation, individuals were assumed to be exposed to contaminated surface water with the same PB-HAP concentration as the TRIM.FaTE screening scenario lake over their entire bodies on a daily basis.

C.2.1 Equations for Estimating Dermal Exposure

The general equation used to estimate dermal absorbed dose (DAD) for water or soil is shown below, and is expressed in milligrams of PB-HAP per kilogram of receptor body weight per day (mg/kg-day). DADs are calculated separately for the water and soil pathways and then added together for each age group.

$$DAD = \frac{DA_{event} \times EV \times ED \times EF \times SA}{BW \times AT}$$

where:

DA_{event} = Absorbed dose per event; chemical-specific; equation for DA_{event} also differs depending on water or soil contact (mg/cm^2-event)

EV = Event frequency (events/day)

ED = Exposure duration (years)

EF = Exposure frequency (days/year)

SA = Skin surface area available for contact (cm^2)

BW = Body weight (kg)

AT = Averaging time; for non-cancer effects, equals ED x 365 days/year; for cancer effects, equals 70 years x 365 days/year (days)

DA_{event} is estimated to be the total dose absorbed through the skin at the end of exposure and the equation for calculation is different for organic and inorganic chemicals in water and for soil. The equations for calculating these chemical-specific DA_{event} values for water contact are provided in RAGS Part E, Chapter 3 (see Equations 3.2 – 3.4). For soil, the equation for calculating these chemical-specific DA_{event} values is provided in RAGS Part E, Chapter 3 (see Equation 3.12).

Water – Organic Chemicals: $$DA_{event} = C_w \times 2 \times FA \times K_p \sqrt{\frac{6 \times \tau_{event} \times t_{event}}{\pi}}$$

Water – Inorganic Chemicals: $$DA_{event} = C_w \times K_p \times t_{event}$$

Soil – All Chemicals: $$DA_{event} = C_s \times AF \times ABS \times CF$$

where:

DA_{event} = Absorbed dose per event (mg/cm^2-event)

C_w / C_s = Chemical concentration in water (mg/cm^3) or soil (mg/kg)

K_p = Chemical-specific dermal permeability coefficient of compound in water (cm/hr)

FA = Chemical-specific fraction absorbed; accounts for loss due to the regular shedding of skin cells of some chemical originally dissolved into skin (unitless)

τ_{event} = Chemical-specific lag time per event (hr/event)

t_{event} = Receptor-specific event duration (hr/event)

AF = Receptor- and activity-specific adherence factor of soil to skin (mg/cm^2-event)

ABS = Chemical-specific dermal absorption fraction (unitless)

CF = Conversion factor (10^{-6} kg/mg)

C.2.2 Exposure Factors and Assumptions

The exposure parameters included in this assessment and their default and other value options are summarized in this subsection. Default values were selected to result in a highly conservative estimate of exposure (i.e., exposures are likely overestimated). Parameter values

were primarily obtained or estimated from RAGS Part E (U.S. EPA 2004) and the Child-Specific Exposure Factors Handbook (CSEFH, U.S. EPA 2008). Receptor-and scenario-specific exposure assumptions are discussed first, and a discussion of chemical-specific parameters values follows. Estimated water and soil exposure concentrations are presented at the end of this subsection.

C.2.3 Receptor-Specific Parameters

Dermal exposures and risks were estimated for the same age groups used in the ingestion exposure assessment: adults (ages 20 to 70 years) and five child age groups: <1 year; 1 to 2 years; 3 to 5 years; 6 to 11 years; and 12 to 19 years. The body weight values used in the ingestion exposure assessment were used in the dermal exposure assessment.

Body surface areas (SAs) for water and soil exposures for adults were calculated using Appendix C, Exhibit C-1, of RAGS Part E. For children, SAs for water and soil exposures for the five children's age groups were estimated using Tables 7-1 and 7-2 of the CSEFH, respectively. For SA (water), individuals were assumed to shower or bathe in the water with 100 percent of their body exposed. For SA (soil), it was assumed that individuals were exposed on a fraction of their total body, specifically their head, forearms, hands, lower legs, and feet. Based on information provided in RAGS Part E, the SA for forearms was calculated using the SA for arms and assuming a forearm-to-arm ratio of 0.45, and the SA for lower legs was estimated using the SA for legs and assuming a lower leg-to-leg ratio of 0.4.

Values for body SA by age group are summarized in Exhibit C-2.

Exhibit C-2. Receptor-Specific Body Surface Area Assumed to be Exposed to Chemicals

Age Group[a] (years)	Surface Area for Water Exposure (cm^2)	Surface Area for Soil Exposure (cm^2)
Adult 20-70	18,150[g]	6,878[h]
Child <1[b]	3,992	1,772
Child 1-2[c]	5,700	2,405
Child 3-5[d]	7,600	3,354
Child 6-11[e]	10,800	4,501
Child 12-19[f]	17,150	6,906

[a]Sources for the child groups included Table 7-1 (total body surface area for SA-Water), and Table 7-2 (fraction of total body surface area for SA-Soil) of the 2008 CSEFH.

[b]Represents a time-weighted average for age groups birth to <1 month, 1 to <3 months, 3 to <6 months, and 6 to <12 months.

[c]Represents a time-weighted average for age groups 1 to <2 years and 2 to <3 years.

[d]Values for age group 3 to <6 years in the 2008 CSEFH.

[e]Values for age group 6 to <11 years in the 2008 CSEFH. Represents a conservative (i.e., slightly low) estimate for ages 6 through 11 years since 11-year olds are not included in this CSEFH age group.

[f]Represents a time-weighted average for age groups 11 to <16 years and 16 to <21 years. Note that estimated values include 11-year-olds and individuals through age 20, which contributes to uncertainty in the estimates for 12 to 19 years.

[g]Represents the average total surface area of adults from Table C-1 of RAGS Part E.

[h]Represents the average surface area of adults for head, forearms, hands, lower legs, and feet from Table C-1 of RAGS Part E.

C.2.4 Scenario-Specific Parameters

Exhibit C-3 summarizes the exposure values related to frequency and duration of contact. In general, these are the recommended defaults for calculating a reasonable maximum exposure (RME) for a residential scenario as proposed by EPA in RAGS Part E, Chapter 3.

Exhibit C-3. Scenario-Specific Exposure Values for Water and Soil Contact

Exposure Parameter	Receptor	Value	Source
Water Contact			
Event Duration (t_{event}) (hr/event)	Child	1	Reasonable maximum exposure scenario for showering/bathing from RAGS Part E, Exhibit 3-2
	Adult	0.58	
Soil Contact			
Soil Adherence Factor (AF) (mg/cm^2)	Child	0.2	For children, value is geometric mean value for children playing (wet soil) and for adults, value is geometric mean value for an adult farmer from RAGS Part E, Exhibit 3-3
	Adult	0.1	
Both Media			
Event Frequency (EV) (events/day)	All	1	Reasonable maximum exposure scenario from RAGS Part E, Exhibits 3-2 and 3-5.
Exposure Frequency (EF) (days/year)	All	350	
Exposure Duration (ED) (years)	Child <1	1	Represents the number of years included in the age group; also used in ingestion exposure calculations.
	Child 1-2	2	
	Child 3-5	3	
	Child 6-11	6	
	Child 12-19	8	
	Adult 20-70	50	
Averaging Time (AT) (days)	For cancer assessment, an AT equal to a lifetime (70 years) × 365 days/year is used. Same value used in ingestion exposure calculations. For non-cancer assessment, an AT equal to the exposure duration (ED) × 365 days/year is used, so AT will vary by receptor group. Same value used in ingestion exposure calculations.		

C.2.5 Chemical-Specific Parameters

The chemical-specific parameters required to quantitatively evaluate dermal pathway exposures are listed in Exhibit C-4. For the water concentration in the dermal analysis, the modeled TRIM.FaTE chemical concentration in the screening scenario pond at the screening threshold emission rate was used. For the soil concentration, the modeled TRIM.FaTE chemical concentration in surface soil in parcel Farm (tilled soil, closest to facility) of the screening scenario at threshold emission rate was used. This same soil concentration was also used in ingestion exposure calculations for soil ingestion.

Dermal absorption of chemicals in water is based on the use of a dermal permeability coefficient (K_p), which measures the rate that a chemical penetrates the skin. Dermal absorption of soil-bound chemicals is based on the use of a dermal absorption fraction (ABS), which is a measure of how much of a chemical the skin absorbs through contact with soil.

Exhibit C-4. Chemical-Specific Dermal Exposure Values for Water and Soil Contact

PB-HAP	Arsenic	Cadmium	Divalent Mercury	2,3,7,8-TCDD	Benzo[a] pyrene	Source
Chemical concentration in Water (Cw) (mg/cm^3)	4.19E-10	7.48E-09	3.85E-10	9.03E-18	1.01E-11	TRIM.FaTE modeled concentration in screening scenario lake
Chemical concentration in Soil (Cs) (mg/kg)	1.66E-02	1.57E-01	3.36E-02	7.41E-10	9.77E-04	TRIM.FaTE modeled concentration in surface soil in farm in screening scenario
Permeability coefficient in water (Kp) (cm/hour)	0.001	0.001	0.001	0.81	0.7	Values from RAGS Part E, Exhibits B-3 (organics) and B-4 (inorganics)
Fraction absorbed water (FA) (unitless)	NA	NA	NA	0.5	1.00	Values from RAGS Part E, Exhibits B-3; only used for organic chemicals
Lag time per event (event) (hr/event)	NA	NA	NA	6.82	2.69	Values from RAGS Part E, Exhibits B-3; only used for organic chemicals
Dermal absorption fraction (ABS) from soil (unitless)	0.03	0.001	0.045[a]	0.03	0.13	Values from RAGS Part E, Exhibit 3-4, unless otherwise noted

[a]Value obtained from Bioavailability in Environmental Risk Assessment (Hrudey et al. 1996).

C.3 Screening-Level Cancer Risks and Non-Cancer Hazard Quotients

Toxicity values were used in conjunction with exposure information to evaluate the potential for cancer risks and non-cancer health hazards. Risk estimation methods are presented below.

C.3.1 Dermal Cancer Risk

Cancer risk for the dermal route was calculated as the product of the age-specific *DAD*s and the absorbed CSF for each chemical, as follows:

$$Dermal\ Cancer\ Risk = DAD \times CSF_{ABS} \qquad \textbf{Eqn. C-1}$$

where:

DAD = Dermal Absorbed Dose (mg/kg-day)

CSF_{ABS} = Absorbed cancer slope factor $(mg/kg\text{-}day)^{-1}$

Lifetime dermal cancer risks were calculated for 2,3,7,8-TCDD and benzo[a]pyrene. The total risk accounts for dermal exposures that an individual might receive from these PB-HAPs in water plus soil over his or her lifetime (70 years).

C.3.2 Dermal Hazard Quotient

Dermal hazard quotient (HQ) was estimated as the ratio of age-specific *DAD*s to the absorbed RfD for each chemical, as shown below:

$$Dermal\ HQ = DAD / RfD_{ABS} \qquad \textbf{Eqn. C-2}$$

where:

DAD = Dermal Absorbed Dose (mg/kg-day)

RfD_{ABS} = Absorbed reference dose (mg/kg-day)

The aggregate HQ accounts for exposures that an individual in a receptor group may receive from the PB-HAP in water and soil over the exposure duration. Non-cancer hazard is not additive *across* the age groups evaluated here.

C.4 Dermal Screening Results

Exhibit C-5 provides estimated dermal non-cancer hazards by age group and Exhibit C-6 provides estimated lifetime cancer risks from dermal exposures. Risks and hazards are summed for exposure to both water and soil. Soil and water concentrations that resulted in the Tier 1 threshold emissions rates are used as the media concentrations; these concentrations resulted in cancer risks of 1E-6 or HQs of 1.0 for a combined farmer and fisher receptor. The highest HQ value for dermal exposures was 0.003, representing divalent mercury exposure for children aged 1 to 2. This is approximately 320 times less than the estimated ingestion HQs associated with the screening scenario (i.e. emissions of divalent mercury in the screening scenario resulted in an ingestion HQ of 1, based on the ingestion of methyl mercury). The highest estimated individual lifetime cancer risk associated with potential dermal exposures was 1.0E-8 for arsenic; this value is approximately 100 times smaller than the estimated Tier 1 ingestion risk (i.e., 1E-06). Exhibit C-7 provides the estimated magnitude of difference between the ingestion risks or HQs and those for dermal exposure for each of the five PB-HAPs. These were calculated as the ratio of the risk or HQ from exposure through ingestion to the risk or HQ from dermal exposures. Although As, BaP, and TCDD have RfDs, as noted in Exhibit C-1, their cancer exposure assessments are more of concern; therefore, only lifetime cancer risks and not HQs are presented in Exhibit C-6.

Exhibit C-5. Dermal Noncancer HQs – Summed for Water and Soil Exposures

PB-HAP	HQ Child 1	HQ Child 2	HQ Child 3	HQ Child 4	HQ Child 5	HQ Adult	Max HQ
Cd	0.0004	0.0004	0.0003	0.0002	0.0002	0.00009	**0.0004**
Hg2+	0.003	0.003	0.002	0.002	0.001	0.0006	**0.003**

Exhibit C-6. Dermal Cancer Risks – Summed for Water and Soil Exposures

PB-HAP	Lifetime Risk
As	1.0E-08
BaP	8.4E-09
TCDD	8.7E-10

Exhibit C-7. Comparison of Ingestion Risk/HQ to Dermal Risk/HQ

PB-HAP	Magnitude of Difference
As	99
BaP	119
Cd	2,400
Hg^{2+}	319
TCDD	1,150

Based on these results and taking into consideration the extremely conservative nature of the dermal exposure calculations, EPA has determined that it is not necessary to incorporate dermal exposures in calculating multipathway screening threshold levels. Specifically, the daily exposure durations of 0.58 hour for adults and 1 hour for children used to calculate dermal exposure from water are highly conservative and assume that the individual is bathing in surface water taken directly from a contaminated lake or is swimming in the lake for 350 days of the year. The exposure frequency of 350 days and corresponding skin surface area available for contact with contaminated soils (i.e., head, hands, arms, legs, and feet) likely also grossly overestimates dermal exposure to soil.

C.5 References

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Attachment D. Summary of TRIM.FaTE Parameters Considered for Inclusion in Tier 2 Assessment

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Exhibit D-1. TRIM.FaTE Parameters Considered for Inclusion in Tier 2 Assessment

Parameter	Mechanism of Potential Influence in TRIM	Assessment of Parameter Significance and Ease in Implementation	Uncertainty in Site-Specific Data for Facilities	Priority for Inclusion
Meteorological Parameters				
Wind direction (% of time wind blows toward the lake and farm)	In previous runs, direct deposition accounted for the bulk of chemical input onto farms and into lakes. Because wind direction is strongly correlated to direct deposition in a given location, media concentrations are potentially highly sensitive to this parameter. Also, because the percentage of time the prevailing wind blows in the direction of lakes and farms can vary considerably across locations, differences in this parameter might also result in significant changes in important environmental concentrations.	**Highly Significant**: Previous sensitivity analyses have confirmed this to be a very sensitive parameter in the Tier 1 Screening modeling set-up. Changing the fraction of time the wind blows toward the lake and farm by a factor of two corresponds to a change in the risk by a factor of two. **Low Effort to Implement:** This variable is relatively straightforward to vary in the Tier 2 screening scenarios.	**Low to Moderate:** The average fraction of time the wind blows in a given direction can be estimated for any surface meteorological station. Then, facilities can be linked to the closest surface meteorological station.	**High**
Wind speed	Wind speed can affect the location of the "peak" concentration and deposition patterns in a given model configuration, as well as the risk-distance profile.	**Highly Significant**: Previous sensitivity analyses have confirmed this to be a very sensitive parameter. However, wind speed does not vary widely across U.S. locations which could reduce its potential influence. **Low Effort to Implement:** This variable is relatively straightforward to vary in the Tier 2 screening scenarios.	**Low to Moderate:** The annually-averaged wind speed can be estimated for any surface meteorological station. Then, facilities can be linked to the closest surface meteorological station.	**High**

Parameter	Mechanism of Potential Influence in TRIM	Assessment of Parameter Significance and Ease in Implementation	Uncertainty in Site-Specific Data for Facilities	Priority for Inclusion
Precipitation	Chemicals for which wet vapor or wet particle deposition processes are important are likely to be sensitive to the assumed level of precipitation.	**Highly Significant**: Previous sensitivity analyses have indicated a relatively high sensitivity of risk to precipitation for most PB-HAPs (PAHs, cadmium, and mercury). **Moderate Effort to Implement:** In implementing changes in precipitation in TRIM, care must be taken to also preserve the overall water balance in the model.	**Low to Moderate:** The annually-averaged precipitation rate can be estimated for the subset of surface meteorological stations that capture rainfall data. Then, facilities can be linked to the closest surface meteorological station with available data.	**High**
Mixing height	Greater mixing heights increase the dispersion of pollutants in the atmosphere and consequently reduce deposition to the ground in the areas around the stack. This is likely to be a highly sensitive parameter if there is a sizeable variation in mixing heights between facilities.	**Highly Significant**: Previous sensitivity analyses have shown risk to be very sensitive to mixing height. **Low Effort to Implement:** This variable is relatively straightforward to vary in the Tier 2 screening scenarios.	**Moderate to High:** Mixing height estimates are available for upper air meteorological stations, and this set of stations is more limited than the set of surface meteorological stations. Each surface station can be linked to the closest upper air station to estimate the average mixing height. Then, facilities can be linked to the closest surface meteorological station. The relative uncertainty in mixing height for a given facility is high, given diurnal variations in mixing height and the smaller number of upper air stations.	**High**

Parameter	Mechanism of Potential Influence in TRIM	Assessment of Parameter Significance and Ease in Implementation	Uncertainty in Site-Specific Data for Facilities	Priority for Inclusion
Configurational Parameters				
Distance of lake from stack	Deposition is known to decrease with distance from stack, although this relationship also depends on meteorological parameters such as wind speed and wind direction.	**Significance Difficult to Determine:** Limited results from previous TRIM model runs show an inconclusive relationship between risk and distance from stack, possibly as a result of limited statistical power. Some studies in the literature show a definite decreasing risk gradient with distance but others report too many confounding factors to isolate the precise relationship. **Moderate Effort to Implement:** This variable requires updates to the layout coordinates and requires more effort to vary in the Tier 2 screening scenarios than the meteorological parameters.	**Low:** The lakes within a given radius of each facility can be found using ArcGIS™.	**High**
Distance of farm from stack	Deposition is known to decrease with distance from stack, although this relationship also depends on meteorological parameters such as wind speed and wind direction.	**Significance Difficult to Determine:** Limited results from previous TRIM model runs show an inconclusive relationship between risk and distance from stack, possibly as a result of limited statistical power. Some studies in the literature show a definite decreasing risk gradient with distance but others report too many confounding factors to isolate the precise relationship. **Moderate Effort to Implement:** This variable requires updates to the layout coordinates and requires more effort to vary in the Tier 2 screening scenarios than the meteorological parameters.	**High:** Although the distance to the farm will likely affect risk, it is difficult to determine the precise land parcels near each facility that are actually used for farming now or in the future.	**Medium**

Parameter	Mechanism of Potential Influence in TRIM	Assessment of Parameter Significance and Ease in Implementation	Uncertainty in Site-Specific Data for Facilities	Priority for Inclusion
Watershed: lake area ratio	A higher watershed:lake area ratio potentially increases the chemical input of water-soluble or particle-attached chemicals into the lake. But the associated higher flush rate will likely reduce this effect.	**Significance Difficult to Determine**: Changes in the watershed to lake ratio affect risk, but the interaction depends on other variables involved in the water balance. **Moderate Effort to Implement:** In implementing changes in the watershed:lake ratios in TRIM, care must be taken to also preserve the overall water balance in the model.	**High:** The portion of land serving as a watershed to a particular lake is difficult to determine.	**Medium**
Area and depth of lake	A higher lake area would capture more deposition but this effect might be counterbalanced by the ensuing larger volume of water, which reduces chemical concentration. Similarly, a deeper lake would also reduce concentrations, but this effect might be counterbalanced by the ensuing lower flush rates at a constant level of precipitation/runoff.	**Significance Difficult to Determine**: The impact of these parameters is inconclusive based on current studies using the TRIM model. **Moderate Effort to Implement:** The lake area variable requires updates to the layout coordinates and requires more effort to vary in the Tier 2 screening scenarios than the meteorological parameters. In implementing changes in these variables in TRIM, care must be taken to also preserve the overall water balance in the model.	**High:** While the area of lakes near a facility can be determined using GIS, the depth cannot.	**Medium**

Parameter	Mechanism of Potential Influence in TRIM	Assessment of Parameter Significance and Ease in Implementation	Uncertainty in Site-Specific Data for Facilities	Priority for Inclusion
Physical Parameters				
Flush rate	A higher flush rate out of the lake would result in a higher rate of chemical output from the lake, assuming constant inflow and volume.	**Significance Difficult to Determine**: The impact of this parameter is inconclusive based on current studies using the TRIM model. **Moderate Effort to Implement:** In implementing changes in the flush rate in TRIM, care must be taken to also preserve the overall water balance in the model.	**High:** The flush rate of a lake cannot be determined easily for any lake found near a facility. In addition, erosion rates, watershed information, and lake depth needed to estimate the flushing rate are not readily available.	**Medium**
Runoff rate and fraction	A higher runoff rate (or fraction) would likely result in greater chemical input into the lake for some chemicals but also potentially a higher flush rate out of the lake.	**Significance Difficult to Determine**: The impact of this parameter is inconclusive based on current studies using the TRIM model. **Moderate Effort to Implement:** In implementing changes in the runoff rate and fraction in TRIM, care must be taken to also preserve the overall water balance in the model.	**High:** As with the flush rate, the runoff rate and fraction for any lake near a facility cannot be readily determined.	**Medium**
Erosion rate and fraction	A higher erosion rate would likely result in greater chemical input into the lake for particle-bound chemicals. It would also result in greater chemical transport onto farmlands, but this might be counterbalanced by equally greater erosion off farmland.	**Highly Significant**: Previous analyses have shown risk to be sensitive to this parameter for some chemicals. **Moderate Effort to Implement:** In implementing changes in the erosion rate and fraction in TRIM, care must be taken to also preserve the overall water balance in the model.	**High:** As with the flush rate, the erosion rate and fraction for any lake near a facility cannot be readily determined.	**Medium**

Parameter	Mechanism of Potential Influence in TRIM	Assessment of Parameter Significance and Ease in Implementation	Uncertainty in Site-Specific Data for Facilities	Priority for Inclusion
Chemical Parameters				
Methylation/ demethylation rates (Hg)	For Hg, methylation and demethylation rates in lake sediment and surface water are potentially sensitive parameters affecting risk. A literature survey has indicated a relatively high range for rate constants describing these processes.	**Highly Significant**: Previous analyses run in TRIM have confirmed the high sensitivity of these parameters for Hg. **Low Effort to Implement:** This variable is relatively straightforward to vary in the Tier 2 screening scenarios.	**High:** The specific methylation / demethylation rates for mercury in the vicinity of a specific facility cannot be readily determined.	**Low**
Total phosphorus levels in the lake	The total phosphorus content of a lake is used as part of the TRIM.FaTE parameterization process to estimate the biomass content of different trophic levels. These biomass levels affect the biomagnification of chemicals up the food chain and potentially risk to human consumers of fish.	**Not Significant**: Previous analyses have shown limited sensitivity to total phosphorus levels. This is likely because the empirical equations predicting biomass in each trophic level depend in similar ways on the level of total phosphorus. So changes in total phosphorus do not significantly affect the ratio of biomass between the different trophic levels. **Low Effort to Implement:** This variable is relatively straightforward to vary in the Tier 2 screening scenarios.	**High:** The total phosphorus levels in lakes near a specific facility cannot be readily determined.	**Low**

Attachment E. Analysis of Lake Size and Sustainable Fish Population

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CONTENTS, ATTACHMENT E

E.1 Overview and Methods E-6
E.1.1 Purpose E-6
E.1.2 Angler Behavior E-9
E.1.3 Fish Populations E-10
E.1.4 Lake Fish Productivity E-14
E.1.5 Proportion of Fish Biomass by Trophic Level E-22
E.1.6 Derivation of Lake Sizes for Sustainable WCC Harvest E-29
E.1.7 References E-35

Exhibits

Exhibit E-1. Aquatic Food Web Simulated in TRIM.FaTE E-7

Exhibit E-2. Distribution of Biomass in Aquatic Compartments E-8

Exhibit E-3. Characteristics of 72 Lakes in Eastern North America E-15

Exhibit E-4. One Trophic Classification Standard for Lakes E-16

Exhibit E-5. Predictions of Biomass (B) of Biotic Components of Lakes with Different Total Phosphorus (TP) Concentrations E-18

Exhibit E-6. Reported Compared With Predicted Fish Biomass for Five Reservoirs E-20

Exhibit E-7. Comparison of Predictions of Total Fish Biomass from Total Phosphorus (TP) E-20

Exhibit E-8. Total Fish Biomass in Reservoirs of the United States by Drainage Area E-21

Exhibit E-9. Total Fish Biomass Density in Reservoirs and Lakes from Different Studies E-22

Exhibit E-10. Distribution of Standing Biomass Among Aquatic Compartments Simulated in the Comprehensive Aquatic Systems Model (CASM) for a Florida lake E-24

Exhibit E-11. Estimated Biomass by Aquatic Compartment in Lake Toya, Japan E-25

Exhibit E-12. Carrying Capacity, Biomass (g ww/m^2) of Fish Supported by Each Food Compartment Across 61 Reservoirs by Drainage Area E-26

Exhibit E-13. Proportion of Total Carrying Capacity, Proportion Fish Biomass Supported by each Food Compartment by Drainage Area E-27

Exhibit E-14. Total Fish Biomass by Trophic Level in Wupper Reservoir, Germany E-28

Exhibit E-15. Number of WCC Adult Fish Supported by Lake Size (surface area in acres) and by Total Fish Biomass (TFB) E-31

Exhibit E-16. Total Standing Biomass of WCC Fish (kg) by Lake Size and Total Fish Biomass (TFB) E-32

Exhibit E-17. Estimated Maximum Fish-fillet-ingestion Rate (g/day) Associated with Sustainable Fishing of WCC by Lake Size and Total Standing Fish Biomass (TFB)..... E-34

Acronyms

BC	benthic carnivore (fish, e.g., large catfish)
BI	benthic (sediment-dwelling) invertebrate
BO	benthic omnivore (fish, e.g., smaller bottom feeding fish)
CASM	Comprehensive Aquatic Systems Model
CPUE	catch per unit effort
DOC	dissolved organic carbon
EPA	U.S. Environmental Protection Agency
HAP	hazardous air pollutant
MEI	morphoedaphic index
MVP	minimum viable population (self-sustaining population over decades)
PB-HAP	persistent and bioaccumulative hazardous air pollutant
RTR	Risk and Technology Review
SD	standard deviation
TLn	trophic level n (e.g., TL1, TL2, TL2.5 . . . TL4.5)
TN	total nitrogen
TP	total phosphorus
WCC	water-column carnivore (fish, e.g., walleye, pike)
WCO	water-column omnivore (fish, e.g., yellow perch, sunfish)
WCH	water-column herbivore (fish, e.g., young-of-the-year fish, minnows)

E.1 Overview and Methods

As stated in Section 1.4.1.1 of the TSD, the tiered risk screen for Risk and Technology Review (RTR) persistent and bioaccumulative hazardous air pollutants (PB-HAPs) includes a subsistence fisher who consumes fish from one or more lakes near a facility. The fish at the top of the aquatic food web in a lake should have substantially higher tissue concentrations of PB-HAPs than concentrations in the water or sediments due to bioaccumulation of the PB-HAPs through the food web links. For the Tier 1 screening scenario, we include a single hypothetical lake near the facility. For Tiers 2 and 3, we include actual lakes near the facility. This attachment provides supporting information for Section 3.5.1 of the Technical Support Document (TSD)—Processing Lake Data for Tier 2 Screen. For the remainder of this attachment, the word "angler" is used to refer to the "human fisher" consuming fish at subsistence levels in order to distinguish it from the mammalian "fisher" (*Martes pennanti*), which is used in the some ecological risk assessments.

E.1.1 Purpose

To develop the screening scenarios with an angler, we needed to address two questions:

1. How large does a lake need to be to provide a self-sustaining population(s) of top-trophic-level fish?
2. How much fish can be harvested sustainably from lakes of different sizes?

The assumed high-end fish ingestion rate for an adult angler is 373 grams fish fillet per day (see Section 1.4.1.2 of TSD). A health-protective assumption is that the angler consumes top-trophic-level fish (allows maximal bioaccumulation). Thus, we needed to estimate, in essence, the fish ingestion rates near trophic level 4 (TL4) supported by lakes of different sizes.

Addressing the first question ensured we did not model an angler harvesting more fish than a lake could provide (e.g., removing several pounds per day, 365 days per year, a rate at which the entire fish population would be fished out within weeks or months). The second question estimates how many lakes of what size(s) would be required to meet the angler's daily fish ingestion rate.

E.1.1.1 Methods – Literature Searches

ICF conducted two searches through online bibliographic databases for information on aquatic food webs and biomass distribution within those webs: one in 2005 and one in 2014. In 2005, using standard literature/citation databases (e.g., Elsevier BIOBASE, Enviroline), ICF's information specialist searched citations for articles published from 1975 to 2005. The following search terms and logical variations of these words were used:

- Aquatic, aquatic ecosystem, fish, fisheries, fisheries population
- Lake, river, reservoir, pond, stream (not marine or estuarine)
- Trophic, pyramid, food web, food chain, trophic community structure
- Biomass, bioaccumulation, biomagnification, accumulation

Fugacity, mass-balance, model

The results of this search yielded an initial list of more than 400 publication titles. These titles were reviewed to develop a list of about 100 articles for which abstracts were retrieved. The abstracts were reviewed and used to select 33 publications for retrieval and review. Some of

these publications cited additional relevant literature that we retrieved for review. Where we have used a secondary source to describe the findings of an original source, we cite both the primary and secondary sources.

We conducted a similar literature search in 2014 to identify relevant references published since 2005. We found an initial list of more than 200 publication titles that we reviewed. We retrieved more than 60 abstracts, and selected 31 for retrieval. We reviewed those studies to supplement this documentation and to determine if any literature contradicted key assumptions we made in 2005. Where we have used a secondary source from our 2014 search to describe the findings of an original source, we provide both the original and secondary citations.

E.1.1.2 Methods – Food Web Simulated in TRIM.FaTE

The food web simulated in TRIM.FaTE is reproduced in Exhibit E-1. Values for attributes of each biotic compartment in the TRIM.FaTE-simulated lake are listed in Exhibit E-2. The total fish biomass per unit area simulated by TRIM.FaTE is 5.7 grams fish wet weight/square meter [g ww/m^2], which is typical of lakes in Maine and southern Ontario. The final two columns in Exhibit E-2 show fish biomass and numbers for purposes of evaluating fish harvesting by anglers. The total fish standing biomass is higher, 40 g ww/m^2, to be more representative of lakes across the United States as described in Section E.1.6. The lower fish biomasses were used for TRIM.FaTE so that the fish compartments did not sequester (remove) large quantities of chemical mass from the water column (and sediments).

With limited removal of chemical mass from water and sediments, the TRIM.FaTE simulation is more similar to other aquatic food-web models that assume bioaccumulation in fish and other biota does not change the concentrations of chemical in water or sediments (e.g., Arnot and Gobas 2004; U.S. EPA 2009 KABAM for predicting pesticide bioaccumulation potential in aquatic systems).

Exhibit E-1. Aquatic Food Web Simulated in TRIM.FaTE

Aquatic Biota Compartments (Consumer Groups)	Percentage of Consumer's Diet									
	Algae	Macrophytes	Zooplankton	Benthic Invertebrates	Water Column Herbivores (WCH)	Benthic Omnivore (BO)	Water Column Omnivore (WCO)	Benthic Carnivore (BC)	Water Column Carnivore (WCC)	Sum of % Total Diet
Zooplankton	100	0	0	0	0	0	0	0	0	100
Benthic Invertebrate (BI)	0	0	0	0	0	0	0	0	0	100
Water Column Herbivore (WCH)	0	0	100	0	0	0	0	0	0	100
Benthic Omnivore (BO)	0	0	0	100	0	0	0	0	0	100
Water Column Omnivore (WCO)	0	0	0	0	100	0	0	0	0	100
Benthic Carnivore (BC)	0	0	0	50	0	50	0	0	0	100
Water Column Carnivore (WCC)	0	0	0	0	0	0	100	0	0	100

Exhibit E-2. Distribution of Biomass in Aquatic Compartments

Organism	Weight Per Individual (kg)	Percent of Total Fish Biomass	For TRIM.FaTE		For Fish Harvesting	
			Biomass (g ww/ m^2)	No. Fish Per Hectare	Biomass (g ww/ m^2)	No. Fish Per Hectare
Macrophytes	na	na	500	na	na	na
Zooplankton	5.70E-08	na	6.4	na	na	na
Benthic Invertebrate (BI)	2.55E-04	na	20	na	na	na
Water Column Herbivore (WCH)	0.025	35	2	800	14	5614
Benthic Omnivore (BO)	0.25	35	2	80	14	561
Water Column Omnivore (WCO)	0.25	8.8	0.5	20	3.5	140
Benthic Carnivore (BC)	2	17.5	1	5	7	35
Water Column Carnivore (WCC)	2	3.5	0.2	1	1.4	7
Total Biomass of All Fish	na	100	5.7	na	40	na

Abbreviations: 1 hectare = 10,000 m^2; na = not applicable; No. = number; ww = wet weight.

Of the trophic compartments in Exhibit E-1, two compartments represent top-trophic-level fish: benthic carnivores (BC) and water-column carnivores (WCC). Benthic carnivores are relatively large (e.g., 2 kg) bottom-feeding fish (e.g., catfish, chub) that consume benthic invertebrates and small benthic fish. The BC compartment thus represents the top-trophic-level fish exposed via trophic transfers to chemicals from the sediment compartment. Water-column carnivores are relatively large (e.g., 2 kg) pelagic piscivores (e.g., walleye, lake trout, northern pike), or "game" fish, that feed primarily on smaller fish in the water column. The WCC thus represents the top-trophic-level fish exposed to chemicals dissolved in the water column or adsorbed to suspended sediment particles and algae.

As shown in Exhibit E-1, for the BC compartment, we assume a diet of 50-percent benthic invertebrates (TL2) and 50-percent smaller benthic fish (TL3) that feed on benthic invertebrates (TL2) that feed on detritus in sediments (TL1, not included in Exhibit E-1). That diet composition averages to TL2.5, which means that the BC compartment represents TL3.5. For the WCC compartment (TL4.5), our simplified food web assumes that 100 percent of the WCC diet consists of water column omnivores (WCO, TL3.5, e.g., "pan" fish such as bluegill, other sunfish, white perch). The diet of the WCO can include various types of prey, but for a simplified food web we assume the diet of WCO is 100-percent minnow-sized fish species and young-of-the-year fish (TL2.5) that feed on zooplankton (TL2) and algae (TL1—treated as a phase of the surface water column) in the water column. In reality, many fish species (e.g., rainbow trout) feed on smaller fish and on invertebrates in both the water column and at the sediment surface.

E.1.1.3 Organization of This Report

Given the necessity of answering the two questions posed in Section E.1.1 for purposes of RTR screening, we evaluated several factors and used several simplifying assumptions. For factors

with high natural variability, and for which we could predict whether high-end or low-end values would increase the angler's exposure to PB-HAPs, we selected values that likely would increase the angler's risk. For factors with high natural variability for which we could not predict which end of the range might result in more or less risk, we selected data or made an informed assumption that we thought would represent a central tendency in conditions across the country. We emphasize, however, that lake productivity, fish predator-prey relationships, and species' population dynamics in lakes across the United States are highly variable.

The remainder of this attachment is organized in six sections:

E.1.2 Assumptions about Angler Behavior

E.1.3 Assumptions about Fish Biology

E.1.4 Lake Fish Productivity

E.1.5 Proportion of Fish Biomass by Trophic Level

E.1.6 Lake Size for Sustainable WCC Harvest

E.1.7 References

E.1.2 Angler Behavior

Assumptions regarding angler behavior drove some of our data selections and assumptions used to answer our two questions.

E.1.2.1 Consumption of Top-trophic-level Fish

As stated in Section 3.3.1, Exhibit 31, of the TSD, the angler consumes only top-trophic-level fish. Although the angler might prefer to catch and consume the WCC (TL4.5) game fish species, individual fish in that group are the least abundant and account for the lowest group biomass of all the fish compartments (Exhibit E-2). Fish in the BC (TL3.5) compartment are more abundant and account for more biomass than the WCC compartment (Exhibit E-2) in most lakes of moderate size (as discussed in Section E.1.5, excludes Great Lakes).

We could not predict a priori whether chemical concentrations in the WCC or in the BC compartment would be higher for any given PB-HAP. Depending on chemical Kow (octanol-water partitioning coefficient) and Kd (soil/sediment-water partitioning coefficient), TRIM.FaTE might estimate higher or lower concentrations in the TL3.5 BC fish than in the TL4.5 WCC fish. Given that unknown, we assumed that the angler catches and consumes a 50:50 ratio of fish from the WCC and BC compartments.

E.1.2.2 Sustainable Fish Harvest Rates

The angler lives in the same location for 50 to 70 years. The lake(s) must support fish harvesting by the angler over that period. In other words, the lake should not be "fished out" by the harvest rate required to meet the angler's fish ingestion rate. The productivity of any particular fishery (local population of a species of fish) and the proportion of adult fish that can be harvested sustainably for human consumption are difficult values to estimate.

Models to predict sustainable harvests of different fisheries are numerous and complex. Species-specific parameters key to such models include fecundity with age and size; survivorship of eggs, fry, and juveniles to sexual maturity (recruitment); natural predation pressures; and temporal variation in food availability. We discuss some of those issues later in

Section E.1.3–Fish Populations, Section E.1.3.4–Sustainable Fish Harvest Rates. Angler behavior related to sustainable fishing is discussed below.

Angler fishing pressure is a product of the number of anglers fishing a lake and the time each angler is willing to spend per unit catch. In reality, those factors are not independent of fish abundance per unit area and total number of fish per lake. For purposes of the RTR assessment, however, we assume a single angler is fishing the lake(s) near a facility. We also assume that the angler harvests fish at a subsistence level. In Tier 1 of the human health risk assessment, we assume the single lake provides fish at that level. In Tiers 2 and 3 of the human health risk assessment, if the lake with the maximum chemical concentrations in fish is too small to provide a sustainable harvest at that level, the angler moves to the next lake with the next highest chemical concentrations, and so on, until the desired harvest is met.

Other influences of angler behavior on fish population density and abundance are not included in RTR assessment. For example, fishing "pressure" does not change the abundance of fish in the lake. In actual lakes, as fishing pressure increases, fish abundance generally decreases. For example, in Wolfe Lake in Alberta, Canada, overfishing of walleye has resulted in a decrease of catch-per-unit-effort or time (CPUE) from 0.25 fish/hour in the early 1980s to 0.02 fish/hr in the mid-1990s (Post et al. 2002). In 1969, catching a pike in Lake Kehiwin took approximately 2.5 hours, whereas in 1995 an estimated 25 hours was required (Post et al. 2002). Stocking lakes has been the solution to allow harvesting at levels well above what wild populations could sustain in many locations. For the RTR screen, interactions among angler effort, fish population size and biomass density, and fishing success are not considered. Instead, we assume certain constants for fish harvesting.

E.1.2.3 Other Assumptions about Angler Behavior

Another assumption about angler behavior is that anglers consume only the fillet portion of a fish. According to Ebert et al. (1993), the edible fraction of fish as a proportion of total fresh body weight is 0.4 for salmon, 0.78 for smelt, and 0.3 for all other species. EPA recommends using 0.30 for the consumable fraction of fish (U.S. EPA 1989). For this assessment, we assume that the edible fraction for top-trophic-level fish is 0.33 (i.e., some proportion of fish consumed are salmon-like). The edible fraction of 0.33 is used in the analyses in Section E.1.6 to estimate total fish biomass required to support specified human fish consumption rates.

A final assumption is that the angler consumes 373 g/day of fish fillet. The value is from Burger's (2002) report on fish ingestion rates for avid sport fishers interviewed at the Palmetto Sportsmen's Classic in South Carolina in March 1998. The ingestion rate of 373 g/person-day is the 99th percentile ingestion rate reported by 107 females. EPA used that value in its *National-scale Assessment of Mercury Risk to Populations with High Consumption of Self-caught Freshwater Fish, in Support of the Appropriate and Necessary Finding for Coal- and Oil-fired Electric Generating Units* (U.S. EPA 2011).

E.1.3 Fish Populations

Our initial question in Section E.1.1 was what is the minimum size of a lake that can support a self-sustaining population of top-trophic-level fish? As stated in Section E.1.2, the RTR screening scenario assumes that an angler consumes 373 g ww fish fillet/day (50:50 ratio of BC to WCC) for 50 to 70 years without stocking to maintain the fish population. This section provides background information required for the lake size analyses in Section E.1.6.

First, some basic principles of fish biology are reviewed (Section E.1.3.1). Next, a brief overview of fish population modeling is presented (Section E.1.3.2). To support calculation of the minimum lake acreage required to support a self-sustaining WCC fish population, an

assumption for the minimum viable population (MVP) size is presented (Section E.1.3.3.). Finally, a sustainable adult fish harvest rate is proposed (Section E.1.3.4).

E.1.3.1 Fish Biology

For persons familiar with human health risk assessment or assessment of risks to terrestrial populations of wildlife (e.g., birds, mammals), some important attributes of fish biology are worth stating.

Fish are cold-blooded (i.e., poikilothermic). Their internal body temperatures vary considerably, particularly with the temperature of ambient water in which they live. Few fish (e.g., open-ocean tuna) are sufficiently active swimmers to maintain a core temperature above ambient water. Nor do fish have significant control over absorption of heat from incident sunlight. Thus, fish growth and reproduction vary considerably with latitude and general climatic factors.

Fish are gape feeders. They consume their prey whole, and thus cannot eat fish larger than their "gape," or mouth opening. This results in the typical aquatic "food chain" of smaller fish being consumed by larger fish, which are consumed by still larger fish. The top piscivorous fish (e.g., walleye, pike) in the water column also tend to have wider or longer gapes, or both, for a given body weight compared to lower trophic-level fish (e.g., perch, sunfish) with a smaller gape relative to their body size.

Fish continue to grow over their lifespan. In northern temperate (and southern temperate) regions like the United States, fish tend to reproduce seasonally (once per year). The fastest growth occurs during the summer months. For all fish species, body size increases with age. For the longer-lived species, growth continues over the lifespan, and the age at first reproduction might be delayed for several years. As growth continues after sexual maturity, larger females can produce more eggs than younger, smaller females.

Many attributes of fish populations are density dependent. Survivorship of young ("recruitment") tends to decrease with increasing abundance of adults and other predatory fish species; conversely, higher mortality among adults can release the young and juveniles from predation and competition for food, allowing higher recruitment and growth rates. Individual fish growth rates depend on density to some extent; growth rates tend to decrease with increasing fish numerical and biomass density due to increasing competition for food.

Approximately 10 percent of energy is lost between tropic levels. Limits to surface water primary productivity and inputs of organic materials from terrestrial ecosystems limit the overall fish carrying capacity (K) of any given lake. Losses of energy from one trophic level of fish to the next tend to be on the order of 90 percent (85–95 percent) (UM 2016); loss of energy from one level to the next for warm blooded animals (birds and mammals) is even higher (95–99 percent) because of the energy spent in maintaining body temperature. Thus, ingestion of 10 grams of fish biomass by another fish usually leads to a 1-gram increase in body weight or in egg production in the consumer fish. Fish standing biomass, therefore, tends to decrease with increasing trophic level.

E.1.3.2 Fish Population Modeling

Population modeling often is used in predicting fisheries responses to management options, including sustainable rates of exploitation. A variety of types of population models have long been used in fisheries management (Vaughan et al. 1984): (1) surplus production models (Shaffer 1968); (2) yield models (Gulland 1969; Ricker 1975); (3) stock-recruitment models (Ricker 1975; DeAngelis and Christensen 1979); (4) Leslie Matrix models (Leslie 1945;

Goodyear and Christensen 1984); and (5) bioenergetics models, which examine factors that affect growth of individual fish (Ursin 1967; Stewart 1980). Leslie matrix models have the advantage of incorporating age-specific survivorship, growth, age at sexual maturity, and fecundity rates for females of a population, which is important for longer-lived top-trophic-level fish.

Use of population models in the field of ecological risk assessment began in the 1990s, but it faces many challenges (Barnthouse et al. 2008). One particularly difficult characteristic of natural populations is variation in key life-history parameter values with changes in population density (i.e., density-dependent population regulation) and fish community structure. In general, some additional adult mortality (e.g., fish harvesting) can be compensated by increased growth rates and increased survival of the young to maturity. Estimating MVP and sustainable harvest rates, given density-dependent compensation in populations, is difficult. Density-dependent predator-prey interactions among fish species in the same lake compound the difficulty. For example, Post et al. (2002) found that in lakes with high walleye harvest rates, populations of cyprinids and other TL3 fish increased. The TL3 fish eventually outcompeted juvenile walleye for food, resulting in loss of walleye altogether (Post et al. 2002).

An example of the Leslie matrix-approach is the Purchase et al. (2005) study of harvest rates of walleye and lake trout compatible with sustained fishing of those species in Lake Erie and in the Upper Kesagami Lake in Ontario, Canada. Purchase and colleagues used a modified age-structured Leslie matrix model (Leslie 1945, Caswell 1989, Hayes 2000) to estimate population sustainability under different fishing pressures. The basic equation using the Leslie matrix can be specified by Equation E-1:

$$1 = \sum_{x=1}^{q} l_x \, m_x \, e^{-rx} \qquad \textbf{\textit{Eqn. E-1}}$$

where:

l_x = age-specific survival rates (per year)

m_x = age-specific fecundity (birth rates, per year)

r = Malthusian parameter (per capita population growth rate)

x = age (years)

q = lifespan (years)

With population- and species-specific life-history data, the maximum value of r (r_{max}) can be estimated. That value corresponds, in theory, with a sustainable harvest rate assuming relatively constant environmental conditions and density-independent values for the specified parameters. The realized value of r for a population must exceed zero for long-term existence.

Purchase et al. (2005) analyzed fisheries data for walleye and lake trout in the two lakes, using published data for age of maturity, relative fecundity, and natural mortality from previous studies of the populations. The annual natural adult mortality rates ranged from 0.11 for walleye in Upper Kesagami to 0.35 for walleye in Lake Erie, while reports of early mortality (for eggs through year 1) ranged from 0.99985 for walleye to 0.9957 for lake trout. Purchase et al. (2005) found that estimates of r_{max} were sensitive to estimates of early mortality, adult mortality, and growth rates. Purchase et al. (2005) found larger differences in modeled population growth rates between two populations of the same species in two different lakes than between the two

different species in the same lake. This level of site-specificity is inappropriate for a screening level, nationwide, risk assessment for thousands of facilities.

Post et al. (2008) demonstrated use of a fish production and harvest model (based on the Gordon-Schaefer model included in Clark 2006), which also depends on the logistic population growth function. The model integrates the density dependence of birth and death rates into the single parameter, *r.* The value of *r* declines with increasing density, approach the carrying capacity of a lake, *K*, at a rate that is density-dependent. The productivity of an environment (and the abiotic characteristics) and species life histories determine *K*. This approach, however, requires knowledge of carrying capacity, which depends on overall lake productivity and size. We therefore moved on to other approaches to estimating MVP (Section E.1.3.3) and lake productivity (Section E.1.4).

E.1.3.3 General Estimates of MVP

The MVP, a concept used frequently in conservation biology for animals, is defined as the smallest population that will persist for a specified duration (e.g., 100, 250, 1,000 years) with a given probability (e.g., 95 percent). To estimate an MVP, one must specify a timeframe of interest and an "acceptable" probability of extinction within that period (e.g., Soulé 1987; Akçakaya et al. 1999).

MVP for any given species and location depends on many attributes of the species' biology (e.g., body size, reproductive rate, home range size, habitat patches, connectivity between habitats, variability in environmental characteristics that impact fecundity and survival, probability of local catastrophes). At lower numbers of breeding individuals, the chance that a local population would go extinct because of random environmental and demographic events is higher (Menzie et al. 2008).

Many textbooks and advanced degrees are dedicated to applied ecology and population modeling to inform conservation or resource management efforts. Much of the initial work on MVP investigated the genetic minima required for short-term survival, continuing adaptation to environmental change, and ultimately, long-term evolution. Consequences of inbreeding have been considered the primary threat to short-term population survival, and genetic drift is the principal threat to losing the genetic variation required for adaptation (Shaffer 1987). Several analyses (Senner 1980; Franklin 1980; Soulé 1980; Frankel and Soulé 1981; Lande and Barrowclough 1987) have led to the conclusion that a minimum "effective" population size of about 50 is required for short-term survival (e.g., several generations, decades). Effective population sizes of approximately 500 are necessary to provide adequate genetic variation for continuing adaptation over the longer term (e.g., tens of generations, centuries for some animals) (Shaffer 1981, 1987; FAO/UNEP 1980).

Effective population size, N_e, is a measure of the rate of genetic drift (loss of genetic diversity or inbreeding), and its definition generally depends on the population in question (Rieman and Allendorf 2001). N_e can be estimated mathematically based on stochastic behavior of gene frequencies in a diploid population. Simple models assume a fixed population size, constant fecundity, specified sex ratio, random mating between individuals, and no overlap between generations (see studies cited in NRC 1986). For animals with 50:50 sex ratios, the effective population size is close to the actual breeding adult population size (Ewens et al. 1987).

The Food and Agriculture Organization of the United Nations Environment Programme (FAO/UNEP 1980) pointed out that if a population is held in check at $N_e = 50$, it will lose about one-fourth of its genetic variation after 20 to 30 generations. Thus, to maintain a particular stock for longer than that, its N_e must be increased. As stated in the report, "a rough rule of thumb is

that *G* is approximately equal to N_e, *G* being the number of generations the stock is likely to retain its fitness at a relatively high level" (FAO/UNEP 1980).

We therefore concluded that a minimum of 50 adult fish of one species in the WCC compartment would be needed for a population to be self-sustaining. Given the large number of factors that influence MVP, Ewens et al. (1987) cautioned against using a "rule of thumb" across circumstances.

E.1.3.4 Sustainable Fish Harvest Rates

In addition to identifying an MVP, we needed to estimate what additional adult mortality might be tolerated by a WCC population due to harvesting by the angler in the RTR screening scenario. This introduces additional density-dependent interactions between the angler and the fish population. From an evaluation of 3,500 rainbow trout populations in British Columbia, Post et al. (2008) concluded that fish population abundance depends on the relationship between fishing effort and fish CPUE for four reasons: (1) harvest equals fishing effort multiplied by catch rate; (2) catch rate correlates with fish abundance; (3) abundance depends on the outcome of the fish population interaction with harvesting; and (4) fishing effort is a function of fish abundance.

Modeling the relationships between angler and fish population would require site-specific data, which is not appropriate for a nationwide screening-level assessment. We therefore searched the literature to find estimates of fish harvest rates that are sufficiently conservative to be tolerated by most fish species.

Allen et al. (2009) used an age-structured model and existing fisheries data to evaluate sustainable recreational harvesting of Murray cod (*Maccullochella peelii peelii*), one of the world's largest freshwater fish in southeastern Australia. They concluded that fishing could be sustained if the exploitation rate is maintained under 0.15 (for the current regulation of 50 cm minimum length to take home) to prevent overfishing. At a higher exploitation rate of 0.30, the minimum fish length would need to be at least 70 cm to be sustainable (i.e., for adequate annual spawning).

Johnson (1980) found that an annual exploitation rate of 0.11 (11 percent) of anadromous arctic charr (*Salvelinus alpinus*) by Inuit in northern Canada led to a steady decline in the size of fish. Based on those data, VanGerwen-Toyne & Tallman (2010) recommended that to ensure sustainability, a harvest rate ≤0.05 per year was needed in this very cold environment (Roux et al. 2011).

In a survey of fish communities in 122 lakes in northern Europe, Håkanson and Boulion (2004) concluded that a typical loss from fishing by birds, mammals, and humans approximates 10 percent of the fish biomass in the prey fish compartment (TL3) and 10 percent of the biomass in the predator fish compartment (TL4).

For our lake size analysis, we assumed that anglers could harvest 10 percent of the biomass of pelagic WCC (TL4.5) adult fish each year without diminishing the WCC fish population size or annual productivity. This harvest rate is low enough to allow density-dependent increased survival and growth rates of young and juvenile fish to balance (compensate for) the additional adult mortality.

E.1.4 Lake Fish Productivity

The first question in Section E.1.1 is: How large does a lake need to be to provide a self-sustaining population(s) of top-trophic-level fish? To phrase the question in another way, what

are the combinations of (a) minimum lake size and (b) fish productivity per unit area that could maintain an MVP of 50 adult breeding fish in the WCC compartment? This section focuses on (b) lake fish productivity per unit area.

We emphasize that lake productivity varies with surface area, depth, temperature, latitude, altitude, nutrient status, local hydrogeology, weather extremes, and other factors. Fish population sustainability also depends on lake primary productivity, inputs of organic materials from land, the relative abundance and diversity of invertebrates and other fish species and their feeding relationships, among other factors. Thus, no "single" answer to either question would be "representative" of lakes across the United States for a screening-level risk assessment.

Nonetheless, for the RTR screen, we established one (Tier 1), or possibly more lake(s) (Tiers 2 and 3), and estimated a WCC harvest in those lakes. As background, we first describe general lake characteristics (Section E.1.4.1). Empirical models of lake productivity as it relates to measurable lake attributes are presented next (Section E.1.4.2). Finally, some of the studies that measured fish productivity in specific locations are included to emphasize similarities and differences among lakes (Section E.1.4.3). All three subsections discuss total fish productivity; we conclude this section with our selection of one lake productivity estimate to use for the RTR screen. Fish productivity by trophic level is investigated in Section E.1.5.

E.1.4.1 Lake Characteristics

Exhibit E-3 provides one summary of physical and chemical characteristics of natural lakes in North America based on a sample of 72 lakes of at least 5 hectares in size, located from the Precambrian shield in Central Ontario through sedimentary basin lakes in the eastern United States (Nürnberg 1996). In this sample, lake surface area ranges over 5 orders of magnitude and the mean depth for each lake ranges from 1.8 to 200 m. Exhibit E-3 is not meant to summarize the characteristics of lakes across all regions of the United States.

Exhibit E-3. Characteristics of 72 Lakes in Eastern North America

Variable	Units	Median	Minimum	Maximum	N
Surface Area (A)	ha	64	5	8.2×10^6	72
	km^2	0.64	0.05	8.2×10^4	
	m^2	640,000	50,000	8.2×10^{10}	
Depth, mean (D)	m	7.6	1.8	200	72
D/A	m/km^2	8.0	0.14	48.1	72
Total Phosphorus (TP)	µg/L	8.1	3.3	107	72
Total Nitrogen (TN)	µg/L	324	149	1,000	63
TN/TP		34	11.6	79	63
Chlorophyll	µg/L	2.9	1.0	40	43
Dissolved Organic Carbon (DOC)	mg/L	3.5	1.5	12.0	62

Note: Lakes from central Ontario in the Precambrian shield, from southern Ontario and Quebec, and from the eastern United States in sedimentary basins. N = number of lakes.
Source: Nürnberg (1996).

Although the maximum total phosphorus (TP) concentration in Exhibit E-3 is 107 µg/L for this sample of lakes, TP concentrations in some lakes are much higher.

Some attributes of lakes vary by latitude. For example, lakes in the southeastern United States are considered monomictic, that is, they turn over[26] once per year in the autumn, whereas northeastern lakes also turn over in the spring when the winter ice cover melts (Osidele and Beck 2003). In addition, the longer growing season in the south promotes higher total phytoplankton and microbial production (and higher turnover rates), which can support higher total biomasses of both non-fish and fish trophic groups (Osidele and Beck 2003).

Lakes have been categorized from a biological perspective into three categories generally related to available nutrients and consequent primary productivity: oligotrophic, mesotrophic, and eutrophic (see text box below). Values for several chemical/physical characteristics of lakes that are associated with these categories have been quantified. For example, Exhibit E-4 presents one lake classification standard and associated values for TP, TN, chlorophyll, and water transparency associated with the three lake trophic categories in Canada (colder than most regions in the United States).

Exhibit E-4. One Trophic Classification Standard for Lakes

Trophic Status	Total Phosphorous (mg/m^3)	Total Nitrogen (mg/m^3)	Chlorophyll a (mg/m^3)	Transparency (m)
Oligotrophic	<15	<400	<3	>4.0
Mesotrophic	15–25	400–600	3–7	2.5–4.0
Eutrophic	>25	>600	>7	<2.5

Measurements are average, epilimnetic (layer of water above the thermocline), summer values, in Canadian lakes.
Source: Forsberg and Ryding (1980) as modified by Canfield et al. (1983).

In most lakes, nitrogen concentrations are more than adequate to support maximal primary production; TP tends to be the limiting nutrient. Thus, the inorganic parameter most often related to lake trophic status is TP concentration. Definitions of TP concentration "cutoffs" between lake trophic categories vary slightly among investigators. Using data from several classification cutoffs reported by Nürnberg (1996), we summarize the definitions of trophic categories for lakes with respect to epilimnetic summer values for TP as:

- Oligotrophic: TP <10–15 µg/L
- Mesotrophic: TP 10–15 to 25–30 µg/L
- Eutrophic: TP 25–30 to 100 µg/L
- Hypertrophic: TP >100 µg/L

Lake Trophic Classification - Definitions

Oligotrophic: Waters lacking in plant nutrients and plants and generally rich in oxygen.

Mesotrophic: Stage between oligotrophic and eutrophic with respect to plant nutrients, plant productivity, and water oxygen content.

Eutrophic: Waters rich in mineral and organic nutrients that promote abundant plant life, particularly algae. As the plant material turns over and decays, dissolved oxygen can decline to levels that support few fish.

[26]During summer, a thermocline generally develops as the surface layer of water warms, becomes less dense, and therefore floats above the bottom layer of colder water (in lakes deep enough to develop a thermocline). In the fall, the surface water layer cools, becomes similar in density to the bottom layer, and they can mix (turn over) with the nutrient-laden bottom waters mixing with the nutrient-depleted surface water. In northern freshwater lakes, ice cover keeps water at the surface colder than in the remainder of the lake; when the ice cover melts, the dense colder surface layer again mixes with the remaining lake waters. TRIM.FaTE does not simulate lake turnovers.

Shallow lakes with large stands of macrophytes can show different relationships between TP and phytoplankton, oxygen, and transparency because of the phosphorus tied up in the macrophytes (Canfield et al. 1983).

The biomass of fish (and the number of trophic levels supported) depends on lake size and the general productivity of a lake per unit area. Lake productivity depends on many factors, including latitude, seasonal temperatures, nutrients supporting algae, and inputs of organic materials (e.g., leaf litter) from terrestrial habitats and from emergent vegetation (allochthonous inputs). For example, in sub-catchments within a 275-hectare watershed in Ontario, Canada, Tanentzap et al. (2014) found that near-shore forested and wetland sub-catchment areas around Daisy Lake export more organic material to the lake than other sub-catchments. They estimated that at least 34 percent of yellow perch (*Perca flavescens*) biomass in the lake is supported by terrestrial primary production via organic inputs that enhance bacterial biomass that enhances biomass in larger zooplankton, which enhances production of young-of-the-year fish. In areas with high forest cover, they estimated that up to 66 percent of fish biomass was supported by organic loading from terrestrial primary production. TRIM.FaTE does not simulate export of organic materials from terrestrial parcels to the lake(s).

E.1.4.2 Predicted Lake Productivity – Nutrient Status and Fish Biomass

As stated above, climatic factors play a large role on a global or hemispheric scale, but at regional scales, many researchers have found “morphometric” (e.g., surface area, maximum depth, mean depth) and “edaphic” (e.g., nutrient content, dissolved oxygen, acidity) indicators for lakes correlate with overall fish productivity. Several versions of the morphoedaphic index (MEI) were developed starting in the 1960s and 1970s to combine lake morphology and nutrient status to estimate fish yields (Cote et al. 2011).

The literature on productivity and standing crop (biomass) of fish and other trophic groups in lakes is extensive and is not reviewed here. As stated in Section E.1.4.1, one physical/chemical attribute of lakes that provides high predictive power for biomass in aquatic ecosystems is the often-limiting nutrient TP. Other characteristics, such as total lake surface area, ratio of surface area to mean depth, dissolved organic carbon (DOC), macrophyte biomass, transparency, and an MEI based on several abiotic and biotic measures, have also been examined for their predictive power. The simplest relationship with high predictive powers, however, relates total fish biomass to lake TP.

Peters (1986) evaluated empirical relationships between TP and biomass in various categories of organisms in lakes developed by other researchers (e.g., Bird and Kalff 1984; Hanson and Legget 1982; Pace 1986). Categories included bacteria, nanoplankton, “net” plankton, microzooplankton (e.g., rotifers and flagellated or ciliated protozoa), and macrozooplankton (e.g., *Daphnia*, copepods, amphipods, fish larvae). Peters converted all biomass to units of grams wet weight per square meter (g ww/m^2). Exhibit E-5 presents those models along with predictions of total biomass for each group for 5, 10, and 50 µg [TP]/L.

Relationships for bacteria and plankton were initially reported in biomass per unit volume. Peters (1986) converted them to biomass per unit area by assuming that bacteria and planktonic organisms occur only in the euphotic zone, the depth of which is given by Equation E-2 from Peters (1986):

$$Depth\_of\_Euphotic\_Zone\ (m) = 24 \times TP\ (mg/m^3)^{-0.28} \qquad \textbf{Eqn. E-2}$$

Note that this equation indicates that the more abundant plankton of more eutrophic lakes should be concentrated in a shallower euphotic zone (the depth of light penetration decreases with increasing concentrations of algae at the surface of more eutrophic lakes). Peters (1986) converted zooplankton dry weight to wet weight assuming a 1:10 ratio and converted bacterial cell counts to wet weight (ww) assuming 0.1 g ww per 10^{12} cells (Peters 1986).

Exhibit E-5. Predictions of Biomass (B) of Biotic Components of Lakes with Different Total Phosphorus (TP) Concentrations

Group	Equation	Biomass (B) (g wet weight/m^2)		
		TP = 5 µg/L	TP = 10 µg/L	TP = 50 µg/L
Bacteria	$B = 2.1 \times TP^{0.37}$	3.8	4.9	8.9
Nanoplankton	$B = 0.40 \times TP^{1.0}$	2.0	4.0	20
Net plankton	$B = 0.20 \times TP^{1.4}$	1.9	5.0	48
Microzooplankton	$B = 4.1 \times TP^{0.29}$	6.5	8.0	13
Macrozooplankton	$B = 4.6 \times TP^{0.37}$	8.3	11	20
Benthos	$B = 0.81 \times TP^{0.71}$	2.5	4.2	13
Fish	$B = 0.59 \times TP^{0.71}$	1.8	3.0	9.5

Source: Adapted from Peters (1986).

In a regression analysis of data on TP and fish biomass for 31 lakes across North America, Europe, and Russia, Nürnberg (1996) summarized the "limits" among three TP-defined lake trophic status categories with respect to total fish wet weight biomass per unit area:

Oligo-meso (TP = 10 µg/L) = 1.9 g ww/m^2

Meso-eutro (TP = 30 µg/L) = 3.7 g ww/m^2

Eutro-hypereutro (TP = 100 µg/L) = 8.5 g ww/m^2

Nürnberg (1996) also summarized total fish biomass limits from Bachmann et al. (1996) for the same lake trophic status categories based on a sample of 60 lakes in Florida:

Oligo-meso (TP = 10 µg/L) = 7.4 g ww/m^2

Meso-eutro (TP = 30 µg/L) = 10.6 g ww/m^2

Eutro-hypereutro (TP = 100 µg/L) = 15.6 g ww/m^2

As expected, for the same TP concentrations, standing fish biomass per unit area in the Florida lakes is two to three times higher than standing fish biomass for more northerly lakes with shorter growing seasons.

Hanson and Legget (1982) evaluated data for 43 lakes ranging in surface area from 0.1 to 82,414 km^2 (10 ha to 8 million ha; 25 acres to 20 million acres), with TP concentrations of 8–540 µg/L and macrobenthos standing crop of 0.48–61.1 g/m^2, and located between 42° and 62° N latitude and 17° E to 117° W longitude. Based on a subset of 21 lakes sampled at the same

time, the best univariate predictor of fish yield was TP; the regression correlation coefficient (r^2) was 0.84 (Equation E-3):

$$FY = 0.792 + 0.072\ (TP) \qquad \textbf{Eqn. E-3}$$

where:

FY = total fish yield (kg/hectare)

TP = total phosphorous (µg/L)

Logarithmic transformation did not improve the predictive power. All but five of the lakes had TP under 100 µg/L and fish yield of less than 1 g ww/m^2. At a 10-percent harvest rate, that would equal 10 g ww biomass/m^2.

Hanson and Legget (1982) also estimated the relationship between macrobenthos biomass and TP and fish standing crop from a sample of 18 to 20 lakes drawn from the same set of 43 lakes. The relationship between TP and total fish standing biomass is shown in Equation E-4 and between standing biomass of benthic invertebrates and fish biomass is shown in Equation E-5.

$$\log_{10}(FSB) = 0.708 \log_{10}(TP) + 0.774 \qquad (r^2 = 0.75,\ n = 18) \qquad \textbf{Eqn. E-4}$$

$$\log_{10}(FSB) = 5.692\ (M/z) + 28.7 \qquad (r^2 = 0.83,\ n = 20) \qquad \textbf{Eqn. E-5}$$

where:

FSB = total fish standing crop or biomass (kg/ha)

TP = total phosphorus (µg/L)

M/z = macrobenthos biomass (kg/ha) divided by mean lake depth (z) (meters)

Hanson and Leggett (1982) compared the predictions of Equation E-4 with Taylor's (1971) data on average TP and total fish biomass from five Tennessee Valley Authority reservoirs following rotenone poisoning. The comparison, presented in Exhibit E-6, produced a reasonable match.

Yurk and Ney (1989) examined the relationship between TP and standing stock of fish in 22 reservoirs in southern Appalachia sampled in 1973. The reservoirs ranged in surface area from 445 to 53,400 hectares, had TP concentrations ranging from 8 to 81 µg/L, with total fish biomass ranging from 3.4 to 232 g ww/m^2. Their logarithmic regression relating total fish standing crop or biomass (*FSB*) to *TP* is presented as Equation E-6.

$$\log_{10}(FSB) = 1.07 + 1.14 \log_{10}(TP) \qquad (r^2 = 0.75,\ n = 22) \qquad \textbf{Eqn. E-6}$$

Predictions of total fish biomass from *TP* from the equation of Yurk and Ney (1989) are compared with the predictions from the equation of Hanson and Legget (1982) in Exhibit E-7. At intermediate TP concentrations, predictions of total fish biomass are similar between the two models.

Exhibit E-6. Reported Compared With Predicted Fish Biomass for Five Reservoirs

Reservoir	Average Total Phosphorus (µg/L)	Reported Fish Biomass (g ww/m^2)[a]	Predicted Fish Biomass (g ww/m^2)	Percent Predicted/Reported
Kentucky	270	28	26	92.5
Cherokee	160	23	19	83.9
Norris	20	15	11	73.3
Nottley	50	14.3	12.8	85.5
Douglas	110	12.5	16.4	131.2

[a] Total fish biomass following rotenone kill as reported by Taylor (1971).
Source: Hanson and Leggett (1982), Table 5; original units for biomass density = kg/hectare; changed to g wet weight biomass/m^2 by dividing by 10.

Exhibit E-7. Comparison of Predictions of Total Fish Biomass from Total Phosphorus (TP)

TP (µg/L)	Total Fish Biomass (g ww/m^2)	
	Hanson and Legget (1982)	Yurk and Ney (1989)
10	3.0	1.6
30	6.6	5.7
80	13.2	17.4
100	15.5	22.4
200	25.4	–
500	48.7	–

"–" indicates that TP is much higher than the TP range for data used to derive the model; thus, estimating fish biomass for those TP values with the Yurk and Ney (1989) model is not appropriate.

For a site-specific, refined risk assessment, one could use these regressions and measured TP concentrations in the lake(s) to predict total fish standing crop or biomass per unit area. For Tiers 1 through 3 of the RTR screening risk assessment, however, we need to assume a single value for fish productivity per unit area where TP concentration is an unknown.

E.1.4.3 Measured Total Fish Standing Biomass

The empirical models provided in Section E.1.4.2 are based on lake data sets for which the original data are only partially published. In this section, we present some studies that measured total fish biomass in lakes of different sizes and from different climates. In reviewing studies of aquatic communities, we excluded data from the Great Lakes, because the size of those systems allows for substantially longer food chains and a more complete segregation between pelagic and benthic food webs than occurs in most freshwater ecosystems of North America. We also excluded lakes less than 5 hectares from our assessment, because they are unlikely to support stable fish communities and therefore generally are not evaluated for bioaccumulative chemicals.

In general, for small lakes in cold climates, relatively low fish productivity is likely. For example, Demers et al. (2001) found total fish standing biomass of 2.73 and 3.81 g ww/m^2 in two lakes of 27 and 22 acres (11 and 9 hectares), respectively, in south-central Ontario. Across 48 lakes in Newfoundland ranging in size from 3.56 hectares to 1,909 hectares, Cote et al. (2011) found

that benthivorous salmonid biomass per unit area varied by more than an order of magnitude (minimum 0.045 g ww/m^2; maximum 1.0 g ww/m^2; mean: 0.40 g ww/m^2). Brook trout (*Salvelinus fontinalis*) biomass was almost 76 percent of total salmonids, but varied by almost two orders of magnitude across lakes.

Brönmark and Weisner (1996) reported fish communities from 44 small ponds in southern Sweden (most were less than 5 hectares, or about 12 acres). All small ponds were dominated by periphyton (algae growing on rock surfaces), which was heavily grazed by freshwater snails. The TL3 fish consumed the snails. The piscivorous fish found in some ponds were all bottom feeders that ate both snails and small fish. Similarly, De Leeuw et al. (2003) found that most Scandinavian and Dutch lakes are dominated by benthivorous fish. The biomass and proportion of benthivores increased significantly with TP primarily due to increase of benthivorous bream (a species of sunfish/cyprinid) >25 cm in length.

The largest freshwater data set from more temperate climates of which we are aware is that of Leidy and Jenkins (1977). They analyzed several large data sets to support modeling of fish productivity and carrying capacity in reservoirs across the United States for the National Reservoir Research Program. The analyses derived from data for 61 reservoirs across the midwestern and eastern United States sampled at different times between 1952 and 1975. Only reservoirs of at least 500 acres (202 hectares) in size were included, with some exceeding 65,000 acres (in the Missouri drainage basin). Considering all 61 reservoirs, the mean total fish biomass density was 41.3 (± 30.4 standard deviation) g ww/m^2 (Exhibit E-8).

Exhibit E-8. Total Fish Biomass in Reservoirs of the United States by Drainage Area

Drainage Area	Number of Reservoirs	Total Fish Biomass (g wet weight/m^2)	
		Mean	SD
Middle Atlantic	1	14.2	
Gulf and South Atlantic	9	18.3	6.2
Ohio Basin	13	26.4	16.3
Lower Mississippi	5	41.1	19.9
Arkansas (Arkansas)	19	68.7	35.1
White (Arkansas)	6	33.4	8.4
Red (Arkansas)	6	30.9	24.6
Rio Grande and Gulf	1	28.3	
Missouri Basin	1	74.1	
All Reservoirs	61	41.3	30.4

Abbreviations: SD = standard deviation.
Source: Appendix B in Leidy and Jenkins (1977).

The minimum and maximum total fish biomass densities were 3.2 and 133.2 g ww/m^2, respectively, and the median value was 30.9 g ww/m^2 (Exhibit E-8). Thus, fish standing biomass per unit area in the reservoirs varied by more than three orders of magnitude.

The fish were sampled using rotenone poisoning of coves ranging in size from 1 to 5 acres after separating the coves from the reservoir using nets, similar to the method of Taylor (1971). To estimate the percentage of fish actually present that were recovered, marked fish were placed in the segregated coves prior to treatment with rotenone. In some cases, divers collected fish that

did not float to the surface. All fish collected were identified to species and weighed. Most cove sampling was performed one time per year in August. Most reservoirs were sampled at least once for 2 or more years between 1952 and 1975, with some being sampled 10 to 20 years during that interval.

Leidy and Jenkins (1977) applied adjustment factors to correct for non-recovery bias (i.e., bottom fish that tend not to float to the surface; small fish that are not recovered) and habitat preference bias (i.e., fish that are more or less abundant in the coves compared with the open water). The combined adjustments for sampling bias ranged from a factor of 0.88 for sunfishes (cyprinids), which were over-represented by sampling in coves, to factors of 3.08 and 3.36 for catostomids and freshwater drum, respectively, which were estimated to be about 2.4 times more prevalent in the open water than in the coves. The use of adjustment factors for some species indicates the uncertainties in the data; however, unadjusted biomass estimates are very likely to be biased.

Exhibit E-9 summarizes the data on total fish biomass in reservoirs and lakes from the literature we reviewed. The table suggests that average fish biomass density for reservoirs, although quite variable, is generally higher than that for lakes. TP concentrations in the reservoirs might be higher on average than TP concentrations in the natural lakes; however, the data are insufficient to test that hypothesis for the studies reviewed. Reservoirs in general might support higher fish biomass densities for a given TP level than do natural lakes because of extensive littoral zones with macrophytes or high quantities of detritus to fuel the BI component of the aquatic food web.

Exhibit E-9. Total Fish Biomass Density in Reservoirs and Lakes from Different Studies

Water Body (Source)	N	Total Fish Biomass (g ww/m^2)				Mean TP µg/L
		Mean	Min	Max	Med.	
Reservoirs of the U.S. > 202 ha (a)	61	41.3	3.2	133	30.9	NR
Appalachian Reservoirs, U.S. (b)	22	64.2	3.4	232	55.0	32
DeGray Lake, Arkansas, U.S. (c)	1	7.5	–	–	–	NR
Ranger & Mouse Lakes, Ontario (d)	2	3.3	2.7	3.8	–	NR
Lakes in U.S. (e)	18	9.4	NR	NR	NR	NR

Abbreviations: NR = not reported; "–" indicates not relevant; TP = total phosphorus.

Sources: (a) Leidy and Jenkins (1977); (b) Yurk and Ney (1989); (c) Ploskey and Jenkins (1982); (d) Demers et al. (2001); (e) Randall et al. (1995) as reanalyzed by Nash et al. (1999).

To estimate the minimum lake size that would support a sustainable WCC fishery, we rounded that value down to a single significant digit of 40 g ww/m^2 as the upper limit for total fish biomass in a lake. That standing biomass is higher than predicted by the regression models of Hanson and Legget (1982), Yurk and Ney (1989), and Nürnberg (1996) at a high TP of 100 µg/L (where phosphorous is the limiting nutrient). Less productive lakes would support fewer fish per unit area, and, therefore, would have to be larger to support a specified fish ingestion rate.

E.1.5 Proportion of Fish Biomass by Trophic Level

Much of the literature on fish communities comes from research on the effects of different trophic elements on aquatic food web structure and consequent productivity of fisheries. Several hypotheses have been developed over the years to explain relationships among trophic levels in lakes and rivers using fundamental ecological concepts.

E.1.5.1 Principles of Trophic Pyramids

As a "rule of thumb" in ecology, 10 percent of the energy produced at one trophic level usually can be converted to biomass in the next trophic level (i.e., approximately 90 percent loss of energy per trophic step) (UM 2016). With different species having different energy assimilation efficiencies, with fat providing approximately twice as many calories as muscle, and with smaller animal species generally having higher turnover rates than larger species, however, the 10-percent energy rule does not necessarily translate into a standing biomass pyramid of similar proportions. In this section, the proportion of fish (based on biomass) that might be expected in the WCC and the BC fish compartments relative to total standing fish biomass are examined assuming that the lake is large enough to support WCC (pelagic TL4.5 fish).

Further complicating prediction of standing biomass at different trophic levels are the relationships among trophic groups. For example, a "classic" trophic cascade hypothesis associated with managing lakes for top-trophic-level fish predicts that increasing piscivore biomass in a lake will result in: (a) decreasing biomass of their prey, including planktivorous fish; (b) increasing biomass of zooplankton, and (c) decreasing biomass of phytoplankton (Carpenter et al. 1985; Carpenter and Kitchell 1996).

An alternative hypothesis about trophic structure is the "top-down/bottom-up" hypothesis, which predicts that the top-down effects of piscivores are strongest at the top of the food web, weakening in trophic groups closer to the primary producers, whereas the phytoplankton are most strongly influenced by nutrient availability (bottom-up). Drenner and Hambright (2002) reported that as of 2002, over 1,900 reports had been published on the effects of fish in lakes. They reviewed 33 experiments and 6 surveys to test these hypotheses, of which only 17 did not include confounding factors. Of those, they concluded that 7 supported the trophic cascade hypothesis and 10 did not.

Drenner and Hambright (2002) found a general pattern of lower chlorophyll concentrations for given TP concentrations in systems containing piscivores (4-link systems) relative to systems with only planktivorous fish (3-link systems). The trophic cascade appears to work where herbivorous fish are dominated by small (vulnerable to predation) species rather than larger herbivores (e.g., shad, carp) that are not vulnerable to predation after reaching larger sizes.

Given the diversity of lake ecosystems and competing hypotheses for fish community structure by trophic level, we investigated two lines of evidence: models of fish biomass at different trophic levels (Section E.1.5.2) and measurements of fish biomass in different trophic groups (Section E.1.5.3). Bioenergetic simulation models of fish community structure are useful because a model can include several species, predator-prey relationships, and age/size classes at one time, using measured values to parameterize the model initially. Measurement of biomass at different trophic levels is difficult because different species and sizes of fish are best caught via different methods. Rotenone killing of all fish in a lake, which can yield the most accurate measurement, is feasible only in relatively small lakes or ponds and is wasteful.

E.1.5.2 Models of Fish Biomass in Different Trophic Groups

Of the recent models that simulate bioaccumulation of toxic chemicals in aquatic food webs identified in the literature search, the one that appeared most similar to the TRIM.FaTE approach in compartmentalizing the fish compartments of the food web is the Comprehensive Aquatic Systems Model (CASM, Version 2.0) developed for Quebec, Canada (DeAngelis et al. 1989). This detailed food-web model includes data sets that provide parameter values for four Canadian aquatic ecosystems: (1) northern lakes/reservoirs, (2) northern rivers, (3) southern lakes/reservoirs, and (4) southern rivers. Northern is defined as between 48° latitude and 55°

latitude, and southern is defined as between 44° and 48° latitude. The parameterization of the model for "southern" locations would apply only to the more northern areas of the United States.

For each aquatic ecosystem, CASM includes three data sets derived from the primary literature: (1) data for the primary producer and consumer populations; (2) definitions of the grazing and predator-prey interactions (diet preferences and assimilation efficiencies); and (3) data on daily incident solar radiation, water temperature, and nutrient inputs. Using those three data sets, CASM can be used to estimate the baseline biomass values in 10 biotic compartments based on factors that affect primary productivity and trophic transfers.

Although CASM and its databases are not publicly available, Bartell et al. (1999) have published baseline biomass estimates in the open literature for a northern river and for a Florida lake. We totaled those biomass estimates for each compartment type and then determined the proportion of the total biomass represented in each compartment type, shown in Exhibit E-10 for the lake. The diets assigned to each species were not reported in the publications, so cannot be evaluated.

Exhibit E-10. Distribution of Standing Biomass Among Aquatic Compartments Simulated in the Comprehensive Aquatic Systems Model (CASM) for a Florida lake

Biotic Compartment	Total Biomass		Percent Biomass	
	g C/m^2	%	Animal	Fish
Phytoplankton	1.38	13	na	na
Periphyton	0.70	6	na	na
Macrophytes (e.g., *Elodea*, *Ceratophyllum*)	7.6	70	na	na
Zooplankton	0.07	1	0.06	na
Benthic Invertebrates	0.44	4	0.39	na
Pelagic Omnivore (e.g., shiners, sunfish)	0.263	2	0.23	0.42
Pelagic Piscivore (e.g., gar, pickerel)	0.059	1	0.05	0.10
Benthic Omnivore (e.g., bullhead, warmouth)	0.275	3	0.24	0.44
Benthic Piscivore (i.e., largemouth bass)	0.022	2	0.02	0.04

Note: We did not identify data for converting dry carbon to wet-weight biomass for the compartments listed.
Abbreviations: C = carbon.
Source: Bartell et al. (1999).

The lake clearly is dominated by macrophytes, including the invasive species from the aquarium trade, rooted or free-floating *Elodea* sp. and *Ceratophyllum* sp., which, unlike phytoplankton, grow in length without harvesting by most fish species (an exception is carp, which can consume both macrophytes). The macrophytes and plankton undoubtedly contribute to detritus in the benthos; however, Bartell et al. (1999) did not report the carbon content of detritus per unit area. The pelagic and benthic omnivores comprise 86 percent of the total fish biomass, while the pelagic and benthic carnivores comprise 14 percent of the total fish biomass. For a lake in Florida without large quantities of invasive macrophytes, the trophic pyramid might look substantially different.

Hossain et al. (2010) evaluated fish biomass and harvest rates for an oligotrophic lake (low productivity) in Southern Hokkaido, Japan (latitude 42°36´ N, longitude 140°51´ E). The lake is volcanic in origin, with surface area 70 km^2, maximum depth 179 m, and mean depth 116 m. A monomictic system, its annual average TP concentration is 3 µg/L and TN is 150 µg/L. Hossain

et al. (2010) used the mass-balance modeling software Ecopath and Ecosim (EwE) (e.g., Christensen and Walters 2004, Christensen et al. 2005), built to simulate coastal fisheries, to investigate whether the level of fish harvests reported for the late 1990s (masu salmon harvest of 2.64 kg/km^2-year and sockeye salmon harvest of 24.45 kg/km^2-yr) are likely sustainable.

Exhibit E-11 lists the estimated biomass, trophic level, annual production/biomass ratio (except for detritus and organic matter), and the percentage of total fish biomass represented in each of their fish compartments. Values in Exhibit E-11 are in line with other estimates (see Exhibit E-1 and Section E.1.5.3): 5.8 percent of total fish biomass estimated at a trophic level higher than 4.0 (masu salmon, *Oncorhynchus masou*), 12 percent of adult sockeye salmon (*Oncorhynchus nerka*) estimated to be at TL3.75 in their model, and 81 percent of fish near TL3 (smelt, *Hypomesus transpacificus nipponensis*, and juvenile sockeye salmon). None of the fish groups are TL2; fish fry are probably represented in the zooplankton compartment. Difficulties interpreting these simulations, however, come from the continual stocking of salmon, fish harvesting above levels that might be sustainable for the sockeye salmon, the complex food web simulated, and migration of some of the fish into and from the lake (anadromous).

Exhibit E-11. Estimated Biomass by Aquatic Compartment in Lake Toya, Japan

Aquatic Compartment	Biomass (kg/km^2)	Biomass (g ww/m^2)	Trophic Level[a]	Production/ Biomass (kg/kg)	Percent Total Fish Biomass
Masu Salmon	22.7	0.023	4.12	0.54	5.8%
Adult Sockeye Salmon	45.5	0.046	3.75	0.33	12%
Juvenile Sockeye Salmon	14.1	0.014	3.16	1.72	4%
Japanese Smelt	303	0.30	3.17	1.24	77%
Other Fish	5.8	0.0058	3.07	1.50	1.5%
Shrimp	5.9	0.0059	2.27	1.83	na
Amphipods	136	0.14	2.32	6.0	na
Insects	110	0.11	2.11	4.2	na
Zooplankton	162	0.16	2.05	33.5	na
Phytoplankton	50.2	0.050	1	365	na
Organic Materials	2000	2.0	1	na	na
Detritus	1000	1.0	1	na	na

Abbreviation: na = not applicable

[a]Trophic level estimated by Hossain et al. (2010) given the food web they characterized.

Source: Hossain et al. (2010).

Exhibit E-11 does illustrate well, however, the relatively low standing biomass of phytoplankton (0.050 g ww/m^2) compared with the other compartments, but its very high annual productivity (365 g/g production/standing biomass) and turnover rates compared with other aquatic compartments. Zooplankton shows the next highest annual productivity rate (33.5 g/g), even though its standing biomass (0.16 g ww/m^2) is less than that of the smelt (0.3 g ww/m^2), which produce 1.24 g/g annually.

Rather than work further with fish biomass and production simulation models, which require substantial data and are not readily transparent, we investigated measurements of fish biomass in different trophic groups (see Section E.1.5.3).

E.1.5.3 Measured Biomass of Fish in Different Trophic Groups

A popular measure of fish productivity for game fish species across lakes is the angler effort required to catch each fish (or catch per unit effort, CPUE). The measure, used in numerous studies (e.g., Gorman et al. 2014; Quiros 1990), provides valuable information for commercial and recreational fisheries applications. It, however, does not provide information on the numeric "trophic pyramid" in lakes or the relative standing biomass of each trophic group needed for TRIM.FaTE modeling. Specifically, CPUE usually misses the smaller fish and untargeted species.

A key difficulty with sampling lakes for total standing fish biomass and for fish biomass at different trophic levels is capturing and measuring the fish in the first place (see Section E.1.4.3). Some lakes have been sampled by killing with rotenone all fish in a lake, which then can be collected and measured. This practice is feasible for relatively small lakes for which state fish and game officials might want to start the lake's trophic structure "over"; however, for larger lakes, it is both impractical and wasteful. Other approaches to making an inventory of fish standing stock include combinations of seine fishing, electroshocking, and other methods; however, each includes some biases against certain species and age-classes that require "correction factors" (e.g., based on total kill inventory methods) or at least acknowledgment of the possible magnitude and direction of biases (Leidy and Jenkins 1977).

Leidy and Jenkins (1977) estimated the biomass of fish supported by various food compartments in the 61 reservoirs included in their survey (Exhibit E-12). Only reservoirs at least 500 acres (202 hectares) in size were included. They did not separate the piscivorous fish species (i.e., the biomass of fish supported by "Fish" in Exhibit E-12) by benthic or pelagic feeding habits. We pulled the data in Exhibit E-12 from Appendix G in Leidy and Jenkins (1977).

Exhibit E-12. Carrying Capacity, Biomass (g ww/m^2) of Fish Supported by Each Food Compartment Across 61 Reservoirs by Drainage Area

Drainage Area	Plants & Detritus	Benthic Inverts.	Zoo-plankton	Fish	Terrest. Inverts.	Total
Gulf and South Atlantic	5.12	3.77	0.55	2.77	0.45	12.67
Green and Cumberland Rivers and Dewey Reservoir	10.60	6.03	1.61	3.09	0.39	21.74
Lower Mississippi Valley	11.54	4.81	4.89	6.77	0.31	28.36
Blue Mountain, Nimrod, and Wister Reservoirs	22.64	8.53	16.03	9.26	0.33	56.72
Arkansas River Basin	25.78	9.63	6.50	7.79	0.44	50.10
Red River Basin	9.01	7.32	0.46	4.40	0.84	22.08
White River Basin	10.46	7.32	2.01	3.43	0.48	23.65
Average	13.59	6.77	4.58	5.36	0.46	30.76
Standard Deviation	7.59	2.05	5.53	2.57	0.18	16.27

Abbreviations: Terrest. Inverts. = terrestrial invertebrates, primarily insects that lay eggs at the water surface or that fall into the reservoir from emergent and terrestrial plants.
Source: Appendix G in Leidy and Jenkins (1977).

We calculated from Exhibit E-12 that, on average, 18 percent of the fish biomass across the 61 reservoirs they examined was piscivorous (minimum of 14 percent and maximum 24 percent, including both benthic and pelagic species; see bold values in Exhibit E-13).

Exhibit E-13. Proportion of Total Carrying Capacity, Proportion Fish Biomass Supported by each Food Compartment by Drainage Area

Drainage Area	Plants & Detritus	Benthic Inverts.	Zoo-plankton	Fish	Terrest. Inverts.	Total
Gulf and South Atlantic	0.40	0.30	0.04	0.22	0.04	1.00
Green and Cumberland Rivers and Dewey Reservoir	0.49	0.28	0.07	0.14	0.02	1.00
Lower Mississippi Valley	0.41	0.17	0.17	0.24	0.01	1.00
Blue Mountain, Nimrod, and Wister Reservoirs	0.40	0.15	0.28	0.16	0.01	1.00
Arkansas River Basin	0.51	0.19	0.13	0.16	0.01	1.00
Red River Basin	0.41	0.33	0.02	0.20	0.04	1.00
White River Basin	0.44	0.31	0.08	0.15	0.02	1.00
Average	0.44	0.25	0.12	**0.18**	0.02	
Standard Deviation	0.05	0.07	0.09	0.04	0.01	
Minimum	0.40	0.15	0.02	**0.14**	0.01	
Maximum	0.51	0.33	0.28	**0.24**	0.04	
Median	0.41	0.28	0.08	0.16	0.02	

Source: Calculated from Exhibit E-12; data from Appendix G in Leidy and Jenkins (1977).

Håkanson and Boulion (2004) created a "distribution coefficient" to indicate what proportion of the total fish biomass in a lake is prey versus predatory fish. Based on data from 122 lakes in Europe and North America, they concluded that 27 percent by biomass is a "normal" portion of predatory fish in a balanced system. They noted further, however, that for eutrophic lakes with TP levels >100 µg/L, the proportion of fish represented by piscivores declined to less than 20 percent. The piscivores included both benthic and pelagic species. We note that most benthic piscivores also consume benthic macroinvertebrates.

Scharf (2008) evaluated the biomass of top predatory fish (TL4 to TL4.5, pike > 20 cm, pikeperch > 40 cm) in a large, deep stratifying reservoir in Germany (Exhibit E-14). Scharf found that over the 20 years of the reservoir's existence, the standing biomass of those fish never exceeded 10 percent of total fish biomass despite stocking and protection efforts. We assigned a TRIM.FaTE compartment (WCC, WCO, WCH, BC, BO) or combination of two compartments to describe the feeding habitat of each fish age/size-class and species and assigned a likely trophic level to each age/size-class based on our experience with estimating fish trophic levels (U.S. EPA 2000). Those compartments and trophic levels also are listed in Exhibit E-14. Our estimate is that the WCC compartment of fish at TL4.5 is 3.4 percent of the total fish biomass and that the combined WCC/BC TL3.5 (perch > 16 cm) is 17.6 percent of the total fish biomass.

Based on data from the reservoir over 20 years, Scharf (2008) concluded that introduction of pikeperch in 1988, which became self-reproducing, helped release perch from competition, which allowed perch to grow larger than >16 cm. At this size, they can consume other fish and become more abundant, accounting for 17.6 percent of the total fish biomass.

Exhibit E-14. Total Fish Biomass by Trophic Level in Wupper Reservoir, Germany

Fish Age-class and Species	Compartment Trophic Level[a]	Biomass Density		Individual Abundance	
		kg ww/ha	Percent	Individuals/ha	Percent
Total Fish Biomass	na	93.6	100	4025	100
Piscivorous Fish Biomass (large pike, pikeperch, perch)	na	25.7	27.5	na	na
Total Fish Biomass without YOY	na	79.4	100	na	na
Piscivorous Fish Biomass	na	25.7	32.4	na	na
Pike > 20 cm in length	WCC 4.5	0.5	**0.5**	1	0.02
Pikeperch YOY (<12 cm)	WCH 2.5	0.2	0.2	30	–
Pikeperch (12 to 40 cm)[b]	WCO 3.5	2.3	2.7	15.5	–
Pikeperch >40 cm	WCC 4.5	2.7	**2.9**	2.5	–
Perch YOY (<10 cm)	WCH 2	12.5	13.4	2.24	56
Perch 1-yr old (10 to <16 cm)	WCO 3	18.6	19.9	677	17
Perch older (>16 cm)	WCC/BC 3.5	16.5	17.6	90	2.2
Cyprinids YOY	WCH 2	1.7	1.8	374	9.3
Cyprinids 1-yr old	WCH 2.5	7	7.5	296	7.4
Cyprinids older (>16 cm)	WCO/BO 3	28.1	30	292	7.3
Eel (benthic carnivore)	BC 3.5	3.5	3.7	6	–
Total of Age Classes		94	100%	1834	100%
Water Column Carnivore (WCC)	4.5	3.2	3.4	na	na
Water Column Carnivore/Benthic Carnivore (WCC/BC) (except eels)	3.5	18.8	17.6	na	na
Water Column Omnivore (WCO/BO)	3.0	46.7	52.6	na	na
Water Column Herbivore (WCH)	2.0–2.5	21.4	22.9	na	na
Benthic Carnivore (BC) (eel)	3.5	3.5	3.7	na	na

Abbreviations: "–" not calculated in Scharf (2008) because body weight distribution across age classes uncertain; na = not applicable (body size varies); YOY = young-of-year (from hatching to <1 yr).

[a] We assigned trophic levels to the group based on general feeding characteristics.

[b] Pikeperch 12 cm to <40 cm in length calculated from row for total pikeperch minus the smaller and larger pikeperch in Table 1 of Scharf (2008).

Source: Scharf (2008), Table 1.

We investigated other studies of fish biomass in lakes; however, most had limitations that meant we could not use them to estimate biomass distribution across fish trophic levels. Moreover, a disproportionate number of studies are for areas with colder climates than most of the continental United States, for which we expect total fish standing biomass to be less than the value of 5.7 g ww/m^2 used for the state of Maine. We list three examples below.

Post et al. (2008) estimated the carrying capacity of south-central British Columbia lakes to be 500 rainbow trout per hectare based on other studies. Individual trout body weight, however, was not reported.

Examining 78 lowland lakes in Germany, Emmrich et al. (2011) found that lake area is positively correlated with the number of fish size classes, with a wider range of fish body size, and with more of the larger sized fish in larger lakes. Raw data were not reported.

For 31 lakes in Newfoundland, Cote et al. (2011) reported a mean brook trout biomass of 0.474 g ww/m^2 (range 0.069–1.01 g ww/m^2) and a mean total salmonid biomass of 0.54 g ww/m^2 (range 0.113–1.01 g ww/m^2).

To summarize, several studies of fish biomass by trophic level indicate that top-trophic-level fish, combining pelagic and benthic carnivorous fish, might comprise approximately 20 percent of the standing fish biomass in many lakes. Ploskey and Jenkins (1982) estimated that piscivorous fish, both those that are generally free swimming or pelagic (e.g., pike, gar, walleye, TL4.5) and those that forage primarily in the benthos (e.g., various species of catfish, suckers, TL3.5) comprise 22 percent of the total fish biomass in DeGray Lake, Arkansas (averaged across several years). Using data from 122 lakes in Europe and North America, Håkanson and Boulion (2004) estimated 27 percent piscivorous fish biomass/total fish biomass for oligotrophic and mesotrophic lakes, declining to 20 percent in lakes with more than 100 µg/L TP. We interpret the data from Leidy and Jenkins (1977) as indicating 18 percent (range 14–24 percent) of the total standing fish biomass in reservoirs to be piscivorous fish (pelagic and benthic). Finally, Scharf (2008) provided data suggesting that 21 percent of the total standing fish biomass represented piscivores, with only 3.4 percent pelagic piscivores (WCC) at TL4.5.

E.1.5.4 Conclusion

Based on the studies listed above, we assume that 3.5 percent of fish standing biomass is in the WCC compartment for purposes of TRIM.FaTE modeling and for simulating angler harvest of WCC from lakes. The remaining distribution of biomass across biotic compartments in TRIM.FaTE, as presented in Exhibit E-2, also is consistent with the data presented here.

E.1.6 Derivation of Lake Sizes for Sustainable WCC Harvest

As stated in Section E.1, this attachment provides supporting information for Section 3.3.1 of the TSD—Accounting for Sustainable Fishing. To develop the screening scenarios with an angler, we needed to address two questions. Question 1—*How large does a lake need to be to provide a self-sustaining population(s) of top-trophic-level fish?*—is answered in Section E.1.6.1. Question 2—*How much fish can be harvested sustainably from lakes of different sizes?*—is answered in Section E.1.6.2.

E.1.6.1 Minimum Lake Size for Self-sustaining Population of WCC

As stated in Section E.1.3.3, we assume that at least 50 adult breeding WCC are needed for a self-sustaining population of WCC in an isolated lake. We derive the minimum lake size from two equations: Equations E-7 and E-8. The standing biomass of WCC in a lake is calculated using Equation E-7. The assumption that the WCC fish compartment represents approximately 3.5 percent of the total fish standing biomass was documented in Section E.1.5.

$$WCC\_SB = Total\_SB \times Fraction\_WCC \qquad \textbf{Eqn. E-7}$$

WCC_SB = Standing biomass of WCC fish (g ww/m^2)

Total_SB = Total standing biomass of all fish (g ww/m^2)

Fraction WCC = Fraction WCC fish biomass of total fish biomass (i.e., 0.035)

Using *WCC_SB* calculated from Equation E-7 and the size of the lake (*Lake_Size*), the total number of WCC fish supported in the lake is calculated using Equation E-8:

$$Number\_WCC = (Lake\_Size \times WCC\_SB \times CF_1) / BW_{WCC} \qquad \textbf{Eqn. E-8}$$

where:

Number_WCC = Total number of adult breeding WCC fish in lake

Lake_Size = Size of lake (acres)

WCC_SB = Standing biomass of WCC fish (g ww/m^2; from Equation E-7)

CF_1 = Unit conversion factor (4047 m^2/acre)

BW_{WCC} = Body weight of adult WCC fish (2000 g ww per individual; assumed)

Based on those two equations, we created a matrix that predicted the *Number_WCC* in a lake as a function of both fish biomass per unit area and the overall lake size in Exhibit E-15. The first vertical column presents the range of total fish biomass found by Leidy and Jenkins (1977) across 61 reservoirs in the United States. The interval between total fish biomass values from one row to the next is not monotonic; finer resolution is provided for the less productive lakes. The second vertical column in Exhibit E-15 presents the corresponding range of WCC biomass estimates assuming that WCC comprises 3.5 percent of the total fish biomass. The remaining columns in Exhibit E-15 present lakes of increasing size (from left to right). Again, the interval in lake size from one column to the next is not monotonic; finer resolution is presented for the smaller lakes. The numbers in each cell of Exhibit E-15 are the number of individual WCC fish predicted for each combination of total fish biomass and lake size.

In Exhibit E-15, all combinations of lake productivity and overall size that would *not* support a population of at least 50 WCC fish are shaded in gray. All combinations of lake productivity and size that might support 500 or more WCC fish, and therefore might be self-sustaining for a century or more, are highlighted in yellow. The unshaded cells represent the number of WCC between 50 and 500 individuals (2 kg each) that might be sustainable for an angler's lifetime.

Exhibit E-15. Number of WCC Adult Fish Supported by Lake Size (surface area in acres) and by Total Fish Biomass (TFB)

TFB	WCC	Number of Adult Water-column Carnivores (WCC) (by lake surface area from 1 to 250 acres)																						
(g ww/m^2)		1	2	3	4	5	7.5	10	15	25	35	40	50	60	70	80	90	100	125	150	175	200	225	250
2	0.070	0	0	0	1	1	1	1	2	4	5	6	7	8	10	11	13	14	18	21	25	28	32	35
3	0.105	0	0	1	1	1	2	2	3	5	7	8	11	13	15	17	19	21	27	32	37	42	48	53
4	0.140	0	1	1	1	1	2	3	4	7	10	11	14	17	20	23	25	28	35	42	50	57	64	71
5.7	0.200	0	1	1	2	2	3	4	6	10	14	16	20	24	28	32	36	40	50	61	71	81	91	101
10	0.350	1	1	2	3	4	5	7	11	18	25	28	35	42	50	57	64	71	89	106	124	142	159	177
15	0.525	1	2	3	4	5	8	11	16	27	37	42	53	64	74	85	96	106	133	159	186	212	239	266
20	0.700	1	3	4	6	7	11	14	21	35	50	57	71	85	99	113	127	142	177	212	248	283	319	354
30	1.05	2	4	6	8	11	16	21	32	53	74	85	106	127	149	170	191	212	266	319	372	425	478	531
35	1.225	2	5	7	10	12	19	25	37	62	87	99	124	149	174	198	223	248	310	372	434	496	558	620
40	1.40	3	6	8	11	14	21	28	42	71	99	113	142	170	198	227	255	283	354	425	496	567	637	708
50	1.75	4	7	11	14	18	27	35	53	89	124	142	177	212	248	283	319	354	443	531	620	708	797	885
60	2.10	4	8	13	17	21	32	42	64	106	149	170	212	255	297	340	382	425	531	637	744	850	956	1062
70	2.45	5	10	15	20	25	37	50	74	124	174	198	248	297	347	397	446	496	620	744	868	992	1115	1239
80	2.80	6	11	17	23	28	42	57	85	142	198	227	283	340	397	453	510	567	708	850	992	1133	1275	1416
90	3.15	6	13	19	25	32	48	64	96	159	223	255	319	382	446	510	574	637	797	956	1115	1275	1434	1594
100	3.50	7	14	21	28	35	53	71	106	177	248	283	354	425	496	567	637	708	885	1062	1239	1416	1594	1771
110	3.85	8	16	23	31	39	58	78	117	195	273	312	390	467	545	623	701	779	974	1169	1363	1558	1753	1948
120	4.20	8	17	25	34	42	64	85	127	212	297	340	425	510	595	680	765	850	1062	1275	1487	1700	1912	2125
130	4.55	9	18	28	37	46	69	92	138	230	322	368	460	552	644	737	829	921	1151	1381	1611	1841	2072	2302

Fish standing biomass for all fish (TFB) and for the WCC fish are provided in the first two columns. The TFB spans 2 to 130 acres in line with Leidy and Jenkins's (1977) estimates of total fish standing biomass per unit area across 61 reservoirs in the United States. The total standing biomass for WCC fish = TFB * 0.035.

Grey shaded area indicates that 50 or fewer WCC fish would be supported at the specified combination of lake size (acres) and TFB. Clear cells represent numbers of individual WCC fish that might be sustainable for an angler's lifetime of 50 to 70 years for lakes of different productivities and size. Yellow cells have populations of WCC that exceed 500, which might be self-sustaining for a century or more.

Note: Exhibit E-16 and Exhibit E-17 retain the same cell shading as Exhibit E-15, which presents the number of individual WCC that might be supported by the combinations of TFB and lake size. Each WCC fish weighs 2 kg.

Exhibit E-16. Total Standing Biomass of WCC Fish (kg) by Lake Size and Total Fish Biomass (TFB)

TFB	WCC	Total Standing Biomass of Water-column Carnivores (WCC) (kg) (by lake surface area from 1 to 250 acres)																						
(g ww/m^2)		1	2	3	4	5	7.5	10	15	25	35	40	50	60	70	80	90	100	125	150	175	200	225	250
2	0.070	0	1	1	1	1	2	3	4	7	10	11	14	17	20	23	25	28	35	42	50	57	64	71
3	0.105	0	1	1	2	2	3	4	6	11	15	17	21	25	30	34	38	42	53	64	74	85	96	106
4	0.140	1	1	2	2	3	4	6	8	14	20	23	28	34	40	45	51	57	71	85	99	113	127	142
5.7	0.200	1	2	2	3	4	6	8	12	20	28	32	40	48	57	65	73	81	101	121	141	161	182	202
10	0.350	1	3	4	6	7	11	14	21	35	50	57	71	85	99	113	127	142	177	212	248	283	319	354
15	0.525	2	4	6	8	11	16	21	32	53	74	85	106	127	149	170	191	212	266	319	372	425	478	531
20	0.700	3	6	8	11	14	21	28	42	71	99	113	142	170	198	227	255	283	354	425	496	567	637	708
30	1.050	4	8	13	17	21	32	42	64	106	149	170	212	255	297	340	382	425	531	637	744	850	956	1062
35	1.225	5	10	15	20	25	37	50	74	124	174	198	248	297	347	397	446	496	620	744	868	992	1115	1239
40	1.40	6	11	17	23	28	42	57	85	142	198	227	283	340	397	453	510	567	708	850	992	1133	1275	1416
50	1.75	7	14	21	28	35	53	71	106	177	248	283	354	425	496	567	637	708	885	1062	1239	1416	1594	1771
60	2.10	8	17	25	34	42	64	85	127	212	297	340	425	510	595	680	765	850	1062	1275	1487	1700	1912	2125
70	2.45	10	20	30	40	50	74	99	149	248	347	397	496	595	694	793	892	992	1239	1487	1735	1983	2231	2479
80	2.80	11	23	34	45	57	85	113	170	283	397	453	567	680	793	907	1020	1133	1416	1700	1983	2266	2550	2833
90	3.15	13	25	38	51	64	96	127	191	319	446	510	637	765	892	1020	1147	1275	1594	1912	2231	2550	2868	3187
100	3.50	14	28	42	57	71	106	142	212	354	496	567	708	850	992	1133	1275	1416	1771	2125	2479	2833	3187	3541
110	3.85	16	31	47	62	78	117	156	234	390	545	623	779	935	1091	1246	1402	1558	1948	2337	2727	3116	3506	3895
120	4.20	17	34	51	68	85	127	170	255	425	595	680	850	1020	1190	1360	1530	1700	2125	2550	2975	3399	3824	4249
130	4.55	18	37	55	74	92	138	184	276	460	644	737	921	1105	1289	1473	1657	1841	2302	2762	3222	3683	4143	4603

Note: Each WCC fish is assumed to weigh 2 kg. The total fish standing biomass used in TRIM.FaTE was 5.7 g ww/m^2 (see Exhibit E-2). Total fish standing biomass of 40 g ww/m^2 (red text) used to assess angler behavior is based on the mean fish standing biomass for 61 reservoirs of 41 g ww/m^2 (Leidy and Jenkins 1977). With a WCC proportion of the total fish biomass of 0.035, the assumed WCC standing fish biomass for the screen is 1.4 g ww/m^2. For example, a 25-acre pond (101,175 m^2) might support an annual average standing biomass of 142 kg WCC at a total fish biomass of 40 g ww/m^2.

E.1.6.2 Maximum Fish Ingestion Rate by Lake Size

The likely annual productivity of WCC fish (kg/year) in a lake is estimated using Equation E-9.

$$Productivity\_WCC = (Lake\_Size \times WCC\_SB \times CF_1) / CF_2 \qquad \textbf{Eqn. E-9}$$

where:

Productivity_WCC = Likely annual productivity of WCC fish (kg/year)
Lake_Size = Size of lake (acres)
WCC_SB = Standing biomass of WCC fish (g ww/m^2; from Equation E-7)
CF1 = Unit conversion factor 1 (4047 m^2/acre)
CF_2 = Unit conversion factor 2 (1000 g/kg)

The maximum daily fish ingestion rate (g/day) for fillet of WCC plus BC associated with sustainable fishing can be predicted using Equation E-10. The equation assumes the angler consumes 50 percent WCC and 50 percent BC, represented by the factor of 2 in Equation E-10:

$$Max\_IR_{(BC+WCC)} = 2 \times (Productivity\_WCC \times FF \times HF \times CF_1) / CF_2 \qquad \textbf{Eqn. E-10}$$

where:

$Max\_IR_{(BC+WCC)}$ = Predicted maximum sustainable ingestion rate for BC and WCC fish (g/day)
Productivity_WCC = Annual productivity of WCC fish in the lake (kg/year; from Equation E-9)
FF = Fillet fraction; represents the assumed edible portion of fish (0.33; unitless)
HF = Annual harvest fraction (0.10; unitless)
CF_2 = Unit conversion factor 2 (1000 g/kg)
CF_3 = Unit conversion factor 3 (365 days/year)

Exhibit E-17 lists the fish-fillet-ingestion rates that could be supported for each combination of lake productivity (standing fish biomass per unit area) and lake size. Exhibit E-17 is similar to Exhibit 26 in the TSD, except that a different series of lake sizes is presented in the columns. *At the assumed total fish standing biomass of 40 g ww/m^2, the ingestion rate of fish fillet (including both WCC and BC fish in a 50:50 ratio) supported by a lake is approximately 1 gram per day per acre. With this assumption, the angler needs to fish from at least 373 acres of lake to support a fish-fillet-ingestion rate of 373 g/day.*

Exhibit E-17. Estimated Maximum Fish-fillet-ingestion Rate (g/day) Associated with Sustainable Fishing of WCC by Lake Size and Total Standing Fish Biomass (TFB)

TFB	WCC	Maximum Fish-fillet-ingestion Rate (g/day) for a Diet of 50% BC Plus 50% WCC Fish (by lake surface area from 1 to 250 acres)																					
(g ww/m^2)		1	2	3	4	5	7.5	10	15	25	35	40	50	60	70	80	90	100	125	150	175	200	225	250
2	0.070	0	0	0	0	0	0	1	1	1	2	2	3	3	4	4	5	5	6	8	9	10	12	13
3	0.105	0	0	0	0	0	1	1	1	2	3	3	4	5	5	6	7	8	10	12	13	15	17	19
4	0.140	0	0	0	0	1	1	1	2	3	4	4	5	6	7	8	9	10	13	15	18	20	23	26
5.7	0.200	0	0	0	1	1	1	1	2	4	5	6	7	9	10	12	13	15	18	22	26	29	33	36
10	0.350	0	1	1	1	1	2	3	4	6	9	10	13	15	18	20	23	26	32	38	45	51	58	64
15	0.525	0	1	1	2	2	3	4	6	10	13	15	19	23	27	31	35	38	48	58	67	77	86	96
20	0.700	1	1	2	2	3	4	5	8	13	18	20	26	31	36	41	46	51	64	77	90	102	115	128
30	1.050	1	2	2	3	4	6	8	12	19	27	31	38	46	54	61	69	77	96	115	134	154	173	192
35	1.225	1	2	3	4	4	7	9	13	22	31	36	45	54	63	72	81	90	112	134	157	179	202	224
40	1.40	1	2	3	4	5	8	10	15	26	36	41	51	61	72	82	92	102	128	154	179	205	231	256
50	1.75	1	3	4	5	6	10	13	19	32	45	51	64	77	90	102	115	128	160	192	224	256	288	320
60	2.10	2	3	5	6	8	12	15	23	38	54	61	77	92	108	123	138	154	192	231	269	307	346	384
70	2.45	2	4	5	7	9	13	18	27	45	63	72	90	108	126	143	161	179	224	269	314	359	403	448
80	2.80	2	4	6	8	10	15	20	31	51	72	82	102	123	143	164	184	205	256	307	359	410	461	512
90	3.15	2	5	7	9	12	17	23	35	58	81	92	115	138	161	184	207	231	288	346	403	461	519	576
100	3.50	3	5	8	10	13	19	26	38	64	90	102	128	154	179	205	231	256	320	384	448	512	576	640
110	3.85	3	6	8	11	14	21	28	42	70	99	113	141	169	197	225	254	282	352	423	493	563	634	704
120	4.20	3	6	9	12	15	23	31	46	77	108	123	154	184	215	246	277	307	384	461	538	615	692	768
130	4.55	3	7	10	13	17	25	33	50	83	117	133	166	200	233	266	300	333	416	499	583	666	749	832

Note: We assume a 10-percent sustainable WCC fish harvest rate for the values in Exhibit E-13. Those values divided by 365 days/year = kg fish harvested/day. Multiplied by 0.33 edible fraction = kg fish fillet/day for one person. The BC fish are more abundant; therefore, if the angler can consume 0.013 kg WCC fish/day, the angler also can consume 0.013 kg BC fish/day. Thus, at a total fish standing biomass of 40 g ww/m^2, a 25-acre lake can support ingestion of 26 g total fish fillet/day (see Equations E-9 and E-10), or 1 g total fish fillet can be harvested per lake acre.

E.1.7 References

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Appendix 7 Protocol for Site-Specific Multipathway Risk Assessment

Protocol for Developing a TRIM.FaTE Model Scenario to Support a Site-specific Multipathway Risk Assessment in the RTR Program

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CONTENTS

1. Introduction 1
   1.1 Regulatory Context and Approach to Risk Assessment for PB-HAPs 1
   1.2 Purpose of this Protocol 2
   1.3 Scope and Limitations 2
   1.4 Caveats 3
   1.5 Protocol Road Map 3
2. A Brief Introduction to TRIM.FaTE Input Requirements 4
   2.1 TRIM.FaTE Input Files and Contents 4
   2.2 Recommended Sequence of Activities for TRIM.FaTE Set Up 5
3. Meteorological Data Development 5
4. Air and Surface-parcel Design 7
   4.1 The Role of Spatial Layouts in TRIM.FaTE 7
   4.2 Recommended Best Practices for Air- and Surface-parcel Design 8
5. Air Modeling Using AERMOD 9
   5.1 Rationale for AERMOD Deposition Inputs 9
   5.2 AERMOD Deposition Modeling 9
      5.2.1 AERMOD Receptor Grid 10
   5.3 Incorporating AERMOD Results into TRIM.FaTE 10
6. Surface Hydrology and Erosion Property Definitions 10
   6.1 Surface-parcel Chemical Transfer Dynamics in TRIM.FaTE 10
   6.2 Estimating Runoff and Erosion Fractions without Sophisticated GIS Software 11
   6.3 Estimating Runoff and Erosion Fractions with Sophisticated GIS Software 12
7. Developing Values of Compartment Properties 13
   7.1 The Role of Properties in TRIM.FaTE 13
   7.2 Approach to Prioritizing Properties for Site-specific Parameterization 13
   7.3 Elimination of Properties from Site-specific Parameterization 14
      7.3.1 Process-based Elimination of Parameters 14
      7.3.2 Data Availability-based Elimination of Parameters 15
      7.3.3 Combination of Data- and Sensitivity-based Elimination of Parameters 15
      7.3.4 Elimination of Physical and Chemical Characteristics 16

7.4 Properties Recommended for Site-specific Parameterization 16
7.5 Properties Recommended for Values Based on Land Use 17
7.6 Properties Recommended for National Values 18

8. Potential Future Improvements 19

References 20

Appendix A. Documentation of Empirical Analyses Used to Prioritize TRIM.FaTE Properties A-1

Appendix B. AERMOD-to-TRIM.FaTE Input Requirements.....B-1

TABLES

Table 3-1. Meteorological Parameters Required for Meteorology Input File for TRIM.FaTE 6

Table 7-1. TRIM.FaTE Properties Recommended for Site-specific Parameterization.....17

Table 7-2. TRIM.FaTE Properties Recommended for Parameterization Based on Land Use 18

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1. Introduction

This document presents a protocol for developing TRIM.FaTE scenarios in support of site-specific multipathway risk assessments conducted within the Risk and Technology Review (RTR) program using the TRIM.FaTE environmental fate-and-transport model.

This section describes the regulatory context, intended purpose of the protocol, the scope and limitations of the protocol, and some caveats to its use. It also presents a road map to the content and structure of this document.

1.1 Regulatory Context and Approach to Risk Assessment for PB-HAPs

Section 112 of the Clean Air Act (CAA) directs the U.S. Environmental Protection Agency (EPA) to assess the risk remaining (residual risk) from emissions of hazardous air pollutants (HAPs) following the implementation of maximum achievable control technology (MACT) standards for emission sources. Such risk assessments for various emission source categories are a major component of EPA's RTR program.

To evaluate multipathway exposures and human health risks for RTR on a source category basis, EPA currently employs an iterative approach. The approach enables EPA to confidently screen out emissions of persistent and bioaccumulative HAPs (PB-HAPs) unlikely to pose health risks above levels of concern and to focus additional resources on sources of greater concern within the category.

Three models are used to estimate multipathway exposure and multipathway risk in the RTR program, as noted below.

- AERMOD is EPA's preferred near-field dispersion model for regulatory applications, and EPA uses it for RTR site-specific multipathway assessments to model the transport of pollutants in air and subsequent dry and wet deposition to soil, plant surfaces, and water.
- EPA uses the Fate, Transport, and Ecological Exposure module of EPA's Total Risk Integrated Methodology (TRIM.FaTE) to model the fate and transport of pollutants deposited to soil, plant surfaces, and water.
- EPA uses the RTR multimedia ingestion risk estimation methodology[1] to estimate transfer and uptake into the farm food chain and exposure to receptors consuming contaminated fish, farm foods, and soil. A subset of media-concentration estimates from AERMOD and TRIM.FaTE serve as inputs for estimating risk, which also depends on other exposure and biotransfer-related input parameters. (This document focuses on the TRIM.FaTE modeling with inputs from AERMOD).

The RTR approach to multipathway assessments is divided into four steps of increasing refinement, which are described below.

1. Tier 1 of the approach identifies facility-level emissions of PB-HAPs within a source category and compares them to the screening threshold emission rates.

2. Tier 2 uses the actual location of the facility emitting PB-HAPs to refine a subset of the assumptions associated with the modeled Tier 1 environmental scenario while maintaining the Tier 1 multipathway exposure-scenario assumptions.

3. Tier 3 uses Web searches on local lakes to determine their fishability and suitability for the approach, and it also uses facility stack parameters and local hourly meteorology to estimate the impacts on potential exposure from plume rise and hour-by-hour variations in meteorological conditions.

[1]The multimedia ingestion risk estimation methodology used for RTR risk assessments is discussed in detail in Attachment B of Appendix 6 to the Risk Report.

4. The final step, for facilities that cannot be screened out based on the Tiers 1–3 screens, is to conduct a more refined, site-specific multipathway risk assessment. A site-specific risk assessment (the subject of this Protocol document) is intended to incorporate location- or facility-specific characteristics regarding the environment to which PB-HAPs are emitted, relevant exposure pathways, ingestion rates or other exposure factors, and other parameters. Site-specific risk assessments require more time and resources to complete than the screens. Unlike the screens, site-specific assessments utilize AERMOD to conduct air-dispersion and deposition modeling of chemical emissions, the results of which are input into TRIM.FaTE.

The methodology that EPA employs for the Tiers 1–3 screening is documented in *Technical Support Document for the TRIM-Based Multipathway Tiered Screening Methodology for RTR*, which is an appendix to the Risk Report.

1.2 Purpose of this Protocol

The site-specific protocol presented in this document is intended to serve as a guiding framework to set up and parameterize scenarios in TRIM.FaTE that support accurate and cost-effective site-specific risk assessments as part of the RTR framework.

The purpose of the protocol is to develop a standard set of guidelines and recommendations for conducting site-specific assessments, providing a streamlined and replicable framework for configuring and parameterizing the TRIM.FaTE model. The protocol aims to balance modeling accuracy with cost-effectiveness in implementation, and to facilitate consistency and transparency across diverse assessments. This protocol is also intended to function as part of the technical documentation for site-specific residual risk assessments by providing a clear and transparent description of the approach to parameterization and some of the relevant sources. Deviations from this protocol would need to be documented on a case-by-case basis.

1.3 Scope and Limitations

The site-specific protocol presented in this document focuses on the fundamental aspects of setting up a scenario in TRIM.FaTE from an RTR perspective. While the *TRIM.FaTE User's Guide* (U.S. EPA 2005) provides guidance on the mechanistic aspects of designing a simulation, the protocol focuses on identifying best practices that optimize model set-up efficiency while maintaining a high level of model precision in the RTR context. Chemical uptake into the farm food chain, average daily doses from ingestion of contaminated media, and subsequent health risks are calculated using the RTR multimedia ingestion risk estimation methodology and are not discussed here[1].

These best practices have been developed with a focus on the impact of alternative model-configuration and model-parameterization approaches on multipathway risk in the RTR process. Thus, if two alternative model-configuration approaches are estimated to have similar impacts on risk estimates in the RTR process, the protocol will recommend the less effort-intensive approach where appropriate. For instance, the protocol identifies only a limited set of TRIM.FaTE model properties as requiring site-specific parameterization, while proposing land-use-specific or nationally representative or health-protective values for others based on the finding that relatively few model parameters substantially influence risk in the RTR context.

However, the protocol is not driven exclusively by considerations of cost-effectiveness. In some instances, the protocol aims to provide superior methods of model configuration based on model accuracy and scientific considerations that were previously not clearly articulated in available TRIM.FaTE guidance and that have a focus on the RTR program.

This protocol is not step-by-step guide to running the model. It is not intended to serve as a substitute for the *TRIM.FaTE User's Guide* (U.S. EPA 2005) or the *TRIM.FaTE Technical Support Document* (U.S. EPA 2002), but it is recommended that the protocol be read in conjunction with those documents to provide a holistic perspective on how the model should be used in site-specific RTR applications.

1.4 Caveats

The findings and recommendations presented in this document are subject to the caveats given below.

- Some of the conclusions presented in this protocol are based on a combination of available empirical evidence, theoretical considerations, and expert judgment. A "brute-force" empirical approach to test an extensive range of scenarios and parameters was not feasible.
- For some model parameters, ICF relied on sensitivity analyses performed on previous configurations of the model. It is possible that the results of previous sensitivity analyses differ somewhat from the current Tier 1 screen model configuration.
- ICF did not test the sensitivity of model parameters in alternative model configurations.
- ICF did not research and identify land use-specific parameter values for soil properties values as part of this protocol, although it recommends their use.

Despite these limitations, the current recommendations are expected to meet the objectives of providing a cost-effective and accurate approach to site-specific multipathway risk assessment in the RTR program. However, users are encouraged to extend site-specific model design and parameterization beyond the levels proposed here as circumstances permit.

1.5 Protocol Road Map

This protocol contains

- best practices for TRIM.FaTE model configuration for use in site-specific RTR applications;
- documentation of the rationale for best-practice recommendations;
- nationally representative or health-protective model parameter values for site-specific applications of TRIM.FaTE; and
- documentation on using AERMOD deposition outputs in TRIM.FaTE.

These distinct elements are woven together in the structure noted below.

- Section 2 sets the context with a summary of TRIM.FaTE input files and their content.
- Section 3 discusses the model's meteorological data requirements, potential data sources, approaches to address missing data, data-processing requirements, and the issue of plume rise.
- Section 4 presents recommendations and rationale for best practices for designing air and surface parcels in TRIM.FaTE.
- Section 5 presents methodology for air-dispersion modeling in AERMOD and incorporation of deposition outputs into TRIM.FaTE.
- Section 6 presents recommendations and rationale for best practices for defining surface hydrology and erosion parameters required by TRIM.FaTE.
- Section 7 identifies parameters recommended for site-specific parameterization.
  - Section 7.5 identifies parameters recommended for land-use specific parameterization.
  - Section 7.6 identifies parameters recommended for national-default parameterization.
- Section 8 discusses potential future improvements and enhancements to the protocol.

2. A Brief Introduction to TRIM.FaTE Input Requirements

TRIM.FaTE is a spatially and temporally explicit multimedia environmental fate-and-transport model that estimates the concentrations of emitted chemicals in biotic and abiotic environmental media. The model uses a compartmental box-model approach to track the movement of chemicals in environmental media. The model is based on representing environmental media as compartments, moving chemical mass between interacting compartments consistent with a set of governing mathematical algorithms that describe physical and chemical processes in the environment, and assuming instantaneous mixing within each compartment.

2.1 TRIM.FaTE Input Files and Contents

TRIM.FaTE requires a variety of inputs from users to define the modeled environment and to quantify the various environmental mass-transfer processes. These inputs are provided to the model in the form of the files noted below.

- A "volume elements" file defines the spatial layout of the modeled domain in terms of three-dimensional abiotic compartments. Each volume element provides a frame of reference for one or more biotic compartments within it.
- A "compartments" file places biotic and abiotic compartments (modeling units containing chemical mass) within the volume elements.
- A "library" file contains all the model algorithms, properties, and emission-source information. Examples of the kinds of properties that are defined in the library file include
  - scenario characteristics (e.g., start/stop time, modeling time parameters, output options);
  - source characteristics (e.g., chemicals emitted, location, emission rate);
  - chemical-specific properties, including physiochemical (e.g., molecular weight, K_{ow}) and abiotic (e.g., degradation half-life);
  - nonchemical-specific characteristics of biota (e.g., body weight, food intake rate);
  - site-specific ecological setting data and characteristics of biota (e.g., type of species present, population and density information, food web relationships); and
  - abiotic environmental setting data such as abiotic media characteristics (e.g., air/water content of soil, pH of surface water, suspended sediment density), runoff/erosion fractions for adjacent surface soil compartments, and water flow between connected surface water compartments.
- A properties file typically contains: (i) simulation- and site-specific property values that are used to overwrite default library values, and (ii) the location of time-varying input files for parameters such as meteorological and vegetation parameters.

For RTR site-specific applications of TRIM.FaTE, EPA uses AERMOD to model the air dispersion of facility emissions and subsequent chemical deposition, rather than the simpler air-transport and deposition algorithms used in TRIM.FaTE (that is, TRIM.FaTE is run without a facility emission rate). The rationale for this is provided in Section 5.

The input files listed above must be developed using syntax that is consistent with TRIM.FaTE requirements. Further detail on the required syntax of the input files, and the process of setting up and running the model using these input files, is available in the *TRIM.FaTE User's Guide* (U.S. EPA, 2005). Supplemental input files are generated that set up AERMOD deposition rates as emission sources for TRIM.FaTE; further details on these input files can be found in Section 5 and Appendix B.

Much of the challenge in a site-specific TRIM.FaTE application lies in designing a spatial layout that is consistent with the nature of the governing algorithms and that reflects the environmental dynamics of the

modeled domain, researching and estimating numerous environmental properties that serve as inputs into the model, finding and preparing appropriate meteorological and climate-related data, and setting up the input files. The following sections discuss the optimal methods of performing these tasks from the perspective of a site-specific RTR application.

2.2 Recommended Sequence of Activities for TRIM.FaTE Set Up

The following sections of this document focus on various aspects of TRIM.FaTE set up as discrete elements in the model-configuration process. There are, however, interconnections between the research required to guide various components of the set-up process. Although there are no firm rules governing the order in which the model's input files must be developed, this protocol recommends the sequence of activities numbered below as a means to enhance efficiency and accuracy in the model-configuration process.

1. Perform qualitative spatial analyses of topography, aerial imagery, hydrography (boundaries of watersheds, flow lines), and land cover around the site. This will aid in identifying meteorology data, modeled lakes and farms or potential farmland, potential residences where home gardening might occur, and the shape of the model domain.

2. Identify meteorological data based on RTR considerations (e.g., what meteorology did the RTR inhalation risk assessment use for the site?), data availability, data quality, and the representativeness of the data and instrument siting with respect to the modeled facility. Create the meteorological file needed for modeling.

3. Identify lakes to model based on lakes evaluated in the Tier 2 screen, lake size, and a preliminary assessment of risk potential and data availability.

4. Identify farms or potential farmland and/or home gardens to model based on a preliminary assessment of risk potential and data availability.

5. Create the modeling spatial layout, including a receptor grid for AERMOD to calculate air concentrations and deposition rates.

6. Run AERMOD with chemical-specific properties and source characteristics to get receptor-specific air concentrations and deposition rates.

7. Estimate and define surface hydrology and erosion dynamics within the layout.

8. Gather data on site-specific properties per the protocol.

9. Generate TRIM.FaTE input files, including the supplemental files that set up surface-deposition rates transformed from AERMOD deposition outputs.

10. Run TRIM.FaTE.

3. Meteorological Data Development

RTR site-specific multipathway assessments use AERMOD to model the air dispersion and deposition to the ground of chemicals emitted by the facility (see Section 5), and then TRIM.FaTE models the re-suspension and re-deposition of chemical (which typically affects a small fraction of the chemical initially deposited) as well as the chemical transfer between terrestrial and aquatic media. Both models in general are highly sensitive to meteorological parameters such as wind speed, wind direction, mixing height, and rainfall rate, among others, and to the interactive effects between those parameters (see U.S. EPA 2009 for a sensitivity analysis for TRIM.FaTE). However, with this configuration of TRIM.FaTE using outputs of AERMOD, meteorological parameters have a greater impact on the results of AERMOD modeling (on the

air transport and initial deposition of facility emissions) than on TRIM.FaTE modeling (on chemical re-suspension and re-deposition, and transfer through compartments).

For these reasons, it is recommended that all meteorological parameters for AERMOD be site-specific at an hourly resolution (averaging to coarser time steps can obscure real trends in the data). For consistency, the TRIM.FaTE meteorological parameters should match those of AERMOD.

The development of meteorological data for the AERMOD modeling should follow established EPA guidance, and the selected surface and upper-air meteorological stations typically should match those used for the facility in the RTR inhalation assessment (which are typically the stations closest to the facility). This section discusses best practices in reformatting the data used in the AERMOD modeling for use in TRIM.FaTE.

TRIM.FaTE meteorology data must include the fields in Table 3-1. The TRIM.FaTE meteorology data file does not require hourly time steps (larger time steps will shorten model run time), although hourly data are used in site-specific assessments.

Table 3-1. Meteorological Parameters Required for Meteorology Input File for TRIM.FaTE

Parameter	Format	Units	Further Description and Notes
Date	M/D/YYYY	NA	NA
Hour	Numeric	NA	NA
Time Zone	e.g., “EST”	NA	NA
Horizontal Wind Speed	Numeric	m/s	NA
Wind Direction	Numeric degrees	degrees clockwise from north; blowing from	e.g., from north is 360 degrees; from east is 90 degrees; from south is 180 degrees; and so on. 0 degrees is reserved for calm winds (e.g., wind speed = 0 m/s), which cannot be used in TRIM.FaTE (minimum wind speed for TRIM.FaTE is 0.75 m/s).
Air Temperature	Numeric	K	NA
Mixing Height	Numeric	m	NA
Rain Rate	Numeric	m/day	NA
Cumulative Rain	Numeric	m	Total precipitation in a precipitation event. A multi-hour event will have equal cumulative rainfall values for each hour.
Is Day	Boolean (i.e., 1 or 0)	NA	Daytime (value of 1; after sunrise) or nighttime (value of 0; after sunset). Calculated using U.S. EPA’s SR-SS.exe program, available with TRIM.FaTE.

The AERMOD meteorology file produces two different calculations of mixing height—one based on convective turbulence, the other based on mechanically-generated turbulence. For use in TRIM.FaTE, these values should be condensed to one mixing height per hour: during convective conditions (when

sensible heat flux is positive) it should be the larger of the two values from the AERMOD meteorology file, and during all other times it should be the mechanical mixing height.

TRIM.FaTE requires that there be no missing data in its meteorology fields. U.S. EPA's recommended guidance for replacing missing meteorology data (U.S. EPA 1992) has a series of objective data replacement steps as a first pass, but those steps might not fill in all missing data. The guidance suggests some subjective procedures for filling in remaining missing data; however, these are manual steps and do not cover all possible cases of missing data (e.g., if more than a few contiguous hours of data are missing). A meteorologist or experienced air quality modeler should perform these subjective data-fill procedures. The user should expect that the quality of substituted values will be worse for longer contiguous periods of missing data versus only a few contiguous hours of missing data. However, as long as the amount of data originally missing is no more than 10 percent, and as long as the substituted values are not out of normal bounds, then substituted data will have only a small impact on modeling results—especially for RTR assessments where (1) AERMOD models primary air dispersion and deposition and (2) the desired TRIM.FaTE outputs are final accumulated media concentrations after several decades of modeling.

ICF developed a tool (AERMET2TRIM), based in Microsoft® Access™, that uses the meteorology file from the AERMOD modeling, fills in missing data in all meteorology fields needed by TRIM.FaTE (based on methods from U.S. EPA 1992), and reformats the data into the format required by TRIM.FaTE. Previous site-specific assessments conducted using TRIM.FaTE typically have used 50-year modeling periods, so the reformatting performed by AERMET2TRIM includes duplicating the one- or several-year meteorology file into a 50-year data period. For example, if the meteorology data represent years 2013–2016, that four-year period is repeated to create 50 years of data (e.g., 1990–2039).

If TRIM.FaTE time steps greater than 1 hour are desired (though not recommended), the user should aggregate the data to conform to the time step. Values of wind speed, air temperature, mixing height, and rain rates should be averaged. For wind direction, the hourly values of wind speed and wind direction should be used to calculate the vector components of the wind (u and v values), those vector components should be averaged, and the averaged vectors should be used to calculate the average wind direction. Calculate new cumulative rain values after averaging the rain rates. Use professional judgment to determine appropriate values for the "Is Day" parameter.

4. Air and Surface-parcel Design

4.1 The Role of Spatial Layouts in TRIM.FaTE

One of the primary inputs required by the TRIM.FaTE model is the specification of a spatial layout using Cartesian coordinates to define the vertices of surface and air parcels and volume elements. This information is input into the model via the "volume elements" input file. To construct the volume elements input file, users are required to divide the modeled domain into two-dimensional air parcels and surface parcels. Air parcels need not line up with surface parcels in all cases. Each parcel is also associated with a height, which may vary in time. The parcel coordinates and height are combined to define three-dimensional abiotic volume elements that contain biotic and abiotic compartments used to model the movement of chemical mass in TRIM.FaTE.

In a site-specific multipathway assessment, the spatial layout should capture the features of interest (farms, home gardens, and/or lakes) at the surface level and also specify how the overlying air domain is to be divided to produce accurate and informative estimates. Although the *TRIM.FaTE User's Guide* (U.S. EPA 2005) provides useful mechanistic guidelines and rules of thumb on the design of air and surface parcels, those recommendations are not specific to the RTR context and are not based on a multipathway risk perspective. The following guidelines, as noted in the introduction to this document, are intended to support site-specific multipathway risk assessments in the RTR program (utilizing AERMOD estimates of air dispersion and deposition of facility emissions) and should be considered in addition to the instructions and recommendations provided in the *TRIM.FaTE User's Guide* and *TRIM.FaTE Technical Support Document* (U.S. EPA 2002).

4.2 Recommended Best Practices for Air- and Surface-parcel Design

The following recommendations for air- and surface-parcel design are intended to maintain a high degree of modeling accuracy while reducing design effort and potentially optimizing computer run time. While these guidelines are intended to facilitate optimal parcel design, every scenario is unique and might require site-specific adjustments beyond the suggested approach provide here.

Step 1: Identify Features of Interest

Several steps are recommended for identifying features of interest, as described below.

- Use geospatial data (e.g., aerial imagery, data on watersheds and water bodies, and remotely-sensed land cover and crop growth) to identify features of potential interest from the RTR perspective, such as farms and lakes.
    - Geospatial data can include Google Earth (Google 2013), the National Hydrography Dataset (USGS 2013), the National Land Cover Database (MRLC 2013), and the Cropland Data Layer (USDA 2013).
- Use the guidance below to finalize the selection of lakes, gardens, and farms for modeling.
    - Features should be within 50 km of the emitting facility.
    - Prefer features closer to the emission source versus those farther away.
    - Prefer features that are frequently downwind from the emission source, if they exist, based on the meteorology data selected for modeling.
    - Prefer features that potentially receive elevated levels of chemical input via runoff and erosion from surrounding areas.
    - Prefer lakes for which preliminary research suggests good availability of modeling data (e.g., flush rates, depth, pH, total phosphorus levels, suspended sediment concentration).
    - Prefer the lake(s) selected in the Tier 2 and Tier 3 screens (all of which are between 25 and 100,000 acres in size).
    - Prefer features that are not very close to other features.
    - Hypothetical home gardens are most appropriately modeled at potential residential locations.

Draw a simplified surface-parcel polygon for each feature, reasonably capturing its true or intended surface area and location while minimizing unneeded complexity (complex feature boundaries will increase model run time while typically not improving modeling accuracy in a significant way).

Step 2: Identify Areas Potentially Impacting Features of Interest

Use geospatial data (elevation contours and watershed boundaries) to identify areas potentially providing non-negligible amounts of chemical runoff and erosion to the features of interest. These areas will include watersheds or sub-watersheds in which the features are located, as well as watersheds or sub-watersheds upslope from those features. These impacting areas should be within 50 km of the emitting facility.

Draw simplified surface-parcel polygons for these impacting areas. It is typically appropriate that they be delineated along watershed or sub-watershed boundaries, which the U.S. Geological Survey (USGS) or other agencies have estimated based on local elevation contours and water-body features that define the flow of rainfall and surface water across the local region. Because TRIM.FaTE requires that surface parcels be characterized based on land cover, it may also be appropriate to further subdivide the polygons based on primary land cover. These polygons should use appropriately simplified boundary definitions to minimize unneeded complexity.

Step 3: Draw Air Parcels

To improve the accuracy of TRIM.FaTE-modeled chemical re-suspension and re-deposition, it is recommended that the air parcels be co-located with surface parcels. This is chiefly important for the features of interest. If model run time is of concern, then air parcels away from the features of interest may be larger than the underlying surface parcels; professional judgment should be used in these cases, as fewer and larger air parcels will degrade the accuracy of calculations of re-suspension and re-deposition.

5. Air Modeling Using AERMOD

AERMOD (the AMS/EPA Regulatory Model) is the preferred model for near-field (less than 50 km) dispersion and it is used in RTR inhalation assessments. Its calculations of chemical air dispersion and deposition are more sophisticated than those of TRIM.FaTE; therefore, RTR site-specific multipathway assessments use AERMOD outputs as inputs into TRIM.FaTE.

5.1 Rationale for AERMOD Deposition Inputs

AERMOD uses a Gaussian plume approach based on a mathematical solution of the advection-dispersion equation with numerous meteorological, thermodynamic, and terrain considerations to calculate air concentrations. The model then calculates deposition using calculated or user-provided deposition velocities; calculated velocities are based on information about meteorological parameters, land cover, viscosities, resistances, particle sizes and densities (for particulate deposition), and pollutant-specific properties (for vapor deposition). It then computes deposition as the product of air concentration and deposition velocity. AERMOD assumes instantaneous steady-state concentrations and is not mass balanced.

The transfer factors in TRIM.FaTE are similar to the algorithms used by AERMOD for some types of deposition. TRIM.FaTE is a dynamic (non-steady-state), system-wide mass-balanced model. AERMOD and TRIM.FaTE are based on very different theoretical assumptions. Although the deposition factors translating air concentrations into deposition rates are approximately similar in some respects, the models are based on entirely different methods of computing air concentrations. Furthermore, in AERMOD, the user defines the particulate- and vapor-phase speciation in the emissions stream (and this distribution is assumed to remain constant during dispersion). By contrast, TRIM.FaTE phase speciation is computed within each air compartment based on chemical equilibration between the solid, aqueous, and air phases of the air compartments. Therefore, it is reasonable to conclude that overall net deposition is computed in very dissimilar ways by AERMOD and TRIM.FaTE.

The pseudo-source method, detailed in this section and Appendix B, combines air-deposition estimates from AERMOD with the surface modeling capabilities of TRIM.FaTE. The method introduces mass deposited from the air (based on AERMOD estimates) directly into the underlying soil, water, and vegetation compartments in the TRIM.FaTE model. Small amounts of chemical re-suspension from the surface, and subsequent re-deposition, are captured by the TRIM.FaTE algorithms; there is potentially a mass-imbalance issue with this approach, although it remains to be evaluated if it involves a significant loss of accuracy.

5.2 AERMOD Deposition Modeling

For the selected facilities of interest, AERMOD is used to estimate deposition rates for input into TRIM.FaTE. AERMOD modeling is completed using guidance from the Guideline on Air Quality Models, also published as Appendix W of 40 CFR Part 51, to determine model set up and application including

- four years of recent, representative meteorological data—the same surface meteorology data used to develop the meteorology file for TRIM.FaTE;
- local terrain elevations for sources and receptors;

- chemical properties and information collection request (ICR) emissions data specific to PB-HAPs of concern; and
- use of urban dispersion-modeling settings when appropriate.

5.2.1 AERMOD Receptor Grid

Deposition rates are calculated for a grid of receptors covering all surface parcels being utilized in TRIM.FaTE. The receptors may use uniform or non-uniform spacing between points. All TRIM.FaTE surface parcels should have at least one corresponding AERMOD receptor; parcels larger than about 1 acre should have at least two AERMOD receptors, and receptor spacing should not exceed 1,000 m.

5.3 Incorporating AERMOD Results into TRIM.FaTE

AERMOD produces deposition estimates for each receptor within a TRIM.FaTE surface compartment. AERMOD deposition rates are calculated on an hourly basis by the model, and then the model aggregates across the entire simulation period to produce a period-total deposition rate (g/m^2/period) for each modeled receptor. To aggregate these outputs for input to TRIM.FaTE, an average deposition rate for each TRIM.FaTE surface parcel is computed from the receptor estimates within each parcel. Area-weighted averages are used to ensure that varying receptor densities between and within parcels are properly accounted for. The units are converted to g/m^2/day for use in TRIM.FaTE (which assumes constant deposition with time).

In order to apply AERMOD average deposition rates directly to soil, plants, and surface water in TRIM.FaTE, several modifications to the model scenarios and libraries are required. A "pseudo-source" is created for each surface parcel to represent the deposition rate for the parcel as derived from AERMOD modeling, with an emission rate (in units g/day) equal to the product of the parcel surface area (m^2) and the spatially averaged deposition rate (g/m^2/day). Separate pseudo-sources are created for each deposition type: wet and dry, vapor-phase and particulate. In this way, the TRIM.FaTE processes of air transport and deposition are replaced by "emissions" directly into surface soil, plant, and surface water compartments. More details on how to input AERMOD deposition values in TRIM.FaTE can be found in Appendix B.

It should be noted that outputs from AERMOD also are used in the multimedia ingestion risk estimation methodology algorithms to estimate exposure and risk. Among other parameters, chemical air concentrations above features of interest and chemical-deposition rates to those features of interest are used. When site-specific RTR multipathway risk assessments use AERMOD for primary dispersion and deposition modeling, the values of air concentration and deposition in TRIM.FaTE reflect only the chemical re-suspended and re-deposited within TRIM.FaTE. Parcel-average AERMOD air concentrations must be calculated and included in estimating exposure and risk, and the parcel-average AERMOD deposition rates input to TRIM.FaTE must also be included in the equations used to estimate exposure and risk.

6. Surface Hydrology and Erosion Property Definitions

6.1 Surface-parcel Chemical Transfer Dynamics in TRIM.FaTE

The TRIM.FaTE model incorporates the ability to account for chemical transfers between adjacent surface parcels via runoff and erosion. The algorithms that model surface runoff and erosion in TRIM.FaTE simulate the advective chemical transfer dynamics between surface parcels without requiring spatial elevation information or land cover details as inputs. Instead, the algorithms depend on inputs explicitly specifying the destination of erosion and runoff from a specific parcel. In other words, for each surface parcel, users must specify the proportion of the erosion and runoff originating in that parcel that reaches specific adjacent parcels. These inputs are known as link properties in TRIM.FaTE and are typically specified in the TRIM.FaTE "properties" file discussed in Section 2. Users must also separately specify the average runoff and erosion rate for each surface parcel. These inputs are combined internally

with estimates of the chemical concentration in surface soil and soil water to estimate mass transfers that occur in conjunction with erosion and runoff processes.

The inter-parcel runoff and erosion parameter inputs in TRIM.FaTE are inherently site-specific because there is no logical default value for the percentage of runoff and erosion from one parcel that reaches an adjacent parcel. Simulations indicate that multipathway risk in the RTR process is sensitive to the choice of these values (refer to Appendix A). This section discusses options for parameterizing these inputs in site-specific TRIM.FaTE applications for RTR.

For users not having access to (or expertise in using) geographical information systems (GIS) software with features to quantitatively analyze surface hydrology and erosion, the recommended method of estimating parcel-to-parcel runoff/erosion fractions is summarized in Section 6.2. If sophisticated GIS software with features to analyze surface hydrology and erosion based on elevation is to be used, the recommended method is summarized in Section 6.3.

6.2 Estimating Runoff and Erosion Fractions without Sophisticated GIS Software

Without a license for sophisticated GIS software, the user can still obtain free GIS viewing tools that allow the user to display multiple layers of geospatial data and that have limited interaction with the data, including querying the data and measuring distances. With such viewing software, the method for estimating runoff/erosion fractions provided in Module 11 of the *TRIM.FaTE User's Guide* (U.S. EPA 2005) is appropriate. This method is summarized briefly here, with some additional tips not provided in the User's Guide.

Step 1: Assemble Hydrological and Elevation Data

The user should obtain geospatial data indicating boundaries of hydrological units relevant to the modeling domain. These hydrological data are available from the USGS National Hydrography Dataset (NHD; USGS 2013). These hydrography data should already have been obtained and used to inform the design of the modeling parcels. The NHD offers several levels of hydrological units, typically from regions (the most spatially coarse) to sub-watersheds (typically the highest spatial resolution). Considering that the typical site-specific TRIM.FaTE modeling domain has a radius less than 50 km and is divided into several surface parcels, watersheds or sub-watersheds will usually offer the most appropriate resolution for use in configuring parcels and estimating runoff/erosion fractions. The NHD also offers directional flow lines of streams, rivers, and other hydrographic features.

The user should also obtain elevation data for the modeling domain. High resolution data are available from the USGS National Elevation Dataset (NED; USGS 2006). These elevation data should already have been obtained and used to help construct the modeling parcels. The data with the highest spatial resolution are not necessary; 30-m resolution usually is appropriate.

Step 2: Relate Model Surface Parcels to Each Other and to Hydrological Units

The user should display the modeling surface parcels along with the appropriate hydrologic unit boundaries from the NHD. For each parcel ("sending parcel"), follow the steps below.

1. For each hydrologic unit that occupies at least part of the sending parcel, estimate (or calculate, if able) the ratio [surface area of the part of the hydrologic unit that is inside the sending parcel] to [surface area of the sending parcel].

2. Identify each neighboring parcel ("receiving parcel"), including sinks where appropriate for the sides of the sending parcel that lie along the outer boundary of the modeling.

3. For each hydrologic unit that occupies at least part of the sending parcel, estimate or calculate the length of each interface between the hydrologic unit and each receiving parcel (not discussed in the *TRIM.FaTE User's Guide*).

4. Estimate or calculate the fraction of the sending parcel's perimeter that interfaces with each receiving parcel (not discussed in the *TRIM.FaTE User's Guide*).

Step 3: Estimate Fraction of Runoff and Erosion

For each hydrologic unit that occupies at least part of a sending parcel, one should use NED elevation data and NHD flow lines to estimate the fraction of runoff that will flow from the hydrologic unit into each receiving parcel and, where appropriate, into sinks outside the modeling domain. A fraction might be 0 if the elevation and flow lines suggest that all water in the hydrologic unit flows away from a receiving parcel.

The *TRIM.FaTE User's Guide* (U.S. EPA 2005) Section A.5 discusses runoff/erosion fractions. Although not discussed there, the NHD flow lines can help estimate the relative distribution of runoff from a sending parcel to its receiving parcels, or from a hydrologic unit in the sending parcel to a receiving parcel. One can examine the flow lines along each sending-receiving boundary to get a sense how much of the boundary has flows from the sending area to the receiving area. This information can be combined with information on how much of the sending area's perimeter interfaces with the receiving area in question, aiding the user in developing runoff/erosion fractions.

As discussed in the *TRIM.FaTE User's Guide* Sections A.5 and A.6—separately for each hydrologic unit in a sending parcel, multiply [the fraction of sending parcel's area covered by the hydrologic unit] by [the runoff/erosion fraction from the hydrologic unit to the receiving parcel] for each of the sending parcel's receiving parcels. Then, for each of these receiving parcels, sum this product across the hydrologic units. This sum provides the final fraction of runoff/erosion from each sending parcel to each receiving parcel. For each sending parcel, the fractions will sum to 1 when sinks are included as appropriate.

Another option is to estimate the runoff and erosion fractions based on visual inspection. This approach does not explicitly relate the area of each hydrologic unit to each sending parcel. Therefore, it does not explicitly assume that water cannot cross the boundaries of hydrologic units. Like the methods described above, this option uses flow lines and the interfacial length between adjacent parcels. In this option, for each sending parcel, the user visually examines the NHD flow lines to see where (if at all) flow lines cross each interfacial boundary and into the receiving parcels. For each sending-receiving pair of parcels, the user should estimate (or measure, if possible) the length of the part of the interfacial boundary that has flow lines crossing into the receiving parcel. Then, divide that length by the total perimeter length of the sending parcel. This ratio provides the fraction of runoff/erosion from the sending parcel into the receiving parcel. Some professional judgment is required to subjectively adjust these fractions based on the relative magnitude of runoff across the various interfacial boundaries. These relative magnitudes can consider the overall terrain and flow patterns throughout the sending parcel (a flow into the receiving parcel with a relatively small fetch will likely carry less chemical into the receiving parcel than a flow with a relatively long fetch).

6.3 Estimating Runoff and Erosion Fractions with Sophisticated GIS Software

The method discussed in this section requires the use of ESRI® ArcGIS™ software. The software license must enable the "Spatial Analyst" extension.

In ArcGIS, select the "Flow Direction" tool of the "Spatial Analyst" extension. Given a raster elevation dataset (such as the NED), this tool will determine the flow direction of each raster cell to the steepest downhill neighboring raster cell. The output of this tool will be a raster, where the value of each raster cell will indicate the flow direction.

Then, select the "Flow Accumulation" tool of the "Spatial Analyst" extension. The input to the "Flow Accumulation" tool is the output of the "Flow Direction" tool described above. Separately for each input raster cell, the "Flow Accumulation" tool will follow the flow direction into the appropriate neighboring cell, and continue following the flow direction of that cell into a third cell, and so on, "connecting the dots" of the flow vectors until an endpoint is reached. This creates flow lines across the raster. Then, the tool

calculates the number of these flow lines that cross each raster cell. This is the "flow accumulation" number produced by this tool. The flow accumulation is unitless, as it does not represent an actual amount of water or chemical flowing from one place to another; the accumulation values should be viewed relative to each other.

For each sending parcel, the user would use the combination of flow-direction and flow-accumulation data from the above tools to calculate the total flow (unitless) from the sending parcel to each receiving parcel. The runoff/erosion fraction from the sending parcel into receiving parcel "A" would be the accumulated flow from the sending parcel to receiving parcel "A" divided by the total accumulated flow from the sending parcel to all its receiving parcels.

7. Developing Values of Compartment Properties

7.1 The Role of Properties in TRIM.FaTE

The TRIM.FaTE model is dependent on hundreds of user-specified properties that describe the biotic and abiotic environments being modeled. Properties in TRIM.FaTE can be broadly divided into the types listed below.

- Nonchemical-specific properties that define biotic compartments (e.g., biomass of game fish in a lake, the length of a leaf on a deciduous plant).
- Nonchemical-specific properties that define abiotic compartments (e.g., porosity of surface soil, the total suspended solids concentration in a lake).
- Chemical-specific properties (including system-wide chemical properties such as the Henry's Law constant, the octanol-water partition coefficient, and compartment-specific chemical properties such as reaction and degradation rate constants in various environmental media).
- Simulation-specific properties (e.g., model run time, model time step).

All user-defined (e.g., non-formula) properties in a TRIM.FaTE scenario can be assigned simulation- or site-specific values. In theory, the more properties that are assigned site-specific values, the more accurately the simulation will represent chemical fate and transport at that location. Following this logic, the user should try to find site-specific values for as many properties as possible. However, although each model property is potentially important in defining a particular environmental fate-and-transport process, it is apparent based on theoretical considerations and empirical evidence (analysis of model results and model evaluations) that there is a subset of model properties that more significantly influences the environmental concentrations that drive the risks of importance in the exposure scenarios evaluated in RTR assessments. The fact that some parameters are more influential on results is true for complex models in general. This is the focus of sensitivity analyses.

In previous site-specific risk assessments using TRIM.FaTE, which were conducted for RTR and in other regulatory applications, a substantial portion of the level of effort required to perform the assessments was directed toward site-specific property parameterization. One of the specific objectives of this protocol is to take advantage of the results of sensitivity analyses and model evaluations conducted of TRIM.FaTE. Based on these results, we have identified those compartment properties that are a high priority for site-specific parameterization, those that can be adequately represented by regional or land-use-specific default values, and those for which nationally representative or health-protective values are adequate. This classification scheme is intended to reduce the level of effort required to adequately parameterize site-specific multipathway assessments while maintaining a high level of accuracy in risk estimates for RTR.

7.2 Approach to Prioritizing Properties for Site-specific Parameterization

ICF relied on a combination of theoretical reasoning and empirical evidence to prioritize TRIM.FaTE properties for the purposes of this protocol. In this way, ICF was able to limit the need for "brute-force"

empirical evaluations (e.g., comprehensive sensitivity analyses that systematically vary all or most of the user-defined inputs, such as those conducted prior to the 2009 EPA Science Advisory Board review of RTR assessments (U.S. EPA 2009)) and additional resource-intensive literature searches. ICF's justification for determining that properties were ***not*** high priority was based on the three lines of evidence discussed below.

1. ICF followed a **"process"-based approach** to rule out a large subset of TRIM.FaTE properties from the need for site-specific parameterization. This approach was founded on the idea that the TRIM.FaTE model produces greater than necessary resolution (in terms of the number of concentrations that are calculated for different environmental media types) when viewed from the RTR perspective. The individual human multipathway exposure scenarios evaluated for RTR rely most directly on results from TRIM.FaTE for surface soil compartments at the location of a farm or garden and fish compartments in a lake of interest. All fate-and-transport processes—and the properties that exclusively define those processes—that do not strongly influence these concentrations can reasonably be ruled out from requiring site-specific parameterization. The implications of this approach will be discussed in greater detail below.

2. ICF also used practical considerations regarding **data availability** to rule out certain properties from site-specific parameterization. Over the course of numerous site-specific assessments and the parameterization of the screening scenarios, ICF has conducted literature searches on numerous TRIM.FaTE properties. ICF used the insight gained from these exercises to identify certain sets of parameters as being too data-scarce to parameterize on a site-specific basis at this time without expending a substantial amount of time and money (for possibly uncertain results).

3. **Physical constants and physicochemical properties** of the modeled PB-HAPs were also ruled out from site-specific parameterization based on their largely unchanging nature in the environment for the chemicals considered for RTR.

ICF evaluated the parameters not eliminated by the above considerations to determine which properties should be the focus of data collection efforts during site-specific TRIM.FaTE modeling for RTR. ICF conducted a limited number of evaluations and used the results of previous sensitivity analyses to decide which of these shortlisted parameters should be prioritized for site-specific parameterization, for land-use-based parameterization, or for regional parameterization.

Other scenario properties, such as emission period and the model's numerical integration time step, are typically not varied between site-specific assessments.

7.3 Elimination of Properties from Site-specific Parameterization

7.3.1 Process-based Elimination of Parameters

The operative principle in the process-based elimination of parameters is that fate-and-transport processes that do not substantially influence concentrations of interest from an RTR perspective are less important to parameterize. ICF used theoretical considerations based on the evaluation of the underlying TRIM.FaTE algorithms, combined with empirical evidence from TRIM.FaTE simulations, to identify the less important fate-and-transport processes and eliminate the need to parameterize those processes on a site-specific basis. The specific processes identified as being of less importance in the RTR context and the underlying justification for ruling them out from site-specific consideration are listed below:

- Chemical transported via water percolation through the sub-surface soil layers (not including surface soil) does not affect surface soil or lake water concentrations. Theoretical considerations suggest that chemical, once transported into the lower soil layers, will not substantially make its way back to the surface compartments of interest.

- Chemical transport via sub-surface soil diffusive processes, although having the potential to transfer mass upwards, is not sizeable in comparison to advective transfer processes. An evaluation of relative mass rate in the Tier 1 screen supports this assertion for all the chemicals evaluated.
- Chemical transport to the lake via horizontal groundwater flow and recharge is negligibly small compared to other advective chemical inputs into the lake. The relative mass rate for this process compared to other advective transfer processes carrying chemical into the lake in the Tier 1 screening scenario supports this assertion for all chemicals evaluated.

Because the RTR user has no intrinsic interest in the concentrations prevailing in the lower soil layers, all of the above processes have been ruled out from consideration for site-specific parameterization. As a consequence, it is possible to rule out all sub-surface soil compartment properties from requiring site-specific parameterization.

7.3.2 Data Availability-based Elimination of Parameters

Chemical-Specific Aquatic Biota Properties: The aquatic biota compartments in TRIM.FaTE—currently including benthic invertebrates and five types of fish—are characterized by several potentially site-specific properties that control algorithms influencing the uptake, degradation, and elimination of chemicals in the aquatic organisms. These chemical-specific properties include the absorption rates of chemical into each type of fish from surface water, elimination rates from fish digestive systems, degradation rates within the fish, and other parameters. In the course of parameterizing TRIM.FaTE for the screens and conducting extensive evaluations of parameter sensitivity, it has become apparent that only a limited number of studies are available for several of these properties for most combinations of chemicals and organisms.

Although these properties may potentially differ in alternative climates and conditions, it appears unlikely that additional literature searches and evaluations would yield better, more appropriate site-specific values than the current defaults. Until such time as more studies on these properties are available, practical considerations suggest that these chemical-specific aquatic biota properties be ruled out from site-specific parameterization.

Chemical-Specific Abiotic Compartment Properties: TRIM.FaTE algorithms model chemical reaction and degradation processes in several abiotic compartments (e.g., surface soil). These algorithms depend on chemical-specific parameters such as degradation rates (or half-lives), transformation rates, and other properties. Literature searches conducted during previous site-specific assessments in the RTR process and other regulatory applications using TRIM.FaTE have suggested that data are limited for these properties.

These chemical properties (with the exception of the oxidation, reduction, and methylation and demethylation rates influencing mercury) therefore are currently ruled out from site-specific consideration. The mercury transformation properties have been shown to be highly risk-influential as well as variable across different ecosystem types and conditions, and these properties are reserved for site- or land-use-specific parameterization in the future, subject to greater data availability and the results of additional evaluations.

7.3.3 Combination of Data- and Sensitivity-based Elimination of Parameters

Terrestrial Vegetation: The terrestrial vegetation compartments in TRIM.FaTE—currently including grass, coniferous forest, deciduous forest, wetland grass, and wetland forest—are not directly part of the RTR risk assessment calculations (chemical concentrations in these compartments are not used as inputs to the ingestion exposure estimation). However, these compartments act as sinks for chemicals and also transfer chemicals from air to soil via leaf litterfall. In this way, the choice of terrestrial vegetation influences surface soil concentrations and, ultimately, risk.

The terrestrial vegetation compartments depend on properties such as the lipid content of leaves, wet density of leaves, area indices of leaves, etc. Although it is possible that these properties differ on a site-specific basis—for instance, the characteristics of coniferous trees in Oregon are different from those of

coniferous trees in North Carolina—these differences are not expected to have a substantial influence on risk. ICF's simulations indicate that the impact on risk of alternative vegetation scenarios is limited after accounting for differences in erosion regimes specific to land-use type (see Appendix A). It is expected that site-specific differences within a single vegetation type would be even lower.

Literature searches during previous site-specific multipathway assessments in the RTR process have indicated that highly intensive literature search would be required to parameterize the full range of terrestrial vegetation parameters required by the TRIM.FaTE algorithms. Based on the limited risk impact of terrestrial vegetation properties, and limited data availability at the site-specific level, these properties are currently ruled out from site-specific parameterization.

7.3.4 Elimination of Physical and Chemical Characteristics

Algorithms in the TRIM.FaTE model frequently depend on physical and chemical parameters, such as the Henry's law constant and the octanol-water partition coefficient, to partition chemicals between phases within a compartment. These properties are, by their nature, relatively unchanging across most standard environmental conditions for the non-ionic organic compounds currently evaluated (dioxins/furans and polycyclic organic matter)[2]. These properties are thus ruled out from requiring site-specific parameterization for the time being.

7.4 Properties Recommended for Site-specific Parameterization

Following the elimination process described above, ICF identified a set of parameters for further evaluation based theoretical considerations as well as higher sensitivity potential displayed in previous sensitivity analyses (e.g., U.S. EPA 2009). To estimate the risk influence of these parameters, ICF performed a limited set of additional sensitivity analyses. The evaluated parameters are listed below, grouped by compartment type.

- Air: dust load, fraction of organic matter.
- Surface Soil: unit soil loss, inter-compartment drainage and erosion fractions, soil-particle density, soil-air fraction, soil organic content, soil pH, soil water content.
- Surface Water and Sediment: suspended solids concentration, bed-sediment density, suspended solids density, bed-sediment porosity.
- Aquatic Biota: biomass of various aquatic biota compartments.
- Terrestrial Vegetation: "Allow exchange" and "Litterfall" file inputs.

Unlike previous analyses, these sensitivity analyses were not based on fixed perturbations from the default values but instead used reasonable high and/or low bounds approximately corresponding to the range found in the environment. The impacts on risk were computed with respect to the Tier 1 screen results at equivalent emission rates. AERMOD was not incorporated into these runs.

ICF extended the scope of the current analyses by also using the results of TRIM.FaTE sensitivity analyses conducted in previous regulatory applications and pertaining to air, surface soil, and surface water and sediment. Although these analyses were performed on a different version of the Tier 1 screen setup, the results are considered informative.

The specific details of the analyses conducted as part of this protocol development are reported in Appendix A, while other supporting evidence has been drawn from previous reports (e.g., U.S. EPA 2009). Based on the results of these analyses, Table 7-1 contains TRIM.FaTE properties recommended for site-specific parameterization in the RTR process. These properties have been further classified as

[2] For mercury, some analogous properties, such as the partition coefficient for mercury in the aqueous phase, do vary according to pH; these relationships are incorporated into the model as formula properties.

high, medium, and low priority to facilitate an appropriate allocation of available resources in the parameterization process.

Table 7-1. TRIM.FaTE Properties Recommended for Site-specific Parameterization

Compartment	Property	Priority	Remark
Surface water	Depth	High	Having depth as well as flush rate helps serve as a check on surface hydrology assumptions.
	Flush rate	High	
	Suspended solids concentration	High	Attempt to find a column-averaged value.
	pH	Moderate	Important for metals.
	Algae density	Moderate	May be estimated from total phosphorus concentrations in the absence of measured values.
	Organic-carbon fraction	Moderate	Important for 2,3,7,8-tetrachlorodibenzo-p-dioxin (U.S. EPA 2009). Data availability may be limited.
	Water temperature	Moderate	Sensitive but unlikely to manifest wide range.
Aquatic biota	Biomass	Moderate	May be estimated from total phosphorus concentrations in absence of measured values.
Surface soil	pH	Moderate	Important for metals.
Terrestrial biota	“Allow Exchange” and “Litterfall” data files	Low	These files govern how long leaves remain open for stomatal exchange during different times of the year and also when the leaves fall off the trees onto the surface soil. Although the impact of these properties has not been empirically tested, theoretical considerations suggest they will have a low impact when estimating average annual risks.

In addition to these values, meteorology parameters, surface hydrology and erosion-related parameters, and the spatial layout are fundamentally site-specific elements of a TRIM.FaTE simulation, as noted in the previous sections.

7.5 Properties Recommended for Values Based on Land Use

In addition to the properties identified in Section 7.4 as desirable for site-specific parameterization, we identified properties that also influence risk substantially but for which the impacts on risk are expected to be largely captured by land-use-specific parameters. In other words, for these properties, accounting for variations that correspond to land use is expected to adequately account for any variation in these parameters (to the extent that they influence risk). Additional variation in parameter values resulting from site-specific variations within a particular land-use category is not expected to be significant. For example, differences in surface soil erosion (as expressed by the unit soil loss rate property in TRIM.FaTE) are expected to be larger between the average deciduous forest and the average parcel of tilled soil than between different types of deciduous forest or between different types of tilled soil. The use of land-use-specific values for such properties is expected, therefore, to adequately capture their impact on risk estimates in the RTR process.

The rationale for identifying properties as land-use-based in this protocol is a combination of risk sensitivity analysis (Appendix A and U.S. EPA 2009), professional judgment about the range exhibited in the environment, and expected data availability at the site-specific level. Table 7-2 lists the TRIM.FaTE parameters that are recommended for land-use-specific parameterization. These parameters are all related to the surface soil parcel and assume distinct values for each of the land-use types modeled in TRIM.FaTE. These land-use types currently include deciduous forest, coniferous forest, grass, agricultural soil, untilled soil, forested wetlands, and grassy wetlands. Land-use type is not an explicit input in TRIM.FaTE but is implicitly reflected in the TRIM.FaTE property values corresponding to each surface parcel.

Table 7-2. TRIM.FaTE Properties Recommended for Parameterization Based on Land Use

Property	Remark
Organic-carbon fraction	Fraction of dry-soil solids that is organic in origin.
Water content	The sum of the water and air content fractions of a soil determines its porosity.
Air content	
Particle density	Refers to the dry density of the average soil particle.
Rainfall/erosivity index	Universal Soil Loss Equation (USLE) properties used to compute each surface soil compartment's average erosion rate.
Soil-erodibility index	
Topographical (LS) factor	
Cover/management factor	
Supporting-practices factor	
Fraction of precipitation that evapotranspires	Water-balance-related property used to compute each surface soil compartment's average runoff rate.
Fraction of precipitation subject to overland runoff	

7.6 Properties Recommended for National Values

Nationally representative or health-protective values are recommended for all TRIM.FaTE properties that are not identified for site-specific or land-use-based parameterization in Sections 7.4 and 7.5 above. These properties are expected either to (1) not substantially influence risk in the RTR process, (2) not have adequate data to support site-specific parameterization, or (3) be relatively constant in the environment, as discussed in greater detail in the approach described earlier in Section 7. These properties have been previously characterized in the RTR Tier 1 and Tier 2 screening threshold derivation analyses by either nationally representative values or health-protective values. The same values are recommended for these properties in site-specific analyses. The national values are documented in the *Technical Support Document for the TRIM-Based Multipathway Tiered Screening Methodology for RTR*, which is an appendix to the Risk Report.

8. Potential Future Improvements

This protocol, documenting the current state of knowledge related to conducting site-specific environmental modeling in support of RTR multipathway risk assessments, could be enhanced in the future by documenting best practices and developing recommendations regarding the issues listed below (among others).

- Identification of land-use-specific parameters for the identified soil properties based on literature review.
- Application of enhanced technical approaches, such as the use of a sensitivity-score approach, to identify the most influential model properties.
- Additional sensitivity tests utilizing the combined AERMOD-TRIM.FaTE approach as well as the current TRIM.FaTE Master Library (now including arsenic) and screening configuration.
- Potential development of regional parameters for a subset of model properties based on the results of further sensitivity analysis and data-availability assessments.
- Greater use of graphics and figures to illustrate model set-up concepts.
- Enhanced technical editing to help the protocol be more self-explanatory and independent of other TRIM.FaTE support documents in its scope.
- Researching the potential for geographically variable biotransfer factors and other parameters used in estimating concentrations of ingested food products.
- Further research and development of GIS-based approaches to surface hydrology and erosion-property parameterization.

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Appendix A. Documentation of Empirical Analyses Used to Prioritize TRIM.FaTE Properties

A.1. Introduction

Several years ago, ICF performed a series of empirical analyses to prioritize TRIM.FaTE model properties for site-specific parameterization. These analyses were based on changing the value of one or more model properties relative to the Tier 1 screen and measuring the relative impact on risk. Unlike in a traditional sensitivity analysis, this analysis changed property values to approximate high- and low-end values within the environmental range of the property of interest, instead of using a fixed perturbation. The measured impacts on risk, the expected range in the environment, and data availability were considered in prioritizing model properties for site-specific parameterization, as discussed in Section 7. These model runs did not utilize AERMOD and did not include arsenic, which was added to RTR multipathway assessments in 2016. The TRIM.FaTE Master Library and Tier 1 configuration used in these empirical analyses may be different from the current Library and configuration.

Table A-1 summarizes the various empirical analyses that were conducted, the risk impact of the scenario modifications, and conclusions from the analyses.

Table A-1. Results and Conclusions from Empirical Analyses Used to Prioritize TRIM.FaTE Properties

Scenario Name	Scenario Description (with respect to Tier 1 Screening Scenario)	Normalized Risk Relative to Tier 1 Screening Scenario				Risk Impact of Scenario Modification	Conclusions
		2,3,7,8-tetrachloro dibenzo-p-dioxin	Benzo(a) pyrene	Cadmium	Methyl Mercury		
Tier 1 SS	Tier 1 Screening Scenario.	1.00	1.00	1.00	1.00	Designed to produce most conservative risk estimate.	All relative risks for modified scenarios are measured relative to the Tier 1 screening scenario.
WF1	Reduce watershed flows (erosion and runoff) to half screening scenario levels. Redirect remainder to sink. Maintain same flow directions as screening scenario.	0.56	0.69	0.21	0.38	Reducing the quantity of runoff and erosion reaching receiving compartments reduces chemical inputs into those compartments, including the lake, and reduces risk.	Surface hydrology and erosion flows (where and how much of the erosion and runoff from a compartment reaches) are potentially highly sensitive properties in the model (influencing risk by up to a factor of 10) and are recommended for site-specific parameterization.
WF2	Reduce watershed flows (erosion and runoff) to 1/10 screening scenario levels. Redirect remainder to sink. Maintain same flow directions as screening scenario.	0.34	0.68	0.11	0.25		
ER0	Switch off erosion.	0.39	1.02	1.10	0.43	Turning off erosion reduces chemical inputs into the lake and reduces chemical removal off the farm.	Although erosion is a relatively important process, its maximum impact on risk is less than a factor of 3, even when

Scenario Name	Scenario Description (with respect to Tier 1 Screening Scenario)	Normalized Risk Relative to Tier 1 Screening Scenario				Risk Impact of Scenario Modification	Conclusions
		2,3,7,8-tetrachlorodibenzo-p-dioxin	Benzo(a)pyrene	Cadmium	Methyl Mercury		
ER1	Double erosion rates.	1.09	0.99	0.86	0.86	Increasing erosion produces competing effects: while it increases chemical inputs into the lake, it also increases the burial rate of sediment and increases chemical removal from the farm.	accounting for variable runoff rates. A land-use-specific parameterization approach is recommended for the average erosion rates of surface soil compartments.
ER-RUN1	Double erosion and runoff rates (same flush rate; higher lake depth).	0.94	0.96	0.74	0.89	Increasing the runoff rate increases the input of soluble chemicals into the lake and decreases the removal of those chemicals from the farm.	
ER-RUN2	Double erosion and runoff rates (higher flush rate; same lake depth).	1.09	0.99	0.74	0.85	Increased runoff rates can be accommodated by means of increased lake depths or increased flush rates.	
RUN1	Switch off runoff; maintain flush rate and depth.	0.99	1.00	0.70	0.96	Nullifying chemical transfer through runoff reduces chemical input into the lake and reduces chemical removal from the farm.	Runoff rates have a limited impact on risk. A land-use-specific parameterization approach is recommended for average runoff rates from surface soil compartments.
RUN2	Implement cumulative runoff regime.	1.02	1.00	1.14	1.10	Assumes runoff from one compartment does not evaporate but contributes to runoff from the receiving compartment.	

Scenario Name	Scenario Description (with respect to Tier 1 Screening Scenario)	Normalized Risk Relative to Tier 1 Screening Scenario				Risk Impact of Scenario Modification	Conclusions
		2,3,7,8-tetrachlorodibenzo-p-dioxin	Benzo(a)pyrene	Cadmium	Methyl Mercury		
FR1	Double lake depth, half flush rate, same rainfall, and same runoff fraction.	0.71	0.96	1.00	1.15	Doubling depth reduces concentrations but halving the flush rate reduces chemical output from the lake.	Lake depth and flush rate have a modest impact on risk. However, knowledge of both these parameters can help guide the surface hydrology and erosion direction flows in the watershed which can more substantially influence risk. Site-specific parameterization is recommended for lake depth and flush rate.
FR2	Half lake depth, double flush rate, same rainfall, and same runoff fraction.	1.28	1.07	1.00	0.92	Halving depth increases concentrations but doubling the flush rate increases chemical output from the lake.	
FR3	Double depth, same flush rate, same rainfall, same runoff fraction (violate water balance in screening scenario).	0.69	0.96	0.58	1.02	Doubling depth reduces lake concentrations for most chemicals.	
FR4	Double flush rate, same depth, same rainfall, and same runoff fraction (violate water balance in screening scenario).	0.95	1.00	0.58	0.89	Doubling flush rate reduces lake concentrations.	
PERC1	Implement balanced percolation regime.	0.99	1.00	0.62	0.99	Assumes runoff from one compartment does not evaporate but percolates in the receiving compartment.	Percolation rate (the fraction of rainfall that is subject to percolation into the sub-surface) has a modest impact on risk. Land-use-based parameterization is recommended for this property.

Scenario Name	Scenario Description (with respect to Tier 1 Screening Scenario)	Normalized Risk Relative to Tier 1 Screening Scenario				Risk Impact of Scenario Modification	Conclusions
		2,3,7,8-tetrachloro dibenzo-p-dioxin	Benzo(a) pyrene	Cadmium	Methyl Mercury		
R1	Reduce rainfall down to 1/3rd SS value; same lake depth; runoff rates and flush rate down to 1/3rd.	0.64	0.59	0.92	0.58	Reducing rainfall reduces chemical washout from air.	This run, when combined with earlier runs focusing on the impacts of flush rate, suggests that the chemical washout impact of rainfall has more influence on risk than the impact of rainfall levels on hydrological properties like flush rate. This reinforces the argument for site-specific meteorological parameters.
V_C	Set all surface compartments except farm to coniferous forests.	0.79	0.75	0.40	0.87	The choice of vegetation in surface soil compartments impacts risk by absorbing chemicals from air and soil and then redepositing them onto the surface soil via litterfall.	Land-use type has a limited impact on risk. Based on these results, terrestrial vegetation parameters are recommended for land-use-specific parameterization. In interpreting these results, it is important to note that these runs have not been normalized for erosion rates. Therefore, the impacts on risk presented here are from a combination of impacts from differential erosion rates and vegetation types.
V_D	Set all surface compartments except farm to deciduous forests.	0.34	0.92	0.49	0.39		
V_G	Set all surface compartments except farm to grassland.	0.88	0.82	0.45	0.92		
V_U	Set all surface compartments except farm to untilled soil.	0.42	0.73	0.37	0.81		
V_WW	Set all surface compartments except farm to forested wetlands.	0.36	0.92	0.49	0.47		

Scenario Name	Scenario Description (with respect to Tier 1 Screening Scenario)	Normalized Risk Relative to Tier 1 Screening Scenario				Risk Impact of Scenario Modification	Conclusions
		2,3,7,8-tetrachloro dibenzo-p-dioxin	Benzo(a) pyrene	Cadmium	Methyl Mercury		
V_WG	Set all surface compartments except farm to grassy wetlands.	0.86	0.83	0.46	0.94		
BM1	Increase aquatic biomass uniformly by a factor of 10.	0.84	0.39	0.90	0.99	Increasing aquatic biomass reduces chemical concentration in biomass as the same amount of chemical is distributed in a higher amount of biomass.	Risk is sensitive to the aquatic biomass levels. These properties are therefore recommended for site-specific parameterization. In interpreting the results of these runs, it may be noted that all biomass levels were uniformly raised. In real applications, the biomass levels of the upper trophic levels may constitute a lower percentage of the total biomass as total biomass increases, suggesting slightly lower risk sensitivity than apparent here.
BM2	Increase aquatic biomass uniformly by a factor of 100.	0.35	0.32	0.29	0.79		
Air_DL1	Increase air dust load by a factor of 10.	2.34	2.31	0.50	0.98	Increasing the dust load in air increases	Although these runs indicate that air dust load

Scenario Name	Scenario Description (with respect to Tier 1 Screening Scenario)	Normalized Risk Relative to Tier 1 Screening Scenario				Risk Impact of Scenario Modification	Conclusions
		2,3,7,8-tetrachloro dibenzo-p-dioxin	Benzo(a) pyrene	Cadmium	Methyl Mercury		
Air_DL2	Increase air dust load by a factor of 100.	4.14	2.71	0.50	0.90	particulate deposition to the surface.	moderately influences risk, literature search indicated that the range manifested by this property is relatively small and the default value used is already in the high end of the observed range in the U.S. Therefore, this property is not recommended for site-specific parameterization.
Air_FOM1	Halve the fraction of organic matter in air solids.	0.87	0.66	0.50	1.00	The organic content of air solids can differentially influence chemical adherence to the solid phase.	Although these runs indicate that the fraction of organic matter in air solids moderately influences risk, literature search indicated that site-specific data may be difficult to obtain. This property is not recommended for site-specific parameterization.
Air_FOM2	Double the fraction of organic matter in air solids.	1.23	1.43	0.50	1.00		
Soil_Air	Double the soil-air content.	1.21	1.29	0.50	1.14	Increasing the soil-air fraction reduces soil solids, which distributes the same amount of chemical over a lower solids content, thereby increasing soil concentrations.	Although these runs indicate that air dust load moderately influences risk, literature search indicated that the range manifested by this property is relatively small and the default value used is already in the high end of the observed range in the U.S. Therefore, this property is not recommended for site-
Soil_FOC	Increase the soil organic fraction content by a factor of 10.	1.04	1.01	0.60	1.00	Increasing soil organic content increases chemical adherence to soil for some chemicals.	

Scenario Name	Scenario Description (with respect to Tier 1 Screening Scenario)	Normalized Risk Relative to Tier 1 Screening Scenario				Risk Impact of Scenario Modification	Conclusions
		2,3,7,8-tetrachlorodibenzo-p-dioxin	Benzo(a)pyrene	Cadmium	Methyl Mercury		
Soil_pH1	Set soil pH at 4.	1.00	1.00	0.39	1.00	Soil pH can influence chemical adherence to soil solids for some chemicals.	specific parameterization.
Soil_pH2	Set soil pH at 10.	1.00	1.00	0.66	1.00		
Soil_Rho	Set soil solids density at 1000 kg/m^3.	1.41	1.43	0.50	1.16	Decreasing soil particle density increases soil concentrations when normalized by soil weight.	
Soil_Water	Double the soil water content.	1.00	1.00	0.50	1.00	Increasing soil water content increases chemical removal by percolation for some chemicals.	
SusSed_TSS1	Increase lake suspended solids concentration by a factor of 2.	0.73	0.98	1.01	0.74	Increasing suspended solids in water causes more chemical to be deposited to sediment.	Suspended solids concentration in lakes has a moderate influence on risk. Due to the wide range potentially exhibited by this property, it has been recommended for site-specific parameterization.
SusSed_TSS2	Increase lake suspended solids concentration by a factor of 10.	0.33	0.98	0.46	0.38		
Sed_Bur	Halve sediment-burial rate; same erosion rate (violate solids balance in screening scenario).	1.11	1.00	1.12	1.31	Decreasing the burial rate reduces the removal of chemicals from the sediment layer.	Sediment properties have a moderate impact on risk, given the limited range of values assumed by them in the environment.

Scenario Name	Scenario Description (with respect to Tier 1 Screening Scenario)	Normalized Risk Relative to Tier 1 Screening Scenario				Risk Impact of Scenario Modification	Conclusions
		2,3,7,8-tetrachlorodibenzo-p-dioxin	Benzo(a)pyrene	Cadmium	Methyl Mercury		
Sed_Rho	Decrease bed-sediment-particle density to 1000 kg/m^3.	1.36	1.00	0.63	2.74	The lower the sediment-particle density, the lower the volumetric re-suspension rate from sediment and the higher the volumetric burial rate.	
Sed_Por	Halve sediment-bed porosity.	0.85	1.00	0.42	0.78	The lower the sediment porosity, the lower the volumetric re-suspension rate from sediment and the lower the volumetric burial rate.	

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Appendix B. AERMOD-to-TRIM.FaTE Input Requirements

As mentioned in Section 5.3, following the pseudo-source methodology of incorporation AERMOD deposition outputs into TRIM.FaTE, pseudo-sources are assigned to placeholder volume elements and linked to transfer mass to the appropriate surface compartments. For water parcels, the entire mass is transferred to the surface water compartment. For land parcels, algorithms apportioning mass between surface soil and leaves were derived from the existing FaTE algorithms for air-soil and air-plant transfers. The mass-transfer rates were time-varying because they rely on factors such as hours of daylight and fraction leaf coverage. The placeholder volume elements are designed to prevent transfer to any compartment other than those prescribed, and each contains only one pseudo-source. These four pseudo-sources (representing dry/wet and vapor/particle deposition) for each parcel, along with the corresponding placeholder compartments, links, and algorithms, are set up in the input files described in Table B-1.

Table B-1. Supplemental Files to Parameterize AERMOD Outputs for TRIM.FaTE

File Description	Contents	Purpose
Pseudo-source deposition-rate properties	Assigns surface-deposition rates ($g/m^2/day$) to each of the placeholder volume elements (dry particle, dry vapor, wet particle, and wet vapor deposition as needed) for each surface parcel and chemical.	This file serves as TRIM.FaTE input file to parameterize the surface-deposition rates for the subsequent fate-and-transport modeling.
Pseudo-source volume elements	Coordinates specifying the spatial dimensions of the placeholder volume elements for each surface parcel.	This file serves as a TRIM.FaTE input file to supplement the defined spatial layout of the modeled domain to include pseudo-source compartments.
Pseudo-source library	Defines supplemental compartment types, property types, and algorithms used in linking placeholder volume elements to targeted surface compartments. This file also includes the definitions and locations of pseudo-sources with emission-rate formulas accounting for parcel surface area.	This file serves as a TRIM.FaTE input file to define additional properties, compartments, and algorithms to initialize the pseudo-source methodology. This file must be manually imported through the TRIM.FaTE graphical interface and saved as a library.
Pseudo-source link properties	Defines the actual links that connect the placeholder elements to their surface targets. This includes determining which water, soil, and/or plant compartments are present on the surface of each parcel.	This file serves as a TRIM.FaTE input file to define links between different compartments, specifically to initiate mass transfer from pseudo-source to compartments on the surface of each parcel.

Appendix 8

Dose-Response Values Used in the RTR Risk Assessments

Appendix 8. Dose-Response Values Used in the RTR Risk Assessments

The dose-response values presented in Table 1 (chronic) and Table 2 (acute) are values used in the Risk and Technology Review program as of June 2018. In some cases, a value in Table 1 or 2 may reflect an update made after a source category-specific risk assessment was conducted. The values used in this risk assessment are presented in Table 3.1-1 in the body of the report.

Definition for Chronic Values

URE (unit risk estimate) = the upper-bound excess lifetime cancer risk estimated to result from continuous exposure to an agent over a lifetime at a concentration of 1 $\mu g/m^3$ in air.
RfC (reference concentration) = an estimate of a continuous inhalation exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of harmful noncancer health effects during a lifetime.
Cancer Slope Factor = an upper-bound estimate of the increased cancer risk from a lifetime oral exposure to an agent.
RfD (reference dose) = an estimate of a continuous oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of harmful noncancer health effects during a lifetime.

Sources:
IRIS = EPA Integrated Risk Information System
ATSDR = US Agency for Toxic Substances Disease Registry
CAL = California EPA Office of Environmental Human Health Assessment
HEAST = EPA Health Effects Assessment Tables
EPA OAQPS = EPA Office of Air Quality Planning and Standards
EPA ORD = EPA Office of Research & Development

Definition of Acute Values

AEGL-1 (acute exposure guideline level 1) = the airborne concentration (expressed as ppm or mg/m3) of a substance above which it is predicted that the general population, including susceptible individuals, could experience notable discomfort, irritation, or certain asymptomatic nonsensory effects. However, the effects are not disabling and are transient and reversible upon cessation of exposure.
AEGL-2 (acute exposure guideline level 2) = the airborne concentration (expressed as ppm or mg/m3) of a substance above which it is predicted that the general population, including susceptible individuals, could experience irreversible or other serious, long-lasting adverse health effects or an impaired ability to escape.
ERPG-1 (emergency response planning guideline 1) = the maximum airborne concentration below which nearly all individuals could be exposed for up to 1 hour without experiencing more than mild, transient health effects or without perceiving a clearly defined objectionable odor.
ERPG-2 (emergency response planning guideline 2) = the maximum airborne concentration below which nearly all individuals could be exposed for up to 1 hour without experiencing or developing irreversible or other serious adverse health effects or symptoms that could impair an individual's ability to take protective action.
REL (reference exposure level) = the concentration level at or below which no adverse health effects are anticipated for a specified exposure duration. RELs are based on the most sensitive, relevant, adverse health effect reported in the medical and toxicological literature and are designed to protect the most sensitive individuals in the population by the inclusion of margins of safety.

Sources:
National Advisory Committee on Acute Exposure Guideline Level for Hazardous Substances, reviewed and published by the National Research Council - AEGL
American Industrial Hygiene Association – ERPG
California EPA Office of Environmental Human Health Assessment - REL

Table 1. Chronic Cancer and Noncancer Inhalation and Oral Dose-Response Values and the Source of Those Values

		Inhalation				Oral (ingestion)[a]	
Pollutant	CAS No.	URE 1/(ug/m^3)	URE Source	RfC (mg/m^3)	RfC Sourc	Cancer Slope Factor (1/(mg/kg/d))	RfD (mg/kg/d)
1,1,1-Trichloroethane	71-55-6			5	IRIS		
1,1,2-Trichloroethane	79-00-5	0.000016	IRIS	0.4	CAL		
Hexachlorocyclohexanes							
alpha-Hexachlorocyclohexane (a- HCH)	319-84-6	0.0018	IRIS	0.02	CAL		
beta-Hexachlorocyclohexane (b- HCH)	319-85-7	0.00053	IRIS	0.002	CAL		
Lindane (gamma-HCH)	58-89-9	0.00031	CAL	0.0003	CAL		
technical Hexachlorocyclohexane (HCH)	608-73-1	0.00051	IRIS				
1,2,4-Trichlorobenzene	120-82-1			0.2			
1,2-Dibromo-3-chloropropane	96-12-8	0.002	CAL	0.0002	IRIS		
1,2-Diphenylhydrazine	122-66-7	0.00022	IRIS				
1,2-Epoxybutane	106-88-7			0.02	IRIS		
1,3-Butadiene	106-99-0	0.00003	IRIS	0.002	IRIS		
1,3-Dichloropropene	542-75-6	0.000004	IRIS	0.02	IRIS		
1,3-Propane sultone	1120-71-4	0.00069	CAL				
p-Dichlorobenzene	106-46-7	0.000011	CAL	0.8	IRIS		
p-Dimethylaminoazobenzene	60-11-7	0.0013	CAL				
1,4-Dioxane	123-91-1	0.000005	IRIS	0.03	IRIS		
2,4,6-Trichlorophenol	88-06-2	0.0000031	IRIS				
2,4-Dinitrotoluene	121-14-2	0.000089	CAL	0.007	CAL		
2,4-Toluene diamine	95-80-7	0.0011	CAL				
2,4/2,6-Toluene diisocyanate mixture (TDI)	26471-62-5	0.000011	CAL	0.00007	IRIS		
2,4-Toluene diisocyanate	584-84-9	0.000011	CAL	0.00007	IRIS		
2-Chloroacetophenone	532-27-4			0.00003	IRIS		
2-Nitropropane	79-46-9	0.0000056	EPA OAQPS	0.02	IRIS		
3,3'-Dichlorobenzidine	91-94-1	0.00034	CAL				
4,4'-Methylene bis(2-chloroaniline)	101-14-4	0.00043	CAL				
4,4'-Methylenedianiline	101-77-9	0.00046	CAL	0.02	CAL		
Methylene diphenyl diisocyanate	101-68-8			0.0006	IRIS		
Acetaldehyde	75-07-0	0.0000022	IRIS	0.009	IRIS		
Acetamide	60-35-5	0.00002	CAL				
Acetonitrile	75-05-8			0.06	IRIS		
Acrolein	107-02-8			0.00035	CAL		
Acrylamide	79-06-1	0.00016	IRIS	0.006	IRIS		
Acrylic acid	79-10-7			0.001	IRIS		
Acrylonitrile	107-13-1	0.000068	IRIS	0.002	IRIS		
Allyl chloride	107-05-1	0.000006	CAL	0.001	IRIS		
Aniline	62-53-3	0.0000016	CAL	0.001	IRIS		
Antimony Compounds							
Antimony compounds	7440-36-0			0.0002	IRIS		
Antimony oxide	1327-33-9			0.0002	IRIS		
Antimony pentafluoride	7783-70-2			0.0002	IRIS		
Antimony pentoxide	1314-60-9			0.0002	IRIS		
Antimony potassium tartrate	304-61-0			0.0002	IRIS		
Antimony tetroxide	1332-81-6			0.0002	IRIS		
Antimony trihydride	7803-52-3			0.0002	IRIS		
Antimony trioxide	1309-64-4			0.0002	IRIS		

		Inhalation				Oral (ingestion)[a]	
Pollutant	**CAS No.**	**URE 1/(ug/m³)**	**URE Source**	**RfC (mg/m³)**	**RfC Sourc**	**Cancer Slope Factor (1/(mg/kg/d))**	**RfD (mg/kg/d)**
Arsenic Compounds							
Arsenic acid	7778-39-4	0.0043	IRIS	0.000015	CAL		
Arsenic as lead arsenate	7784-40-9	0.0043	IRIS	0.000015	CAL		
Arsenic chloride	7784-34-1	0.0043	IRIS	0.000015	CAL		
Arsenic compounds	7440-38-2	0.0043	IRIS	0.000015	CAL	1.5	
Arsenic pentoxide	1303-28-2	0.0043	IRIS	0.000015	CAL		
Arsenic trioxide	1327-53-3	0.0043	IRIS	0.000015	CAL		
Arsine	7784-42-1			0.00005	IRIS		
Benzene	71-43-2	0.0000078[d]	IRIS	0.03	IRIS		
Benzidine	92-87-5	0.1072	IRIS	0.01	CAL		
Benzyl chloride	100-44-7	0.000049	CAL				
Beryllium Compounds							
Beryllium chloride	7787-47-5	0.0024	IRIS	0.00002	IRIS		
Beryllium compounds	7440-41-7	0.0024	IRIS	0.00002	IRIS		
Beryllium fluoride	7787-49-7	0.0024	IRIS	0.00002	IRIS		
Beryllium nitrate	13597-99-4	0.0024	IRIS	0.00002	IRIS		
Beryllium oxide	1304-56-9	0.0024	IRIS	0.00002	IRIS		
Bis(2-ethylhexyl) phthalate	117-81-7	0.0000024	CAL	0.01	CAL		
Bis(chloromethyl)ether	542-88-1	0.062	IRIS				
Bromoform	75-25-2	0.0000011	IRIS				
Cadmium Compounds							
Cadmium acetate	543-90-8	0.0018	IRIS	0.00001	ATSDR		
Cadmium compounds	7440-43-9	0.0018	IRIS	0.00001	ATSDR		0.001
Cadmium as cadmium cyanamide	20654-10-8	0.0018	IRIS	0.00001	ATSDR		
Cadmium nitrate	10325-94-7	0.0018	IRIS	0.00001	ATSDR		
Cadmium oxide	1306-19-0	0.0018	IRIS	0.00001	ATSDR		
Cadmium stearate	2223-93-0	0.0018	IRIS	0.00001	ATSDR		
Carbon disulfide	75-15-0			0.7	IRIS		
Carbon tetrachloride	56-23-5	0.000006	IRIS	0.1	IRIS		
Carbonyl sulfide	463-58-1			0.163[f]	EPA ORD		
Chlordane	57-74-9	0.0001	IRIS	0.0007	IRIS		
Chlorine	7782-50-5			0.00015	ATSDR		
Chlorobenzene	108-90-7			1	CAL		
Chlorobenzilate	510-15-6	0.000078	HEAST				
Chloroform	67-66-3			0.098	ATSDR		
Chloroprene	126-99-8	0.00048	IRIS	0.02	IRIS		
Chromium Compounds							
Ammonium chromate	7788-98-9	0.012	IRIS	0.0001	IRIS		
Ammonium dichromate	7789-09-5	0.012	IRIS	0.0001	IRIS		
Barium chromate	10294-40-3	0.012	IRIS	0.0001	IRIS		
Calcium chromate	13765-19-0	0.012	IRIS	0.0001	IRIS		
Chromic acid (VI)	7738-94-5	0.012	IRIS	0.0001	IRIS		
Chromic sulfuric acid	13530-68-2	0.012	IRIS	0.0001	IRIS		
Chromium (VI) as lead chromate	7758-97-6	0.012	IRIS	0.0001	IRIS		
Chromium (VI) as lead chromate oxide	18454-12-1	0.012	IRIS	0.0001	IRIS		
Chromium (VI) compounds	18540-29-9	0.012	IRIS	0.0001	IRIS		
Chromium (VI) trioxide, chromic acid mist	11115-74-5	0.012	IRIS	0.000008	IRIS		
Chromium compounds	7440-47-3	0.012	IRIS	0.0001	IRIS		
Chromium dioxide	12018-01-8	0.012	IRIS	0.0001	IRIS		
Potassium chromate	7789-00-6	0.012	IRIS	0.0001	IRIS		
Potassium dichromate	7778-50-9	0.012	IRIS	0.0001	IRIS		
Sodium chromate	7775-11-3	0.012	IRIS	0.0001	IRIS		
Sodium dichromate	10588-01-9	0.012	IRIS	0.0001	IRIS		

		Inhalation				Oral (ingestion)[a]	
Pollutant	**CAS No.**	**URE 1/(ug/m³)**	**URE Source**	**RfC (mg/m³)**	**RfC Source**	**Cancer Slope Factor (1/(mg/kg/d))**	**RfD (mg/kg/d)**
Strontium chromate	7789-06-2	0.012	IRIS	0.0001	IRIS		
Zinc chromate	13530-65-9	0.012	IRIS	0.0001	IRIS		
Zinc potassium chromate	11103-86-9	0.012	IRIS	0.0001	IRIS		
Cobalt Compounds							
Cobalt aluminate	1345-16-0			0.0001	ATSDR		
Cobalt bromide	7789-43-7			0.0001	ATSDR		
Cobalt carbonate	513-79-1			0.0001	ATSDR		
Cobalt carbonyl	10210-68-1			0.0001	ATSDR		
Cobalt chloride	7646-79-9			0.0001	ATSDR		
Cobalt compounds	7440-48-4			0.0001	ATSDR		
Cobalt hydrocarbonyl	16842-03-8			0.0001	ATSDR		
Cobalt naphtha	61789-51-3			0.0001	ATSDR		
Cobalt nitrate	Co Nitrate			0.0001	ATSDR		
Cobalt oxide	1307-96-6			0.0001	ATSDR		
Cobalt oxide (II, III)	1308-06-1			0.0001	ATSDR		
Hexanoic acid, 2-ethyl-, cobalt (2+) salt	136-52-7			0.0001	ATSDR		
Coke Oven Emissions							
Benzene soluble organics (BSO)	141	0.00099	IRIS				
Coke oven emissions	8007-45-2	0.00099	IRIS				
Methylene chloride soluble organics (MCSO)	142	0.00099	IRIS				
Cresols							
Cresols (mixed)	1319-77-3			0.6	CAL		
m-Cresol (3-methylphenol)	108-39-4			0.6	CAL		
o-Cresol	95-48-7			0.6	CAL		
p-Cresol (4-methy phenol)	106-44-5			0.6	CAL		
Cumene	98-82-8			0.4	IRIS		
Cyanide Compounds							
Acetone cyanohydrin	75-86-5			0.01	HEAST		
Barium cyanide	542-62-1			0.0008	IRIS		
Calcium cyanamide	156-62-7			0.0008	IRIS		
Calcium cyanide	592-01-8			0.0008	IRIS		
Copper cyanide	544-92-3			0.0008	IRIS		
Cyanazine	21725-46-2			0.0008	IRIS		
Cyanide as Cadmium cyanamide	20654-10-8			0.0008	IRIS		
Cyanide compounds	57-12-5			0.0008	IRIS		
Cyanogen	460-19-5			0.0008	IRIS		
Cyanogen bromide	506-68-3			0.0008	IRIS		
Cyanogen chloride	506-77-4			0.0008	IRIS		
Cyanogen iodide	506-78-5			0.0008	IRIS		
Cyanophos	2636-26-2			0.0008	IRIS		
Cyanuric fluoride	675-14-9			0.0008	IRIS		
Ethylene cyanohydrin	109-78-4			0.0008	IRIS		
Hydrogen cyanide	74-90-8			0.0008	IRIS		
Isopropyl cyanide	78-82-0			0.0008	IRIS		
Potassium cyanide	151-50-8			0.0008	IRIS		
Potassium silver cyanide	506-61-6			0.0008	IRIS		
Potassium thiocyanate	333-20-0			0.0008	IRIS		
Silver cyanide	506-64-9			0.0008	IRIS		
Sodium cyanide	143-33-9			0.0008	IRIS		
Thiocyanate	Thiocyanate			0.0008	IRIS		
Thiocyanic acid	21564-17-0			0.0008	IRIS		
Zinc cyanide	557-21-1			0.0008	IRIS		
Dichloroethyl ether	111-44-4	0.00033	IRIS				
Dichlorvos	62-73-7			0.0005	IRIS		
Diesel engine emissions	Diesel emis			0.005	IRIS		

		Inhalation				Oral (ingestion)[a]	
Pollutant	CAS No.	URE 1/(ug/m^3)	URE Source	RfC (mg/m^3)	RfC Source	Cancer Slope Factor (1/(mg/kg/d))	RfD (mg/kg/d)
Diethanolamine	111-42-2			0.003	CAL		
Dimethyl formamide	68-12-2			0.03	CAL		
Dioxins and Furans							
1,2,3,4,6,7,8,9-Octachlorodibenzo-p-dioxin	3268-87-9	0.0099	EPA ORD	0.00013	CAL	45	
1,2,3,4,6,7,8,9-Octachlorodibenzofuran	39001-02-0	0.0099	EPA ORD	0.00013	CAL	45	
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	35822-46-9	0.33	EPA ORD	0.000004	CAL	1500	
1,2,3,4,6,7,8-Heptachlorodibenzofuran	67562-39-4	0.33	EPA ORD	0.000004	CAL	1500	
1,2,3,4,7,8,9-Heptachlorodibenzofuran	55673-89-7	0.33	EPA ORD	0.000004	CAL	1500	
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	39227-28-6	3.3	EPA ORD	0.0000004	CAL	15000	
1,2,3,4,7,8-Hexachlorodibenzofuran	70648-26-9	3.3	EPA ORD	0.0000004	CAL	15000	
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	57653-85-7	3.3	EPA ORD	0.0000004	CAL	6200	
1,2,3,6,7,8-Hexachlorodibenzofuran	57117-44-9	3.3	EPA ORD	0.0000004	CAL	15000	
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin	19408-74-3	3.3	EPA ORD	0.0000004	CAL	6200	
1,2,3,7,8,9-Hexachlorodibenzofuran	72918-21-9	3.3	EPA ORD	0.0000004	CAL	15000	
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	40321-76-4	33	EPA ORD	0.00000004	CAL	150000	
1,2,3,7,8-Pentachlorodibenzofuran	57117-41-6	0.99	EPA ORD	0.0000013	CAL	4500	
2,3,4,6,7,8-Hexachlorodibenzofuran	60851-34-5	3.3	EPA ORD	0.0000004	CAL	15000	
2,3,4,7,8-Pentachlorodibenzofuran	57117-31-4	9.9	EPA ORD	0.00000013	CAL	45000	
2,3,7,8-Tetrachlorodibenzo-p-dioxin	1746-01-6	33	EPA ORD	0.00000004	CAL	150000	
2,3,7,8-Tetrachlorodibenzofuran	51207-31-9	3.3	EPA ORD	0.0000004	CAL	15000	
Hexachlorodibenzo-p-dioxin	34465-46-8	3.3	EPA ORD	0.0000004	CAL	15000	
Epichlorohydrin	106-89-8	0.0000012	IRIS	0.001	IRIS		
Ethyl benzene	100-41-4	0.0000025	CAL	1	IRIS		
Ethyl carbamate	51-79-6	0.000464	CAL				
Ethyl chloride	75-00-3			10	IRIS		
Ethylene dibromide	106-93-4	0.0006	IRIS	0.009	IRIS		
Ethylene dichloride	107-06-2	0.000026	IRIS	2.4	ATSDR		
Ethylene glycol	107-21-1			0.4	CAL		
Ethylene oxide	75-21-8	0.005	IRIS	0.03	CAL		
Ethylene thiourea	96-45-7	0.000013	CAL	0.003	CAL		
Ethylidene dichloride	75-34-3	0.0000016	CAL	0.5	HEAST		
Formaldehyde	50-00-0	0.000013	IRIS	0.0098	ATSDR		
Glycol ethers							
1,2-Dimethoxyethane	110-71-4			0.02	IRIS		
2-Butoxyethyl acetate	112-07-2			0.02	IRIS		
2-(Hexyloxy)ethanol	112-25-4			0.02	IRIS		
2-Propoxyethyl acetate	20706-25-6			0.02	IRIS		
Butyl carbitol acetate	124-17-4			0.02	IRIS		
Carbitol acetate	112-15-2			0.02	IRIS		
Diethylene glycol diethyl ether	112-36-7			0.02	IRIS		
Diethylene glycol dimethyl ether	111-96-6			0.02	IRIS		

		Inhalation				Oral (ingestion)[a]	
Pollutant	CAS No.	URE 1/(ug/m^3)	URE Source	RfC (mg/m^3)	RfC Source	Cancer Slope Factor (1/(mg/kg/d))	RfD (mg/kg/d)
Diethylene glycol ethyl methyl ether	1002-67-1			0.02	IRIS		
Diethylene glycol monobutyl ether	112-34-5			0.02	HEAST		
Diethylene glycol monoethyl ether	111-90-0			0.02	IRIS		
Diethylene glycol monomethyl ether	111-77-3			0.02	IRIS		
Ethoxytriglycol	112-50-5			0.02	IRIS		
Ethylene glycol diethyl ether	629-14-1			0.02	IRIS		
Ethylene glycol ethyl ether	110-80-5			0.2	IRIS		
Ethylene glycol ethyl ether acetate	111-15-9			0.3	CAL		
Ethylene glycol methyl ether	109-86-4			0.02	IRIS		
Ethylene glycol methyl ether acetate	110-49-6			0.09	CAL		
Ethylene glycol mono-sec-butyl ether	7795-91-7			0.02	IRIS		
Glycol ethers	171			0.02	IRIS		
Methoxytriglycol	112-35-6			0.02	IRIS		
Methyl Cellosolve Acrylate	3121-61-7			0.02	IRIS		
N-Hexyl carbitol	112-59-4			0.02	IRIS		
Phenyl cellosolve	122-99-6			0.02	IRIS		
Propyl cellosolve	2807-30-9			0.02	IRIS		
Triethylene glycol dimethyl ether	112-49-2			0.02	IRIS		
Triethylene glycol monohexyl ether	25961-89-1			0.02	IRIS		
Triglycol monobutyl ether	143-22-6			0.02	IRIS		
Heptachlor	76-44-8	0.0013	IRIS				
Hexachlorobenzene	118-74-1	0.00046	IRIS	0.003	CAL		
Hexachlorobutadiene	87-68-3	0.000022	IRIS	0.09	CAL		
Hexachlorocyclopentadiene	77-47-4			0.0002	IRIS		
Hexachloroethane	67-72-1			0.03	IRIS		
Hexamethylene-1,6-diisocyanate	822-06-0			0.00001	IRIS		
n-Hexane	110-54-3			0.7	IRIS		
Hydrazine	302-01-2	0.0049	IRIS	0.0002	CAL		
Hydrochloric acid	7647-01-0			0.02	IRIS		
Hydrofluoric acid	7664-39-3			0.014	CAL		
Isophorone	78-59-1			2	CAL		
Lead Compounds[e]							
Lead (II) oxide	1317-36-8			0.00015	EPA OAQPS		
Lead acetate	301-04-2			0.00015	EPA OAQPS		
Lead as lead arsenate	7784-40-9			0.00015	EPA OAQPS		
Lead as lead chromate	7758-97-6			0.00015	EPA OAQPS		
Lead as lead chromate oxide	18454-12-1			0.00015	EPA OAQPS		
Lead chloride	7758-95-4			0.00015	EPA OAQPS		
Lead compounds	7439-92-1			0.00015	EPA OAQPS		
Lead compounds (other than inorganic)	603			0.00015	EPA OAQPS		
Lead dioxide	1309-60-0			0.00015	EPA OAQPS		
Lead nitrate	10099-74-8			0.00015	EPA OAQPS		

		Inhalation				Oral (ingestion)[a]	
Pollutant	**CAS No.**	**URE 1/(ug/m³)**	**URE Source**	**RfC (mg/m³)**	**RfC Source**	**Cancer Slope Factor (1/(mg/kg/d))**	**RfD (mg/kg/d)**
Lead subacetate	1335-32-6			0.00015	EPA OAQPS		
Lead sulfate	7446-14-2			0.00015	EPA OAQPS		
Tetraethyl lead	78-00-2			0.00015	EPA OAQPS		
Tetramethyl lead	75-74-1			0.00015	EPA OAQPS		
Maleic anhydride	108-31-6			0.0007	CAL		
Manganese Compounds							
Manganese chloride	2145-07-6			0.0003	ATSDR		
Manganese compounds	7439-96-5			0.0003	ATSDR		
Manganese dioxide	1313-13-9			0.0003	ATSDR		
Manganese nitrate	10377-66-9			0.0003	ATSDR		
Manganese oxide	1317-35-7			0.0003	ATSDR		
Manganese sulfate	7785-87-7			0.0003	ATSDR		
Manganese tetroxide	1317-35-7			0.0003	ATSDR		
Manganese tricarbonyl (eta.5-2,4-cyclopentadien-1-yl)-	12079-65-1			0.0003	ATSDR		
Manganese trioxide	1317-34-6			0.0003	ATSDR		
Mercury compounds							
Gaseous divalent mercury	201			0.0003	IRIS		0.0001
Mercuric acetate	1600-27-7			0.0003	IRIS		0.0001
Mercuric chloride	7487-94-7			0.0003	IRIS		0.0001
Mercuric nitrate	10045-64-0			0.0003	IRIS		0.0001
Mercuric oxide	21908-53-2			0.0003	IRIS		0.0001
Mercury (elemental)	7439-97-6			0.0003	IRIS		c
Mercury (organic)	22967-92-6			0.0003	IRIS		c
Mercury compounds	HGCMPDS			0.0003	IRIS		c*
Methoxyethylmercuric acetate	151-38-2			0.0003	IRIS		0.0001
Methyl mercury	22967-92-6			0.0003	IRIS		c
Methylmercuric dicyanamide	502-39-6			0.0003	IRIS		0.0001
Particulate divalent mercury	202			0.0003	IRIS		0.0001
Phenylmercuric acetate	62-38-4			0.0003	IRIS		0.0001
Methanol	67-56-1			20	IRIS		
Methyl bromide	74-83-9			0.005	IRIS		
Methyl chloride	74-87-3			0.09	IRIS		
Methyl isobutyl ketone	108-10-1			3	IRIS		
Methyl isocyanate	624-83-9			0.001	CAL		
Methyl methacrylate	80-62-6			0.7	IRIS		
Methyl tert-butyl ether	1634-04-4	0.00000026	CAL	3	IRIS		
Methylene chloride	75-09-2	0.000000016	IRIS	0.6	IRIS		
Naphthalene	91-20-3	0.000034	CAL	0.003	IRIS		
Nickel compounds							
Nickel (II) sulfate hexahydrate	10101-97-0	0.00048	EPA OAQPS	0.00009	ATSDR		
Nickel acetate	373-02-4	0.00048	EPA OAQPS	0.00009	ATSDR		
Nickel carbonyl	13463-39-3	0.00048	EPA OAQPS	0.00009	ATSDR		
Nickel chloride	7718-54-9	0.00048	EPA OAQPS	0.00009	ATSDR		
Nickel compounds	7440-02-0	0.00048	EPA OAQPS	0.00009	ATSDR		
Nickel nitrate	13138-45-9	0.00048	EPA OAQPS	0.00009	ATSDR		
Nickel oxide	1313-99-1	0.00048	EPA OAQPS	0.00002	CAL		

		Inhalation				Oral (ingestion)[a]	
Pollutant	**CAS No.**	**URE 1/(ug/m³)**	**URE Source**	**RfC (mg/m³)**	**RfC Source**	**Cancer Slope Factor (1/(mg/kg/d))**	**RfD (mg/kg/d)**
Nickel refinery dust	NI_DUST	0.00024	IRIS				
Nickel subsulfide	12035-72-2	0.00048	IRIS	0.00009	ATSDR		
Nickel sulfamate	13770-89-3	0.00048	EPA OAQPS	0.00009	ATSDR		
Nickel sulfate	7786-81-4	0.00048	EPA OAQPS	0.00009	ATSDR		
Nitrobenzene	98-95-3	0.00004	IRIS	0.009	IRIS		
Nitrosodimethylamine	62-75-9	0.022	IRIS				
N-Nitrosomorpholine	59-89-2	0.0019	CAL				
o-Toluidine	95-53-4	0.000051	CAL				
Pentachlorophenol	87-86-5	0.0000051	CAL	0.1	CAL		
Phenol	108-95-2			0.2	CAL		
Phosgene	75-44-5			0.0003	IRIS		
Phosphine	7803-51-2			0.0003	IRIS		
Phthalic anhydride	85-44-9			0.02	CAL		
Polychlorinated Biphenyls							
Aroclor 1016	12674-11-2	0.0001	IRIS				
Aroclor 1221	11104-28-2	0.0001	IRIS				
Aroclor 1242	53469-21-9	0.0001	IRIS				
Aroclor 1248	12672-29-6	0.0001	IRIS				
Aroclor 1254	11097-69-1	0.0001	IRIS				
Aroclor 1260	11096-82-5	0.0001	IRIS				
Polychlorinated biphenyls	1336-36-3	0.0001	IRIS				
2,4,4'-Trichlorobiphenyl (PCB-28)	7012-37-5	0.0001	IRIS				
4,4'-Dichlorobiphenyl (PCB-15)	2050-68-2	0.0001	IRIS				
Decachlorobiphenyl (PCB-209)	2051-60-7	0.0001	IRIS				
Heptachlorobiphenyl	28655-71-2	0.0001	IRIS				
Hexachlorobiphenyl	26601-64-9	0.0001	IRIS				
Pentachlorobiphenyl	25429-29-2	0.0001	IRIS				
Tetrachlorobiphenyl	26914-33-0	0.0001	IRIS				
Polycyclic Organic Matter							
POM 71002		0.000048	CAL				
16-PAH	40	0.000048	CAL				
PAH, total	234	0.000048	CAL			0.05	
Polycyclic organic matter	246	0.000048	CAL			0.05	
POM 72002		0.000048	CAL				
1-Methylnaphthalene	90-12-0	0.000048	CAL			0.05	
1-Methylphenanthrene	832-69-9	0.000048	CAL				
2-Methylphenanthrene	2531-84-2	0.000048	CAL				
1-Methylpyrene	2381-21-7	0.000048	CAL				
2-Methylnaphthalene	91-57-6	0.000048	CAL			0.05	
2-Naphthylamine	91-59-8	0.000048	CAL				
12-Methylbenz(a)anthracene	2422-79-9	0.000048	CAL				
beta-Chloronaphthalene	91-58-7	0.000048	CAL			0.05	
Acenaphthene	83-32-9	0.000048	CAL			0.05	
Acenaphthylene	208-96-8	0.000048	CAL			0.05	
Anthracene	120-12-7	b	IRIS				
Benzo(a)fluoranthene	203-33-8	0.000048	CAL			0.05	
Benzo(g,h,i)fluoranthene	203-12-3	0.000048	CAL			0.05	
Benzo(c)phenanthrene	195-19-7	0.000048	CAL			0.05	
Benzo(e)pyrene	192-97-2	0.000048	CAL			0.05	
Benzofluoranthenes	56832-73-6	0.000048	CAL			0.05	
Benzo(ghi)perylene	191-24-2	0.000048	CAL			0.05	
Coal tar	8007-45-2	0.00099	CAL				
Coronene	191-07-1	0.000048	CAL				
Extractable organic matter (EOM)	284	0.000048	CAL				
Fluoranthene	206-44-0	0.000048	CAL			0.05	
Fluorene	86-73-7	0.000048	CAL			0.05	

		Inhalation				Oral (ingestion)[a]	
Pollutant	**CAS No.**	**URE 1/(ug/m^3)**	**URE Source**	**RfC (mg/m^3)**	**RfC Source**	**Cancer Slope Factor (1/(mg/kg/d))**	**RfD (mg/kg/d)**
Indene	95-13-6	0.000048	CAL				
Methylanthracene	26914-18-1	0.000048	CAL				
9-Methylanthracene	779-02-2	0.000048	CAL				
Methylbenzopyrene	65357-69-9	0.000048	CAL				
Octabromodiphenyl ether	32536-52-0	0.000048	CAL				
Perylene	198-55-0	0.000048	CAL			0.05	
Phenanthrene	85-01-8	b	IRIS				
Pyrene	129-00-0	b	IRIS				
POM 73002		0.096	CAL				
7,12-Dimethylbenz[a]anthracene	57-97-6	0.1136	CAL			250	
POM 74002		0.0096	CAL				
1,6-Dinitropyrene	42397-64-8	0.0096	CAL				
3-Methylcholanthrene	56-49-5	0.01008	CAL			22	
6-Nitrochrysene	7496-02-8	0.0096	CAL				
Dibenzo[a,h]pyrene	189-64-0	0.0096	CAL				
Dibenzo[a,i]pyrene	189-55-9	0.0096	CAL			10	
Dibenzo[a,l]pyrene	191-30-0	0.0096	CAL				
POM 75002		0.00096	CAL				
1,8-Dinitropyrene	42397-65-9	0.00096	CAL				
2-Acetylaminofluorene	53-96-3	0.00208	CAL			1	
Methylchrysene	41637-90-5	0.00096	CAL				
5-Methylchrysene	3697-24-3	0.00096	CAL				
7H-Dibenzo[c,g]carbazole	194-59-2	0.00096	CAL				
Benzo[a]pyrene	50-32-8	0.00096	EPA			1	
Dibenzo[a,e]pyrene	192-65-4	0.00096	CAL				
Dibenzo[a,h]anthracene	53-70-3	0.00096	EPA			1	
Polycyclic aromatic hydrocarbon as B(a)P TEQ		0.00096	CAL				
POM 76002		0.000096	CAL				
1-Nitropyrene	5522-43-0	0.000096	CAL				
4-Nitropyrene	57835-92-4	0.000096	CAL				
5-Nitroacenaphthene	602-87-9	0.0000592	CAL				
Benz[a]anthracene	56-55-3	0.000096	EPA			0.1	
Benzo[b]fluoranthene	205-99-2	0.000096	EPA			0.1	
Benzo[b+k]fluoranthene	102	0.000096	CAL			0.1	
Benzo[j]fluoranthene	205-82-3	0.000096	CAL			0.1	
Dibenz[a,h]acridine	226-36-8	0.000096	CAL				
Dibenz[a,j]acridine	224-42-0	0.000096	CAL			0.1	
Indeno[1,2,3-c,d]pyrene	193-39-5	0.000096	EPA			0.1	
POM 77002		0.0000096	CAL				
Benzo[k]fluoranthene	207-08-9	0.0000096	EPA			0.01	
2-Aminoanthraquinone	117-79-3	0.000015	CAL				
2-Nitrofluorene	607-57-8	0.0000096	CAL				
Carbazole	86-74-8	0.0000096	CAL			0.02	
Chrysene	218-01-9	0.00000096	EPA			0.001	
POM 78002		0.000176	CAL				
7-PAH	75	0.000176	CAL			0.05	
Propionaldehyde	123-38-6			0.008	IRIS		
Propylene dichloride	78-87-5			0.004	IRIS		
Propylene oxide	75-56-9	0.0000037	IRIS	0.03	IRIS		
Radionuclides							
Uranium, insoluble	7440-61-1			0.0008	ATSDR		
Uranium (IV) dioxide	1344-57-6			0.0008	ATSDR		
Uranium compounds	7440-61-1			0.0008	ATSDR		
Uranium hexafluoride	7783-81-5			0.00004	ATSDR		
Uranium oxide	1344-59-8			0.0008	ATSDR		
Uranium, soluble	UranSol			0.00004	ATSDR		

		Inhalation				Oral (ingestion)[a]	
Pollutant	**CAS No.**	**URE 1/(ug/m^3)**	**URE Source**	**RfC (mg/m^3)**	**RfC Source**	**Cancer Slope Factor (1/(mg/kg/d))**	**RfD (mg/kg/d)**
Uranyl acetate dihydrate	541-09-3			0.00004	ATSDR		
Uranyl nitrate hexahydrate	13520-83-7			0.00004	ATSDR		
Selenium Compounds							
Hydrogen selenide	7783-07-5			0.00008	CAL		
Potassium selenate	7790-59-2			0.02	CAL		
Selenious acid	7783-00-8			0.02	CAL		
Selenium compounds	7782-49-2			0.02	CAL		
Selenium dioxide	7446-08-4			0.02	CAL		
Selenium disulfide	7488-56-4			0.02	CAL		
Selenium hexafluoride	7783-79-1			0.02	CAL		
Selenium oxide	12640-89-0			0.02	CAL		
Selenium oxychloride	7791-23-3			0.02	CAL		
Selenium sulfide	7446-34-6			0.02	CAL		
Selenourea	630-10-4			0.02	CAL		
Sodium selenate	13410-01-0			0.02	CAL		
Sodium selenite	10102-18-8			0.02	CAL		
Styrene	100-42-5			1	IRIS		
Styrene oxide	96-09-3			0.006	CAL		
Tetrachloroethene	127-18-4	0.00000026	IRIS	0.04	IRIS		
Titanium tetrachloride	7550-45-0			0.0001	ATSDR		
Toluene	108-88-3			5	IRIS		
Toxaphene	8001-35-2	0.00032	IRIS				
Trichloroethylene	79-01-6	0.0000048	IRIS	0.002	IRIS		
Triethylamine	121-44-8			0.007	IRIS		
Vinyl acetate	108-05-4			0.2	IRIS		
Vinyl bromide	593-60-2	0.000032	HEAST	0.003	IRIS		
Vinyl chloride	75-01-4	0.0000088	IRIS	0.1	IRIS		
Vinylidene chloride	75-35-4			0.2	IRIS		
Xylenes							
m-Xylene	108-38-3			0.1	IRIS		
o-Xylene	95-47-6			0.1	IRIS		
p-Xylene	106-42-3			0.1	IRIS		
Xylenes (mixed)	1330-20-7			0.1	IRIS		

Notes:

[a] Benchmark values are provided only for those PB-HAPs for which multipathway risk is assessed (via TRIM). There may be other PB-HAPs in this table, even though no benchmark is presented.

[b] IRIS has determined this POM to be not carcinogenic.

[c] The predominant form of mercury assessed in our multipathway risk screening is methyl mercury, which is a transformation product of divalent mercury and accumulates in fish. While elemental mercury emissions can convert to divalent mercury in the atmosphere, such transformations generally occur beyond the 50 km modeling domain around the emissions sources in our assessment. *Emissions reported as "mercury compounds" is speciated into elemental, particulate divalent, and gaseous divalent and modeled accordingly in the multipathway screening assessment.

[d] The EPA IRIS assessment for benzene provides a range of plausible UREs. This assessment used the highest value in that range, 7.8E-06 µg/m3. The low end of the range is 2.2E-06 µg/m3.

[e] There is no reference concentration for lead. In considering noncancer hazards for lead in this assessment, we compared rolling three-month average exposure estimates to the National Ambient Air Quality Standard (NAAQS) for lead (0.15 µg/m3). The primary (health-based) standard is a maximum or not-to-be-exceeded, rolling three-month average, measured as total suspended particles (TSP). The secondary (welfare-based) standard is identical to the primary standard.

[f] A chronic screening level of 0.163 mg/m3 was developed for carbonyl sulfide by EPA ORD from a No Observed Adverse Effects Level of 200 ppm based on brain lesions and neurophysiological alterations in rodents.

Table 2. Acute Dose-Response Values

Pollutant	CAS No.	AEGL-1 (1-hr) (mg/m^3)	AEGL-2 (1-hr) (mg/m^3)	ERPG-1 (mg/m^3)	ERPG-2 (mg/m^3)	MRL (mg/m^3)	REL (1-hr) (mg/m^3)
1,1,1-Trichloroethane	71-55-6	1300	3300	1900	3800	11	68
1,1-Dimethylhydrazine	57-14-7		7.4				
1,2-Epoxybutane	106-88-7	210	410				
1,2-Propyleneimine	75-55-8		28				
1,3-Butadiene	106-99-0	1500	12000	22	1100		[a]
1,4-Dichlorobenzene	106-46-7					12	
1,4-Dioxane	123-91-1	61	1200			7.2	3
2,4/2,6-Toluene diisocyanate mixture	26471-62-5	0.14	0.59	0.071	1.1	0.000071	0.002
2,4-Toluene diisocyanate	584-84-9	0.14	0.59	0.071	1.1	0.000071	0.002
Methylene diphenyl diisocyanate	101-68-8				5		0.012
Acetaldehyde	75-07-0	81	490	18	360		0.47
Acetonitrile	75-05-8	22	84				
Acrolein	107-02-8	0.069	0.23	0.11	0.34	0.0069	0.0025
Acrylic acid	79-10-7	4.4	140	2.9	150		6
Acrylonitrile	107-13-1		3.7	22	76	0.22	
Allyl chloride	107-05-1	8.8	170	9.4	130		
Aniline	62-53-3	30	46				
Antimony Compounds							
Antimony compounds	7440-36-0					0.001	
Antimony oxide	1327-33-9					0.001	
Antimony pentafluoride	7783-70-2					0.001	
Antimony pentoxide	1314-60-9					0.001	
Antimony potassium tartrate	304-61-0					0.001	
Antimony tetroxide	1332-81-6					0.001	
Antimony trihydride	7803-52-3		7.7		2.6	0.001	
Antimony trioxide	1309-64-4					0.001	
Arsenic Compounds							
Arsenic acid	7778-39-4						0.0002
Arsenic as lead arsenate	7784-40-9						0.0002
Arsenic chloride	7784-34-1						0.0002
Arsenic compounds	7440-38-2						0.0002
Arsenic pentoxide	1303-28-2						0.0002
Arsenic trioxide	1327-53-3		3.0				0.0002
Arsine	7784-42-1		0.54		1.6		0.0002
Benzene	71-43-2	170	2600	160	480	0.029	[b]
Benzyl chloride	100-44-7			5.2	52		0.24
Beryllium Compounds							
Beryllium compounds	7440-41-7				0.025		
Biphenyl	92-52-4		61				
Bis(chloromethyl)ether	542-88-1		0.21		0.47		
Cadmium compounds	7440-43-9	0.1	0.76			0.00003	
Carbon disulfide	75-15-0	40	500	3.1	160		6.2
Carbon tetrachloride	56-23-5		82	130	630		1.9
Carbonyl sulfide	463-58-1		140				
Chlorine	7782-50-5	1.5	5.8	2.9	8.7	0.17	0.21
Chloroacetic acid	79-11-8		26				
Chlorobenzene	108-90-7	46	690				
Chloroform	67-66-3		310		240	0.49	0.15
Chloromethyl methyl ether	107-30-2		1.6		3.3		
Cobalt Compounds							
Cobalt hydrocarbonyl	16842-03-8				0.9		
Cumene	98-82-8	250	1500				

Pollutant	CAS No.	AEGL-1 (1-hr) (mg/m^3)	AEGL-2 (1-hr) (mg/m^3)	ERPG-1 (mg/m^3)	ERPG-2 (mg/m^3)	MRL (mg/m^3)	REL (1-hr) (mg/m^3)
Cyanide Compounds							
Acetone cyanohydrin	75-86-5	7	25				
Calcium cyanide	592-01-8	3.8	13				
Cyanogen	460-19-5	4.3	18				
Cyanogen chloride	506-77-4				0.13		
Hydrogen cyanide	74-90-8	2.2	7.8		11		0.34
Isopropyl cyanide	78-82-0		5.7		85		
Potassium cyanide	151-50-8	5.3	19				
Sodium cyanide	143-33-9	4.0	14				
Dichlorvos	62-73-7	0.99	5.1			0.018	
Dimethyl formamide	68-12-2		270	6	300		
Dimethyl sulfate	77-78-1	0.12	0.62				
Epichlorohydrin	106-89-8	6.4	91	19	76		1.3
Ethyl acrylate	140-88-5	34	150	0.041	120		
Ethyl benzene	100-41-4	140	4800			22	
Ethyl chloride	75-00-3					40	
Ethylene dibromide	106-93-4	130	180				
Ethylene dichloride	107-06-2			200	810		
Ethylene glycol	107-21-1					2	
Ethylene imine (aziridine)	151-56-4		8.1				
Ethylene oxide	75-21-8		81		90		
Formaldehyde	50-00-0	1.1	17	1.2	12	0.049	0.055
Glycol ethers							
1,2-Dimethoxyethane	110-71-4						0.093
2-Butoxyethyl acetate	112-07-2						0.093
2-(Hexyloxy)ethanol	112-25-4						0.093
2-Propoxyoethyl acetate	20706-25-6						0.093
Butyl carbitol acetate	124-17-4						0.093
Carbitol acetate	112-15-2						0.093
Diethylene glycol diethyl ether	112-36-7						0.093
Diethylene glycol dimethyl ether	111-96-6						0.093
Diethylene glycol ethyl methyl ether	1002-67-1						0.093
Diethylene glycol monobutyl ether	112-34-5						0.093
Diethylene glycol monoethyl ether	111-90-0						0.093
Diethylene glycol monomethyl ether	111-77-3						0.093
Ethoxytriglycol	112-50-5						0.093
Ethylene glycol diethyl ether	629-14-1						0.093
Ethylene glycol ethyl ether	110-80-5						0.37
Ethylene glycol ethyl ether acetate	111-15-9						0.14
Ethylene glycol methyl ether	109-86-4						0.093
Ethylene glycol methyl ether acetate	110-49-6						0.093
Ethylene glycol mono-sec-butyl	7795-91-7						0.093
Glycol ethers	171						0.093
Methoxytriglycol	112-35-6						0.093
Methyl cellosolve acrylate	3121-61-7						0.093
N-Hexyl carbitol	112-59-4						0.093
Phenyl cellosolve	122-99-6						0.093
Propyl cellosolve	2807-30-9						0.093
Triethylene glycol dimethyl ether	112-49-2						0.093
Triethylene glycol monohexyl ether	25961-89-1						0.093
Triglycol monobutyl ether	143-22-6						0.093
Hexachlorobutadiene	87-68-3			11	32		
Hexachloroethane	67-72-1					58	
n-Hexane	110-54-3		10000				
Hydrazine	302-01-2	0.13	17	0.66	6.6		
Hydrochloric acid	7647-01-0	2.7	33	4.5	30		2.1

Pollutant	CAS No.	AEGL-1 (1-hr) (mg/m^3)	AEGL-2 (1-hr) (mg/m^3)	ERPG-1 (mg/m^3)	ERPG-2 (mg/m^3)	MRL (mg/m^3)	REL (1-hr) (mg/m^3)
Hydrofluoric acid	7664-39-3	0.82	20	1.6	16	0.016	0.24
Maleic anhydride	108-31-6			0.8	20		
Mercury compounds							
Mercury (elemental)	7439-97-6		1.7		2		0.0006
Methanol	67-56-1	690	2700	260	1300		28
Methyl bromide	74-83-9		820		190	0.19	3.9
Methyl chloride	74-87-3		1900	310	2100	1	
Methyl hydrazine	60-34-4		1.7				
Methyl iodide	74-88-4	130	480	150	290		
Methyl isocyanate	624-83-9		0.16	0.058	0.58		
Methyl methacrylate	80-62-6	70	490				
Methyl tert-butyl ether	1634-04-4	180	2100	180	3600	7.2	
Methylene chloride	75-09-2	690	1900	1000	2600	2.1	14
Nickel compounds							[c]
Nickel carbonyl	13463-39-3		0.25				
Parathion	56-38-2		1.5				
Phenol	108-95-2	58	89	38	190		5.8
Phosgene	75-44-5		1.2		2.0		0.004
Phosphine	7803-51-2		2.8		0.7		
Phosphorus	7723-14-0	3.7	11				
Propionaldehyde	123-38-6	110	620				
Propylene dichloride	78-87-5					0.23	
Propylene oxide	75-56-9	170	690	120	590		3.1
Radionuclides							
Uranium (IV) dioxide	1344-57-6				10		
Uranium hexafluoride	7783-81-5	3.6	9.6	5	15		
Uranium oxide	1344-59-8				10		
Selenium Compounds							
Hydrogen selenide	7783-07-5		0.36		0.66		0.005
Selenium hexafluoride	7783-79-1	0.42	0.69				
Styrene	100-42-5	85	550	210	1100	21	21
Tetrachloroethene	127-18-4	240	1600	680	1400	0.041	20
Titanium tetrachloride	7550-45-0		7.8	5	20		
Toluene	108-88-3	250	2100	190	1100	7.5	37
Trichloroethylene	79-01-6	700	2400	540	2700		
Triethylamine	121-44-8						2.8
Vinyl acetate	108-05-4	24	130	18	260		
Vinyl chloride	75-01-4	640	3100	1300	13000	1.3	180
Vinylidene chloride	75-35-4				2000		
Xylenes							
m-Xylene	108-38-3						22
o-Xylene	95-47-6						22
p-Xylene	106-42-3						22
xylenes (mixed)	1330-20-7	560	4000			8.7	22

06/2018

Notes:

[a] Based on examination of California EPA's acute (1-hour) REL for 1,3-butadiene, and taking into account aspects of the methodology used in the derivation of the value, we have decided not to use this value to support EPA's risk and technology review rules.

[b] Based on examination of California EPA's acute (1-hour) REL for benzene, and taking into account aspects of the methodology used in the derivation of the value and how this assessment stands in comparison to the ATSDR toxicological assessment, we have decided not to use this value to support

EPA's risk and technology review rules.

[c] Based on an in-depth examination of the available acute value for nickel [California EPA's acute (1-hour) REL], we have concluded that this value is not appropriate to use to support EPA's risk and technology review rules. This conclusion takes into account: the effect on which the acute REL is based; aspects of the methodology used in its derivation; and how this assessment stands in comparison to the ATSDR toxicological assessment, which considered the broader nickel health effects database. (79 FR 60247-8; October 6, 2014)

Appendix 9

Environmental Risk Screening Assessment

Technical Support Document for the Environmental Risk Screen for RTR

July 2017

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U.S. Environmental Protection Agency
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CONTENTS

FIGURES iv

TABLES iv

ACRONYMS/ABBREVIATIONS v

1 Overview 1

2 Key Components of the Environmental Risk Screen 2
- 2.1 Environmental HAPs 2
- 2.2 Assessment Endpoints 7
- 2.3 Environmental Risk Screening Approach 9
  - 2.3.1 PB-HAPs 10
  - 2.3.2 Lead 12
  - 2.3.3 Acid Gases 13

3 Effects Assessment 14
- 3.1 Benchmarks for PB-HAPs 17
  - 3.1.1 Types of Benchmarks for PB-HAPs 18
  - 3.1.2 Preferred Sources for PB-HAP Benchmark Values 21
  - 3.1.3 Selected PB-HAP Benchmarks 25
- 3.2 Benchmarks for Acid Gases 26
  - 3.2.1 Hydrogen Chloride 26
  - 3.2.2 Hydrogen Fluoride 28

4 Exposure Assessment 31
- 4.1 Environmental Risk Screen for PB-HAPs 32
  - 4.1.1 Tier 1 Exposure Assessment 34
  - 4.1.2 Tier 2 Environmental Risk Screen 38
  - 4.1.3 Tier 3 Exposure Assessment 40
- 4.2 Environmental Risk Screen for Lead and Acid Gases 43
  - 4.2.1 Lead 43
  - 4.2.2 Acid Gases 43

5 Environmental Risk Characterization/Screening Results 45
- 5.1 Environmental Risk Screen Metrics for PB-HAPs 45
  - 5.1.1 Tier 1 45
  - 5.1.2 Tiers 2 and 3 46
- 5.2 Environmental Risk Screen Metrics for Acid Gases 48
- 5.3 Environmental Risk Screen Metrics for Lead 49

6 References 49

FIGURES

Figure 2-1. Overview of the Environmental Risk Screen for PB-HAPs 11
Figure 2-2. Overview of the Environmental Risk Screen for Acid Gases 14
Figure 4-1. Approach for Tier 1 Environmental Risk Screen for PB-HAPs 34
Figure 4-2. TRIM.FaTE Lake-centric (Top) and Farm-centric (Bottom) Surface Layouts for the Tier 1 Screen 36
Figure 4-3. Approach for Tier 2 Environmental Risk Screen for PB-HAPs 39
Figure 4-4. Approach for Tier 3 Environmental Risk Screen for PB-HAPs 42

TABLES

Table 2-1. Summary of HAPs Considered for Inclusion in Environmental Risk Screen 4
Table 2-2. Generic Ecological Assessment Endpoints Used in the Environmental Risk Screen 8
Table 3-1. Ecological Benchmarks Used in the Environmental Risk Screen for each PB-HAP and Assessment Endpoint 26
Table 5-1. Summary of PB-HAP Environmental Risk Screen Metrics 47
Table 5-2. Summary of Acid Gas Environmental Risk Screen Metrics 49

ACRONYMS/ABBREVIATIONS

AERMOD	American Meteorological Society/EPA Regulatory Model
AWQB	ambient water quality benchmarks
AWQC	ambient water quality criteria
BaP	benzo[a]pyrene
BM	benchmark
BW	body weight
CAA	Clean Air Act
DOE	Department of Energy
EcoEEF	ecological exposure equivalency factor
Eco-SSL	ecological soil screening level (Superfund)
EcoTEF	ecological toxic equivalency factor
EEF	exposure equivalency factor
EPA	Environmental Protection Agency
GEAE	generic ecological assessment endpoints
GLWQI	Great Lakes Water Quality Initiative
HAP	hazardous air pollutant
HCl	hydrogen chloride
HEM	Human Exposure Model
HF	hydrogen fluoride
LOAEL	lowest observed adverse effect level
LOEL	lowest observed effect level
MACT	maximum achievable control technology
NAAQS	national ambient air quality standards
NAWQC-ALC	national ambient water quality criteria–aquatic life criteria
NEL	no effect level
NOAEL	no observed adverse effect level
OAQPS	Office of Air Quality Planning and Standards (U.S. EPA)
ORNL	Oak Ridge National Laboratory
OSWER	Office of Solid Waste and Emergency Response (U.S. EPA) (currently Office of Land and Emergency Management
PAH	polycyclic aromatic hydrocarbon
PB-HAP	persistent and bioaccumulative HAP
PEL	probable effect level
POM	polycyclic organic matter
RAIS	Risk Assessment Information System (ORNL)
RTR	Risk and Technology Review
SAB	Science Advisory Board
SEB	soil ecotoxicity benchmark
SQB	sediment quality benchmark
SV	screening value
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin, termed “dioxin” in this report
TCE	trichloroethylene
TEF	toxic equivalency factor
TEL	threshold effect level
TRIM.FaTE	TRIM’s Fate, Transport, and Ecological Exposure module
TRV	toxicity reference value
UF	uncertainty factor

1 Overview

The environmental risk screen was developed to examine the potential for "adverse environmental effects" as required under Section 112(f)(2)(A) of the Clean Air Act (CAA). Section 112(a)(7) of the CAA defines an "adverse environmental effect" as:

> *"any significant and widespread adverse effect, which may reasonably be anticipated, to wildlife, aquatic life, or other natural resources, including adverse impacts on populations of endangered or threatened species or significant degradation of environmental quality over broad areas."*

The environmental risk screen was developed as a systematic, scientifically defensible, and efficient approach that the U.S Environmental Protection Agency (EPA) can use to screen for potential adverse environmental effects associated with emissions of hazardous air pollutants (HAPs) from facilities in Risk and Technology Review (RTR) source categories. The environmental risk screen is designed so it can be used effectively for large source categories, some with more than one thousand facilities, and for facilities located in any part of the United States.

The screen can be run quickly and with minimal additional data gathering by drawing on existing data, models, and modeling results, including those developed for the human health multipathway risk screen. The environmental risk screen uses the same TRIM.FaTE (Total Risk Integrated Methodology's Fate, Transport, and Ecological Exposure module) multipathway modeling and American Meteorological Society/EPA Regulatory Model (AERMOD) air dispersion modeling used for the human health risk assessment. In addition, the environmental risk screen applies ecological assessment endpoints and ecological health benchmarks to the same tiered screen design used for the human multipathway screen.

The environmental risk screen was developed to ensure consistency with the following EPA guidance and peer-review comments:

- EPA's *1998 Guidelines for Ecological Risk Assessment* (U.S. EPA 1998)

- EPA's Scientific Advisory Board Comments (U.S. EPA SAB 2010) on the Portland Cement manufacturing case study and the Petroleum Refining case study provided to the panel for review of RTR methods (U.S. EPA 2009)
- Participant comments and feedback provided to EPA during the Office of Air Quality Planning and Standards (OAQPS) Workshop on Ecological Risk Assessment of Air Toxics, June 2006 (ICF 2006).

Below, we summarize the design and key features of the environmental risk screen. In Section 2, we present the environmental risk screen conceptual model, the HAPs included in the screen, and the endpoints for which environmental risk are screened. Section 3 presents the approach used to identify ecological benchmarks for each HAP for each assessment endpoint. Section 4 describes the methods used to estimate HAP exposures in the environment. This section also describes how we used the benchmarks identified in Section 3 to calculate "screening threshold emission rates" and how we compared those thresholds to exposure estimates to screen for adverse environmental effects. Section 5 presents the outputs and analyses of the risk screening results.

2 Key Components of the Environmental Risk Screen

2.1 ENVIRONMENTAL HAPS

When considering which HAPs should be included in the environmental risk screen, we narrowed the list of 189 HAPs to the 31 environmental HAPs suggested in EPA's 2006 Ecological Risk Workshop. The workshop participants developed a list of 31 suggested environmental HAPs by starting with the 14 PB-HAPs identified for the RTR program (See the second column of Table 2-1) and then adding the following 17 pollutants for the reasons indicted below (OAQPS Workshop on Ecological Risk Assessment of Air Toxics June of 2006; ICF 2006):

- Hydrogen chloride (HCl), hydrogen fluoride (HF), and trichloroethylene (TCE) – toxicity to plants
- Hexachlorobutadiene and pentachlorophenol – toxicity to plants and aquatic species
- Phthalates – dibutyl phthalate, dimethyl phthalate, and bis-(2-ethylhexyl) phthalate (DEHP) – endocrine disruptors

- HAP metal compounds – antimony compounds, arsenic compounds, beryllium compounds, chromium compounds, cobalt compounds, manganese compounds, nickel compounds, and selenium compounds – persistence
- Cyanide compounds – highly toxic.

We evaluated the 31 suggested environmental HAPs for inclusion in the environmental screen based on the criteria shown in Table 2-1:

- Persistence and bioaccumulation potential
- Inclusion in the TRIM.FaTE multipathway model
- Magnitude of emissions
- Relative environmental toxicity – based on toxicity to wildlife, soil communities, and aquatic biota.

Table 2-1. Summary of HAPs Considered for Inclusion in Environmental Risk Screen

Pollutant	*RTR PB-HAP*	In Multi-pathway Model	2005 NEI Point Source Emissions (TPY)	Environmental Criteria			Included/Excluded – Rationale
				Wildlife NOAEL for Mink (mg/kg/d)[a]	Soil Screening BM (mg/kg)[b]	Water Quality Criteria (µg/L)[c]	
Antimony compounds			54	0.052	0.142[d]	80[d]	**Excluded** – persistent, but not bioaccumulative.
Arsenic compounds		X	181	0.052	100	150	**Included** – persistent but not bioaccumulative; low toxicity to aquatic biota and soil communities; but high relative wildlife toxicity.
Beryllium compounds			12	0.51	1.06[d]	3.6[d]	**Excluded** – persistent, but not bioaccumulative.
Bis-(2-ethylhexyl) phthalate (DEHP)			266	7.6	0.925[d]	0.3[d]	**Excluded** – not bioaccumulative; low relative wildlife toxicity.
Cadmium compounds	X	X	34	0.742	20	0.25	**Included** – PB-HAP in multipathway model; moderate wildlife and aquatic toxicity.
Chlordane	X		0.01	14	0.22[d]	0.0043	**Excluded** – PB-HAP, but very low emissions.
Chlorinated dibenzodioxins and furans (2,3,7,8-TCDD)	X	X	NA	0.0000008	1.2E-06[e]	1.0E-05[e]	**Included** – PB-HAP in multipathway model, high relative toxicity.
Chromium compounds			4,025	2.52 (Cr6) 2,105 (Cr3)	10	11 (Cr6) 74 (Cr3)	**Excluded** – persistent, but not bioaccumulative; low relative wildlife and water toxicity.
Cobalt compounds			77	7.33[g]	0.14[d]	24[d]	**Excluded** – persistent, but not bioaccumulative.
Cyanide compounds			290	49.7	1.33[d]	5.2	**Excluded** – not PB-HAP.
DDE dichlorodiphenyldichloroethylene	X		0.005	0.62	0.596[d]	4.5E-9[d]	**Excluded** – PB-HAP, but very low emissions.
Dibutyl phthalate			89	229	0.15[d]	9.7[d]	**Excluded** – not PB-HAP, low relative wildlife toxicity.
Dimethyl phthalate			248	NA	734[d]	330[e]	**Excluded** – not PB-HAP; low relative toxicity.
Heptachlor	X		0.002	0.1	0.0060[d]	0.0038	**Excluded** – very low emissions.
Hexachlorobenzene	X		0.61	0.08[i]	0.20[d]	0.0003[d]	**Excluded** – PB-HAP, but low emissions.
Hexachlorobutadiene			0.77	NA	0.040[d]	0.053[d]	**Excluded** – not PB-HAP, low emissions.
Hexachlorocyclohexane (all isomers)	X		0.01	NA	NA	NA	**Excluded** – PB-HAP, but low emissions, no BM.
Hydrogen chloride			396,069	NA	NA	NA	**Included** – high vapor emissions and high toxicity to terrestrial plants.

Pollutant	*RTR PB-HAP*	In Multi-pathway Model	2005 NEI Point Source Emissions (TPY)	Environmental Criteria			Included/Excluded – Rationale
				Wildlife NOAEL for Mink (mg/kg/d)[a]	Soil Screening BM (mg/kg)[b]	Water Quality Criteria (µg/L)[c]	
Hydrogen fluoride			60,238	NA	NA	NA	**Included** – high vapor emissions and high toxicity to terrestrial plants.
Lead compounds	X		307	6.15	900	2.5	**Included** – PB-HAP, Secondary NAAQS Standard.
Manganese compounds			1,386	68	100	120[h]	**Excluded** – persistent, but not bioaccumulative; low relative toxicity.
Mercury compounds	X	X	33	1.0	30	0.77	**Included** – PB-HAP, in multipathway model; methylmercury highly bioaccumulative and toxic.
Methoxychlor	X		0.001	3.1	NA	0.03	**Excluded** – PB-HAP, but very low emissions.
Nickel compounds			566	30.77	90	52	**Excluded** – persistent, but not bioaccumulative; low relative toxicity.
Pentachlorophenol			3	0.185	400	15	**Excluded** – not PB-HAP, low emissions.
Polychlorinated biphenyls	X		0.6	0.14[j]	0.000332[d]	0.014	**Excluded** – PB-HAP, but low emissions.
Polycyclic organic matter (BaP)	X	X	181	0.42	1.52[d]	0.014[g]	**Included** – PB-HAP, in multipathway model, and high relative toxicity.
Selenium compounds			496	0.154	100	5	**Excluded** – not PB-HAP; low relative toxicity.
Toxaphene	X		0.003	6.2	0.119[d]	0.0002	**Excluded** – very low emissions.
Trichloroethylene			4,291	0.291	12.4[d]	47[d]	**Excluded** – not PB-HAP; low relative toxicity.
Trifluralin	X		1	NA	NA	0.2[h]	**Excluded** – PB-HAP, but low relative toxicity to wildlife and no BM for soils.

Acronyms and abbreviations: BaP – benzoapyrene; BM – benchmark, NA – Not Available; NAAQS – National Ambient Air Quality Standards; PB-HAP – persistent bioaccumulative hazardous air pollutant, NEI = National Emissions Inventory, TPY = short tons per a year

[a]Sample et al. (1996). U.S. Department of Energy. ES/ER/TM-86/R3.

[b]Efroymson et al. (1997a,b). U.S. Department of Energy. ES/ER/TM-126/R2.

[c]U.S. EPA (2016b) National Aquatic Life Criteria Table. http://water.epa.gov/scitech/swguidance/standards/criteria/current/index.cfm

[d]U.S. EPA (2003a) Region 5 "RCRA [Resource Conservation and Recovery Act] Ecological Screening Levels" for soil and water.

[e]U.S. EPA Region 6 recommends using Texas Natural Resource Conservation Commission values (TNRCC 2001).

[f]U.S. EPA (2005c) "Ecological Soil Screening Levels for Cobalt, Interim Final" OSWER Directive 9285.7-67

[g]Suter and Tsao (1996). U.S. Department of Energy. ES/ER/TM-96/R2.

[h]U.S. EPA (2006). Region 3 Biological Technical Assistance Group (BTAG) Freshwater Sediment Screening Benchmarks.

The far right column of Table 2-1 shows the rationale for each HAP's inclusion or exclusion from the current environmental HAP risk screen. The following eight environmental HAPs are included in the environmental risk screen:

- Six persistent and potentially bioaccumulative HAPs:
  - arsenic
  - cadmium
  - dioxins/furans
  - polycyclic organic matter
  - mercury (both inorganic mercury and methyl mercury)
  - lead
- Two acid gases:
  - hydrogen chloride
  - hydrogen fluoride.

HAPs that persist in the environment and bioaccumulate through food chains are of particular environmental concern. They can accumulate in soils and sediments, with subsequent releases to pore water and surface waters where they can be taken up by plants or by animals (e.g., small fish) near the base of food webs, with possible further concentration by animals at higher trophic levels. Table 2-1 shows that cadmium, dioxins/furans, mercury, and POM all have relatively high environmental toxicity values (i.e., threshold-for-effect benchmarks are relatively low). Lead was included in the screen because it is a PB-HAP and because we can use the secondary lead National Ambient Air Quality Standards (NAAQS) as a reasonable measure for determining whether an adverse environmental effect occurs. According to the 2011 National Air Toxics Assessment of stationary sources, the six PB-HAPs we include in the screening analysis (arsenic, cadmium, mercury, lead, dioxins, POM) account for 99.9 percent of national emissions of PB-HAPs (the 14 in the RTR list cited above plus arsenic).

The acid gases HCl and HF were included due to their well-documented potential to cause direct damage to terrestrial plant foliage. In addition, when HF concentrations are above those at which plant damage is first seen, HF can cause fluorosis in livestock feeding on exposed forage. According to the 2005 National Emissions Inventory, HCl and HF account for about 99 percent (on a mass basis) of national acid gas emissions from stationary sources. We acknowledge that

other HAPs beyond the eight discussed above might have potential to cause adverse environmental effects. Therefore, EPA might add other HAPs to its environmental risk screen in the future, as risk assessment methods and resources allow.

2.2 ASSESSMENT ENDPOINTS

For the RTR environmental risk screen, we use conventional generic ecological assessment endpoints (GEAEs) (U.S. EPA 2003b, 2016b; Suter et al. 2004). EPA's 1998 *Guidelines for Ecological Risk Assessment* defines an ecological assessment endpoint as "an explicit expression of the environmental value to be protected and is defined operationally as an ecological entity (e.g., individual organisms, specified populations of species, biological communities or assemblages, and ecosystems) and its attributes (e.g., frequency of mortality, average fecundity, species abundance, community diversity)" (U.S. EPA 1998). Although EPA developed GEAEs to improve the scientific basis for ecological risk management decisions at EPA, GEAEs are used frequently for ecological risk assessments conducted outside the Agency.

For the RTR assessment, all emissions of HAPs are to the air from point sources (i.e., facilities) in the evaluated source categories. Consequently, all environmental media can be exposed to the HAPs. For the ecological HAPs that partition primarily to air (e.g., HCl, HF), we evaluate risks to the environment from direct contact with the airborne HAPs. For HAPs that can deposit on and partition to ground-level environmental media, and from there partition to other media and accumulate along biological food chains (i.e., PB-HAPs), we evaluate multimedia risks to the environment.

In the environmental risk screen, we evaluate the following four exposure media: terrestrial soils, surface water bodies, fish consumed by wildlife, and air. Within these four exposure media, we evaluate the nine GEAEs shown in Table 2-2. The GEAEs reflect the overall "health" of aquatic and terrestrial ecosystems and any important biota or community types that could be exposed in those ecosystems. For PB-HAPs, the generic set of receptors includes both community-level and population-level endpoints. For acid gases, the receptors are terrestrial plant communities. Selection of species for the population-level assessments for PB-HAPs was based on those organisms likely to be the most highly exposed due to bioaccumulation of the PB-HAP through

aquatic and terrestrial food chains. Exposure scenarios assumed for all GEAEs are chronic. For each GEAE listed in Table 2-2, we identified ecotoxicity benchmarks as discussed in Section 3.

Table 2-2. Generic Ecological Assessment Endpoints Used in the Environmental Risk Screen

Exposure Media	No.	Assessment Endpoint	Entities	Relevant Attributes	Benchmark[a]
Terrestrial Soils	1	Maintain structure/function of soil invertebrate communities (e.g., for nutrient recycling, soil aeration)	Assemblages of earthworms, insect grubs, nematodes	Species abundance and diversity; species composition; and survival and reproduction of those species' populations	Soil ecotoxicity benchmark (SEB): Invertebrates
	2	Maintain structure/function of terrestrial plant communities (e.g., for food and habitat for wildlife)	Assemblages of plant species: trees, herbs, grasses	Species abundance and diversity; survival, growth, and productivity of those species	SEB: Plants
	3	Maintain local bird populations that feed on soil invertebrates	Woodcock, robins, thrashers	Individual survival, growth, reproduction and development; area contaminated	SEB: Birds
	4	Maintain local mammal populations that feed on soil invertebrates	Shrews, moles, voles	Individual survival, growth, reproduction and development; area contaminated	SEB: Mammals
Surface Water Bodies	5	Maintain benthic community structure/function (sediment-dwelling organisms)	Assemblages of aquatic insects, amphipods, isopods, crayfish	Species abundance and diversity; survival, growth, development, and reproduction of those species	Sediment quality benchmark (SQB)
	6	Maintain aquatic community structure/function (water-column community to support fisheries)	Assemblages of fish and invertebrates in water column	Species abundance and diversity; survival, growth, development, and reproduction of those species	Ambient water quality benchmark (AWQB)
Fish (consumed by wildlife)	7	Maintain local populations of birds that feed on fish and other aquatic prey	Common merganser, belted kingfisher, herons, gulls, loons	Individual survival, growth and development, reproduction; area contaminated	Wildlife Toxicity Reference Value (TRV)
	8	Maintain local populations of mammals that feed on fish and other aquatic prey	Mink, otter, raccoon	Survival, growth and development, reproduction at the individual level; proportion habitat contaminated	Wildlife TRV
Air	9	Maintain community structure/ function of plants with foliage exposed to HAPs in air (e.g., food and habitat for wildlife)	Trees, shrubs, herbs, grasses, crops	Abundance; productivity	Air ecotoxicity benchmark: Plants

[a]A soil ecotoxicity benchmark (SEB) is a generic term used here to indicate any type of soil benchmark for ecological risk assessment. A sediment quality benchmark (SQB) also is a generic term, as is the term ambient water quality benchmark (AWQB). Different agencies, states, and offices have named and defined their own particular SEBs, SQBs, and AWQBs in different ways.

As mentioned above, the GEAEs used in the environmental risk screen include both population-level and community-level endpoints. Column 3 ("Assessment Endpoint") of Table 2-2 indicates whether an endpoint is population based or community based.

An assessment *population* is a group of organisms belonging to the same species that occupy the area defined as relevant to the ecological risk assessment (U.S. EPA 2003b). For the RTR risk screen, that area is defined as the modeling domain surrounding each facility. Endpoints 3, 4, 7, and 8 in Table 2-2 represent population-based GEAEs. Specifically, these population endpoints include bird and mammal populations that feed on soil invertebrates and aquatic prey (e.g., fish). Impairment of individual growth, development, reproduction, or survival could reduce population size and productivity and increase the probability of local extirpation if the impairment occurs in a significant proportion of the exposed or local population.

An assessment *community* is a multispecies group of organisms occupying the area defined as relevant to the assessment (U.S. EPA 2003b). For the RTR risk screen, that area is defined as the modeling domain surrounding each facility. Endpoints 1, 2, 5, 6, and 9 in Table 2-2 represent community-based GEAEs. Specifically, these community endpoints include the following communities: soil invertebrate, terrestrial plant, benthic, and aquatic.

2.3 ENVIRONMENTAL RISK SCREENING APPROACH

EPA conducts the environmental risk screen if any facilities in the source category emit any of the eight environmental HAPs. Specifically, if one or more of the eight environmental HAPs are emitted by at least one facility in the source category, the Agency conducts the environmental risk screen. Because of the unique properties and environmental effects of the HAPs, the environmental risk screen differs for three groups of the eight environmental HAPs:

- PB-HAPs – arsenic, cadmium, mercury, POM, and dioxin/furans
- Lead
- Acid gases – HCl and HF.

An overview of the environmental risk screen for each group is provided below.

2.3.1 PB-HAPs

For the five PB-HAPs—arsenic, cadmium, mercury, POM, and dioxins/furans—the environmental risk screen consists of three tiers (Figure 2-1). The tiered design used for the environmental risk screen is the same as that used for the human multipathway screen described Appendix 6 of the Risk Report.[1] Each tier uses a different conceptual model for the spatial relationship of the facility to surface waters and terrestrial environments, and each tier uses different parameter inputs. All three tiers of the environmental risk screen for PB-HAPs use the same assessment endpoint benchmarks (see Section 2.2).

The first tier of the screen is based on a hypothetical facility for which the surrounding environment was designed to encompass a health-protective environmental layout that would maximize PB-HAP concentrations in fish and in terrestrial environments in the immediate vicinity of a facility. This conceptual model is the same as used for the Tier 1 human health screen.

Section 4 provides further description of the conceptual model as applied in the environmental risk screen. TRIM.FaTE simulations were used to back-calculate Tier 1 screening threshold emission rates that correspond to the assessment endpoint benchmarks for each PB-HAP. In other words, each Tier 1 screening threshold emission rate represents the emission rate in tons per year that results in media concentrations at the hypothetical facility that equal the relevant ecological benchmarks.[2]

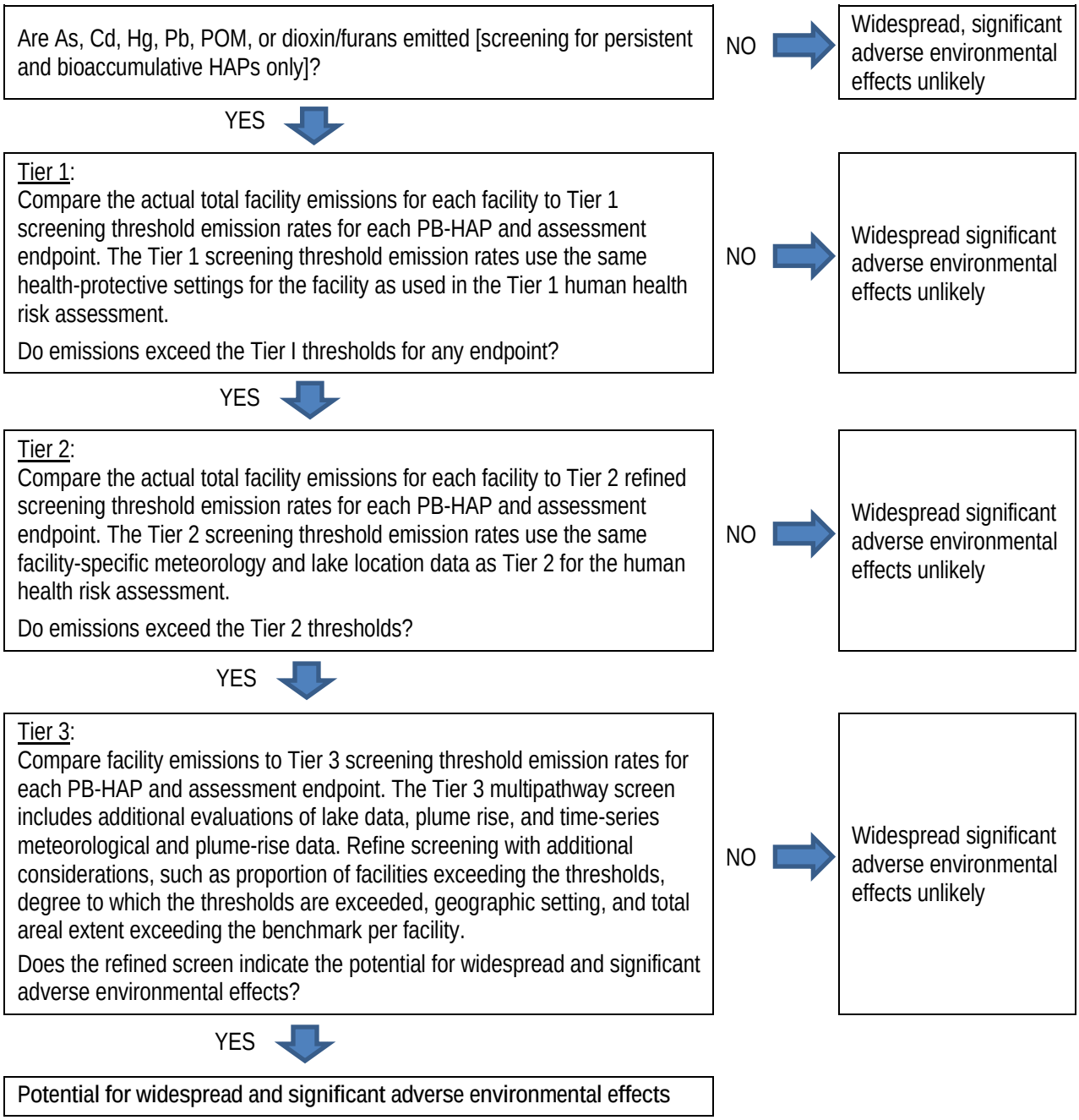
The Tier 1 environmental risk screen is performed by comparing the reported emission rate for each facility in tons per year to the Tier 1 screening threshold emission rate in tons per year for each PB-HAP, GEAE, and effect level if more than one is identified. If none of the emissions from a facility exceed these chemical-specific Tier 1 screening threshold emission rates, the facility "screens out" and therefore is not evaluated further under the environmental risk screen.

[1] Appendix 6 to the Risk Report is the *Technical Support Document for the TRIM-Based Multipathway Tiered Screening Methodology for RTR*.

[2] See Section 3, Effects Assessment, for discussion of the ecological benchmarks and wildlife toxicity reference values used for all three tiers of the environmental assessment.

If emissions from a facility exceed any of the Tier 1 screening threshold emission rates, the facility could be further evaluated in Tier 2.

Figure 2-1. Overview of the Environmental Risk Screen for PB-HAPs



In Tier 2 of the environmental risk screen, the screening threshold emission rates are refined to account for facility-specific meteorology and the actual location of lakes near facilities that did not pass the Tier 1 screen. If emissions from a facility do not exceed the Tier 2 screening

threshold emission rates, the facility "screens out" and is not evaluated further under the environmental risk screen. If emissions from a facility exceed the Tier 2 screening threshold emission rates, the facility could be further evaluated in Tier 3.

In Tier 3 of the environmental risk screen, the screening threshold emission rates are refined to account for lake data and time-series meteorological and plume-rise data (see Section 4 of Appendix 6 of the Risk Report for more information on the Tier 3 methods). If emissions from a facility do not exceed the Tier 3 screening threshold emission rates, the facility "screens out" and is not evaluated further. If emissions from a facility exceed the Tier 3 screening threshold emission rates, the facility could be further evaluated to consider the degree to which the screening threshold emission rates are exceeded, which endpoints and effect levels are exceeded, the geographic setting, and the total area exceeding the screening threshold emission rates. If, after additional refinement, the facility still exceeds the screening threshold emission rates, the facility might cause adverse environmental effects.

As with the multipathway human health risk assessment, a site-specific assessment could be conducted if the Tier 3 screening results indicate a potential for adverse environmental effects. The site-specific assessment uses model parameter values and scenario designs intended to better represent the modeled facility—aspects such as local terrain (influencing runoff and erosion patterns), watersheds, actual lake boundaries and water retention rates, soil types, and land cover. Site-specific assessments are not presented in this report.

2.3.2 Lead

The environmental risk screen for lead consists of one tier. For lead compounds, we currently do not have the ability to calculate concentrations in soils, surface waters, and sediments using the TRIM.FaTE model. Therefore, to evaluate the potential for adverse environmental effects from lead compounds, we compare the Human Exposure Model (HEM)/AERMOD-modeled air concentrations of lead for each facility in the source category to the level of the secondary

NAAQS for lead.[3] We consider values below the level of the secondary lead NAAQS to be unlikely to cause adverse environmental effects.[4]

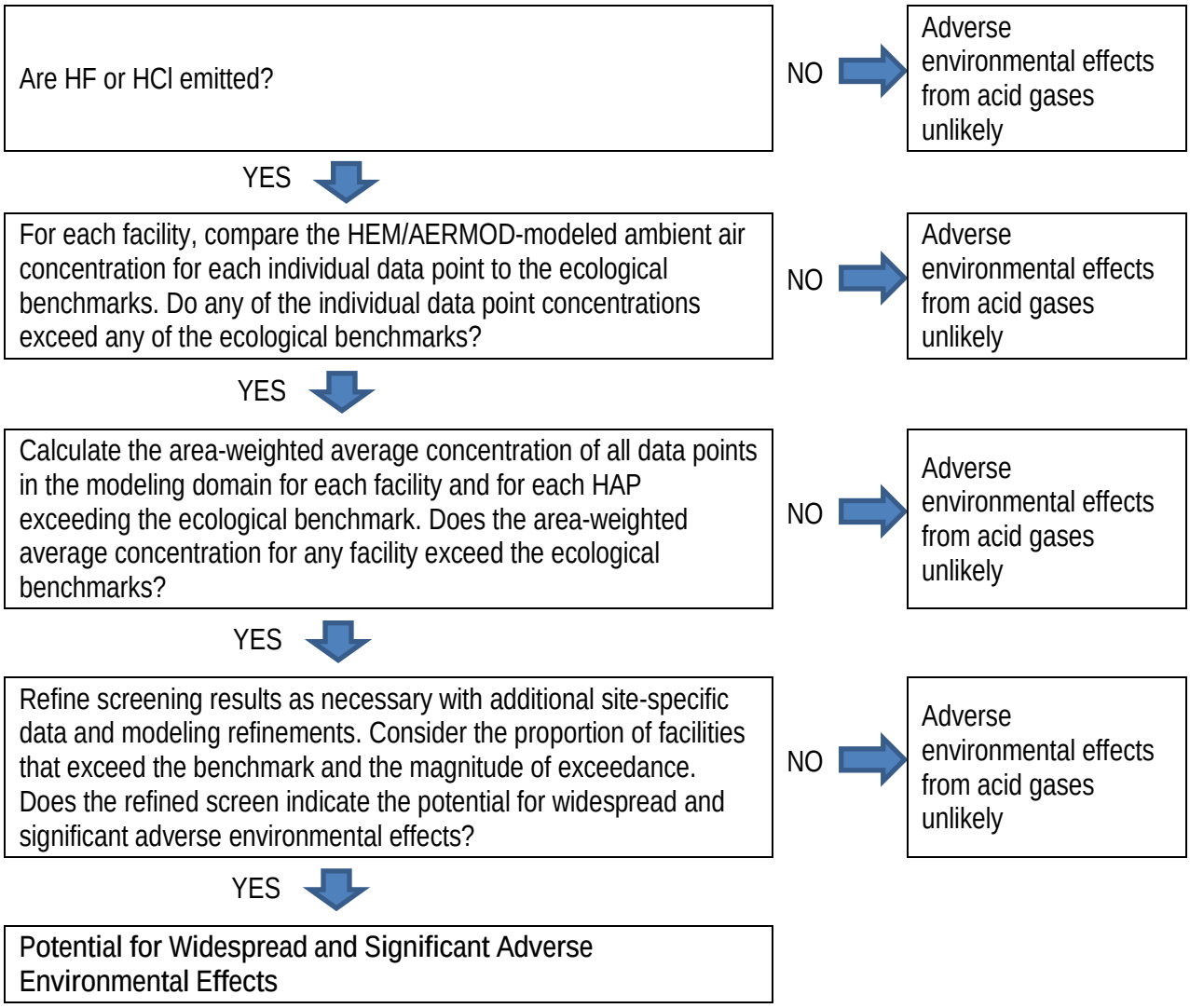
2.3.3 Acid Gases

For HF and HCl, the environmental risk screen evaluates potential phytotoxicity (i.e., poisonous to plants) and reduced productivity of plants due to chronic exposure. For each acid gas, the environmental risk screen compares the HEM/AERMOD-modeled ambient air concentrations in the modeling domain around each facility to ecological benchmarks (Figure 2-2). If the average concentration of a given HAP in the modeling domain around a facility exceeds the ecological benchmark, the facility does not pass the screen and, therefore, might cause adverse environmental effects.

[3] The secondary lead NAAQS is a reasonable measure of determining whether an adverse environmental effect exists because it was established considering "effects on soils, water, crops, vegetation, man-made materials, animals, wildlife, weather, visibility and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being."

[4] On October 18, 2016, EPA issued its final decision to retain the 2008 NAAQS for lead (U.S. EPA 2016a).

Figure 2-2. Overview of the Environmental Risk Screen for Acid Gases



3 Effects Assessment

To assess effects, we identified appropriate ecological benchmarks to compare to exposure concentrations. As indicated in Section 2.2, we searched for available ecological benchmarks for each assessment endpoint listed in Table 2-2. Specifically, we sought benchmarks for chronic exposure to each HAP included in the environmental risk screen, except for lead, which was screened using the secondary NAAQS.

Ecological benchmarks represent a level of exposure to a chemical in the environment that has been linked to a particular environmental effect level (e.g., a no-effect level or a threshold effect level) through scientific study. The three general metrics for ecological benchmarks are listed below.

- **Dose-based –** Dose-based benchmarks are expressed as a dose of chemical ingested per day per kg of animal body weight, typically mg/kg-day, which has been linked to a particular effect level. This type of benchmark usually is used when evaluating risks to wildlife via ingestion pathways. TRVs for terrestrial animals (e.g., wildlife) are an example of a dose-based benchmark.
- **Concentration-based –** Concentration-based benchmarks are expressed as the concentration of a chemical in an environmental medium (e.g., µg of HAP per liter of water) that has been linked to a particular environmental effect level. Concentration-based benchmarks usually are used when evaluating risks to receptors that have direct contact with the contaminated medium (e.g., fish in water, plant roots in soil, plant foliage in air).
- **Tissue-based –** Tissue-based benchmarks are expressed in units of the amount of chemical per mass of tissue in the exposed receptor (e.g., mg of cadmium per kg of tissue). This type of benchmark can be used to assess almost any type of consumer animal (e.g., fish, benthic invertebrates, birds, mammals).

To evaluate risk in the RTR program, we use reported emissions data that include the mass of HAPs emitted from each facility in the source category being examined. The emissions data are used as inputs to the TRIM.FaTE multipathway model to estimate HAP concentrations in soil, surface water bodies, and fish, and using the HEM/AERMOD model to estimate HAP concentrations in air. These estimates are best suited to the use of dose-based or concentration-based benchmarks. Tissue-based benchmarks have little utility for the RTR program because site-specific data for the concentrations of HAPs in animal tissues (e.g., liver, kidney) are not available. Therefore, the identification of benchmarks for the environmental risk screen focused entirely on dose-based and concentration-based benchmarks.

Based on a review of available ecological benchmarks, where possible, we identified existing ecological benchmarks at three generic effect levels:

- **Probable effect level (PEL):** The level above which adverse effects at both population and community levels are expected to occur frequently. In general, local extirpation or absence of populations of key community species is likely, compromising community structure and function.
- **Threshold effect level (TEL):** The level at which some adverse effects might occur in a minority fraction (e.g., up to 20 percent) of the exposed proportion of a specified population (e.g., mink, merganser) or at which few species (e.g., 5 percent aquatic animal species) might be lost from a community. Losses are not expected to influence either population survival over its range at the county or state level or the overall structure and function of the community near the facility. To screen risks to wildlife populations, we use lowest-observed-adverse-effect levels (LOAELs) from scientific toxicity tests that assess survival, growth, reproduction, and development to calculate assessment population benchmarks from the same taxonomic class that represents TELs. LOAELs are the lowest test exposure level at which statistically significant adverse effects on survival, growth, reproduction, or development occurred in the test organisms of the toxicity study considered key.[5]
- **No effect level (NEL):** The highest exposure level at which no biologically significant increases occur in the frequency or severity of (1) adverse effects on community structure or (2) adverse effects on assessment populations. To screen risks to wildlife populations, we use no-observed-adverse-effect levels (NOAELs) from a key toxicity test[6] that assessed growth, reproduction, or survival species from the same taxonomic class to calculate assessment population benchmarks that represent NELs.[7]

We identified preferred benchmark sources to allow selection of benchmarks for each environmental HAP for each ecological assessment endpoint. In general, we used EPA sources at a programmatic level (e.g., Office of Water, Superfund Program), if available. If not, we used

[5]Many ecological risk assessors use the geometric mean of the LOAEL and NOAEL to represent a "threshold" acceptable exposure level. For the RTR assessment, we use the LOAEL to represent a threshold for potential "significant" (biologically) adverse effect in keeping with Section 112(a)(7) of the CAA.

[6]A key toxicity test is one selected from the set of adequately conducted and documented tests to represent a sensitive species and sensitive endpoint, given the experimental data set as a whole.

[7]No-effect-level benchmarks are generally used to assess risks to threatened and endangered species (e.g., U.S. EPA 2004), although additional "safety" factors might be applied to account for species-to-species variation in chemical sensitivity and for extrapolation from laboratory to field conditions.

EPA benchmarks used in regional programs (e.g., region-specific Superfund). If benchmarks were not available at a programmatic or regional level, we used benchmarks developed by other federal agencies (e.g., National Oceanic and Atmospheric Administration), state agencies, or Canada. Section 3.1.2 discusses the preferred benchmark sources in detail.

Benchmarks for all effect levels are not available for all combinations of environmental HAPs and assessment endpoints. In cases where benchmarks representing multiple effect levels, as defined above, were available for a particular environmental HAP and assessment endpoint, we used all three available effect levels. We believe this best informs conclusions regarding whether ecological risks exist and, if so, whether the risks could be considered significant and widespread. Probable-effect-level benchmarks generally are not available except for benthic community sediment benchmarks for some chemicals.

We have organized the remainder of this section into two sections: Section 3.1 – Benchmarks for PB-HAPs and Section 3.2 – Benchmarks for Acid Gases. Attachment A contains additional discussion about the ecological benchmarks, wildlife toxicity reference values, and toxic equivalency factors (TEFs). Attachment A also includes additional tables and citations to those presented in this section.

3.1 BENCHMARKS FOR PB-HAPS

This section identifies ecological toxicity (ecotoxicity) benchmarks, expressed as concentrations of chemicals in environmental exposure media, for the five PB-HAPs included in the environmental risk screen (Note, lead is the sixth PB-HAP. It is screened using the secondary NAAQS level for lead). It also includes TRVs for wildlife. The PB-HAPs included in the ecological effects assessment (i.e., benchmark assessment) are mercury (as methyl mercury or inorganic divalent mercury), cadmium, POM, arsenic, and dioxins/furans. We evaluated POM and dioxins/furans by relating each compound to an "index" compound within the group. Specifically, we identified both toxicity benchmarks for the "index" chemicals (i.e., benzo[a]pyrene, or BaP, and 2,3,7,8-tetrachlorodibenzo-p-dioxin or TCDD) and TEFs for the remaining chemicals in each category relative to the appropriate index chemical.

3.1.1 Types of Benchmarks for PB-HAPs

In this section, we define the benchmarks selected for the combinations of assessment populations and communities and exposure media listed in Table 2-2 (Section 2.2), focusing on PB-HAPs. We also briefly reiterate the three generic effect levels for which we sought benchmarks.

3.1.1.1 Surface Soil Benchmarks

Across the Agency, up to two distinct types of soil communities and two groups of wildlife species have been used to derive soil ecotoxicity benchmarks (SEBs): (1) invertebrate community, (2) plant community, (3) birds that feed on soil invertebrates, and (4) mammals that feed on soil invertebrates. The latter two assessment endpoints are included specifically for PB-HAPs because the soil invertebrates might bioaccumulate these chemicals, resulting in higher exposures for the ground-feeding birds and mammals compared with chemicals that do not bioaccumulate.

SEBs are expressed as milligrams (mg) or micrograms (µg) of chemical per kilogram (kg) dry soil. To screen a location for possible risks to one or more of the soil assessment endpoints, estimates of surface soil contamination of PB-HAPs are compared with available corresponding benchmark values. TRIM.FaTE estimates concentrations for surface soil compartments at several successive distances from the source up to 10 kilometers (km). The TRIM.FaTE estimate of surface soil compartment chemical mass per unit volume is converted to a dry weight soil concentration by multiplying the volume of the compartment by the fraction of the volume that is in solid phase (0.57) and dividing the volume of the compartment by the mass-density of soil particles (2.6 kg/L soil).

For SEBs for avian or mammalian wildlife that EPA already has calculated for the Superfund or Resource Conservation and Recovery Act programs, we accepted the SEB as is. Implicit in the SEB is the TRV for the bird or mammal used by the office to back-calculate the SEB.[8]

[8]EPA "back-calculates" an SEB for a ground-feeding bird (e.g., woodcock) or mammal (e.g., shrew) as a concentration of chemical in soil that would result in the bird or mammal ingesting an amount of chemical equal to its TRV in mg/kg-day. A chemical-specific bioaccumulation factor relates the concentration in the food (e.g., earthworms) to the concentration in the soil. For PB-HAPs, the SEBs are lowest for wildlife species that ingest soil

For its derivation of ecological soil screening levels (Eco-SSLs), EPA's Superfund Office uses both bounded and unbounded NOAELs to establish a TRV for birds and for mammals based on the geometric mean of NOAELs across different toxicity tests for growth and reproductive effects for each taxon (U.S. EPA 2003c). The method also uses bounded LOAELs to check the final geometric mean NOAEL for plausibility. The geometric mean calculation gives equal weight to each result from multiple studies of the same endpoint (e.g., clutch size) for the same species (e.g., chicken) as for single studies of a different endpoint (e.g., weight gain by chicks) with a different species (e.g., mallard). We therefore conclude that the final geometric mean NOAEL does not account for interspecies variation in sensitivity (i.e., NOAEL is biased toward the species tested most often) and does not necessarily correspond to the most sensitive effect (i.e., NOAELs are averaged across growth and reproduction endpoints even if reproduction is the most sensitive endpoint).

3.1.1.2 Surface Water Body Benchmarks

Some EPA programs and regions (e.g., Superfund, Office of Water, Office of Pesticide Programs, various EPA Regions) also have developed aquatic benchmarks for two environmental "compartments" of aquatic ecosystems that might be in disequilibrium with each other: benthic sediments and the water column. Thus, benchmarks have been derived for aquatic communities in both compartments: the benthic community and the water-column community (Endpoints 5 and 6, respectively, in Table 2-2). The benthic community consists primarily of macroinvertebrates in contact with the sediments that consume detritus or graze on algae (e.g., amphipods, annelid worms, snails, aquatic larval stages of many insect species), but also can include filter feeders (e.g., mussels), predatory invertebrates (e.g., dragonfly nymphs), and invertebrate scavengers (e.g., crayfish). Benthic organisms are exposed through direct contact with contaminants in sediments and sediment pore water and by consumption of contaminated detritus/prey in the sediments. Benchmarks for the benthic community generally are called sediment quality benchmarks (SQB) and usually are expressed in units of mg chemical per

invertebrates (e.g., earthworms); other chemicals might be accumulated more by plants than by soil invertebrates. To calculate an SEB, EPA uses species-specific values for wildlife body weight, diet, food ingestion rate, and incidental soil ingestion as described in its guidance (U.S. EPA 2003c).

kilogram (kg) dry-weight sediment (Jones et al. 1997). Some SQBs are normalized to the total organic carbon content of the sediments (Jones et al. 1997).

The "aquatic" biota in the water-column compartment include plankton (i.e., microscopic algal cells and zooplankton such as water fleas and copepods) and free-swimming fish and some larger invertebrates (e.g., shrimp-like crustaceans). The water-column organisms are exposed by direct contact with the water (and water through their gills for respiration) and by ingestion of chemicals in their food. The food of free-swimming animals can be obtained from the water-column, the benthos, or both, depending on species of consumer. For that reason, the two compartments are not strictly separable when considering aquatic food webs. EPA Office of Water benchmarks for the water-column community of organisms generally are called ambient water quality benchmarks (AWQB) or criteria (AWQC) for the protection of aquatic life and are expressed as water concentrations in micrograms per liter of water.

3.1.1.3 Benchmarks for Wildlife that Feed on Contaminated Fish

For bioaccumulative chemicals, animals that feed at the top of food webs (i.e., top predators) are likely to experience the highest exposures of animal species in geographic area/ecosystem. For chemicals that bioaccumulate through aquatic food chains, the top predators in many geographic areas are wildlife that feed on aquatic prey. Thus, for PB-HAPs, EPA usually assesses risks to fish-eating (i.e., piscivorous) birds and mammals when evaluating ecological risks (e.g., see Great Lakes Water Quality Initiative, U.S. EPA 1995b).

EPA selected the American merganser (*Mergus merganser americana*), a bird of intermediate body size that regularly consumes relatively larger fish (up to 36 cm, Mallory and Mertz 1999), to represent highly exposed piscivorous birds. Many species of birds are piscivorous (Table 2-2, Endpoint 7). The belted kingfisher often is evaluated in ecological risk assessments; however, the maximum size of fish (and hence the top trophic level of fish they can consume) that belted kingfishers can consume is relatively small (generally no larger than 18 cm, Salyer and Lagler 1946).

EPA selected mink for screening of piscivorous mammals. Few mammals (see Table 2-2, Endpoint 8) are piscivores. Both river otters and mink commonly are assessed for risks from persistent and bioaccumulative chemicals (e.g., DDT [dichlorodiphenyltrichloroethane], DDE

[dichlorodiphenyldichloroethylene], PCBs [polychlorinated biphenyls], and other chemicals released directly to surface waters). Mink in some locations consume fish almost exclusively, and their smaller body size (i.e., 0.68–1.4 kg) compared with otters (i.e., 4.5–11 kg) (Burt and Grossenheider 1980) means that mink have a higher metabolic rate and so consume more fish per unit body weight than do otters. Both species consume primarily trophic level 3 fish (i.e., minnows, shiners, small trout, perch), although river otters capture larger fish on occasion. In addition, mink tend to be more abundant than otters and have smaller home ranges (U.S. EPA 1993a,b).

Note that geographic range was not a criterion that distinguished one species from another for the options listed above. The overall range of belted kingfishers and the common merganser spans North America from coast to coast, although the summer breeding ranges generally are more northerly while the overwintering ranges are more southerly. Similarly, the overall range of mink and river otters spans North America from coast to coast.

To assess risks to piscivorous wildlife from consuming contaminated fish for the environmental risk screen, we calculated TRVs, expressed as a dose, to compare with the total chemical intake of each wildlife species from its aquatic prey. To estimate exposures as total chemical intake, we used the Tier 1 (or Tier 2) screening TRIM.FaTE scenario to estimate the concentration of chemicals in the aquatic biota (compartments) included. Species-specific data for the mink and common merganser were used to estimate their food ingestion rates and the proportion of their diets likely obtained from each biotic compartment. For the latter, literature on the size of fish captured was consulted for both mink and merganser.

3.1.2 Preferred Sources for PB-HAP Benchmark Values

Available community-level benchmarks for sediments, surface waters, and soils were identified using the Oak Ridge National Laboratory (ORNL) Risk Assessment Information System (RAIS) (http://rais.ornl.gov/). The Department of Energy (DOE) maintains the ORNL RAIS database for use in its risk assessments at hazardous waste sites. RAIS identifies virtually all toxicity reference values and benchmarks developed to date by federal and some state agencies in the United States and by other countries (e.g., Canada) for human health and ecological risk assessment. RAIS therefore allows quick identification of available ecotoxicity benchmarks.

RAIS includes all screening-level ecological benchmarks available from Suter and Tsao (1996; benchmarks developed at ORNL for use at DOE Superfund sites), which was a key source of benchmarks for the Coke Oven MACT (maximum achievable control technology) Residual Risk Assessment (U.S. EPA 2003d). RAIS also includes the other sources of benchmarks used in that assessment (e.g., U.S. EPA National Ambient Water Quality Criteria, EPA Region 4 values, National Oceanic and Atmospheric Administration benchmarks, Florida Department of Environmental Protection benchmarks).

We established a hierarchy of preferred benchmark sources to allow selection of benchmarks for chemicals and environmental media for which numerous benchmarks are listed in RAIS. In general, EPA benchmarks used at a programmatic level (e.g., Office of Water, Superfund Program) are preferred, if available. If not, EPA Regional benchmarks as used in regional programs (e.g., Superfund) are used, if available. If benchmarks are not available from EPA at a regional level, we consider the benchmarks developed by other agencies (e.g., DOE), by states, or by Canada.

In all cases, we reviewed available benchmarks to find one to represent each of the three levels of effect specified above (i.e., NEL, TEL, PEL). For some media/chemical combinations, we could identify benchmarks for all three effect levels, but for most, we could not. In several cases, only a single benchmark was available, generally a threshold for effects.

3.1.2.1 Soil Ecotoxicity Benchmarks (SEB)

For soils, EPA's national Superfund Program Eco-Soil Screening Levels (Eco-SSLs, U.S. EPA 2005a) were selected, if available, as the SEBs for the RTR environmental risk screen. These Superfund Eco-SSLs (from EPA's Office of Solid Waste and Emergency Response, OSWER [currently Office of Land and Emergency Management) are the only peer-reviewed and EPA-vetted ecological toxicity screening benchmarks for soils established for use by the Agency nationwide. For chemicals for which no Eco-SSLs were available, EPA Regional sources of soil ecotoxicity benchmarks (SEBs) were reviewed (e.g., Regions 4, 5, and 6). The general methods for deriving those benchmarks can differ from the methods EPA used to derive Eco-SSLs.

For some chemicals, the Regions use SEBs developed by other agencies such as DOE or by a state within the region. If not specified in published information, we assumed that whichever

group of organisms was most sensitive to the chemical in soil (e.g., earthworms, insect larvae, plant roots, ground-feeding wildlife consuming soil invertebrates, and in some cases herbivorous animals consuming plants grown in the contaminated soil) was likely to have been the basis for the criterion. If an EPA Region and another non-EPA agency use the same numeric benchmark, all sources that designated that value are acknowledged in the tables presenting the RTR ecotoxicity benchmarks. Finally, if the only source providing a screening-level benchmark for soils was not an EPA office or Region (e.g., DOE, Environment Canada, or a state), that value is used.

3.1.2.2 Aquatic Sediment Quality Benchmarks

For the benthic community residing in and on the sediments of a water body, the preferred benchmarks were the national-level sediment quality criteria published by EPA's Office of Water (U.S. EPA 1993a, 2001b, 2003e, 2008), if they were available or readily usable.

If national sediment quality benchmarks were not available from EPA's Office of Water, we selected sediment benchmarks from those available from EPA's Superfund Program and Regions 4 and 5, as available. If EPA-vetted sediment benchmarks were not available, other benchmarks were used (e.g., from the State of Florida, ORNL, and MacDonald et al. [2000]).

3.1.2.3 Ambient Water-Column Benchmarks

For organisms that live primarily in the water-column of aquatic ecosystems, EPA's National Ambient Water Quality Criteria, Aquatic Life Criteria (NAWQC-ALC) were used, as available (Stephan et al. 1985, U.S. EPA 2002). According to Suter and Tsao (1996), the *acute* NAWQC-ALC are considered "upper" screening levels in EPA's Superfund program—which we interpret to mean *probable effect levels* if associated with continuous long-term (chronic) exposures. The *chronic* NAWQC-ALC are considered "lower" screening-level benchmarks in EPA's Superfund program (Suter and Tsao 1996). Given the methods by which both acute and chronic NAWQC-ALC are derived, we interpret the chronic NAWQC-ALC to represent a threshold for adverse effects in aquatic communities (water-column compartment) rather than a no-effect level. At the NAWQC-ALC, 5 percent of species typical of the ecosystem might be lost; however, substantial changes in aquatic community structure and function are not expected because of functional redundancies among species in aquatic communities.

For chemicals for which NAWQC-ALC and Tier II secondary values were not available, we turned to benchmarks developed by EPA Regions 4, 5, or 6.

3.1.2.4 Avian and Mammalian Toxicity Reference Values

To assess risks to piscivorous (i.e., fish-eating) wildlife, one must identify a TRV for the wildlife species, expressed as an oral dose, and estimate dietary exposure via the chemical in prey (i.e., in fish and invertebrates consumed). The estimated total chemical intake via all types of prey in the diet, expressed as mg[chemical]/kg[wildlife body weight]/day (mg[chem]/kg bw-day), then can be compared with the TRV (expressed in the same units). An emission rate, back-calculated to match the TRV, then is used to screen facilities in Tiers 1, 2, and 3 environmental risk screens.

Two types of avian and mammalian TRVs were included in the environmental risk screen. The first type of TRV is incorporated into the EPA OSWER derivation of the Eco-SSLs intended to protect wildlife that feed on soil invertebrates (see Section 3.1.1). We indirectly use those TRVs by using the Eco-SSLs as soil benchmarks. We calculated separate TRVs to use for wildlife that consume fish using an approach similar to that developed for the EPA Great Lakes Water Quality Initiative (GLWQI, U.S. EPA 1995b). Those calculations are presented in Attachment A, Section A.3.

EPA OSWER developed TRVs for the EcoSSLs using an approach unique to those benchmarks. The EcoSSL TRVs are based on NOAELs, and they are calculated as the geometric mean of all NOAELs from adequately performed and reported studies for growth and for reproductive effects across all species of birds (or mammals). Thus, even unbounded NOAELs, which might be well below an effect level (because no effect level was identified), are included in calculating the geometric mean. That method of calculating a wildlife TRV has some limitations, as discussed by several investigators (e.g., Allard et al. 2010; Mayfield and Fairbrother 2013; Sample et al. 2014a,b).

For purposes of the RTR assessment of fish-eating wildlife, we prefer the GLWQI approach to developing a TRV for wildlife (U.S. EPA 1995b), which is to select a key study that represents a sensitive species and endpoint from among the available, adequately conducted and reported, studies. Moreover, we prefer to scale doses between experimental animals and wildlife species

based on relative body weight (U.S. EPA 2011). The derivation of TRVs for PB-HAPs for piscivorous wildlife are presented in Attachment A, Section A.3.1.

For the GLWQI approach, the available toxicity data are examined to determine the magnitude of uncertainty factors (UFs) that might be needed for three types of data gaps: to estimate a NOAEL from a LOAEL, to extrapolate from subchronic to chronic exposure, or to account for differences in sensitivity of test species. For most chemicals (including PB-HAPs, particularly dioxins and POM), only a few species of birds (e.g., quail, mallard, chicken, pheasant) and a few species of mammals (e.g., mice, rats, hamster, mink) have been tested sufficiently to provide both a LOAEL and a NOAEL for effects resulting from chronic exposures. Uncertainty factors can range from 1 to 10 for each type of uncertainty listed above, depending on the apparent magnitude of the data gap. A joint uncertainty value (the product of all three types of UF) exceeding 100 indicates that a TRV cannot be derived (U.S. EPA 1995b). Typically, a value of 1, 3, or 10 (not values in between) is used for each UF. The appropriate UFs are applied as divisors of the original toxicity value (e.g., LOAEL).

To estimate TRVs for piscivorous wildlife, we used the LOAELs and NOAELs from a single key study (most sensitive effect and species). If only an unbounded LOAEL were available (no NOAEL), the LOAEL could be divided by a factor of 10 to extrapolate to a NOAEL or an EPA-derived UF could be applied. The subchronic-to-chronic uncertainty factor was not applied, because all TRVs calculated for the PB-HAPs are based on chronic or gestational exposures. Neither was an interspecies UF used, except for the case of methyl mercury, for which EPA had already published a joint LOAEL-to-NOAEL and an interspecies UF for birds (Attachment A, Section A.3.2). For the other PB-HAPs, doses were scaled between a test species and the assessment species based on relative body weight to the ¾ power (U.S. EPA 2011).

3.1.3 Selected PB-HAP Benchmarks

Table 3-1 shows the ecological benchmarks used in the environmental risk screen for each PB-HAP and assessment endpoint. A discussion of the TEFs used to adjust each POM chemical relative to BaP and to adjust each dioxin congener relative to TCDD is presented in Attachment A, Section A.4.

3.2 BENCHMARKS FOR ACID GASES

3.2.1 Hydrogen Chloride

For HCl, EPA identified chronic benchmark concentrations as described in Appendix K to EPA's (2009) *Risk and Technology Review (RTR) Risk Assessment Methodologies: For Review by the EPA's Science Advisory Board. Case Studies – MACT I Petroleum Refining Sources, Portland Cement Manufacturing*. Substantial data were available for short-term exposures of plants to HCl; however, data to relate chronic exposures of plants to adverse effects on growth and productivity were lacking.

The chronic benchmark for HCl was based on a lowest-observed-effect level (LOEL) from a short-term exposure (20 minutes) that related HCl concentration to "changes" in the leaves of 7 of 8 plant species as reported by Lerman et al. (1976). The benchmark was the lowest exposure concentration at which effects of any type were observed (visible injury to some proportion of leaves). Haber's law (see Attachment A, Section A.2.3.2) was used to extrapolate the 1.5-mg/m^3 LOEL concentration after 20 minutes of exposure to a 0.5-mg/m^3 concentration expected to produce the same effect after 1 hour. To extrapolate from a 1-hr estimated LOEL to a chronic benchmark, they divided by a factor of 10 to yield 0.050 mg/m^3 or 50 µg/m^3.

We recognize that the uncertainty associated with extrapolating from a 20-minute exposure with minimally defined visual effects on foliage to a chronic exposure scenario with plant productivity as the assessment endpoint is very high. Thus, 50 µg/m^3 cannot be assumed to represent a benchmark with a known effect level for chronic exposures. EPA does consider the benchmark, however, to likely represent a NEL for exposures of plants to HCl.

Table 3-1. Ecological Benchmarks Used in the Environmental Risk Screen for each PB-HAP and Assessment Endpoint

Eco-HAP	Assessment Endpoint	Benchmark Effects Level	Benchmark Value	Benchmark Source
Cadmium	Fish-eating birds feeding from lake	NOAEL–common merganser	0.7 (mg/kg BW/day)	CA DTSC HERD 2009
		LOAEL–common merganser	1 (mg/kg BW/day)	
	Fish-eating mammals feeding from lake	NOAEL–mink	0.742 (mg/kg BW/day)	Sample et al. 1996 from Sutou et al. 1980
		LOAEL–mink	7.42 (mg/kg BW/day)	
Cadmium	Lake benthic sediment community	No-effect level	0.33 (mg/kg dry sediment)	CCME 1999a
		Threshold level	1.2 (mg/kg dry sediment)	U.S. EPA 1996a
		Probable-effect level	3.5 (mg/kg dry sediment)	CCME 1999a

Eco-HAP	Assessment Endpoint	Benchmark Effects Level	Benchmark Value	Benchmark Source
Cadmium	Surface soil – birds and mammals that consume soil invertebrates; soil plant and invertebrate communities	Threshold–shrew	0.36 (mg/kg dry soil)	U.S. EPA 2005d, OSWER Eco-SSLs
		Threshold–woodcock	0.77 (mg/kg dry soil)	
		Threshold–plant community	32 (mg/kg dry soil)	
		Threshold–invert. community	140 (mg/kg dry soil)	
Cadmium	Water-column community	Threshold level (chronic)	0.72 (µg/L)	U.S. EPA 2001b, revised 2016b
		Frank-effect level (acute)	1.8 (µg/L)	
Arsenic	Fish-eating birds feeding from lake	NOAEL–common merganser	0.15 (mg/kg BW/day)	Sample et al. 1996 from Camardese et al. 1990
		LOAEL–common merganser	1.5 (mg/kg BW/day)	
Arsenic	Fish-eating mammals feeding from lake	NOAEL–mink	0.052 (mg/kg BW/day)	Sample et al. 1996 from Schroeder and Mitchener 1971
		LOAEL–mink	0.52 (mg/kg BW/day)	
Arsenic	Lake benthic sediment community	Threshold level	8.2 (mg/kg dry sediment)	U.S. EPA 1996a
		Probable-effect level	33 (mg/kg dry sediment)	U.S. EPA 1996b
Arsenic	Surface soil – birds and mammals that consume soil invertebrates; soil plant community	Threshold–shrew	46 (mg/kg dry soil)	U.S. EPA 2005b, OSWER Eco-SSLs
		Threshold–woodcock	43 (mg/kg dry soil)	
		Threshold–plant community	18 (mg/kg dry soil)	
Arsenic	Water-column community	Threshold level (chronic)	150 (µg/L)	U.S. EPA 1995a OW
		Frank-effect level (acute)	340 (µg/L)	
2,3,7,8-TCDD	Fish-eating birds feeding from lake	NOAEL–common merganser	0.0000014 (mg/kg BW/day)	U.S. EPA 1995b, GLWQI, from Nosek et al. 1992a,b
		LOAEL–common merganser	0.000014 (mg/kg BW/day)	
2,3,7,8-TCDD	Fish-eating mammals feeding from lake	NOAEL–mink	0.000000771 (mg/kg BW/day)	U.S. EPA 1995b, GLWQI, from Murray et al. 1979
		LOAEL–mink	0.00000771 (mg/kg BW/day)	
2,3,7,8-TCDD	Lake benthic sediment community	Threshold level	0.00000116 (mg/kg dry sediment)	Average of U.S. EPA 2001a, 2003a, 2006 (Regions 3, 4, and 5)
2,3,7,8-TCDD	Surface soil – mammals that consume soil invertebrates	Threshold – shrew	0.0000002 (mg/kg dry soil)	U.S. EPA 2003a, Region 5
2,3,7,8-TCDD	Water-column community	Threshold level (chronic)	0.000012 (µg/L)	U.S. EPA 2001a, Region 4
		Frank-effect level (acute)	0.1 (µg/L)	
Mercuric chloride	Lake benthic sediment community	Threshold level	0.16 (mg/kg dry sediment)	Average of 8*
		Probable-effect level	0.84 (mg/kg dry sediment)	Average of 4**
Mercuric chloride	Surface soil plant and invertebrate communities	Threshold–plant community	0.3 (mg/kg dry soil)	U.S. EPA Region 6 cites Efroymson et al. 1997a
		Threshold–invert. community	0.1 (mg/kg dry soil)	U.S. EPA 2015, Region 4
Mercuric chloride	Water-column community	Threshold level (chronic)	0.77 (µg/L)	U.S. EPA 1993c, 1995a, 2015, OW
		Frank-effect level (acute)	1.4 (µg/L)	
Mercury (methyl)	Fish-eating birds feeding from lake	NOAEL–common merganser	0.013 (mg/kg BW/day)	U.S. EPA 1995b from Heinz 1974, 1975, 1976a,b, 1979
		LOAEL–common merganser	0.078 (mg/kg BW/day)	

Eco-HAP	Assessment Endpoint	Benchmark Effects Level	Benchmark Value	Benchmark Source
Mercury (methyl)	Fish-eating mammals feeding from lake	NOAEL–mink	0.0247 (mg/kg BW/day)	Sample et al. 1996 from Verschuuren et al. 1976
		LOAEL–mink	0.123 (mg/kg BW/day)	
Mercury (methyl)	Lake benthic sediment community	Threshold level	0.2 (mg/kg dry sediment)	MacDonald et al. 2000
		Probable-effect level	1 (mg/kg dry sediment)	
Mercury (methyl)	Surface soil – birds and mammals that consume soil invertebrates; soil plant and invertebrate communities	Threshold–montane shrew	0.0068 (mg/kg dry soil)	U.S. EPA 2015, Region 4
		Threshold–American robin	0.0011 (mg/kg dry soil)	
		Threshold–plant community	0.3 (mg/kg dry soil)	U.S. EPA Region 6 cites Efroymson et al. 1997a
		Threshold–invert. community	0.1 (mg/kg dry soil)	U.S. EPA 2015, Region 4
Mercury (methyl)	Water-column community	Threshold level (chronic)	0.0028 (µg/L)	U.S. EPA 2015, Region 4, cites Suter and Tsao 1996
		Frank-effect level (acute)	0.099 (µg/L)	
Benzo[a]-pyrene	Fish-eating mammals feeding from lake	NOAEL–mink	0.417 (mg/kg BW/day)	Sample et al. 1996, from Mackenzie and Angevine 1981
		LOAEL–mink	4.17 (mg/kg BW/day)	
Benzo[a]-pyrene	Lake benthic sediment community	No-effect level	0.032 (mg/kg dry sediment)	CCME 2012
		Threshold level	0.15 (mg/kg dry sediment)	U.S. EPA 1996b, 2006
		Probable-effect level	1.45 (mg/kg dry sediment)	
Benzo[a]-pyrene	Surface soil – mammals that consume soil invertebrates	Threshold–masked shrew	1.52 (mg/kg dry soil)	U.S. EPA 2003a, Region 5
Benzo[a]-pyrene	Water-column community	Threshold level (chronic)	0.014 (µg/L)	U.S. EPA 2003a, Region 5, from Suter and Tsao 1996
		Frank-effect level (acute)	0.24 (µg/L)	Suter and Tsao 1996
Lead	Ambient Air	NAAQS Secondary Standard	0.15 (µg/m^3)	U.S. EPA 2016a

Acronyms/abbreviations: BW = avian or mammalian body weight; invert. = invertebrates; CCME = Canadian Council of Ministers of the Environment; GLWQI = Great Lakes Water Quality Initiative; NAAQS = National Ambient Air Quality Standards; LOAEL = lowest-observed-adverse-effect level; NOAEL = no-observed-adverse-effect-level; OW = EPA's Office of Water; TCDD = 2,3,7,8-tetrachlorodibenzo-p-dioxin.

* Average of 8 threshold-effect levels: U.S. EPA (1996b) 0.18 mg/kg dry sediment; MacDonald et al. (2000) 0.18 mg/kg; Florida Department of Environmental Protection (FDEP, MacDonald 1994) 0.13 mg/kg; U.S. EPA (1996a) 0.15 mg/kg; U.S. EPA (2015) 0.13 mg/kg; U.S. EPA (2006) 0.18 mg/kg; U.S. EPA (2003a) 0.174 mg/kg); and Region 6 (TNRCC 2001) 0.174 mg/kg.

** Average of 4 probable-effect levels: U.S. EPA (1996b) 1.06 mg/kg; MacDonald et al. (2000) 1.06 mg/kg; FDEP (MacDonald 1994) 0.70 mg/kg; and CCME (2001) 0.486 mg/kg.

3.2.2 Hydrogen Fluoride

HF is one of the most phytotoxic air pollutants. It is 10 to 1000 times more toxic to plants than ozone, and many species of plants are more sensitive to the chronic effects of HF than are humans (APIS 2010). Reports from decades ago document commercially significant injuries to plants near facilities that emitted fluoride. The damages included "commercially significant" reductions in crops of citrus fruits (Wander and McBride 1956); grapes (Brewer et al. 1957; Wann 1953); Italian prunes (Miller et al. 1948; Wann 1953); peaches (Daines et al. 1952); ponderosa pine (Adams et al. 1956); apricots (Wann 1946; De Ong 1946); and many varieties of

gladioli (Johnson et al. 1950; Miller et al. 1953) (examples cited by Hill and Pack 1983). In an area around one industrial emitter of HF, before installation of control equipment, a high proportion of the ponderosa pine trees surrounding the facility had died (Adams et al. 1956). Incidents like this in the United States, however, have declined; no publications describing similar events in the past few decades were identified in our literature search.

Atmospheric fluoride ion accumulates in the leaves of plants, entering through stomata on the underside of leaves. Atmospheric fluoride deposition to soils also can occur, but most soil fluoride changes to insoluble forms that are not readily bioavailable to plants. Several researchers have concluded that the limited amounts of fluoride that reach soils from contaminated atmospheres do not affect plant uptake overall (MacIntire et al. 1949; Hansen et al. 1958). Researchers also have demonstrated that leaves can absorb the fluoride from soluble fluoride particles (such as calcium fluoride, which yields a fluoride ion), particularly when the leaves are moist with dew. Nonetheless, fluoride as gaseous HF is the most bioavailable and causes much greater injuries to plants (Hill and Pack 1983).

Gas-phase HF is particularly hazardous to plants because of its tendency to accumulate over time in foliar tissue. Plants can accumulate HF to concentrations 1,000,000 times higher than ambient atmospheric concentrations. Thus, unlike many pollutants, HF is expected to cause injury to plants primarily from exposures over weeks to months, and the longer the exposure, the more severe the effects (Hill 1969).

Susceptibility to HF varies widely among plant species and varieties. Species known to be sensitive to HF exposure include gladioli, apricots, prunes, sorghum, corn, grapes, and conifers (Hill and Pack 1983). Species that are relatively insensitive to HF exposure include cotton, celery, alfalfa, and tomatoes (Hill and Pack 1983). Relatively low air concentrations can damage sensitive species, while less sensitive species can exhibit little to no damage at somewhat higher concentrations (TCEQ 2009; CEPA/FPAC WGAQOG 1996; Hill 1969). Several monocotyledons rank among the most sensitive taxa, including the genera *Gladiolus*, *Allium*, *Crocus*, *Tulipa*, *Lilium*, and *Polygonatum* (APIS 2010, citing Weinstein et al. 1998).

Attachment A, Section A.2.3 contains the following background sections.

- Section A.2.3.1: Methods for Establishing HF Benchmarks – presents three bases that can be used to establish HF regulatory standards.
- Section A.2.3.2: HF Regulatory Levels – summarizes atmospheric (air concentration) criteria and regulatory levels that states and other countries have established for HF for the protection of vegetation and other endpoints.
- Section A.2.3.3: Studies Showing the Effects of HF Exposure on Plants – discusses the bulk of readily available data relating HF exposures to plant responses based on atmospheric concentrations. Those data are presented to assist EPA risk managers in interpreting the results of screens of HF emissions. Comparisons of the criteria for protecting productivity of agricultural plants and livestock from fluorosis to those available for protecting human health indicate that air concentration benchmarks for HF developed for plants are lower than those developed to protect livestock and human health.

Two HF benchmarks are used for the environmental risk screen. The value of 0.5 μg HF/m^3 is based on the Washington State criterion for gaseous HF. The value of 0.4 μg HF/m^3, which is 20 percent lower, is based on the Environment Canada criterion. Both criteria were developed for 90-day averaging periods during the growing season.

For HF, we model annual estimates of facility emissions in HEM/AERMOD to obtain average annual HF air concentrations. When screening for chronic HF risks to plants in the environmental risk screen, we compare the average *annual* HF air concentrations from the HEM/AERMOD runs to the 90-day criteria. If exposures are not the same during the growing season and the nongrowing season, the use of annual average exposures could underestimate or overestimate risks. An additional uncertainty in evaluating chronic HF risks to plants is the wide variation in plant sensitivity to airborne HF and the relatively few nonagricultural plants that have been tested (Attachment A, Section A.2.3.2).

Empirical models that relate exposure concentration, exposure duration, and plant response for different plant groups are not simple mathematical relationships (e.g., see McCune et al. (1991) equations to predict severity *and* incidence of foliar injury from HF exposure concentration *and*

duration). In other words, although plant foliage accumulates fluoride from HF in air over time, effects on plants are not proportional to air concentration only, nor are they proportional to the simple product of average exposure concentration and duration (e.g., a time-weighted average exposure concentration). This lack of proportionality could be due to factors such as more frequent periods of rain wash-off that can leach fluoride from leaves over longer exposure periods and slower fluoride absorption rates as fluoride concentrations in plant leaves increase.

Short-term exposure data and criteria were not used to assess risk to plant communities from HF for several reasons. Characterizing possible adverse effects on the assessment endpoints of plant productivity and community structure (e.g., as habitat for wildlife, agricultural productivity) over the long term from data on species-specific effects on plants from short-term exposures (or short-term exposure criteria) would require many assumptions and include major uncertainties. Data are lacking to link effects like "foliar markings" and mild leaf necrosis to plant reproduction and productivity over the long term. Also lacking are data on the recovery of plants after short-term exposures and the frequency of high short-term exposures that could be tolerated if time needed for recovery is adequate. In addition, some long-term effects (e.g., annual seed production) that might result from short-term exposures would occur only if a short-term peak in HF concentration occurred during the few days of a sensitive life-stage of the plant (e.g., flowering).

4 Exposure Assessment

This section presents the models and methods used to estimate HAP exposures in the environment. We describe how to use the effect levels to calculate emission "screening thresholds" and how these thresholds are compared to facility emissions to screen for adverse environmental effects.

The first step in the ecological exposure assessment is to determine whether any facilities in the source category of interest emit any of the eight environmental HAPs (see Figure 2-1 and Figure 2-2 in Section 2.3). This step is performed by querying the emissions data for the source category in question. Typically, emissions data are obtained from the National Emissions Inventory, section 114 surveys of the industry, or from facility stack emissions tests. Emissions data for facilities identified in this step are used to perform the environmental risk screen, as described in this section. The approach for the overall environmental risk screen uses separate methods to

assess ecological exposures to PB-HAPs, lead, and acid gases. Section 4.1 details the exposure assessment methods for PB-HAPs. Section 4.2 details the exposure assessment methods for lead and the acid gases.

4.1 ENVIRONMENTAL RISK SCREEN FOR PB-HAPS

Figure 2-1 in Section 2.3 provides an overview of the approach for the environmental risk screen for PB-HAPs. This approach includes three tiers of assessment designed for implementation with a minimum of required site-specific or other assessment-specific inputs. The Tier 1, Tier 2, and Tier 3 approaches are discussed in further detail in Sections 4.1.1, 4.1.2, and 4.1.3. See Section 5 for further discussion of outputs from an environmental risk screen.

Possible exposure pathways from facility air emissions to biological receptors of concern were identified from HAP-specific chemical properties, the conceptual model of multimedia fate and transport, and the GEAEs in Table 2-2. The wildlife populations most highly exposed to PB-HAPs would be those that consume aquatic or terrestrial biota that have bioaccumulated the chemical along food chains. Thus, we assumed that some local populations of birds or mammals could be exposed to PB-HAPs that have bioaccumulated in food chains to relatively high concentrations in fish and in terrestrial prey. Additionally, persistent HAPs could accumulate over time in surface soils and reach concentrations toxic to terrestrial plants and to invertebrate communities in soils (e.g., earthworms).

The biotic compartments in the lake(s) for which TRIM.FaTE simulates whole-organism contaminant concentrations in Tiers 1, 2, and 3 are described below.

1. Phytoplankton, suspended algae in the water column, is modeled as a “phase” of the water column.
2. Zooplankton are modeled as a compartment in the water column that is in chemical equilibrium with the phases in the water column, including aqueous and algal phases.
3. Macrophytes in a lake can accumulate and “sequester” some chemicals and are modeled as a separate compartment in the water column.

4. Benthic invertebrates such as mollusks, crustacea, and aquatic insect nymphs that consume periphyton and detritus are modeled as a compartment in chemical equilibrium with bottom sediments.

5. Benthivorous fish are bottom-feeding fish (e.g., young catfish) that consume primarily benthic invertebrates.

6. Bottom-feeding carnivores (e.g., adult catfish) consume both benthic invertebrates and young benthivorous fish.

7. Water-column planktivores, such as young-of-the-year for many species and other small fish (e.g., shiners, minnows), consume primarily planktonic organisms.

8. Water-column omnivores are larger fish that consume invertebrates and smaller fish from both the benthic and pelagic environments (e.g., "panfish" like bluegill, yellow perch, and young age classes of the game species).

9. Water-column piscivores are larger game-fish species that primarily consume smaller fish in pelagic or benthic environments (e.g., walleye, largemouth bass).

The same aquatic food webs developed in TRIM.FaTE for the human health screen for fish ingestion are used to estimate doses to fish-eating wildlife species chosen as assessment populations for the environmental risk screen. The parameterization of those compartments is described in Appendix 6 to the Risk Report.

For wildlife exposed to PB-HAPs via consumption of aquatic life, we assume that the assessment populations obtain 100 percent of their diet from the appropriate biotic compartments corresponding to the different types of aquatic prey they consume. Parameterization of the wildlife diets and other relevant exposure factors (e.g., body weight) is described in Attachment A, Section A.6.

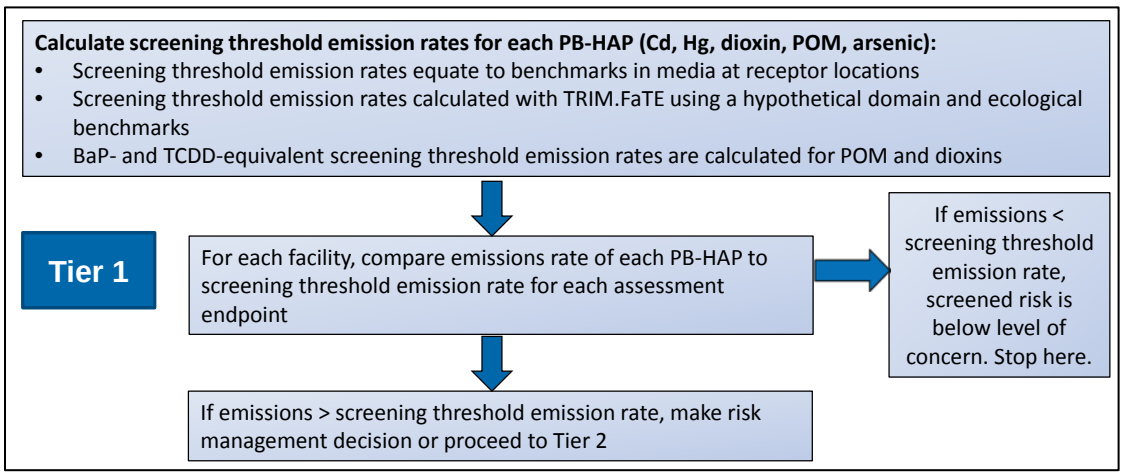
We also assumed that ground-feeding birds and mammals that consume primarily soil invertebrates (e.g., earthworms, grubs) could be exposed to PB-HAPs that have bioaccumulated in the invertebrates from ingestion of or contact with soils. We assumed that the assessment populations obtained 100 percent of their diet from the assessment area (radius of 10 km). We did not assess risks to higher-level carnivores (e.g., wolves, eagles) because their feeding ranges generally are large and difficult to link to specific facilities.

For benthic and water-column aquatic communities, we estimate exposure to PB-HAPs using the TRIM.FaTE-model-estimated concentrations in sediments and the water column, respectively, for the lake(s) situated in Tiers 1, 2, and 3.

4.1.1 Tier 1 Exposure Assessment

Figure 4-1 summarizes the Tier 1 screening approach for PB-HAPs. The Tier 1 assessments for all source categories use ecological screening threshold emission rates for each GEAE and PB-HAP. The screening threshold emission rates (in tons per year) yield concentrations in environmental media at receptor locations in the hypothetical TRIM.FaTE environmental setting that equal the ecological benchmarks. The ratio of a facility's PB-HAP emissions to the corresponding screening threshold emission rate is called the "screening value" (SV). When rounded to one significant figure, SVs greater than 1 indicate that adverse ecological effects within 10 km of the facility cannot be ruled out, and further assessment (e.g., Tier 2, see Section 4.1.2) might be needed.

Figure 4-1. Approach for Tier 1 Environmental Risk Screen for PB-HAPs



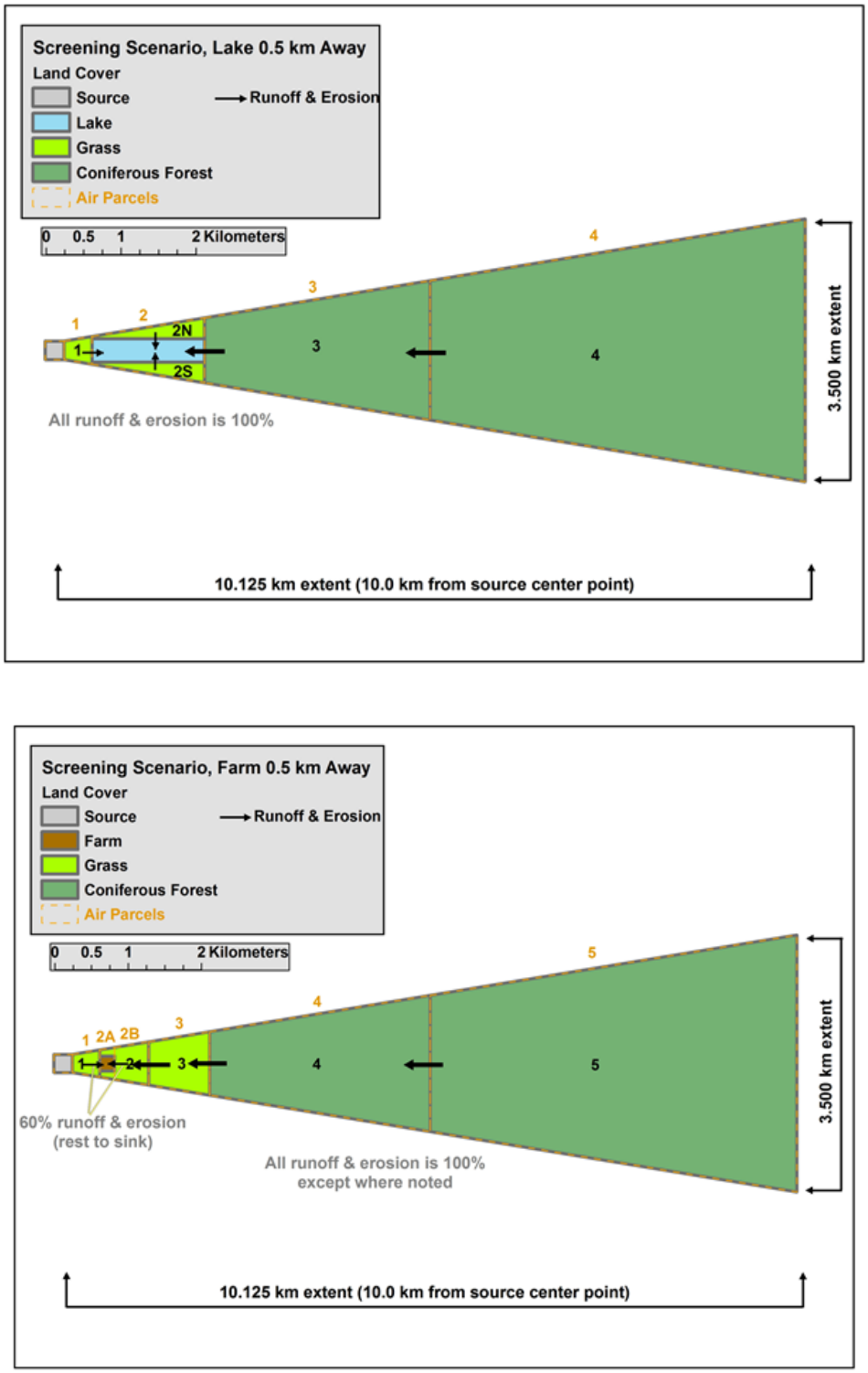
The hypothetical environmental settings are the same as used in the human health risk screen. The lake-centric setting (top panel of Figure 4-2) is used to assess fish and other biota in surface water and sediment. The nonfarm (i.e., grass and forest) parcels in the farm-centric setting (bottom panel of Figure 4-2) are used for the environmental risk screen related to soil.[9] Both

[9]The farm itself is not used in the environmental risk screen.

spatial layouts include an emission source on the west side and several modeling compartments extending to 10 km east of the source. The compartments are shown with arbitrary names (e.g., 1, 2, 3) and are modeled with the indicated land-cover properties and runoff patterns. The assessment of aquatic-related endpoints uses modeled concentrations for water, sediment, fish tissue, and benthic invertebrates at a lake close to the facility (see top panel of Figure 4-2). The assessment of soil-related endpoints uses the modeled surface soil concentrations at five distances from the facility, up to 7.5 km (see bottom panel of Figure 4-2), not including the farming parcel.

The Tier 1 environmental modeling scenario was parameterized to include hypothetical environmental conditions that would provide conservatively high PB-HAP concentration estimates. For example, in the Tier 1 scenario, emissions blow from the facility into the narrow wedge depicted for both settings in Figure 4-2 for 3 days per week, or 43 percent of the time—an unusually consistent long-term wind pattern but not unrealistic (e.g., similar to wind direction patterns in Yakima, Washington). Model settings maximize runoff from terrestrial parcels into the hypothetical lake (for aquatic-related assessment), which in turn maximizes the chemical concentrations in the water, sediments, and fish. The lake situated near the facility also would receive relatively high levels of direct air-to-surface wet and dry deposition. Further details of the Tier 1 TRIM.FaTE environmental modeling scenario, including a description of the aquatic food web, are available in Appendix 6 to the Risk Report. EPA's Science Advisory Board reviewed the approach to parameterizing the hypothetical environmental setting, and other aspects of the TRIM-based modeling used to develop screening threshold emission rates, in 2009/2010.

Figure 4-2. TRIM.FaTE Lake-centric (Top) and Farm-centric (Bottom) Surface Layouts for the Tier 1 Screen



Note: For the environmental risk screen, the lake-centric layout is used for fish, surface water, and sediment endpoints, while the grass and forest parcels of the farm-centric layout are used for soil endpoints.

To calculate the environmental screening threshold emission rates, we ran TRIM.FaTE with a standardized emission rate of 1 g/day for each PB-HAP and saved the resulting PB-HAP concentrations in media at receptor locations throughout the hypothetical environment. We then calculated the environmental screening threshold emission rates by multiplying the 1 g/day emission rate by the ratio of ecological benchmark concentrations to modeled media concentrations. This approach is possible because, for any single period and location (all things being held constant), changes in TRIM.FaTE-predicted PB-HAP concentrations are linear with changes in emission rate. Attachment A provides the final Tier 1 environmental screening threshold emission rates.

Two of the six PB-HAPs for which environmental screening threshold emission rates have been developed (POM and dioxins) are chemical groups comprising numerous individual compounds. For example, for POM, emissions reported include various chemicals, such as benz[a]anthracene, 2-methylnaphthalene, and chrysene, and a few nonspecific entries, such as "PAH, total." As explained below, the results for individual compounds in the POM and dioxin groups are summed, using a TEF approach (see Appendix 6 of the Risk Report for additional information) and an exposure equivalency factor (EEF) approach (described below), to provide one POM result in BaP-equivalents and one dioxin result in 2,3,7,8-TCDD-equivalents.

For POM and dioxins, ecological exposure equivalency factors (EcoEEFs) are calculated for surface water, soil, and sediment by dividing the media concentrations predicted by TRIM.FaTE for each chemical by the predicted concentration of the reference (index) chemical for each group. For example, the EcoEEF for chrysene in soil is calculated as the TRIM.FaTE-estimated concentration of chrysene in soil divided by the estimated concentration of BaP in soil at the same location.

Application of EcoEEFs for POMs and dioxins for piscivorous wildlife differs from the approach described above for surface water, soil, and sediment because TRIM.FaTE does not model PB-HAP exposure doses for the representative animal fish-eating wildlife (i.e., mink, American merganser). The exposure doses for each individual chemical (arsenic, cadmium, mercury, each congener of the POM and dioxin groups) are calculated outside of TRIM.FaTE using the TRIM.FaTE-estimated concentrations in fish and using fish ingestion rates and body weights

specific to the mink and merganser. Each chemical's EcoEEF then is calculated as the ratio of its exposure dose to the exposure dose of the index chemical. The wildlife exposure doses vary across chemicals because the relative concentrations of individual chemicals in each food type consumed (e.g., different fish compartments) vary across chemicals relative to the index chemical due to the variation in chemical-specific assimilation efficiencies, among other factors, for a given fish compartment. The wildlife-specific characteristics influencing the types and quantity of aquatic biota consumed are described in Attachment A, Section A.6, including the data used to assess ingestion of chemicals from each dietary component for mink and common mergansers.

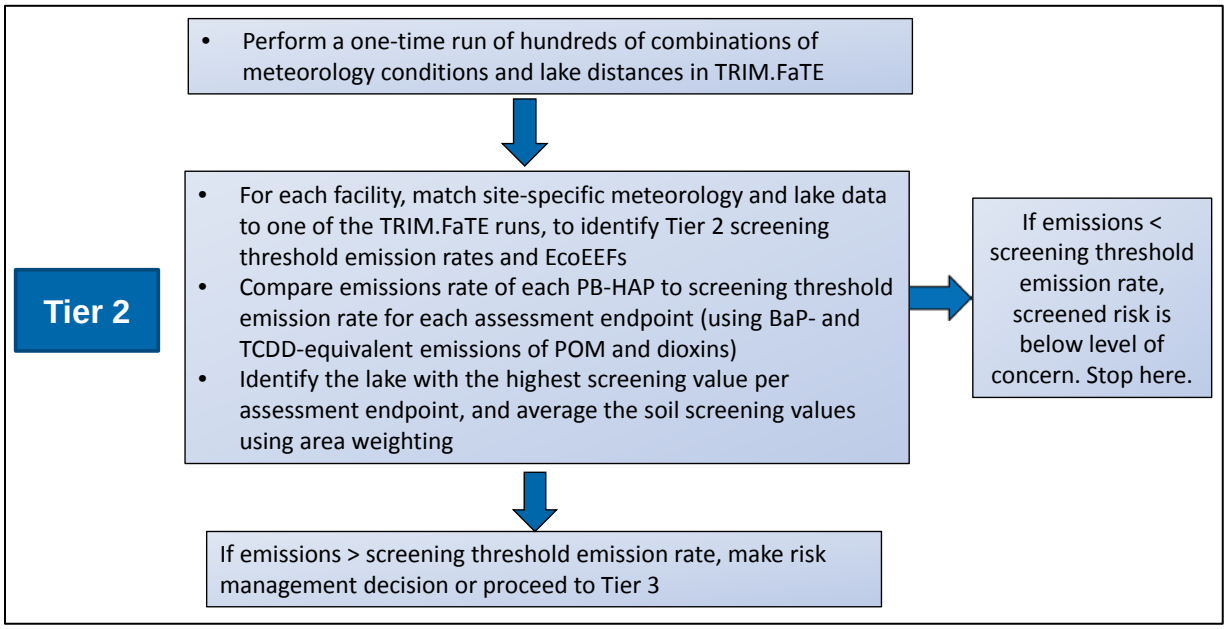
For wildlife-consuming aquatic biota, no adjustments were needed for variation in chemical assimilation efficiency among POM and dioxin/furan congeners, respectively. All toxicity data used to estimate TRVs for POM and dioxin/furan congeners for birds and mammals were based on "administered" doses (the amount of chemical ingested with food, not the amount absorbed into the blood stream). Thus, no adjustments for absorption are needed; differences in absorption among congeners are reflected in the TRVs. That is in contrast to the aquatic food chain modeling, for which congener-specific absorption, metabolic degradation, and elimination rates were estimated for fish and invertebrates and incorporated into the TRIM.FaTE compartment models to estimate bioaccumulation through the aquatic food chains more accurately when calculating EEFs.

The Tier 1 SV for a chemical's emissions from a facility is calculated as Emissions × ecological toxic equivalency factor (EcoTEF) × EcoEEF ÷ Screening Threshold Emission Rate. For each assessment endpoint and benchmark, the SVs are summed for all POM congeners at a facility (into a total BaP-equivalent SV), and the SVs are summed for all dioxin congeners at a facility (into a total 2,3,7,8-TCDD-equivalent SV).

4.1.2 Tier 2 Environmental Risk Screen

After reviewing the results of the Tier 1 environmental risk screen, EPA might choose to evaluate sources with HAP emissions above the Tier 1 screening threshold emission rates (with SVs of 2 or more when rounded to one significant figure). The Tier 2 environmental screening approach, summarized in Figure 4-3, consists of the following steps.

Figure 4-3. Approach for Tier 2 Environmental Risk Screen for PB-HAPs



First, TRIM.FaTE is used to estimate environmental concentrations associated with an emission rate of 1-g/day for 64 combinations of meteorological conditions (see Section 3 of Appendix 6 of the Risk Report for more information). We assess five different distances of the lake from the facility (see Section 3 in Appendix 6 of the Risk Report for a discussion on modeling domain sizes, including modeled lake location values). For the soil endpoints, we use the Tier 1 farm-centric layout (locations of soil endpoints are unchanged from Tier 1). All other attributes of the TRIM.FaTE runs for the Tier 2 environmental risk screen are identical to those of Tier 1. The Tier 2 TRIM.FaTE runs are performed once, for use in both the human health and ecological risk screening.

Second, for aquatic-related endpoints, each lake near the facility that meets inclusion criteria is identified by its location relative to the facility and by its surface area (see Section 3 of Appendix 6 of the Risk Report for more information). Section 3 of Appendix 6 of the Risk Report also describes the lake database used to identify appropriate lakes. Several lake-selection criteria used in the human health assessment (not swampy or covered in algae, not closed to public access) are not used as criteria for the environmental assessments. Facility-specific meteorology and lake location data are used to identify which combination of meteorological conditions and lake distance is most similar to that of the facility and each individual lake.

Third, for soil endpoints, facility-specific meteorological data are used to identify which combination of meteorological conditions is most similar to that of the facility, and the corresponding chemical-specific environmental screening threshold emission rates and EcoEEFs are identified for each of the five soil locations.

The second-pass Tier 2 SV is based on additional adjustments for how frequently the wind blows toward the lake or soil locations of interest (compared with Tier 1) and for the relationship between site-specific air mixing height.

The third-pass Tier 2 screen accounts for multifacility chemical loading to lakes (e.g., two facilities from the same source category located within 100 km of each other, each contributes chemical mass to the same lake). For each ecological assessment endpoint and benchmark effects level, the SVs are summed for all POM congeners (into a total BaP-equivalent SV) and the SVs are summed for all dioxin congeners (into a total 2,3,7,8-TCDD-equivalent SV).

For each facility, for each assessment endpoint, benchmark, and PB-HAP (with POM and dioxins summed to BaP- and 2,3,7,8-TCDD equivalents, respectively), we identify the lake with the largest Tier 2 SV—the final Tier 2 SV for that facility, endpoint, benchmark, and PB-HAP.

For each facility, endpoint, benchmark, and PB-HAP (with POM and dioxins summed to BaP- and 2,3,7,8-TCDD equivalents, respectively), we average the Tier 2 SVs across all 40 soil locations (8 directional octants × 5 soil distances). Each estimate is area weighted (points distant from the source represent larger soil areas than nearer points in the radial domain) to obtain an area-weighted average soil SV.

If Tier 2 SVs are less than or equal to 1, after rounding to one significant figure, the facility screens out (the emissions are below environmental screening threshold emission rates), and it is typically not evaluated further. If Tier 2 SVs after rounding to one significant figure are greater than 1, the facility might be evaluated further with additional site-specific data and modeling refinements as described for Tier 3.

4.1.3 Tier 3 Exposure Assessment

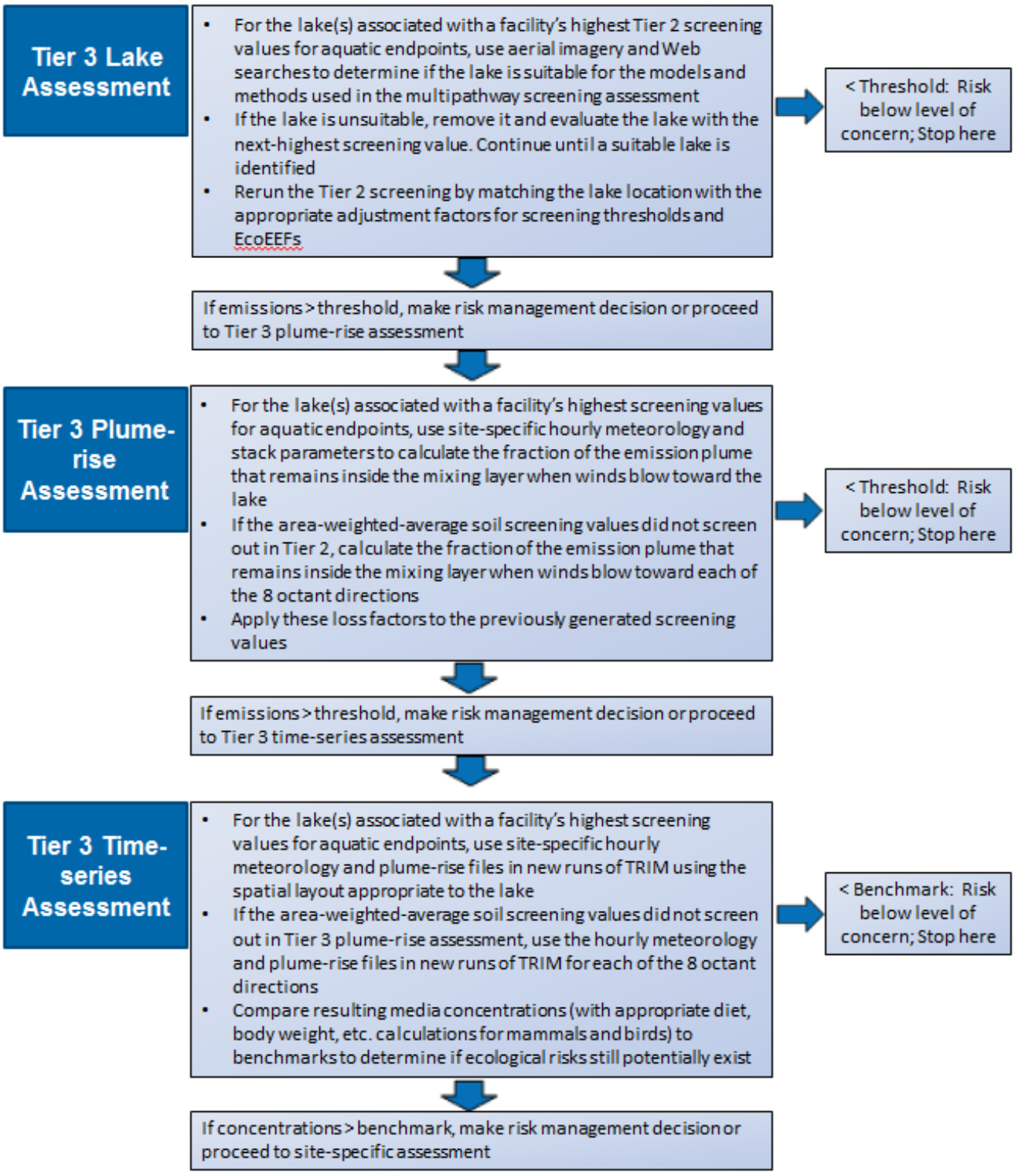
A Tier 3 screen can be conducted on facilities that do not screen out in Tier 2. The Tier 3 screening approach consists of three individual assessments (shown in Figure 4-4 and described

in more detail in Section 4 of Appendix 6 of the Risk Report) that further refine the screening scenario (beyond Tier 2) based on additional site-specific data and evaluations. The refinements are conducted in a step-wise fashion, and all three might not always be needed (e.g., a facility might screen out after the first refinement in Tier 3).

In the first step of the Tier 3 assessment (the lake assessment), we investigate further the lakes assessed in Tier 2 (the lake at each facility associated with the largest aquatic-related SVs per PB-HAP). If we modify, add, or remove any lakes from the assessment, we also modify the lake database and rerun the Tier 2 assessment (e.g., identify a new, more appropriate lake for assessment). If SVs still exceed 1, in the second step of the Tier 3 assessment (i.e., the plume-rise assessment), we estimate how often the chemical plume rises above the mixing layer and, therefore, disperses out of the modeling domain (no ground-level exposures). Finally, if SVs still exceed 1, in the third step of the Tier 3 assessment (the time-series-meteorology assessment), we conduct new runs of TRIM.FaTE and the Multimedia Ingestion Risk Calculator with time-series data for meteorology and plume rise. This last set of SVs typically is smaller than those produced by the Tier 3 plume-rise assessment.

Information about the number and proportion of facilities in a source category exceeding the environmental screening threshold emission rates (SVs >1), proportion and absolute area over which soil-based screening threshold emission rates are exceeded, and magnitude of those SVs help EPA decide whether adverse ecological effects are potentially widespread and significant. If a facility exceeds Tier 3 screening threshold emission rates, it could be further evaluated to consider the degree to which the emission rates are exceeded, which endpoints and effect levels are exceeded, the geographic setting (e.g., proximity to protected areas and resources), and the total area exceeding the screening threshold emission rates. If, after additional refinement, the facility still exceeds the screening threshold emission rates, a site-specific assessment could be conducted. The site-specific assessment uses model parameter values and scenario designs intended to better represent the modeled facility—aspects such as local terrain (influencing runoff and erosion patterns), watersheds, actual lake boundaries and water retention rates, soil types, and land cover. Site-specific environmental assessments are not presented in this report.

Figure 4-4. Approach for Tier 3 Environmental Risk Screen for PB-HAPs



4.2 ENVIRONMENTAL RISK SCREEN FOR LEAD AND ACID GASES

4.2.1 Lead

The level of the primary and secondary NAAQS for lead, 0.15 µg/m^3, is intended to protect humans from both excess inhalation and ingestion exposures and, secondarily, to protect the environment from adverse effects (U.S. EPA 2016a).[10] Therefore, RTR multipathway assessments evaluate modeled air concentrations of lead compounds against the NAAQS level directly, without additional fate, transport, and exposure modeling. We compare the AERMOD-modeled air concentrations of lead for each individual emission point for each facility in the source category to the 0.15-µg/m^3 level of the secondary NAAQS for lead. The environmental risk screen for lead consists of this single tier. We consider air concentrations below the level of the secondary lead NAAQS unlikely to cause adverse environmental effects.

4.2.2 Acid Gases

We needed a separate approach for exposure modeling for acid gases because TRIM.FaTE does not explicitly model gas-phase dispersion in ambient air around a source and the estimated ground-level ambient concentrations are uncertain, particularly with respect to relatively fine spatial resolution. Based on the nature of the GEAE selected for acid gases and the mode of exposure for these chemicals (direct contact of plant foliage with acid gases present in ambient air), EPA used AERMOD (an air dispersion model), which is the same model used in the human inhalation risk assessment. The typical defaults for AERMOD are to model 13 concentric rings at various distances from the facility with 16 concentration data points equally spaced across each ring for 208 modeled air concentrations.

Relative to the PB-HAPs exposure estimates, the acid gas exposure estimates are less health protective and more facility specific, primarily due to the characteristics of the acid gas analysis:

- Only one environmental medium assessed (air only in contrast to air, soil, and water)
- Direct contact of the chemical in air with plant foliage, which eliminates the need for multimedia modeling of chemical transfers

[10] The secondary lead NAAQS (U.S. EPA 2016a) is a reasonable measure of determining whether an adverse environmental effect is present because it was established considering "effects on soils, water, crops, vegetation, man-made materials, animals, wildlife, weather, visibility and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being."

- Refined air modeling approach, using hourly meteorology data and multiple emission sources.

The environmental risk screen for acid gases includes a single tier. Screening compares modeled ambient air concentrations of each acid gas, HF and HCl, to the air concentration benchmarks for terrestrial plants. For HF, we assume that all HF emitted by facilities would remain in the atmosphere in vapor phase; none would be adsorbed to particles that also might be emitted by the facility. That assumption could substantially overestimate the HF concentrations to which terrestrial plant foliage might be exposed.

Because modeled air concentrations are compared directly to the acid-gas ecological benchmarks expressed as air concentrations, emission-based screening thresholds are not calculated for acid gases as they are in the environmental risk screen for PB-HAPs.

For HF, the exposure durations for the available benchmarks (Section 3.2.2) do not correspond precisely to the exposure averaging times of the HEM/AERMOD results. The benchmarks for HF are equivalent to the 90-day Washington State criterion (0.5 µg HF/m^3) and the 90-day Canadian Ambient Air Quality Objective for the growing season (0.4 µg HF/m^3). Although some risk assessors would consider those two values to be essentially equivalent, and might propose using the more health-protective (lower) value, others might consider the 20-percent difference between the two values an important distinction and propose using a value applied within the United States. We therefore have retained both benchmarks for now. The exposure averaging time output from HEM/AERMOD for chronic scenarios is an annual average. Given that 90 days (the approximate growing season when foliage is present and exposed to air) is the longest duration for which HF criteria are available for jurisdictions within North America, the 90-day criteria are considered the best available benchmarks for direct comparison to annual average concentrations from HEM/AERMOD.

For HCl, our calculations to estimate a chronic benchmark for terrestrial plants expressed as air concentrations are the same as described in Appendix K of the 2009 SAB report (U.S. EPA 2009). Specifically, as summarized on page 3-23 of that report:

> *"We extrapolated the LOEL and LOAEL exposures to 1-hour equivalent concentrations of 0.5 and 1 mg/m^3, respectively using the common application of Haber's law, as*

> *modified by Ten Berge et al. [1986]. Lacking long-term study data, we applied an additional uncertainty factor of 10 to extrapolate the lower of the two acute thresholds (0.5 mg/m^3) from a 1-hour to a 1-year exposure threshold of 0.05 mg/m^3."*

Therefore, in our environmental risk screen, we compare the annual average HEM/AERMOD concentrations to the HCl benchmark of 0.05 mg/m^3 (50 µg/m^3).

5 Environmental Risk Characterization/Screening Results

In this section, we discuss the outputs and analyses generated as part of the environmental risk screen.

5.1 ENVIRONMENTAL RISK SCREEN METRICS FOR PB-HAPS

5.1.1 Tier 1

The modeling domain for Tier 1 consists of a health-protective set of conditions (see Section 4.1.1). The modeled area for Tier 1 does not fully extend around the facility but, rather, is a single, downwind wedge. The wedge includes point locations (centroids for modeled surface soil compartments) for estimating chemical concentrations in untilled surface soils at five locations (at 312 m, 850 m, 1500 m, 3500 m, and 7500 m from the facility; measured from the facility center point to the center point of the parcel). The wedge contains one freshwater lake or pond at approximately 500 m from the facility (with one compartment each for surface water, sediment, benthic invertebrates, and five categories of fish). Therefore, for water, sediment, and fish tissues, the Tier 1 environmental risk screen for a PB-HAP is based on the TRIM.FaTE-modeled chemical concentration in the lake water-column compartment, in the lake sediment compartment, and in each of the six aquatic animal compartments in the one lake, respectively. For surface soils, the Tier 1 environmental risk screen is based on the location with the highest soil concentration. Use of the highest soil concentration for Tier 1 is consistent with Tier 1 being a health-protective scenario.

The Tier 1 environmental risk screen for a PB-HAP is performed with a computational tool that automates steps from assembling emissions data to presenting results in preformatted spreadsheet tables. The tool calculates the Tier 1 SV, which is the emissions of the PB-HAP from a facility (adjusted to the BaP and 2,3,7,8-TCDD index chemicals for POM and dioxins, respectively) divided by the environmental screening threshold emission rate for that PB-HAP.

An SV less than or equal to 1 (after rounding to 1 significant figure) indicates that the facility screened out; an SV greater than 1 indicates the potential for adverse environmental effects cannot be ruled out. Outputs provided by this tool include the Tier 1 SVs, the number of facilities that did not screen out in the Tier 1 screen, and the highest Tier 1 SV. The SVs also can be presented for each facility, PB-HAP, assessment endpoint, and benchmark effects level and can be summarized across all facilities. See Table 5-1 for a summary of PB-HAP environmental risk screen metrics.

Facilities not passing the Tier 1 screen for any PB-HAP, assessment endpoint, or benchmark effects level are evaluated in Tier 2. Facilities that screened out of the Tier 1 screen are not evaluated further for potential environmental effects.

5.1.2 Tiers 2 and 3

For Tier 2, TRIM.FaTE was run once with hundreds of combinations of meteorological conditions (from 823 meteorological stations) and lake locations [five distances in eight octants (wedges) that together fully surround the source]. For each combination, environmental screening threshold emission rates are calculated for each PB-HAP and assessment endpoint. For a Tier 2 assessment for a given source category, each facility not ruled out by Tier 1 can be evaluated. First, a computational tool identifies which combination of meteorological conditions and lake location best matches the facility. The SVs for the facility equal the ratio of the facility's emissions to the environmental screening threshold emission rate for that combination from the Tier 2 TRIM.FaTE runs. Tier 2 soil calculations use the same five facility-to-soil distances as in Tier 1, but in all eight directional octants.

As in Tier 1, the Tier 2 environmental risk screen for PB-HAPs uses a computational tool that automates the steps described above. The Tier 2 environmental SVs are tabulated by facility, PB-HAP group, and assessment endpoint. For aquatic assessment endpoints, the final Tier 2 SVs are for the lake with the highest chemical concentrations (a protective setting). For soil-based assessment endpoints, the final Tier 2 SV is the average of the area-weighted SVs across all 40 surface soil compartments (5 distances in each of 8 octants). Facility-level results include the percentage of the total modeled soil area not passing the screen for each facility and each PB-

HAP. The tool also identifies lake names, sizes (acres), and locations. Table 5-1 summarizes the PB-HAP environmental risk screen metrics.

Table 5-1. Summary of PB-HAP Environmental Risk Screen Metrics

Tier	Modeling Domain	Source Category Results
Tier 1	Soils and Lake	• Tier 1 emission screening value (SV) for each facility for each combination of PB-HAP, assessment endpoint, and benchmark effects level. [For soils, the SV is based on the highest concentration from among the five soil locations.] • Number of facilities that do not screen out (for each combination of PB-HAP, assessment endpoint, and benchmark effects level; associated with SVs of 2 or more). • Highest Tier 1 screening ratio for the category (for each combination of PB-HAP, assessment endpoint, and benchmark effects level).
Tier 2	Soils	• Tier 2 SVs for each facility for each combination of PB-HAP, assessment endpoint, and benchmark effects level. [Overall SV is based on the area-weighted average for all 40 calculated soil concentrations within a 7.5-km radius.] • Number of facilities that do not screen out (for each combination of PB-HAP, assessment endpoint, and benchmark effects level; associated with SVs of 2 or more). • Highest Tier 2 SV for the category (for each combination of PB-HAP, assessment endpoint, and benchmark effects level). • Percentage of the total soil area with an SV of 2 or more for each facility (if at all).
	Lakes	• Tier 2 SVs for each facility for each combination of PB-HAP, assessment endpoint, and benchmark effects. [SV is based on the highest lake concentrations, after accounting for possible multifacility chemical loading.] • Number of facilities that do not screen out (for each combination of PB-HAP, assessment endpoint, and benchmark effects level; associated with SVs of 2 or more). • Highest Tier 2 SV for the category (for each combination of PB-HAP, assessment endpoint, and benchmark effects level). • For each modeled lake: lake name, lake surface area (acres), facility-to-lake distance, and latitude/longitude of the lake.
Tier 3	Same as Tier 2.	

Facilities that screen out of the Tier 2 screen for all assessment endpoints are not evaluated further. Facilities that do not screen out might be evaluated further with additional site-specific data and modeling refinements as described for Tier 3 (see Section 4.1.3).

The Tier 3 screening approach consists of three individual assessments (shown in Figure 4-4 and described in more detail in Section 4 of Appendix 6 of the Risk Report) that further refine the screening scenario (beyond Tier 2) based on additional site-specific data and evaluations. The refinements are conducted in a step-wise fashion, and all three might not always be needed (e.g.,

a facility might screen out after the first refinement in Tier 3). The three tier 3 assessments include the lake assessment, plume-rise assessment, and time-series meteorological assessment. The environmental risk screen metrics for Tier 3 are the same as for Tier 2.

As with the multipathway human health risk assessment, a site-specific assessment could be conducted if the Tier 3 screening results indicate a potential for adverse environmental effects. The site-specific assessment uses model parameter values and scenario designs intended to better represent the modeled facility—aspects such as local terrain (influencing runoff and erosion patterns), watersheds, actual lake boundaries and water retention rates, soil types, and land cover. This report does not present site-specific assessments.

5.2 ENVIRONMENTAL RISK SCREEN METRICS FOR ACID GASES

The HEM/AERMOD domain extends 50 km from the center of the facility. The HEM/AERMOD approach includes 13 concentric rings at various distances (out to 50 km) from the facility with 16 locations, each separated by 22.5 degrees on each ring. Therefore, the HEM/AERMOD model generates 208 point estimates of acid gas concentration.

Although an SV could be calculated for all 208 point estimates of air concentration, an SV for a single data point would have little meaning in the context of assessing "significant and widespread" effects over "broad areas" as specified in the CAA definition of "adverse environmental effects." For example, the area of a parcel close to the facility is only a few acres in size. Therefore, in the context of the statutory definition of adverse environmental effects, we use the metrics shown in Table 5-2 to identify effects that are significant and widespread (covering broad areas).

For acid gases, we report the following:

- If individual locations with an SV of 2 or more are present around a facility, we indicate the percentage of the modeling area that had an SV of 2 or more.
- If all locations (i.e., 208 modeled locations) for which HEM/AERMOD estimated acid gas concentrations had SVs less than 2, we indicate that all estimated concentrations around the facility are below the ecological benchmarks for acid gases in air.

Table 5-2. Summary of Acid Gas Environmental Risk Screen Metrics

	Metric	Description
Facility	Modeled area exceeding the ecological benchmarks, in acres and km^2	• All 208 modeled acid gas concentrations in air are compared with the ecological benchmarks (concentration/benchmark = screening value). Those SVs of 2 or more do not screen out. • The total modeled area with an SV of 2 or more.
	Percentage of the modeled area exceeding the ecological benchmarks	• The total modeled area with an SV of 2 or more divided by the total area of the 50-km (radius) modeling domain.
	Area-weighted average SV	• The area-weighted average concentration of all 208 modeled data points divided by the ecological benchmark.
Source Category	Number of facilities with exceedances	• The number of facilities in the category that did not screen out according to area-weighted averaging.

5.3 ENVIRONMENTAL RISK SCREEN METRICS FOR LEAD

For lead compounds, we currently have no ability to calculate concentrations in multiple environmental media using the TRIM.FaTE model. Therefore, to evaluate the potential for adverse environmental effects from lead compounds, we compare the HEM/AERMOD air concentrations of lead around each facility in the source category to the 0.15-µg/m^3 level of the secondary NAAQS for lead (U.S. EPA 2016a). The environmental risk screen for lead consists of one tier. We consider values below the level of the secondary lead NAAQS unlikely to cause adverse environmental effects.

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Attachment A. Environmental Effects Assessment

CONTENTS

TABLES A-2

ACRONYMS A-3

A.1 Introduction A-5
A.2 Ecological Benchmarks A-6
A.2.1 Population-level and Community-level Benchmarks A-7
A.2.2 Ecological Benchmarks for Persistent and Bioaccumulative Hazardous Air Pollutants (PB-HAPs) A-13
A.2.3 Hydrogen Fluoride (HF) Air Benchmarks for Terrestrial Plants A-28
A.3 Wildlife Toxicity Reference Values A-41
A.3.1 Derivation of TRVs for Piscivorous Wildlife in RTR Assessment A-42
A.3.2 Chemical-specific Wildlife TRVs for PB-HAPs A-44
A.4 Derivation of Ecological TEFs for POM and Dioxin Benchmarks A-50
A.4.1 TEFs for POM for Surface Water, Sediments, and Soils A-50
A.4.2 TEFs for Dioxins for Surface Water, Sediments, and Soils A-62
A.5 TEFs for Wildlife TRVs A-62
A.5.1 TEFs for Wildlife for POM A-62
A.5.2 TEFs for Wildlife for Dioxins A-66
A.6 Piscivorous Wildlife Exposure Factors A-66
A.6.1 Mink Exposure Factor Values and Assumed Diet A-68
A.6.2 Merganser Exposure Factor Values and Assumed Diet A-70
A.7 Derivation of Bioaccumulation Factors for Arsenic A-71
A.7.1 Differences between Freshwater and Marine Fish A-72
A.7.2 BAFs for Arsenic in Freshwater Fish A-74
A.7.3 BSAFs for Arsenic in Freshwater Benthic Invertebrates and Fish A-80
A.8 Environmental Screening Threshold Emission Rates A-81
A.9 References A-86

TABLES

Table A-1. Generic Ecological Assessment Endpoints Not Used in the Nationwide RTR Environmental Risk Screen A-12
Table A-2. Ecological Freshwater Benchmarks for Dissolved Inorganic Arsenic (µg/L) A-15
Table A-3. Sediment Screening Benchmarks Identified in ORNL RAIS Database A-18
Table A-4. Ecological Soil Benchmarks for Inorganic Arsenic, CAS No. 7440-38-2 ..... A-23
Table A-5. Screening Soil Benchmarks for Cadmium, Thresholds for Effect A-25
Table A-6. Soil Screeing Benchmarks for Methyl Mercury A-26
Table A-7. Soil Ecological Screening Levels for Methyl Mercury from Los Alamos National Laboratory A-27
Table A-8. Overview of Three Approaches to Hydrogen Fluoride (HF) Environmental Standards (Hill 1969) A-30
Table A-9. Governmental Air Criteria for Hydrogen Fluoride (HF) to Protect Plants A-32
Table A-10. Adverse Effects in Terrestrial Plants Following Short-term Exposures to HF A-35
Table A-11. Adverse Effects in Terrestrial Plants Following Longer-term Exposure to HF A-37
Table A-12. Toxicity Equivalency Factors (TEFs) for Surface Waters, Soils, Sediments, and Mammalian Wildlife—POM Compounds Relative to BaP A-51
Table A-13. Toxicity Equivalency Factors (TEFs) for Surface Waters, Soils, Sediments, and Mammalian and Avian Wildlife—Dioxins Relative to 2,3,7,8-TCDD A-63
Table A-14. TEFs for Oral Exposures of Mammalian Wildlife—POM Congeners Relative to BaP A-64
Table A-15. Mink Exposure Factor Values A-68
Table A-16. Mink Diet Assumptions[a] A-69
Table A-17. Common Merganser Exposure Factor Values A-70
Table A-18. Common Merganser Diet Assumptions[a] A-70
Table A-19. Marine and Freshwater Fish Tissue Concentrations A-73
Table A-20. BAF/BCF Values for Freshwater Fish Exposed to Different Water Concentrations of Arsenic A-77
Table A-21. Tier 1 Environmental Screening Threshold Emission Rates (ESTER) for each PB-HAP and each Benchmark Assessed in the Environmental Risk Screen A-82

ACRONYMS

AE	assimilation efficiency
ALC	aquatic life criteria
ARCS	Assessment and Remediation of Contaminated Sediments
As	arsenic
BaF	benzo[a]fluoranthene
BAF	bioaccumulation factor
BaP	benzo[a]pyrene
BeP	benzo[e]pyrene
BCF	bioconcentration factor
BbF	benzo[b]fluoranthene
BjF	benzo[j]fluoranthene
BMD	benchmark dose
BSAF	biota-sediment accumulation factor
BTAG	Biological Technical Assistance Group
bw	body weight
CAS	Chemical Abstracts Service
CCME	Canadian Council of Ministers of the Environment
Cd	cadmium
DaP	dibenzo[a,i]pyrene
DMA	dimethylarsenic acid
DOE	Department of Energy
dw	dry weight
ECxx	effective concentration (xx-percent response)
Eco-TEFs	ecological toxicity equivalency factors
EPA	U.S. Environmental Protection Agency
EqP	equilibrium partitioning
ESL	ecological screening level
ESTER	environmental screening threshold emission rate
F	fluorine
FIR	food ingestion rate
FMR	free-living metabolic rate
GE	gross energy
GEAE	generic ecological assessment endpoint
GLNPO	Great Lakes National Program Office
GLWQI	Great Lakes Water Quality Initiative
GMAT	geometric mean
GMATC	geometric mean maximum acceptable toxic concentration
HAP	hazardous air pollutant
HCl	hydrogen chloride
HF	hydrogen fluoride
Hg	mercury
Hg++	divalent mercury
HMW	high-molecular-weight
ISQG	interim sediment quality guideline
Koc	organic carbon partitioning factor
Kow	octanol-water partitioning coefficient
LANL	Los Alamos National Laboratory
LC50	lethal concentration for 50 percent of animals tested
LEL	lowest-effect level
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed-effect concentration
LOEL	lowest-observed-effect level
MACT	maximum achievable control technology
MATC	maximum allowable toxicant concentration
MATL	maximum allowable toxicant level
ME	metabolizable energy
MeHg	methyl mercury
MOA	mode of action
MMA	monomethylarsonic acid
NAWQC	National Ambient Water Quality Criteria
NEC	no-effect concentration
NEL	no-effect level
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NOEL	no-observed-effect level
OEHHA	California Office of Environment and Health Hazard Assessment
OME	Ontario Ministry for the Environment

ORNL	Oak Ridge National Laboratory
OSWER	Office of Solid Waste and Emergency Response
OW	Office of Water
PAH	polycyclic aromatic hydrocarbon
PB-HAP	persistent bioaccumulative HAP
PCDD	polychlorinated dibenzo-p-dioxin
PCDF	polychlorinated dibenzofurans
PEL	probable-effect level
PHE	phenanthrene
PMC	photomodification
POM	polycyclic organic matter
PSC	photosensitization
PYR	pyrene
QSAR	quantitative structure-activity relationship
RAIS	Risk Assessment Information System
RCRA	Resource Conservation and Recovery Act
RfD	reference dose
RTR	Risk Technology and Review
SAV	secondary acute value
SCV	secondary chronic value
SESL	soil ecological screening level
SQB	sediment quality benchmark
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
TCEQ	Texas Commission on Environmental Quality
TEC	threshold-effects concentration
TEF	toxicity equivalency factor
TEL	threshold-effect level
TL	trophic level
TMA	trimethylarsenic
TOC	total organic carbon
TPY	tons per year
TRIM.FaTE	Total Risk Integrated Methodology, Environmental Fate, Transport, and Ecological Exposure
TRV	toxicity reference value
UF	uncertainty factor
WEFH	Wildlife Exposure Factors Handbook
WHO	World Health Organization
WQB	water quality benchmark
WQG	water quality guideline
ww	wet weight
YOY	young of the year

A.1 Introduction

Pursuant to the Clean Air Act, the U.S. Environmental Protection Agency (EPA) developed both human health and environmental risk screens under its Risk Technology and Review (RTR) program. The program assesses risk remaining (i.e., residual risk) from emissions of hazardous air pollutants (HAPs) following the implementation of maximum achievable control technology (MACT) standards for emission sources. This attachment provides materials supporting EPA's approach to the effects assessment, as described in Section 3 of the main report.

EPA developed the environmental risk screen to examine the potential for adverse environmental effects as required under Section 112(f)(2)(A) of the Clean Air Act. Section 112(a)(7) of the Act defines "adverse environmental effect" as "any significant and widespread adverse effect, which may reasonably be anticipated, to wildlife, aquatic life, or other natural resources, including adverse impacts on populations of endangered or threatened species or significant degradation of environmental quality over broad areas."

The environmental risk screen includes eight HAPs, which we refer to as "environmental HAPs": six persistent bioaccumulative HAPs (PB-HAPs) and two acid gases. The six PB-HAPs are arsenic; cadmium; mercury (both inorganic mercury and methyl mercury); dioxins/furans (referred to herein as dioxins); polycyclic organic matter (POM); and lead. The two acid gases are hydrogen chloride (HCl) and hydrogen fluoride (HF). The remainder of this attachment is organized in eight sections.

Section A.2. We first provide supplemental information for the derivation of ecological benchmarks for surface waters, sediment, surface soils, and air. Benchmarks are expressed as the concentrations of individual chemicals in the environmental media listed above. The benchmarks are compared with exposure estimates to screen for risks to generic ecological assessment endpoints (GEAEs).

Section A.3. For POM and dioxins, we discuss derivation of toxicity equivalency factors (TEFs) for each group relative to their index chemicals, benzo[a]pyrene (BaP) and 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), respectively, for surface waters, sediment, and surface soils.

Section A.4. We describe the derivation of toxicity reference values (TRVs) for two wildlife species—mink and common (American) merganser—intended to represent fish-eating mammals and fish-eating birds, respectively. In contrast to ecological benchmarks, which are expressed as concentrations of chemicals in environmental media, TRVs are expressed as ingested doses in milligrams chemical ingested per kilogram wildlife body weight per day. TRVs are calculated for mink and American merganser based on key toxicity studies in the literature.

Section A.5. We discuss derivation of ecological toxicity equivalency factors (Eco-TEFs) for POM and dioxins relative to their index chemicals, BaP and TCDD, respectively, for TRVs for birds and mammals.

Section A.6. Data on dietary habits and values for exposures factors (e.g., ingestion rates, body weight) are provided for mink and American merganser. Those data are used to estimate exposure doses for wildlife from estimates of chemical concentrations in smaller and larger fish made with the Total Risk Integrated Methodology, Environmental Fate, Transport, and Ecological Exposure module (TRIM.FaTE).

Section A.7. Empirical data by which a bioaccumulation factor (BAF) was derived for arsenic in the water column and in benthic sediments are presented.

Section A.8. Screening emission rate thresholds, expressed as tons of chemical per year (TPY) released by a facility, are presented for each chemical, assessment endpoint, and environmental medium evaluated in the environmental risk screen.

Section A.9. This attachment concludes with a list of the references cited.

A.2 Ecological Benchmarks

Benchmark concentrations are derived for several GEAEs (U.S. EPA 2003a, 2016a) that are relevant to the different environmental media. GEAEs can be defined for individual organisms, specified populations of species, biological communities or assemblages, and ecosystems. Effects at the population or community level usually are inferred from scientific measurement of adverse effects at the individual or population level, respectively. Table 2-2 in the main report presents a list of GEAEs used in the RTR screen. We assess both populations (e.g., mink,

merganser) and communities (e.g., sediment benthic invertebrates, soil communities, water column communities) for the RTR assessment.

In this section, we provide supplemental information for the ecological benchmarks described in Section 3 of the main report. Section A.2.1 describes differences between "population-level" and "community-level" benchmarks in more detail than in the main report. Section A.2.2 provides supplemental information supporting the derivation of ecological benchmarks for PB-HAPs. Section A.2.3 provides background information and data from original studies used to derive the air concentration benchmarks for plants exposed to HF in air. Derivation of air concentration benchmarks for plants exposed to HCl was presented in materials prepared for the previous 2009 EPA Science Advisory Board review of the RTR assessment risk screens (U.S. EPA 2009a) and is not repeated here.

A.2.1 Population-level and Community-level Benchmarks

For readers familiar with EPA human health risk assessment, for which EPA identifies benchmarks and TRVs intended to protect individual humans from adverse health effects (e.g., noncancer effects) or to ensure risks (e.g., of cancer) are no higher than 1-in-ten thousand to 1-in-one million, the basis of ecological benchmarks and TRVs can be confusing. Federal risk assessments for endangered or threatened species might be conducted with individual-level TRVs, as is done for humans.

For nonthreatened wildlife, risks of losing local populations of economically important, "ecological indicator" species or most "exposed species" often are assessed (Section A.2.1.1). For other biota, such as invertebrates in aquatic sediments or in soils, community assemblages generally are assessed for their ability to provide habitat or other ecosystems services (Section A.2.1.2). Three of EPA's twelve GEAEs (USEPA 2003a) were not used because established benchmarks are not available (Section A.2.1.3).

Similar to the situation for human health risk assessment, we prefer to use previously established and peer-reviewed ecological benchmarks and TRVs (Section A.2.1.4). We also considered three effect levels that could assist EPA decision-makers in interpreting the results of RTR environmental risk screens (Section A.2.1.5).

A.2.1.1 Population-level Benchmarks

In general, population-level effects are inferred from available single-species toxicity tests for the assessment species (or the most closely related species as data allow). The results of single-species chronic toxicity tests with animals usually have been reported as NOELs (no-observed-effect levels) and LOELs (lowest-observed-effect levels) for a specified effect. The NOEL and LOEL (NOEC and LOEC where the C stands for "concentration" instead of level) are identified by hypothesis testing. The LOEL is the lowest exposure level at which the test-group response differs from the response of the control group with a probability, *p* (usually <0.05), that the difference is due to chance alone. The NOEL is the highest exposure level at which the test group response does not statistically differ from that of the control group.

For nonhuman biota, "health" usually is assessed at the population level (Biddinger et al. 2008). Therefore, generally only effects that readily can be linked to negative population-level consequences (or higher level impacts such as on communities or ecosystems) have been considered to represent lowest-observed-*adverse*-effect levels (LOAELs) in ecological risk assessments. Four effect categories for individual-level effects are considered closely linked with population-level effects: survival, reproduction, development, and growth (Rodier and Zeeman 1994; U.S. EPA 1998). When using both the statistical and biological definitions of "significant" effects, distinguishing biological significance (e.g., average weight loss of the test group of 10 percent is considered biologically significant) from statistical significance (i.e., less than a 5-percent chance that the difference from the control or reference area is due to chance alone) is important.

For a given species, if different sensitivities are associated with different lifestages, results from tests of the most sensitive lifestage are used to represent the species in chronic exposure scenarios. If some effects occur at lower concentrations than others (e.g., impaired reproductive success compared with growth), the most sensitive effect is used. If multiple studies on the same species' most sensitive lifestage report the same most sensitive effect, the geometric mean of the no-observed-adverse-effect level (NOAEL) values and the geometric mean of the LOAEL values across tests can be used to represent the NOAEL and LOAEL, respectively, for the species and endpoint. Otherwise, well-tested species could be over-represented in criteria or benchmark development (Stephan et al. 1985; U.S. EPA 1999).

Because of costs, fewer exposure levels typically are used in chronic toxicity tests than are used in acute toxicity tests. That has resulted in many reports of tests in which the LOAEL is the lowest exposure level tested or in which the NOAEL is the highest concentration tested (i.e., "unbounded" LOAEL and NOAEL values, respectively). The numeric values for unbounded LOAELs and NOAELs generally have the "<" and ">" signs, respectively, included. Tests in which both a NOAEL and a LOAEL are identified provide "bounded" values amenable to evaluating toxicity to the species used in that test.

A recent trend with the advent of the benchmark dose (BMD) approach is to evaluate the response at all chronic exposure concentrations. The BMD approach now is preferred to establish points of departure for toxicity when deriving reference values protective of human health, provided that available data are adequate to use the approach (U.S. EPA 2012a). Similarly, for ecotoxicity testing, particularly as reported in peer-reviewed journals, the trend is to report several points along the exposure-response curve for sublethal effects of chronic exposures, for example an EC_{05} or EC_{10}, an EC_{20}, EC_{25}, or EC_{30}, as well as an EC_{50}. An EC_{xx} is the "effective concentration" at which a specified effect is observed in xx percent of the test animals.

When EC values are available or can be calculated, and when the lower percent-effect concentrations have not been extrapolated "too far" below the range of observed responses, risk assessors consider an EC_{05} or EC_{10} to be roughly equivalent to historical NOECs or NOAECs in aquatic animal toxicity testing (SETAC 1994, p. 6; Sijm et al. 2002, p. 234). The effect level considered equivalent to LOECs or LOAECs is greater than an EC_{10}, with some risk assessors citing an EC_{20} (Anderson and Norberg-King 1991; Sijm et al. 2002, p. 235) and others indicating that LOAECs can be equivalent to EC_{25} or higher EC values, depending on many factors (e.g., number of animals per exposure group, number of exposure groups, spacing of exposure concentrations or doses) (Suter et al. 2000, 2003). The advantages of using all exposure-response data to fit exposure-response models to estimate low-effect levels instead of using NOAELs and LOAELs determined by hypothesis testing have been discussed in several texts and EPA guidance documents (e.g., Efroymson et al. 1997a,b; Suter 1993; U.S. EPA 1998, 2005a).

Assuming the availability of a robust toxicity test for a species of similar or greater sensitivity than the assessment species, usually a NOAEL (or EC_{05}–EC_{15}) and a LOAEL (or EC_{15}–EC_{25}) can be defined. For environmental screens, some EPA program offices prefer to use a NOAEL-

based benchmark (e.g., Superfund). Other offices have preferred using a GMAT—the geometric mean of the NOAEL and LOAEL, often referred to as a maximum allowable toxicant level (MATL) or concentration (MATC). The MATC is roughly equivalent to a "threshold for effects." The LOAEC often is associated with an effect level (e.g., 20–25 percent) that might not be sustainable for a local population, depending on species, its life history, sample sizes in the toxicity experiment, and other factors (Suter et al. 2000, 2003). Generally, however, the NOAEL and LOAEL are within one order of magnitude of each other in chronic experiments; hence, the utility of calculating the geometric mean between them is limited.

A.2.1.2 Community-level Benchmarks

Usually, ecological communities are valued by humans for the services they provide to humans, to wildlife, to valued species, to landscapes, or to functioning of ecosystems in general (Daily 1997; NRC 2004). For example, soil invertebrate communities are needed to recycle nutrients and to aerate soils. Measureable attributes of a soil invertebrate community that might influence its provision of those services include the presence and abundance of one (or more) key organism(s) (e.g., earthworms) or a diversity of organisms. Benthic (sediment-dwelling) invertebrates in lakes and rivers are important for recycling detritus and in providing food for fish communities.

Protection of ecosystem services provided by ecological communities usually requires an adequate number, abundance, and diversity of different species present to perform key ecological functions despite natural variation in local conditions (e.g., weather). For example, soil invertebrate communities generally require earthworms for soil aeration and conditioning to support plant life adequately; however, a diversity of other soil invertebrates assist. Benthic communities often require invertebrates that graze on algae or detritus to support higher tropic levels. To support fisheries, surface waters require a diversity of potential prey species, including smaller fish (e.g., minnows), young-of-year fish, and invertebrates (e.g., aquatic insect larvae such as midge and mayfly larvae).

For most ecological communities to provide an appropriate structure (e.g., tree canopy with understory) and to serve various functions (e.g., as bird habitat, flood protection), not all species in the community are required. In most ecosystems, several species perform similar or overlapping functions, and loss of one does not necessarily mean loss of the ecological service it

provides (this is particularly true of benthic invertebrates and plant communities). Some keystone species, however, are critical to community structure and function. Loss of those (e.g., sea otters consuming sea urchins in kelp beds, blue mussels occupying space in the intertidal zone, wolves feeding on other mammals on the prairies) can profoundly change the presence and abundance of other major species and thus profoundly change the structure of the ecosystem.

For sediments, exposure-effect data for some chemicals directly relate measures of benthic community structure (e.g., related to species diversity and abundance) to the concentration of specific chemicals. For water-column and soil-based communities, on the other hand, exposure-response functions generally are not available for community structure or function. Thus, EPA has used the premise that community structure (and therefore function) is unlikely to be affected if fewer than 5 percent of species (Office of Water [OW], U.S. EPA 1998; Stephan et al. 1985) or 10 percent of species (Solomon and Takacs 2002; Efroymson et al. 1997a,b) in the community might be locally extirpated. The rationale for allowing 5 or 10 percent of species to be affected, and potentially to disappear from a local community, is the concept of ecosystem resiliency, that is, the functional redundancy of groups of species (Solomon and Takacs 2002; van Straalen and van Leeuwen 2002).

Functional redundancy in most ecosystems has evolved owing to natural fluctuations in environmental conditions and has been demonstrated in several experimental multispecies tests (Solomon and Takacs 2002). In general, such experiments suggest that the 5-percent species-protection level does protect ecosystem structure and function against significant changes (Posthuma et al. 2002). Identifying upper percentile species “protection” benchmarks, however, requires testing of many phylogenetically distinct species; therefore, derivation of community-level benchmarks often is precluded for chemicals for which few species have been tested.

A.2.1.3 Assessment Endpoints Not Used in RTR Environmental Screen

Nine GEAEs (U.S. EPA 2003a) used in the RTR environmental screen are listed in Table 2-2 of the main report. We evaluated, but did not use, the remaining three EPA GEAEs for the environmental risk screen (Table A-1):

- Animals exposed to airborne HAPs by inhalation,
- Microbial community in soils, and
- Amphibians and reptiles in their respective habitats.

Table A-1. Generic Ecological Assessment Endpoints Not Used in the Nationwide RTR Environmental Risk Screen

Exposure Media	No.	Assessment Endpoint	Entities	Relevant Attributes	Benchmark
Air	10	Maintain local populations of wildlife and aboveground invertebrates exposed to airborne HAPs via inhalation	Birds, mammals, bees, butterflies, etc.	Individual survival, growth and development; area contaminated	No avian or invertebrate data available
Other	11	Maintain microbial function in soils (e.g., nitrogen fixation, decomposition of detritus to nutrients)	Assemblages of bacteria, fungi	Species diversity; decomposition rate for leaf litter; "soil" oxygen consumption rates; area contaminated	No consensus benchmarks available
	12	Maintain local populations of amphibians and reptiles (aquatic-stage amphibia should be covered by ambient water criteria)	Frogs, salamanders, toads, turtles, lizards	Individual survival, growth and development; area contaminated	No consensus benchmarks available; cold blooded, food ingestion rates substantially lower than for birds and mammals

A.2.1.4 Preferred Sources of Benchmarks

We prefer to use established and peer-reviewed ecological benchmarks when available. Benchmarks for sediments, surface waters, and soils initially were identified using the Oak Ridge National Laboratory (ORNL) Risk Assessment Information System (RAIS) (http://rais.ornl.gov/). The ORNL RAIS database is maintained by the Department of Energy (DOE) for use in its risk assessments at hazardous waste sites. It includes virtually all TRVs and benchmarks developed to date that might be used by federal agencies in the United States and several other countries to assess risks to human health and the environment (ecological receptors). RAIS therefore provides "one-stop shopping" to identify the availability of and values for ecotoxicity benchmarks for chemicals of concern to U.S. regulatory communities.

All screening-level benchmarks available from Suter and Tsao (1996), which was a key source of benchmarks for the Coke Oven MACT Residual Risk Assessment (U.S. EPA 2003b), are included in RAIS, as are the other sources of benchmarks used in that assessment (e.g., U.S. EPA National Ambient Water Quality Criteria [NAWQA], EPA Region 4 values, National Oceanic and Atmospheric Administration benchmarks, Florida Department of Environmental Protection benchmarks). Once we identified ecological benchmarks in RAIS, we obtained the original sources to confirm values. Our most recent query of RAIS was in August 2016, to check for updates and possibly new benchmarks; we found both.

Finally, we established a hierarchy of preferred benchmark sources to enable selection of benchmarks for each environmental HAP for each ecological assessment endpoint. In general, we used EPA sources at a programmatic level (e.g., OW, Superfund Program), if available. If not, we used EPA benchmarks used in regional programs (e.g., region-specific Superfund). If benchmarks were not available at a programmatic or regional level, we used benchmarks developed by other federal agencies (e.g., DOE), state agencies, or Canada.

A.2.1.5 Effect Levels

In our review of existing benchmarks derived by EPA program offices, EPA regions, other agencies, and states, we found that for some environmental media, notably sediments, benchmarks had been established for two or three different effect levels, not just a "threshold for effects." Several physical attributes of sediments can modify the response of biota living in them. These include pH, sediment particle size, interstitial pore size, organic carbon content, acid volatile sulfide, content, sediment depth, and characteristics of benthic organisms (e.g., sizes, method of feeding, depth of burial, mobility). Therefore, over the past several decades, sediment benchmarks often have been defined at three different levels of effect: no-effect level (NEL: low probability of changes in the structure or function of the benthic community); threshold-effect level (TEL: concentrations above threshold might cause adverse effects in structure and function of benthic community); and probable-effect level (PEL: high probability of frank changes in community structure, function, and provision of ecosystem services).

We therefore decided to look for benchmarks that might represent all three effect levels (i.e., NEL, TEL, and PEL) for each exposure medium/GEAE/chemical combination. Only TELs were available for most benchmarks; we included NEL and PEL values, if available, to provide more information to EPA decision-makers who need to consider whether adverse ecological effects are significant and widespread.

A.2.2 Ecological Benchmarks for Persistent and Bioaccumulative Hazardous Air Pollutants (PB-HAPs)

Ecological benchmarks for PB-HAPs are needed for three environmental media: the water column in lakes (Section A.2.2.1), the sediment bed in lakes (Section A.2.2.2), and surface soils in terrestrial environments (Section A.2.2.3).

A.2.2.1 Water-Column Benchmarks

For organisms that live primarily in the water-column of aquatic ecosystems, EPA's NAWQC-ALC (aquatic life criteria) are used as available (Stephan 1985, 2002; U.S. EPA 2002, 2016b). According to Suter and Tsao (1996), the *acute* NAWQC-ALC are considered "upper" screening levels in EPA's Superfund program—which we interpret to mean *probable effect levels* if associated with continuous long-term (chronic) exposures. The *chronic* NAWQC-ALC are considered "lower" screening-level benchmarks in EPA's Superfund program (Suter and Tsao 1996). Given the methods by which both acute and chronic NAWQC-ALC are derived, we interpret the chronic NAWQC-ALC to represent a threshold for adverse effects in aquatic communities (water-column compartment) rather than an NEL.

For chemicals for which available data do not cover the taxonomic groups required to establish NAWQC, EPA's OW established a Tier II approach (not to be confused with the RTR ecological or human health Tier 2 assessment) allowing derivation of a secondary acute value (SAV) and a secondary chronic value (SCV) based on toxicity data for fewer taxonomic groups than the eight specified for NAWQC. The Tier II approach was developed for the Great Lakes Water Quality Initiative (GLWQI) (U.S. EPA 1993a). Depending on the number of taxa for which acute toxicity data are available, a sliding scale of uncertainty factors (UFs) is applied to the lowest acute and chronic toxicity values to estimate the Tier II SAVs and SCVs. EPA's Superfund program adopted the Tier II SAV methodology from the GLWQI, but on occasion varies its approach to calculating SCVs from SAVs when chronic aquatic toxicity data are limited.

For chemicals for which NAWQC-ALC and Tier II secondary values were not available, we turned to benchmarks developed by EPA Regions 3, 4, 5, or 6.

We describe the sources of the TELs and PELs (acute and chronic criteria) for the PB-HAPs below. For arsenic, we present our review of available data in detail to document our approach. For cadmium, mercury (divalent and methyl), POM, and dioxins, we simply present the benchmarks selected based on the preferred hierarchy of sources.

Arsenic (As) Surface Water Column Screening Benchmarks

EPA derived NAWQC-ALC for arsenic (III). No data are available to determine whether the effects of arsenic (III) and (IV) are additive (U.S. EPA 1995a). Therefore, the values are applied

to total dissolved inorganic arsenic. The multiple freshwater benchmarks identified in DOE ORNL RAIS are listed in Table A-2.

Table A-2. Ecological Freshwater Benchmarks for Dissolved Inorganic Arsenic (µg/L)

Name of Benchmark	Arsenic (III)	Arsenic, Inorganic
Canadian WQG Surface Water Screening Benchmark	NA	5
U.S. EPA Region 4 Acute Surface Water Screening Benchmark	360	360
U.S. EPA Region 4 Chronic Surface Water Screening Benchmark	190	190
U.S. EPA NAWQC Acute Criterion	NA	340
U.S. EPA NAWQC Chronic Criterion	NA	150
U.S. EPA OSWER (Superfund) Water Quality Screening Level	NA	190
U.S. EPA Region 5 ESL Surface Water Screening Benchmark	NA	148
U.S. EPA Region 6 FW Surface Water Screening Benchmark	NA	190

Abbreviations and Acronyms: ESL = ecological screening level; FW = freshwater; NAWQC = National Ambient Water Quality Criteria (U.S. EPA, for the protection of aquatic life); OSWER = Office of Solid Waste and Emergency Response (Superfund, U.S. EPA); NA = not available; µg/L = micrograms per liter; WQG = water quality guideline

Source: Department of Energy (DOE) Oak Ridge National Laboratory (ORNL) Risk Assessment Information System (RAIS) Ecological Benchmark Tool. Listed in order of RAIS output. Marine values excluded.

The acute and chronic NAWQC for freshwater aquatic life, 340 and 150 µg/L, respectively, are applicable nationwide. Therefore, they were selected as the PEL and TEL freshwater benchmarks for arsenic (listed in Table 5-1 of the main report).

Most benchmarks identified by RAIS are similar for acute and chronic exposures; an exception is the Canadian water quality guideline (WQG) of 5 µg/L. It was derived from the 50-µg/L arsenic concentration that reduced growth in one algal species by 50 percent (Vocke et al. 1980). That value was multiplied by a safety factor of 0.1 to calculate the Canadian WQG (Canadian Council of Ministers of the Environment [CCME] 1991). In surface waters, many different algal species can provide the same ecological services. Thus, in the field, the loss of a single algal species does not necessarily alter the ecological structure or function of the aquatic community. We therefore considered the Canadian WQG to be too conservative for the RTR assessment.

Cadmium (Cd) Surface Water Column Screening Benchmarks

Cadmium is one chemical for which we found a 2016 revision to the NAWQC in our review of benchmarks in RAIS: chronic criterion (TEL) of 0.72 µg/L and acute criterion (PEL) of 1.8 µg/L dissolved Cd assuming water hardness of 100 mg/L as $CaCO_3$ (in Table 3-1 of the main report;

U.S. EPA 2016c). EPA's NAWQC for the protection of aquatic life for cadmium depend on water hardness (U.S. EPA 2001a).

Divalent Mercury (Hg^{++}) Surface Water Column Screening Benchmarks

For inorganic, divalent mercury (e.g., dissolved mercuric chloride), EPA's NAWQC are 0.77 µg/L for the chronic criterion and 1.4 µg/L for the acute criterion (U.S. EPA 2016b) (listed in Table 3-1 of the main report). The 1995 criteria (U.S. EPA 1995b) were updated by multiplying the criteria by 0.85 to account for the fraction dissolved in water, as per guidance (U.S. EPA 1993b) that was not widely available in 1995.

Methyl Mercury (MeHg) Surface Water Column Screening Benchmarks

Facilities in RTR source categories emit inorganic mercury, which deposits to surface waters and soils, and from soils, runoff and erosion transport it to the lake, where it enters sediments. Although the divalent mercury is methylated primarily in sediments, some net methylation also occurs in surface soils. TRIM.FaTE estimates bioaccumulation of MeHg through the aquatic food chain, predicting concentrations in the various biotic compartments, particularly fish.

EPA's OW decided to publish its NAWQC criteria for MeHg as concentrations in fish rather than as concentrations in water, because measured BAFs for MeHg in surface waters vary substantially across lakes. Thus, we could have compared TRIM.FaTE-estimated concentrations of MeHg in fish with the NAWQC MeHg concentrations in fish. Instead, we chose to identify MeHg concentrations in the water column to serve as benchmarks for the aquatic community.

EPA Region 4 cites Suter and Tsao (1996) as its source for a Tier II SCV (chronic, TEL) of 0.0028 µg/L and Tier II SAV (acute or PEL) of 0.099 µg/L (U.S. EPA 2015) (listed in Table 5-1 of the main report). Suter and Tsao (1996) followed the EPA GLWQI guidance for deriving Tier II SCV and SAV values (U.S. EPA 1995b).

POM—Benzo[a]pyrene (BaP) Surface Water Column Screening Benchmarks

Data available for BaP are insufficient for deriving an EPA NAWQC. BaP is highly lipophilic; thus, toxicity testing for aquatic organisms is difficult because toxicity might not be reached at the limit of solubility. Suter and Tsao (1996) calculated a Tier II SCV and SAV using EPA GLWQI (1993a) guidance, and other groups have adopted their values. The SCV (chronic TEL) of 0.014 µg/L has been adopted by EPA Region 5 (U.S. EPA 2003c) and the State of Texas

(TNRCC 2001), and Region 6 recommends its use to its risk assessors (ORNL RAIS).[11] The SAV (acute, PEL) of 0.24 µg/L, calculated by Suter and Tsao (1996) has not been adopted by the EPA regions, but is included in the RTR ecological benchmarks rather than having no PEL freshwater benchmark.

Dioxins—2,3,7,8-TCDD Surface Water Column Screening Benchmarks

Dioxins also are lipophilic and difficult to test for aquatic toxicity; thus, no NAWQC are available for TCDD. Nonetheless, EPA Region 4 developed chronic and acute freshwater screening values for TCDD of 1E-05 µg/L and 0.1 µg/L, respectively (U.S. EPA 2001b) (in Table 3-1 of the main report).

A.2.2.2 Sediment Benchmarks

This section describes the selection of sediment benchmarks for arsenic, cadmium, divalent mercury, methyl mercury, BaP for POM, and 2,3,7,8-TCDD for dioxins. We demonstrate our approach using arsenic, and provide briefer accounts for the remaining five PB-HAPs.

Arsenic (As) Sediment Screening Benchmarks

Many groups and investigators have developed chronic sediment quality criteria for arsenic, including those for freshwater sediments listed in Table A-3. Further, many different acronyms and terms are used to describe the same concepts within sediment benchmark terminology. For example, some sediment criteria experts consider a TEL or threshold-effects concentration (TEC) to be a level below which adverse effects are unlikely to occur (MacDonald et al. 2000), while others define a lowest-effect level (LEL) or a minimal effect threshold as the 15th percentile of species-specific threshold concentrations across diverse taxa (Jones et al. 1997). Few studies have examined the success/failure rate of sediment benchmarks to predict sediment toxicity accurately in the field. The Canadian Council of Ministers of the Environment (CCME 1999a,b) reported that the incidence of effects in sediment samples below the Canadian *interim* sediment quality guideline (ISQG) concentration for arsenic (i.e., 5.9 mg/kg dry sediment) is only 3 percent, which is close to a no-effect incidence rate (CCME 1999a,b).

[11] EPA recently merged Regional Screening Levels for Chemical Contaminants at Superfund Sites for Regions 3, 6, and 9 at a single website (https://www.epa.gov/risk/regional-screening-levels-rsls).

Table A-3. Sediment Screening Benchmarks Identified in ORNL RAIS Database

Sediment Screening Benchmark	mg/kg dw	Rationale for Not Using*
U.S. EPA ARCS highest NEC (similar to Washington State MAEL)	92.9	biological meaning of *highest* NEC for sediment communities is unclear
U.S. EPA ARCS PEC	33	*selected for use for RTR as probable effect level
U.S. EPA ARCS TEC	9.79	lower "threshold" available
Canadian ISQG	5.9	Canadian
Canadian PEL	17	lowest PEL
Consensus PEC (MacDonald et al. 2000)	33	*selected for use for RTR as probable effect level
Consensus TEC (MacDonald et al. 2000)	9.79	lower "threshold" available
Florida Department of Environmental Protection PEL	41.6	Florida conditions unusual
Florida Department of Environmental Protection TEL	7.24	Florida conditions unusual
Ontario Low (Persaud et al. 1993)	6	Canadian
Ontario Severe (Persaud et al. 1993)	33	*selected for use for RTR as probable effect level
U.S. EPA OSWER (Superfund) ERL	8.2	*selected for use for RTR as threshold for effect
U.S. EPA Region 4 TEL	7.24	could not verify online
U.S. EPA Region 5 RCRA ESL	9.79	lower "threshold" available
U.S. EPA Region 6 freshwater	5.9	could not verify online
Washington State freshwater MAEL	93	higher than PELs & PECs
Washington State freshwater NEL	57	higher than other threshold levels
U.S. EPA Region 3 BTAG, freshwater	9.8	lower "threshold" available

Acronyms: ARCS = Assessment and Remediation of Contaminated Sediments (Program); BTAG = Biological Technical Assistance Group (Superfund); EPA = Environmental Protection Agency; ERL = effects range – low; ESL = ecological screening level; ISQG = interim sediment quality guideline; MAEL = Sediment Impact Zone Maximum Level; NEC = no-effects concentration; NEL = no-effect level; ORNL = Oak Ridge National Laboratory; PEC = probable effects concentration; PEL = probable effect level; RCRA = Resource Conservation and Recovery Act; RAIS = risk assessment information system (Department of Energy); TEC = threshold effects concentration
Abbreviations: mg/kg dw = milligrams arsenic per kilogram dry weight sediment
* Value selected for use in RTR screens; see text.

For purposes of RTR assessments, we selected a threshold-effects benchmark of 8.2 mg[As]/kg dry weight sediment from EPA's Superfund program, because it is an EPA benchmark (preferred over DOE ORNL and state and Canadian benchmarks), and we could verify its derivation (U.S. EPA 1996a). Not all benchmarks included in RAIS can be verified using original sources, because several sources do not explain derivation of the benchmarks.

Values in Table A-3 associated with benchmark names suggesting that adverse effects are "probable," likely to be "frequent," or likely to be "severe" range from 33 to 93 mg[As]/kg[dry

weight (dw) sediment]. With three different groups identifying 33 mg/kg dw sediment as a probable or severe effects level [i.e., U.S. EPA Assessment and Remediation of Contaminated Sediments (ARCS program); MacDonald et al. 2000; Ontario (Persaud et al. 1993)], we recommend 33 mg[As]/kg dw sediment to represent the probable-effect benchmark.

Cadmium (Cd) Sediment Screening Benchmarks

Although we would prefer to have NEL, TEL, and PEL benchmarks from the same source for sediments; that was not possible for cadmium. EPA has recommended only a TEL (U.S. EPA 1996a, OSWER – Superfund Program) of 1.2 mg[Cd]/kg dw sediment. The CCME (1999b) had recommended an ISQG of 0.6 mg[Cd]/kg dw sediment, but more recently defined an even lower effect level, called the rare-effect level (EC & MDQuébec 2007) of 0.33 mg[Cd]/kg dw sediment. The CCME (1999b) PEL is 3.5 mg[Cd]/kg dw sediment.

Divalent Mercury (Hg^{++}) Sediment Screening Benchmarks

For Hg^{++}, we found no benchmarks representing an NEL but many benchmarks that could be interpreted as TELs and PELs. Given the similarity of the benchmarks, we could not clearly recommend one over another and decided, in this case, to average the values across sources to develop a TEL and a PEL for sediments.

For a TEL, we calculated the arithmetic mean of eight available benchmarks for inorganic (or total) mercury. That approach gives equal weight to the eight sediment benchmarks:

- U.S. EPA (1996b) – Great Lakes National Program Office (GLNPO) Assessment and Remediation of Contaminated Sediments (ARCS) program – 0.18 mg/kg[dry weight sediment] (mg/kg dw);
- MacDonald et al. (2000) – Consensus Threshold Effects Concentration – 0.18 mg/kg dw;
- Florida Department of Environmental Protection (MacDonald 1994) – sediment screening benchmark – 0.13 mg/kg dw;
- U.S. EPA (1996a) OSWER – Ecotox Threshold sediment screening level – 0.15 mg/kg dw;
- U.S. EPA (2015) Region 4 – sediment screening benchmark – 0.13 mg/kg dw;
- U.S. EPA (2006a) Region 3 Biological Technical Assistance Group (BTAG) – sediment screening benchmark – 0.18 mg/kg dw;
- U.S. EPA (2003c) Region 5 – Resource Conservation and Recovery Act (RCRA) – sediment screening benchmark – 0.174 mg/kg dw; and
- U.S. EPA Region 6 (TNRCC 2001) – sediment screening benchmark – 0.174 mg/kg dw.

The benchmarks listed above range from 0.13 mg/kg dw sediment to 0.18 mg/kg dw, with arithmetic mean 0.16 mg/kg dw sediment (in Table 3-1 of the main report).

For a PEL, we averaged the four values available for freshwater sediment probable effect levels:

- U.S. EPA (1996b) – GLNPO ARCS probable effects concentration – 1.06 mg/kg dw;
- MacDonald et al. (2000) – Consensus probable effects concentration – 1.06 mg/kg dw;
- Florida Department of Environmental Protection (MacDonald 1994) – PEL – 0.70 mg/kg dw; and
- CCME (2001) – PEL – 0.486 mg/kg dw sediment.

The benchmarks listed above range from 0.486 mg[Hg]/kg dry sediments to 1.06 mg[Hg]/kg dw, with arithmetic mean 0.84 mg [Hg]/kg dry sediments (in Table 3-1 of the main report).

Methyl Mercury (MeHg) Sediment Screening Benchmarks

We identified no benchmarks for MeHg in sediments. MacDonald et al. (2000) estimated a consensus TEC of 0.2 mg[total Hg]/kg dry sediments and a PEC of 1 mg[total Hg]/kg dry sediments (rounded to one significant digit). MeHg generally is 4 percent (range 1 to 11 percent) of total Hg in sediments (Krabbenhoft et al. 1999). Thus, we could have set benchmarks at 0.005 and 0.04 mg[MeHg]/kg dry sediments if we had confidence in the proportion of MeHg in sediments. TRIM.FaTE, however, estimates mercury transformations between Hg^{++} and MeHg for the environmental input parameters (e.g., pH, chloride ions, fraction organic carbon) and empirical values for equilibrium partitioning between aqueous phase and particulate phase chemical. Thus, over-riding those calculations based on the data reported by Krabbenhoft et al. (1999) would not have been reasonable. We therefore kept the TEC and PEC values estimated by MacDonald et al. (2000). Because the TEC and PEC values for Hg^{++} (see previous paragraph) are lower than for MeHg, and because most Hg in sediments is likely to be inorganic, the sediment benchmarks for Hg^{++} are the limiting benchmarks. Effectively, we have no benchmarks for MeHg in sediments.

POM—Benzo[a]pyrene [BaP] Sediment Screening Benchmarks

Several freshwater sediment benchmarks are available for BaP for the NEL, TEL, and PEL. For the NEL, we used the value of 0.032 mg[BaP]/kg dry sediments, which is recommended by CCME (1999b) and Region 6 (TNRCC 2001). Three sources recommend a TEL of 0.15 mg[BaP]/kg dry sediments: GLNPO ARCS (U.S. EPA 1996b); Region 3 BTAG (U.S. EPA

2006); and MacDonald et al. (2000). The same three sources recommend a PEL of 1.5 mg[BaP]/kg dry sediments (in Table 3-1 of the main report).

Dioxins—2,3,7,8-TCDD Sediment Screening Benchmarks

Dioxins are difficult to test for aquatic toxicity, because they basically do not partition to the water column or to sediment pore water. In addition, they are toxic at very low concentrations that are difficult to measure. We did identify TELs for 2,3,7,8-TCDD in sediments of 8.5E-07 mg/kg dry sediment (U.S. EPA 2006, Region 3), 2.5E-06 mg/kg dw (U.S. EPA 2001b, Region 4), and 1.2E-07 mg/kg dw (U.S. EPA 2003c, Region 5). The arithmetic mean of those three benchmarks rounded to two significant digits is 1.2E-06 mg/kg dw (in Table 3-1 of the main report). A geometric mean would be more conservative; however, the Los Alamos National Laboratory (LANL) recently has made its database of benchmarks available via the internet, recommending a screening LOAEL value of 8.5E-06 mg/kg dw (LANL 2015). We attempted to verify the derivation of that value; however, the references are to previous LANL versions of the database (e.g., LANL 2012 and earlier), rather than to original toxicity studies. Thus, we retain the arithmetic mean of three EPA TCDD benchmarks for sediments.

Initially, we found no benchmarks for an NEL or a PEL for TCDD. In 2016, we found a NOAEL of 8.5E-07 mg/kg dw in the LANL (2015) database and a PEL of 0.022 (rounded to two significant digits) mg/kg dw for Canadian sediments (CCME 2001; previously overlooked). We have not verified the derivation of the NEL or PEL; therefore, they each represent a single point-estimate of a sediment benchmark, in contrast to the TEL, which represents three separate point-estimates of a sediment benchmark.

A.2.2.3 Soil Benchmarks

For soils, EPA's national Superfund Program (formerly called the Office of Solid Waste and Emergency Response or OSWER) Eco-Soil Screening Levels (Eco-SSLs, U.S. EPA 2005c) were selected, if available, as the soil ecological benchmarks for the ecological risk environmental screens for the RTR assessment. The OSWER Eco-SSLs are the only EPA-vetted ecological toxicity screening benchmarks for soils established for use by the Agency nationwide. For chemicals for which no Eco-SSLs were available, EPA regional sources of soil ecotoxicity benchmarks were sought (e.g., Regions 4, 5, and 6). The general methods for deriving those

benchmarks differ from the methods EPA used to derive Eco-SSLs, and some are not available via the internet.

For some chemicals, EPA regions use soil ecological benchmarks developed by other agencies such as DOE or one of the states in the region. If not specified in published information, we assumed that whichever group of organisms was most sensitive to the chemical in soil (e.g., earthworms, insect larvae, plant roots, and in some cases herbivorous animals consuming plants grown in the contaminated soil) was likely to have been the basis for a soil screening criterion. If an EPA region and another non-EPA agency were identified as using the same numeric benchmark value, the sources that designated that value are acknowledged. Finally, if the only source providing a screening-level benchmark for soils was not an EPA office or region (e.g., DOE, ORNL, Environment Canada, a state), the value was used as last priority.

Arsenic (As) Soil Screening Benchmarks

Arsenic has not been demonstrated to bioaccumulate significantly in soil invertebrates. Data compiled to develop and validate bioaccumulation models for earthworms indicate that arsenic concentrations in earthworms tend to be approximately one order of magnitude lower than the concentration in soils on a mg/kg dry weight basis (i.e., both soils and earthworm arsenic concentrations measured per unit dry weight; Sample et al. 1998). Thus, for arsenic, the Eco-SSL for plants is lower than the Eco-SSLs for ground-feeding birds and mammals that ingest soil invertebrates. In contrast, the most appropriate Eco-SSLs for bioaccumulative substances (e.g., mercury, cadmium) are for birds or mammals consuming soil invertebrates. The lowest arsenic Eco-SSL value for plants, 18 mg[As]/kg[dry weight soil] (Table A-4), is the geometric mean of the maximum allowable toxicant concentration (MATC) for three plant studies (with ryegrass, cotton, and rice) that EPA judged to have appropriate arsenic bioavailability.

The three studies included both a low pH (5.6) and organic matter content (0.7%) and a higher pH (7.9) and organic matter content (1.1%) (Table 3.1 in U.S. EPA 2005b). For each of the three plant species, the MATC represents the geometric mean of the experimentally determined LOAEL and the NOAEL for plant growth.

Table A-4. Ecological Soil Benchmarks for Inorganic Arsenic, CAS No. 7440-38-2

Name of Benchmark	mg/kg dw soil
U.S. EPA OSWER Eco-SSL Plants	18
U.S. EPA OSWER Eco-SSL Avian	43
U.S. EPA OSWER Eco-SSL Invertebrate	NA
U.S. EPA OSWER Eco-SSL Mammalian	46
U.S. EPA Region 6 Earthworms Surface Soil Screening Benchmark	60
U.S. EPA Region 6 Plants Surface Soil Screening Benchmark	37
U.S. DOE ORNL Invertebrates Soil Screening Benchmark	60
U.S. DOE ORNL Microbes Soil Screening Benchmark	100
U.S. DOE ORNL Plants Screening Benchmark	10

Abbreviations and Acronyms: CAS = Chemical Abstracts Service; dw = dry weight; Eco-SSL = U.S. EPA Ecological Soil Screening Level (Superfund); ORNL = Oak Ridge National Laboratory; DOE =Department of Energy; mg/kg dw soil = milligrams arsenic per kilogram dry weight soil; NA = not available

The avian Eco-SSL (woodcock) is based on one of four toxicity experiments that both met EPA's criteria for study acceptability and examined growth and reproduction in birds. Of those, only one experiment identified NOAELs for both growth and reproduction at 2.24 mg[As]/kg [body weight]-day (arsenate oxide) in domestic chickens (Holcman and Stibilj 1997, as cited in U.S. EPA 2005b). Camardese et al. (1990) identified a lower LOAEL of 1.49 mg/kg-day (arsenate) for growth for mallard duck; however, because that study did not identify a NOAEL, EPA used 2.24 mg/kg-day as a TRV for birds (U.S. EPA 2005b). Using that TRV and back-calculating a soil concentration based on woodcock consumption of arsenic with a diet of earthworms yields an Eco-SSL for ground-feeding birds of 43 mg/kg dw soil (U.S. EPA 2005b).

More toxicity studies of acceptable quality were available for mammals than for birds. From 55 mammalian studies, over 100 toxicity values were identified. EPA calculated the geometric mean of 27 bounded[12] NOAELs for reproduction and growth to be 2.47 mg[As]/kg-day. One study using beagle dogs (initially 7–8 months old) identified a bounded LOAEL of 1.66 mg/kg-day (Neiger and Osweiler 1989, as cited in U.S. EPA 2005b), which is lower than 2.47 mg/kg-day. EPA therefore used the NOAEL associated with the dog study, 1.04 mg/kg-day, to calculate a

[12]A bounded NOAEL is one from a study in which a LOAEL was identified. A bounded LOAEL is one from a study in which a NOAEL was identified.

TRV for mammals. Back-calculation of a soil concentration for a shrew that consumes invertebrates in soils yielded an Eco-SSL for ground-feeding mammals of 46 mg[As]/kg dw soil.

Five other LOAELs are from studies [two in mice measuring growth and reproduction (total four LOAELs), and one in Guinea pigs measuring growth] that did not identify a NOAEL and for which the LOAELs for reproduction, growth, or survival were lower than 2.47 mg[As]/kg-day (see Figure 6.1 in USEPA 2005b). Those were not considered in deriving the Eco-SSL for mammals because they were not bounded by a NOAEL identified in the same experiment. Thus, the Eco-SSL for soils for shrews might be based on a NOAEL that is not necessarily protective of some sensitive species or sensitive effect endpoints.

Cadmium (Cd) Soil Screening Benchmarks

EPA has derived four Eco-SSLs for cadmium (Table A-5). As is often the case for Eco-SSLs for bioaccumulative substances, the benchmarks protective of birds and mammals that feed on soil invertebrates are lower (more restrictive) than those for plants and invertebrates. That is because chemicals bioaccumulate from soils to the soil invertebrates that then are consumed by the wildlife. Although nominally based on NOAELs for adverse effects on reproduction and growth, the Eco-SSLs for insectivorous wildlife are based on the geometric mean of NOAELs across both types of effect and across all species for which data are available within each group, birds or mammals, respectively.

For the cadmium Eco-SSL for birds, most (15/20) NOAELs used to derive the geometric mean NOAEL came from toxicity tests using chickens and quail (Order Galliformes) with a minority (4/20) of toxicity values from mallard duck and one value from wood duck (Order Anseriformes includes ducks, mergansers, and other waterfowl). The avian geometric mean NOAEL calculated for the Eco-SSL is 1.47 mg[Cd]/kg bw-day. Back-calculating a soil concentration that corresponds to the avian TRV for woodcock consuming 100% earthworms yields an Eco-SSL of 0.77 mg[Cd]/kg dw soil (U.S. EPA 2005d) (Table A-5). We calculated a NOAEL and LOAEL for piscivorous birds (in Section A.4 below) as 1.0 and 0.7 mg[Cd]/kg bw-day, respectively. We conclude that the avian Eco-SSL, based on the geometric mean of NOAELs across species and endpoints, is similar to a LOAEL for ducks and mergansers as discussed in Section A.4.

Table A-5. Screening Soil Benchmarks for Cadmium, Thresholds for Effect

Benchmark Type	Value	Units	Benchmark	Reference
Mammals (shrew)	0.36	mg[total Cd]/kg dry weight soil	Eco-SSL for four soil communities specified under Benchmark Type	U.S. EPA 2005d, OSWER
Birds (American woodcock)	0.77			
Plants	32			
Invertebrates	140			

Acronym: Eco-SSL = U.S. EPA Ecological Soil Screening Level (Superfund)

For the cadmium Eco-SSL for mammals, the geometric mean of 23 NOAELs for reproduction (21 from rats and 2 from mice) and 59 NOAELs for growth (most from rats, but a few from mice, cattle, sheep, pigs, dogs, and voles) of 1.86 mg[Cd]/kg bw-day turned out to be higher than the highest bounded NOAEL (0.77 mg cadmium/kg bw-day) below the lowest bounded LOAEL. EPA therefore set the TRV used to calculate the Eco-SSL for mammals to 0.77 mg cadmium/kg bw-day. Back-calculating the corresponding soil concentrations for shrews that consume 100% earthworms resulted in an Eco-SSL of 0.36 mg[Cd]/kg dry soil (U.S. EPA 2005d). The values we identified as the LOAEL and NOAEL for mammals for a sensitive species and endpoint (in Section A.4) are 7.42 and 0.742 mg[Cd]/kg bw-day, respectively. Thus, in this case, the mammalian Eco-SSL is based on a TRV that is similar to a NOAEL for a sensitive mammalian species and endpoint.

Divalent Mercury (Hg^{++}) Soil Screening Benchmarks

EPA has not estimated Eco-SSLs for divalent mercury in soils. Inorganic mercury is not expected to bioaccumulate. Thus, the only soil screening levels that we identified were the EPA Region 6 recommendation of 0.3 mg[Hg]/kg dry soil for plants (Efroymson et al. 1997a) and the EPA Regions 4 and 6 recommendation of 0.1 mg[Hg]/kg dry soil for earthworms in soil (U.S. EPA 2015 and Efroymson et al. 1997b, respectively). See Table 3-1 of the main report.

Methyl Mercury (MeHg) Soil Screening Benchmarks

Methyl mercury is expected to bioaccumulate; however, its concentrations in soils that receive air deposition of divalent mercury are expected to be low. Nonetheless, some methylation of mercury can occur in soils, so in 2016, we sought benchmarks for MeHg in soils (Table A-6).

Table A-6. Soil Screeing Benchmarks for Methyl Mercury

Benchmark Type	Units	Value	Source: Benchmark Name [Comment]
Mammals (shrew)	mg/kg dry soil	0.0068	GMATC values calculated from U.S. EPA (2015) Region 4 SESLs for mammals and birds [LANL (2012) ECORISK Database Version 3.2]
Birds (robin)		0.0011	
Plants		0.3	U.S. EPA (2015) Region 4 cites Efroymson et al. (1997a)
Invertebrates		0.1	U.S. EPA (2015) Region 4

Acronyms: GMATC = geometric mean maximum allowable toxicant concentration = geometric mean of LOAEL and NOAEL, SESL = soil ecological screening level

In a recent update of its ecological screening benchmarks, Region 4 cited the September 2012 release of the LANL ECORISK (Version 3.2) database as its source of estimated soil-screening levels for MeHg protective of ground-feeding birds and mammals (U.S. EPA 2015). As of August 20, 2016, a more recent version of the ECORISK database (Version 3.3) was available (LANL 2015), which we checked for MeHg soil ecological screening levels (SESLs). We provide a summary of the derivation of the LANL ECORISK SESLs below. Unlike the EPA Eco-SSLs, which use a geometric mean of all NOAELs from all studies and all avian species of acceptable quality for growth and reproduction for which both a NOAEL and LOAEL were identified, the LANL ECORISK SESLs are based on a single critical study, a sensitive species, and sensitive endpoints (i.e., according to U.S. EPA 1995b GLWQI Guidelines). After LANL has selected TRVs for sensitive endpoints and species from the available data, it uses the TRVs to back-calculate SESLs, as does EPA when deriving Eco-SSLs.

For birds, LANL uses American robin (wide habitat and geographic range) instead of woodcock as the ground-feeding bird for which to back-calculate a soil concentration. As shown in Table A-7, the lowest SESLs for American robin are associated with a diet consisting entirely (100%) of soil invertebrates. That is the same diet assumed for woodcock for U.S. EPA (2007) Eco-SSLs. Both LANL and EPA assume that the soil invertebrates are earthworms, which bioaccumulate MeHg from the soils.

LANL (2015) cited the Heinz et al. (1979) study of mallard duck exposed to MeHg in the diet for three generations. A significant decrease in egg and duckling production was observed at that 0.5 ppm in the diet. Sample et al. (1996) used the food consumption rate from Heinz et al. (1979) and typical body weights for growing mallards from another data source to convert the 0.5-ppm MeHg concentration in food to a TRV dose of 0.064 mg/kg-day. Using a LOAEL-to-NOAEL UF of 10, Sample et al. (1996) estimated a NOAEL of 0.0064 mg/kg-day. Back-calculating the

corresponding soil concentration for a robin consuming 100 percent earthworms that had bioaccumulated MeHg from the soil, LANL (2015) estimated a NOAEL and LOAEL of 0.00035 and 0.0035 mg/kg dry soil, respectively (Table A-7).

Table A-7. Soil Ecological Screening Levels for Methyl Mercury from Los Alamos National Laboratory

Species	Diet	Soil Ecological Screening Level (mg/kg dry soil)		
		SESL NOAEL	SESL LOAEL	GMATC
American robin (avian ground-feeding bird)	100% plants (berries)	0.075	0.75	0.2372
	100% soil invertebrates	0.00035	0.0035	**0.0011**
	50:50 plants/soil invertebrates	0.00071	0.0071	0.0022
American kestrel (avian top carnivore)	100% small mammal flesh	0.0078	0.078	0.0247
	50:50 small mammals and soil invertebrates	0.0017	0.017	0.0054
Deer mouse	50:50 soil invertebrates and seeds	0.0063	0.031	0.0140
Montane shrew	100% soil invertebrates	0.0031	0.015	**0.0068**

Acronyms: GMATC = geometric mean maximum acceptable toxic concentration; calculated in this table as the geometric mean of the SESL NOAEL and LOAEL. SESL = soil ecological screening level

Because the EPA Superfund Eco-SSLs provide a single SSL for each assessment endpoint, and because we are using each Eco-SSL as a TEL, having two different LANL SESLs (i.e., a NOAEL and a LOAEL) would be inconsistent. We therefore calculated the geometric mean of the NOAEL and LOAEL SESLs (i.e., the GMATC) to represent a TEL for the robin (0.0011 mg/kg dw soil, value in bold in Table A-7).

For mammals, LANL used a montane shrew to represent ground-feeding small mammals. LANL cited the Verschuuren et al. (1976) toxicity study of rat exposed to MeHg in the diet for three generations at three dietary concentrations—0.1-, 0.5-, and 2.5-ppm MeHgCl, where Hg makes up 79.9% of the compound. Reduced pup viability was observed in the 2.5-ppm MeHgCl, and no adverse effects were observed in the other two groups. LANL (2015) cited Sample et al. (1996) for the conversion of dietary concentrations to ingested doses based on rat food ingestion rates (FIRs) and body weights: the chronic TRV NOAEL is 0.032 mg[Hg]/kg bw-day and the chronic TRV LOAEL is 0.16 mg[Hg]/kg bw-day. Back-calculating the corresponding soil concentrations for montane shrew consuming 100-percent earthworms that had bioaccumulated MeHg from the soil, LANL (2015) estimated a NOAEL of 0.0031 mg[Hg]/kg dry soil and a LOAEL of 0.015 mg[Hg]/kg dry soil (listed in the last row of Table A-6). Again, we calculated the geometric

mean of the NOAEL and LOAEL (i.e., the GMATC) to represent a TEL for the shrew, 0.0068 mg[Hg]/kg dry soil (in bold in Table A-7).

POM—Benzo[a]pyrene (BaP) Soil Screening Benchmark

EPA has developed no Eco-SSLs for BaP, although it has estimated an Eco-SSL for mammals and an Eco-SSL for invertebrates for HMW polycyclic aromatic hydrocarbons (PAHs) (i.e., 4 or more fused benzene rings) of 1.1 mg/kg dry soil and 18 mg/kg dry soil, respectively (U.S. EPA 2007). EPA Region 5 has developed a soil screening value for BaP for masked shrew of 1.52 mg/kg dry soil. Because we are using the toxicity equivalency approach to evaluate the joint toxicity of POM based on their BaP-toxic equivalents, we use the EPA Region 5 value for BaP.

Dioxins—2,3,7,8-TCDD Soil Screening Benchmark

EPA Region 5 estimated an ESL for soils of 2.0E-07 mg/kg dry soil to protect masked shrews that consume earthworms contaminated with TCDD from soils (U.S. EPA 2003c). LANL (2015) lists its soil screening levels for montane shrew as a NOAEL of 2.9E-07 mg/kg dry soil and LOAEL of 1.9E-06 mg/kg dry soil. Those values bracket the Region 5 ESL; therefore, we use the Region 5 value to represent a TEL for the shrew.

No screening benchmarks were identified for birds or plants exposed to TCDD in soils. For invertebrates, LANL (2015) calculated a NOAEL of 5 mg/kg dry soil and a LOAEL of 10 mg/kg dry soil for SELS for TCDD. The geometric mean of 5 and 10 equals 7.1 mg/kg dry soil, which we use for the soil invertebrate community TEL benchmark.

A.2.3 Hydrogen Fluoride (HF) Air Benchmarks for Terrestrial Plants

Gaseous fluorides, such as HF, are phytotoxic (i.e., toxic to plants). Gaseous HF enters leaves of plants through the stomata, which generally are open during daylight hours and closed at night. Gaseous HF is much more rapidly absorbed than fluoride associated with particulates, which do not diffuse through the stomata. Fluoride absorption is fairly uniform over the entire leaf under-surface. It readily dissolves and is then transported in ionic form through the apoplastic aqueous spaces of the mesophyll cell walls driven by transpiration. Thus, fluoride moves via translocation to the leaf tip and edges where cell necrosis occurs first (Hill and Pack 1983). Leaf tips can contain up to 100 times more fluoride than the leaf basal section after long-term exposure (Hill 1969; Hill and Pack 1983).

The most common initial symptoms of fluoride injury are necrotic lesions at leaf tips and edges, extending toward the leaf base as exposure continues (Hill and Pack 1983). In a few species, including corn and citrus, chlorosis (i.e., loss of chlorophyll and green color) is evident before necrosis appears. Although loss of functional leaf area can reduce growth and yield in many species of plants, a few species show little effect on yield depending on the part of the plant harvested and the stages at which exposures occurred (e.g., some species are most sensitive during rapid growth of seedlings or during flowering) (Hill and Pack 1983).

Susceptibility to HF also varies with lifestage of the plant and abiotic factors. For broadleaf plants, several studies indicate that damage from HF exposure is more pronounced when plant tissues are expanding or elongating (WHO 2002; Hill 1969). Some pine species are included among species of concern due to their sensitivity to HF during needle growth (Adams et al. 1956; APIS 2010). Abiotic factors, such as humidity, air temperature, wind (speed and direction), and soil water content, can influence exposure by modifying the rate of HF absorption by plants. For example, dry conditions reduce HF absorption due to reduced transpiration and stomatal conductance (APIS 2010). Excessive rain also can reduce exposure due to "washing," while light rain can effectively increase the amount of fluoride deposited on the leaves (Hill 1969). Abiotic factors also can affect inherent plant sensitivity to HF exposures. In the field, some plants stressed by unfavorable conditions of low fertility and limited water are more sensitive to HF exposure than the same species grown under more favorable conditions (Hill and Pack 1983).

The remainder of this section provides background information for the derivation of HF air concentration benchmarks for terrestrial plant communities (Section 3.2.2 of the main report). Section A.2.3.1 discusses three distinct approaches to setting limits for plant exposures to HF. Section A.2.3.2 lists existing regulatory benchmarks for HF in the United States and other countries. Section A.2.3.3 summarizes exposure-response data for effects of air HF on plants, both for short-term exposures (e.g., 1-day maximum concentration) and over the longer term (e.g., average 4-month concentration).

A.2.3.1 Methods of Establishing HF Benchmarks

In theory, environmental standards for HF effects on vegetation could be defined in at least three ways (Hill 1969): atmospheric fluoride concentrations, vegetation fluoride concentrations, or the

presence of leaf necrosis or chlorosis. Table A-8 presents the pros and cons of each method outlined by Hill (1969).

Table A-8. Overview of Three Approaches to Hydrogen Fluoride (HF) Environmental Standards (Hill 1969)

Approach	Traditional Use/Benefit	Complicating Factors with Hydrogen Fluoride
Atmospheric concentration	Used frequently in air quality standards Simple Ease of use for control programs	Inter- and intraspecies variation in effects (lack of data for levels that are protective of large majority of species for site-specific assessments) Need to understand contribution of exposure duration Variation in responses associated with abiotic factors (e.g., rainfall, humidity, temperature) Atmospheric fluoride includes total soluble inorganic fluorides (speciation data and effects data for various species lacking) and might include fluoride adsorbed to particles in the air
Vegetation concentration	Useful for protecting wildlife (or livestock) via plant consumption Leaves accumulate most HF (compared with other plant parts) Leaf sampling relatively simple and cost effective	Need for standardization in: Leaf age (at time of exposure) Lifestage (at time of exposure, e.g., fast growth, flower set) Time of sampling Species/varieties sampled Random selection of leaves Method of analysis Need to remove F from plant surfaces without leaching F from leaf interior Fluoride content concentrated along leaf margins and tips
Leaf appearance (necrosis or chlorosis)	Time effective Summary outcome (no detailed analysis of complex variables)	Need qualified/trained personnel Leaf appearance can be influenced by other (non-HF-related) factors Need fluoride analysis to confirm

Most existing HF standards are based on plant concentration data collected for what have been identified thus far as particularly sensitive species and for livestock forage. Hill (1969) noted that adequate data generally are not available to develop site-specific HF air benchmarks for the protection of plants. To estimate fluoride concentrations in plants, however, TRIM.FaTE would need to be parameterized for plant uptake of fluoride from the air and possibly from uptake through the roots. That level of effort is beyond a Tier 1 or 2 screen for ecological risks.

For RTR ecological risk screens of acid gases, which are conducted using modeled estimates of ambient air concentrations based on emissions from the regulated source, the most expedient expression of an air benchmark for HF for plants is as an air concentration. The remaining sections of this document, therefore, focus on the relationship between HF air concentrations and

adverse effects in plants. In addition, for purposes of the RTR ecological risk screen, chronic benchmarks are relevant to the chronic exposure scenarios evaluated.

A.2.3.2 HF Regulatory Levels

Although EPA has not established environmental criteria for HF, at least 13 other countries have established national environmental criteria or standards (Newman 1984). In the United States, at least 12 states have established criteria or standards, most based on protecting forage grasses from accumulating fluoride to concentrations exceeding 35–40 mg[F]/kg dry weight plant. Some data suggest higher concentrations in forage result in development of fluorosis in cattle/calves (Newman 1984).

Most of the available criteria or standards are expressed as concentrations in plants, not as atmospheric concentrations, particularly if the intent is to protect livestock from fluorosis from fluoride in their forage. Use of plant-based HF concentrations would require a plant-fluoride uptake model. At this time, TRIM.FaTE is not parameterized for HF uptake in plant compartments. The remaining discussion focuses on criteria and standards expressed as concentrations of gas-phase HF in air, not total fluoride in plants. The criteria or standards that were readily available from Canada and several U.S. states are summarized in Table A-9.

Guidelines to protect vegetation from exposures to HF expressed as air concentrations were first developed in Canada under the Canadian Environmental Protection Act by Bourgeau and colleagues in 1996 (EC 1996). To protect vegetation from adverse effects resulting from HF exposure, CCME (1999c) recommends HF concentrations not exceed 0.4 µg/m^3 air over 30 to 90 days (Alberta Environment 2006; HF concentrations can be higher for shorter exposures).

Environment Canada (EC 1996; CCME 1999c) defined the criteria as:

> "The level above which there are demonstrated effects on human health and/or the environment. It is scientifically based and defines the boundary between the LOAEL and the NOAEL. It is considered to be the level of exposure just below that most likely to result in a defined and identifiable but minimal effect. The reference levels have no safety factors applied to them, as they are related directly to the LOAEL, and are the most conservative estimates of the effect level." (emphasis added; CCME 1999c)

Table A-9. Governmental Air Criteria for Hydrogen Fluoride (HF) to Protect Plants

Reference	Specific Information[a]	Air Criteria for Hydrogen Fluoride (µg HF/m³) for Specified Duration (Averaging Period)						
		30 min	12 h	24 h	7 d	30 d	70 d	90 d
Canada (EC 1996; CCME 1999c)	Gaseous, growing season	–	–	1.1	0.5	0.4	–	0.4
Alberta and Manitoba (Alberta Environment 2006)	Gaseous	–	–	0.85	0.55	0.35	0.2	0.2
Ontario (OME 2004)	Gaseous, growing season	4.3	–	0.86	–	0.34	–	–
Ontario (OME 2004)	Total, growing season	8.6	–	1.72	–	0.69	–	–
Ontario (OME 2004)	Total, nongrowing season	17.2	–	3.44	–	1.38	–	–
Texas Commission on Environmental Quality (TCEQ 2009)[b]	Gaseous	–	–	3.0	–	–	–	0.6
Kentucky, Jefferson County[c]	Gaseous	–	3.68	2.86	0.80	0.50	–	–
New York State[d]	Gaseous	–	3.7	2.85	1.65	0.80	–	–
Washington State[e]	Gaseous	–		2.9	1.7	0.84	0.5	0.5
Tennessee[f]	Not specified	–	3.7	2.9	1.6	1.2	–	–

Abbreviations: min = minutes; h = hours; d = days; "–" means no criterion for that exposure duration
[a]"Total" atmospheric HF includes both gaseous and particulate-bound HF.
[b]Air quality standards for the State of Texas were removed in 2000.
[c]See http://www.epa.gov/region4/air/sips/ky/lou/3.04.pdf.
[d]See http://www.dec.ny.gov/regs/4146.html.
[e]See https://fortress.wa.gov/ecy/publications/publications/wac173481.pdf; the bold highlighted values are HF benchmarks for RTR environmental risk screening (see text).
[f]See http://www.state.tn.us/sos/rules/1200/1200-03/1200-03-03.pdf.

The Environment Canada criteria were based on regression analysis of exposure-concentration vs. exposure-duration data from the studies shown in Section A.2.3.3 and from additional unpublished studies.[13] The linear regression model used log(exposure concentration × duration), specified as "dose," as the dependent variable. Log(exposure duration) was the independent variable. Environment Canada pointed out, however, the selection of data to include in the regression was based on expert judgment, and the data set used did not meet some assumptions associated with estimating confidence intervals for the regression equation. Also, the value for

[13]References cited by Environment Canada (1996) from conference proceedings abstracts or other nonpeer-reviewed/nonpublished sources are not included in this report.

"dose" is not independent of the duration value, violating a key assumption for simple regression analyses.

Most investigators plot a specified effect level (e.g., initial evidence of leaf necrosis) for each study using log(exposure concentration) for the y-axis and log(exposure duration) for the x-axis. If Haber's rule applies, a straight line with a slope of 1.0 would result across all exposure durations. Haber's rule states that response is directly proportional to the exposure duration multiplied by the exposure concentration. Over the short term (i.e., a few days), the accumulation of HF in plants generally follows Haber's rule (data presented in McCune 1969a). The slope of the relationship decreases (becomes more horizontal; more dependent on concentration and less dependent on exposure duration) as exposure duration increases beyond 1 or 2 days (McCune 1969a). Thus, for chronic exposures, only exposure concentration need be specified.

Provincial guidelines for Alberta include a 30-day average limit of 0.35 µg HF/m^3 and 70- and 90-day average limits of 0.20 µg HF/m^3. Although Alberta Environment did not specify the level of effect associated with 0.2 µg HF/m^3 (see Table A-11), given the available data, only grapevines might be expected to show some evidence of injury at that concentration, and the significance of that injury to grape productivity is unknown. Thus, we conclude that the provincial guidelines for Alberta are similar to an NEL for plant communities and populations, including the most HF-sensitive commercial crops.

The Ontario Ministry for the Environment (OME 2004) has established provincial guidelines for Ontario that distinguish between the growing season and the nongrowing season and between total HF in air (including particulates) and gaseous HF only. The 30-day criterion for gas-phase HF during the growing season is 0.34 µg HF/m^3; longer-duration criteria were not established. This criterion and other air concentration criteria for HF established in Canada are listed in Table A-9. The criteria are based on studies of agricultural crops, horticultural plants, and coniferous trees, as described in Section A.2.3.3.

In the United States, for the states having ambient air quality standards or criteria for gaseous HF, the values are generally less than 1.0 µg/m^3 as a 30-day limit. Examples for several states are included in Table A-9. The Texas Commission on Environmental Quality (TCEQ) established effect screening levels for the protection of vegetation, cattle, and human health (TCEQ 2009, Table A-9). The TCEQ chronic (90-day) criterion was based on a LOAEL for

soybean productivity; nonagricultural plants were not evaluated. The other state and county standards or criteria included in Table A-9 are similar in magnitude to the TCEQ values for 90-day durations.

For purposes of the RTR environmental risk screen, the two benchmarks for HF were evaluated as representing an LEL: the 90-day criterion from Washington State of 0.5 µg HF/m^3 and the Environment Canada 90-day criterion of 0.4 µg HF/m^3. Both criteria are presented in bold and highlighted in Table A-9. Section A.2.3.3 below includes summaries of original data on HF toxicity to plants.

For comparison with long-term human health criteria, the California Office of Environment and Health Hazard Assessment (OEHHA) has recommended a chronic inhalation reference exposure limit for humans of 14 µg/m^3 based on the occurrence of skeletal fluorosis.[14] Thus, the 90-day criteria for plants are lower than the reference exposure limit to protect human health from inhalation toxicity.

A.2.3.3 Hydrogen Fluoride (HF) Exposure-Response for Plants

Critical concentrations cited in criteria documents often are based on the prevention of visible injury to plants by HF rather than on measured reductions in plant productivity as measured by vegetative growth and seed yield, for two reasons. First, data on effects of HF on plant growth and productivity are limited. Second, concentrations inducing visible injury are lower than those affecting growth and are therefore protective of both endpoints (APIS 2010).

Short-term Exposures

Short-term exposure to HF typically results in leaf lesions and necrosis along the tips and margins of leaves where fluoride has accumulated. Table A-10 summarizes information on the phytotoxic effects of short-term exposure to HF available from the literature. Consequently, a longer averaging time (e.g., 24 hours) is more relevant than a shorter averaging time (e.g., 30 minutes, 1 hour).

[14] See http://oehha.ca.gov/air/chronic_rels/HyFluoCREL.html.

Table A-10. Adverse Effects in Terrestrial Plants Following Short-term Exposures to HF

Species Tested	Study Protocol[a]	Results	LOEL ($\mu g/m^3$)	C × D ($\mu g/m^3$-d)[b]	Reference
Ponderosa pine (*Pinus ponderosa*)	1.46 µg F/m^3 for 24 h	Leaf injury index = 0.5, that is 50% of the length of needles injured	1.46	1.46	Adams et al. (1956)
Jerusalem cherry (*Solannum pseudo-capsicum*)	0.9 or 4.0 µg F/m^3 for 4 d in dark	Mild leaf necrosis in "sensitive" clone during exposure in dark, which became "severe" (40–60%), leaf necrosis after plant was exposed to light	0.9	0.9	MacLean et al. (1982)
Gladiolus sp.	0.17 µg F/m^3 for 9 d	Necrotic leaf tips (% not specified)	0.17	1.53	Hitchcock et al. (1962), as cited in WHO (2002)
Wheat (*Triticum aestivum*)	0.9 µg or 2.9 µg F/m^3 for 4 d	Reduced mean yield by 25% dry weight in grain spikes when exposure occurs during anthesis (i.e., flowering)	0.9	3.6	MacLean and Schneider (1981)
Sorghum sp. Northrup King 22A hybrid	"0"[c],1.6, 2.2, 2.8, or 3.3 µg/m^3 (mean concentration over 9 d); experiment varied the order in which different exposure concentrations (1.5, 1.8, 3.2, or 3.6 µg F/m^3) were applied over three successive 3-day periods	Reduced total dry weight biomass at harvest by 20% after 72-d exposure and reduced grain dry weight yield by 9% with exposures at 1.5, 3.2, then 1.8 µg/m^3 for three successive 3-d periods	2.2	20	MacLean et al. (1984)
Black spruce (*Picea mariana*); 2 years old	0.3, 2.3, 4.2, or 8.1 µg F/m^3 for 78 h, observed 20 d after exposure ceased	At 2.3 µg/m^3, 23% of trees exhibited slight (12%), moderate (10%), or severe (1%) injury to needles. At 4.2 µg/m^3, 61% of trees exhibited needle injury. At 8.1 µg/m^3, 96% of trees exhibited needle injury, and 72% of injury was moderate to severe.	2.3	7.5	McCune et al. (1991)
White spruce (*Picea glauca*); 3 years old	0.3, 2.6, 5.2, or 11.1 µg F/m^3 for 50 h, observed 20 d after exposure ceased	At 5.2 µg/m^3, 9% of trees categorized with needle injury. At 11 µg/m^3, 40% of trees with needle injury: 32% categorized with moderate to severe needle necrosis; remaining 8% with slight needle necrosis	5.2	11	McCune et al. (1991)
Tobacco (*Nicotiana tabacum* L.)	0.5 or 45 µg HF/m^3 for 1 d	Growth (plant height) reduced by 50% at 45 µg/m^3 compared with controls, accompanied by 63% reduction in chlorophyll content	45	45	Döğeroğlu et al. (2003)
Tobacco (*Nicotiana tabacum*)	0.5 or 45 µg HF/m^3 for 3 d	Growth (plant height) reduced by 70% at 45 µg/m^3 compared with controls, accompanied by 85% reduction in chlorophyll	45	135	Döğeroğlu et al. (2003)

Species Tested	Study Protocol[a]	Results	LOEL (µg/m^3)	C × D (µg/m^3-d)[b]	Reference
"Conifers"	Summary of dose-response relationships from the available literature based on 24-h average HF concentrations	Increased foliar markings	3.0[d]	3.0	McCune (1969b)
"Fruit Trees"		Increased foliar markings	4.5[d]	4.5	
Gladiolus		Reduced growth or yield	6.0[d]	6.0	
Corn		Reduced growth or yield	10.5[d]	10.5	
Tomato		Increased foliar markings	12[d]	12	

[a]Concentrations can be reported for hydrogen fluoride (HF) or the fluoride ion (F) only. Atomic weight of H = 1 g/mole, and F = 19 g/mole. Thus, the difference in an air concentration expressed as µg HF/m^3 and an air concentration expressed as µg F/m^3 is only 5%. For comparison with other measurements of HF concentrations in air, note that 1 µg/m^3 of fluoride (F) is equal to 0.874 ppb (parts per billion) fluoride by weight or 1.33 ppb by volume of any gas containing 1 fluorine atom per molecule. These conversions hold true at an atmospheric pressure of 29.9 inches of Hg and 60 °F (Hill and Pack 1983).
[b]C × D = exposure concentration multiplied by exposure duration, assuming Haber's rule applies over short-term exposures.
[c]The authors stated that "no HF" exposure occurred for this group, but a background concentration around 0.01–0.03 µg/m^3 likely was used for this group.
[d]Values reported by McCune (1969) are 24-hour mean threshold concentrations based on an evaluation of the available literature (exposure concentration, duration, and plant-response data plotted with curves).

Concentrations listed in the LOEL column of Table A-10 represent the lowest concentration at which statistically significant effects on growth, yield, or leaf necrosis were evident when the test group exposed at the LOEL was compared with the control group of plants. We use LOEL instead of LOAEL terminology because the significance of low levels of leaf necrosis and several other types of effects on plant productivity has not been quantified. The study protocol column includes a list of the exposure concentrations tested. In some cases, the lowest concentration listed is the "background" concentration the control plants experience. The highest concentration listed in the study-protocol column that is lower than the concentration listed in the LOEL column represents a NOEL. Effects, if present, at a NOEL were not statistically different from effects shown in the control plants (or the NOEL represents the control plants). Table A-10 indicates that effects evident after short-term exposures include foliar chlorosis and necrosis and, in some tests, reduced plant growth rates.

Longer-term Exposures

Longer-term (i.e., greater than 30 days) exposures of plants to HF usually result in leaf chlorosis and necrosis and can result in reduced growth and productivity even when leaf damage is not apparent. More data are available for longer-term exposures of plants to HF (Table A-11) than for short-term exposures.

Table A-11. Adverse Effects in Terrestrial Plants Following Longer-term Exposure to HF

Species Tested	Study Protocol	Results	LOEL (µg/m^3)	Reference
Tendergreen bean	0.6 µg F/m^3 for 43 d	Reduced number and mass of marketable pods by 20% and 25%, respectively; no influence on growth or foliar appearance	0.6	MacLean et al. (1977)
Tomato (Fireball 861 VR)	0.6 µg F/m^3 for 93 d	No effect on growth or fruiting	–	MacLean et al. (1977)
Soybean	0.64, 2.1, or 5.0 µg F/m^3, 10–16 h/d for 98 d	Reduced number of fruit per pot by more than 90%	<0.64	Pack and Sulzbach (1976)
Bell pepper	0.01, 0.63, 2.2, 4.5, or 10 µg F/m^3, 10–16 h/d for 112 d	Reduced number of peppers by more than 65%	2.2	Pack and Sulzbach (1976)
Sorghum	0.01, 0.53, 2.2, 4.7, or 10.6 µg F/m^3, 10–16 h/d for 114 d	Slightly reduced weight per seed; at 4.7 µg F/m^3, number of seeds reduced by 85%	2.2	Pack and Sulzbach (1976)
Sweet corn	0.01, 0.54, 2.0, 2.3, or 8.7 µg F/m^3, 10–16 h/d for 97 d	Seed production totally (100%) inhibited (after anthers released, ears and seeds did not develop)	2.0	Pack and Sulzbach (1976)
Cucumber	0.01, 0.61, 2.3, 4.4, 4.6, 5.5, 7.8, or 8.9 µg F/m^3, 10–16 h/d for 104 d	Reduced number of fruit by 24%	4.6	Pack and Sulzback (1976)
Pea	0.01, 2.1, 4.4, 5.3, or 9.0 µg F/m^3, 10–16 h/d for 56 d	Reduced number of seeds per fruit by approximately 5%	4.4	Pack and Sulzback (1976)
Wheat	0.01, 5.0, or 8.2 µg F/m^3, 10–16 h/d for 130 d	Reduced number of seeds by 50%; reduced weight per seed by 18%	8.2	Pack and Sulzback (1976)
Oat	0.01, 2.2, 4.3, or 9.1 µg F/m^3, 10–16 h/d for 147 d	Reduced seed production (proportion not specified)	9.1	Pack and Sulzbach (1976)
Cotton	0.01, 3.1, 5.0, or 8.0 µg F/m^3, 10–16 h/d for 164 d	No significant differences for all measured parameters	>8	Pack and Sulzbach (1976)
Snow princess gladiolus (*Gladiolus grandiflorus*)	0.03, 0.35, 0.36, 0.41, 0.44, 0.50 and higher up to 1.85 µg F/m^3 for up to 117 d	Leaf necrosis (65% of leaves); 117 d	0.36	Hill and Pack (1983)
Freesia sp. (commercial flower)	Continuous fumigation at 0.5 µg HF/m^3 for 5 mo OR intermittent fumigation with 0.3 µg HF/m^3 (6 h/d, 3 or 4 times/wk) for 18 wk	Leaf necrosis over 30% of exposed leaf surface area compared with 5% in control plants	0.3	Wolting (1975)
Gladiolus sp. (commercial flower)	0.35 or 0.76 µg F/m^3 for 40 d	Increased necrosis by 46% and increased respiration by 39%	0.76	Hill et al. (1959)
Apple (*Malis domestica* Borkh)	0.03, 0.44, 0.82 µg HF/m^3 for 164 d	Slightly reduced growth and necrosis (see text)	0.44	Hill and Pack (1983)
Pole bean (*Phaseolus vulgaris*)	0.03, 0.54, 0.79 µg HF/m^3 for 83 d	Fruit set reduced by 80%	0.54	Hill and Pack (1983)

Species Tested	Study Protocol	Results	LOEL (µg/m³)	Reference
Grapevine (*Vitis vinifera*)	0.07 (control), 0.17, or 0.27 µg/m³ for 189 d	Foliar necrosis after 99 and 83 d at 0.17 and 0.27 µg/m³, respectively; reduced chlorophyll a and total chlorophyll noted	0.17	Murray (1984)
Grapevine (3 varieties)	0.37 µg F/m³ to 6.0 µg F/m³ for four growing seasons (season duration varied from 54–159 d) for 12 different exposure concentration/duration combinations	No substantial effects up to 1.5 µg F/m³ for 54 d; exposure at 2.2 µg F/m³ for 60 d reduced leaf area by up to 45%	2.2	Doley (1986)
Wheat (*Triticum aestivum*)	0.03 or 0.38 µg HF/m³ for 90 d	No effects on yield	–	Murray and Wilson (1988a)
Barley (*Hordeum vulgare*)	0.03 or 0.38 µg HF/m³ for 90 d	Increase in grain protein concentration; not necessarily an adverse effect	0.38	Murray and Wilson (1988a)
Tendergreen bean plant	0.58, 2.1, 9.1, or 10.5 µg F/m³ seedling to maturity to next generation	At 2.1 µg F/m³, lower starch content of seeds (15–21%) compared with controls (35% starch) resulting in reduced F_1 generation plant height (−17%) and leaf surface area (−23%) and increased (+137%) proportion abnormal trifoliate leaves	2.1[a]	Pack (1971)
Eucalyptus (*Eucalyptus tereticornis*)	0.03 or 0.38 µg F/m³ for 90 d in open-top chambers	Reduced leaf surface area and weight in mature and immature leaves	0.38	Murray and Wilson (1988b)
Marri (*E. calophylla*)	0.03 or 0.39 µg F/m³ for 120 d	Reduced leaf surface area and weight in immature leaves, reduced surface area in mature leaves	0.39	Murray and Wilson (1988c)
Tuart (*E. gomphocephala*)	0.03 or 0.39 µg F/m³ for 120 d	Reduced leaf surface area and weight in mature and immature leaves	0.39	Murray and Wilson (1988c)
Jarrah (*E. marginata*)	0.03 or 0.39 µg F/m³ for 120 d	Reduced leaf surface area and weight in immature leaves only	0.39	Murray and Wilson (1988c)

[a]Primary leaves of some F1 progeny noted as being severely stunted and distorted at 2.1 µg/m³ (dosing protocol unclear).

Although many plant species do not exhibit adverse effects from short-term exposures at ambient air concentrations less than 1 µg F/m^3, several do show effects after longer-term exposures at

concentrations of 0.5 µg F/m^3 or less (Table A-11).[15] Several studies of plants of agricultural importance are described in more detail below.

Pack and Sulzbach (1976) fumigated nine species of agricultural crops with HF gas from seed through flowering to the time of harvest. Table A-11 lists the exposure concentrations and the exposure durations associated with the LOEL concentration for each crop. The crop that was most sensitive to HF was soybean, with a 90-percent reduction in the number of bean pods at the lowest exposure concentration tested (0.64 µg F/m^3). The next most sensitive crop appears to be sweet corn. Although no effects other than brown streaks through the plant leaves were observed at 0.54 µg F/m^3, at the next higher exposure concentration (2.0 µg F/m^3), ears and seeds failed to develop in all corn plants. Cotton was the most resistant to fumigation with HF of the plants tested, with no effects observed at a concentration of 8.0 µg F/m^3 for 164 days.

Hill and Pack (1983) grew apples (1-year-old whips of the delicious variety) in three greenhouses, starting HF exposures 5 weeks after planting and continuing for 164 days.

Air was filtered in two greenhouses to remove gaseous (and particulate) fluoride. One of those greenhouses served as a "clean air" control (0.03 µg HF/m^3), while HF was added to another greenhouse to achieve an air concentration of 0.44 µg HF/m^3. The third greenhouse received ambient air with an average concentration of 0.83 µg HF/m^3. The group exposed at 0.44 µg HF/m^3 exhibited an 11-percent reduction in leaf length and a 6-percent reduction in leaf width ($p < 0.01$) compared with the control. In addition, leaves exposed during their expansion sporadically exhibited leaf tip necrosis and chlorosis, with leaf growth ceasing once necrosis was visible. Leaf injury often was apparent soon after 24-h air sample readings of up to 0.99 µg HF/m^3.

Hill and Pack (1983) also examined the response of Chinese apricot trees fumigated with HF during three growing seasons (Experiments A, B, and C). Experiments A and B used higher exposure concentrations over shorter durations than did experiment C. In trials B and C, both ambient air and test air HF concentrations were 0.35 µg HF/m^3, and the "clean" air greenhouse (at 0.03 µg HF/m^3) served as the control. Trees were exposed as soon as they began to develop

[15]Air concentrations are variously reported as µg HF/m^3 or µg F/m^3. We report the original units without adjusting one to the other. The atomic weight of F is approximately 95% of the molecular weight of HF.

leaves. Necrosis of leaf tips and edges, necrotic spots on leaves, leaf curling, and increased leaf drop were observed. In Experiment C, after 117 days of exposure at 0.35 µg HF/m^3, leaf drop averaged 18 percent, average tree trunk diameters were 53 percent that of controls, and average shoot length was 54 percent that of controls.

Peach trees exposed to gaseous HF under conditions similar to those described above appeared to be even more sensitive to HF. Specifically, leaves of HF-exposed peach trees tended to be smaller than those of controls and also tended to drop prematurely (Hill and Pack 1983). In one part of the study, leaves on trees exposed at 0.41 µg HF/m^3 for 73 days were 24-percent smaller than leaves on control trees. In another part of the study, 1,768 leaves dropped from trees exposed at 0.34 µg HF/m^3 for 110 days, while only 102 leaves dropped from the control trees.

Pack (1971) evaluated effects on tendergreen bean plants grown from seeding to maturity under continuous exposure to HF gas at 0.58, 2.1, 9.1, or 10.5 µg F/m^3. No significant growth or yield effects were observed at any test concentration, with the exception of a 15- to 21-percent reduction in bean starch content at the three highest concentrations tested. Beans from the exposed parental generation (F0) then were planted and grown in "clean" air to produce the F1 generation. For the plants exposed at 2.1 µg F/m^3, the F1 generation plants exhibited a 17-percent reduction in plant height and a 23-percent reduction in leaf surface area. Subsequent plantings of F2 and F3 generations (grown in clean air) indicated that the traits exhibited in the F1 generation were not heritable.

Murray and Wilson (1988c) evaluated adverse effects from 120 days of HF exposure for three eucalyptus species by conducting an analysis of variance for the exposed (0.39 µg HF/m^3) versus control plants (background concentration of 0.03 µg HF/m^3) for several parameters. For immature leaves, reduced leaf area and reduced leaf weight were significant at $p = 0.001$ for *Eucalyptus calophylla*. For *E. marginata*, reduced immature leaf area was significant at $p = 0.01$, and reduced immature leaf weight was significant at $p = 0.05$. For *E. gomphocephala*, reduced immature leaf area was significant at $p = 0.05$, and reduced immature leaf weight was significant at $p = 0.01$. In contrast, for mature leaves, only *E. gomphocephala* showed both significantly reduced leave area ($p = 0.01$) and weight ($p = 0.001$).

Murray and Wilson (1988c) also estimated visible foliar injury for three eucalyptus species using two factors: "*A*" (the proportion of necrotic leaf area on damaged leaves) and "*L*" (the proportion of all damaged leaves). The injury index (*I*) formula then was calculated using Equation A-1:

$$I = (A \times L)^{0.5} \quad \text{Eq. A-1}$$

Analysis of variance for the exposed plants compared with control plants indicated that *E. calophylla* was significantly affected at 0.39 µg/m^3 (p = 0.001). Murray and Wilson (1988c) did not report actual measurements for leaf area, weight, or necrotic leaves.

Considering the data as a whole, a benchmark of 0.4 or 0.5 µg HF/m^3 air would appear protective of most plant species included in the table, but not some species of commercial flowers or ornamental plants (see gladiolus Table A-10) and freesia, grapevine, and eucalyptus (Table A-11). Streaking of leaves is an adverse effect for plants bred for their appearance. Thus, air HF concentration benchmark of 0.4 or 0.5 µg HF/m^3 air appears consistent with a TEL for assessing plant communities for wildlife food and habitat and for agricultural crops. Some species of HF-sensitive ornamental plants would not be protected at that level.

A.3 Wildlife Toxicity Reference Values

To assess risks to piscivorous wildlife, a TRV for wildlife, expressed as an oral dose, is needed for comparison with estimated dietary exposures via the chemical in prey (i.e., in fish and invertebrates consumed). The estimated total chemical intake via all types of prey in the diet, expressed as mg[chemical]/kg[wildlife body weight]/day (mg/kg-day), can be compared with the TRV (expressed in the same units) to estimate a hazard quotient. An emission rate that corresponds to a hazard quotient of 1.0 (i.e., the emissions screening threshold rate) then is used to screen facilities in Tiers 1 through 3 of the RTR ecological risk environmental screens.

Avian and mammalian TRVs are included in the RTR ecological assessment in two contexts. The first is in OSWER's derivation of Eco-SSLs, expressed as chemical concentrations in soil, to protect wildlife that feed on soil invertebrates (Section A.2.2.3 and Section A.3.2). The second is use of TRVs, expressed as chemical doses to avian and mammalian wildlife (mg/kg-day), to compare with their estimated ingestion of chemicals in fish from the onsite lake. These TRVs are calculated in this section using an approach similar to that used for the EPA GLWQI (U.S. EPA

1995b) (Section A.2.2.1). One exception is allometric scaling of dose from a test animal to dose for the wildlife species based on relative body weights instead of using an interspecies UF of 10.

An interspecies UF generally has two components: a toxicokinetic component and a toxicodynamic component (U.S. EPA 2005a, 2011a). The toxicokinetic component generally can be represented by scaling the toxicity value for the test species to the assessment species on the basis of relative body weights to the ¾ power. That scaling is based on the allometric relationship of metabolic rate to body weight for mammals in general (U.S. EPA 1993c) and assumes that much of the toxicokinetic difference among species scales to metabolic rate. Toxicodynamic differences are associated with taxonomic differences in physiology between the test and assessment species that might affect sensitivity to a toxicant. Such differences generally increase in magnitude with increasing taxonomic distance (Brown et al. 2000); for example, rodents might be more sensitive to some plant toxins than ungulates such as cows or goats or other herbivores that have evolved metabolic pathways to detoxify those compounds.

Given the maximum value for the interspecies UF is 10, a common recent EPA practice has been to assign each component, toxicokinetic and toxicodynamic, a UF of 3 (U.S. EPA 2011). Thus, if toxicokinetic differences are accounted for by scaling to relative body weight, the maximum value of the remaining UF would equal 3 (i.e., $3 \times 3 = 9$; close to 10). The approach is consistent with that used most recently by EPA to estimate reference doses (RfDs) for humans (U.S. EPA 2011) and that EPA has used for some time in estimating cancer potency factors for humans from animal data (U.S. EPA 2005a). For purposes of clarity and simplicity, however, we did not apply a UF of 3 if the test species taxon differs from the assessment species taxon at the level of order (e.g., test species is a rodent [rat] and assessment species is a carnivore [mink]; both are mammals).

A.3.1 Derivation of TRVs for Piscivorous Wildlife in RTR Assessment

As described in Section 3.1.1 of the main report, we selected mink (*Mustela vison*, recently renamed *Neovison vison* based on cytogenetic and biochemical data that distinguish it from other members of the genus *Mustela*; Wozencraft 2005) to represent fish-eating mammals and common (American) merganser (*Mergus merganser americana*) to represent fish-eating birds. These two species are of moderate size (moderate metabolic rate and FIR per kg body weight), but can catch and consume larger fish than other moderate-sized mammals or birds, respectively.

The TRVs used to assess risk to piscivorous wildlife for Tiers 1 through 3 of the RTR ecological risk screen were calculated for the RTR assessment using the methods developed for the GLWQI (U.S. EPA 1995b). In the GLWQI approach, the most sensitive type of effect of the most sensitive of the species tested is used to identify a LOAEL and NOAEL, which then can be used to calculate TRVs. For most chemicals, only a few (e.g., 2–7) species of birds (e.g., quail, mallard, chicken, pheasant) and a few species of mammals (e.g., mice, rats, hamster, mink) have been tested sufficiently to provide both a LOAEL and a NOAEL for effects resulting from chronic exposures. Thus, using toxicity data from the most sensitive of the few species tested is not necessarily an overly protective approach.

For wildlife, chronic TRVs were derived after reviewing the following sources of toxicity study summaries:

- chronic (or reproductive) toxicity studies of mammals and birds as compiled by EPA for the GLWQI (U.S. EPA 1995b),
- studies compiled by Sample et al. (1996),
- studies reported in Eco-SSL documents for individual chemicals (i.e., cadmium, U.S. EPA 2005d), or
- studies identified by a literature search for TRVs or toxicity benchmarks for the six PB-HAPs for birds and mammals (e.g., CA DTSC HERD 2009).

For each source listed above, the author(s) had evaluated the individual toxicity study reports for scientific adequacy. We used the study for the most sensitive species showing an adverse effect on survival, growth and development, or reproduction to identify the lowest LOAEL and lowest NOAEL for use as wildlife TRVs. When not available, a NOAEL was set equal to the LOAEL divided by a UF of 10 (U.S. EPA 1995b). For TRVs obtained from the GLWQI documentation, we used the LOAELs and NOAELs from the key study without application of any UFs, which is consistent with Sample et al. (1996), maximizes clarity, and minimizes the number of assumptions used in developing the TRVs.

To estimate TRVs for the RTR piscivorous wildlife risk screen, we used the LOAELs and NOAELs from a single key study (most sensitive effect and species). Doses were scaled between a test species and the assessment species on the basis of relative body weight to the ¾ power (U.S. EPA 2011), as described below if the difference in body weight was more than 20 percent.

For mammals, for which the test species, usually rats (350 g) or mice (30 g), is of smaller body size and higher metabolic rate than mink (1,000 g), dose conversions from the test animal to mink were based on allometric scaling of metabolic rate between mammalian species (U.S. EPA 1993c,1995b; Equation A-2 below):

$$Dose_{wildlife} = Dose_{test\text{-}species} \times (BW_{test\text{-}species}/BW_{wildlife})^{1/4} \quad \text{Eq. A-2}$$

where

Dose = chemical ingestion (mg[chemical]/kg[wildlife BW]-day)

BW = body weight

For birds, given the similarity of the body weight of test species (e.g., chicken, pheasant, mallard duck about 1 kg) to American merganser (1.27 kg), no dose conversions were performed.

A.3.2 Chemical-specific Wildlife TRVs for PB-HAPs

In the main report, Table 3-1 lists the TRVs, both the NOAEL and the LOAEL, used for fish-eating birds and mammals for each PB-HAP. Further details are provided below. The discussion for arsenic demonstrates our approach. The remaining derivations are described more briefly.

A.3.2.1 Arsenic (As) Wildlife TRVs

Data were available to calculate a TRV for both (1) mink (*Mustela vison, or Neovison vison*) and (2) American merganser (*Mergus merganser americana*).

Mink Arsenic (As) TRV

We based our wildlife TRV for arsenic toxicity to mink on a three-generation study of mice. Schroeder and Mitchener (1971) administered a soluble arsenite (AsO_3^{-3}) salt in drinking water of mice at 3 ppm (or 5 mg[As]/L). They found a statistically significantly reduced litter size (25 percent, 8 percent, and 23 percent for generations 1, 2, and 3, respectively) for female mice ingesting the arsenite in drinking water. Because arsenic is naturally occurring, feed for both control and experimental mice contained 0.06 ppm arsenic. Sample et al. (1996) calculated the dose at the LOAEL to be 1.26 mg[As]/kg[mouse body weight]-d. To estimate a NOAEL, the LOAEL is divided by a UF of 10 (GLWQI, U.S. EPA 1995b).

LOAEL for mouse = 1.26 mg[As]/kg-day

NOAEL for mouse = LOAEL/10

= 0.126 mg[As]/kg-day

LOAEL for mink = daily dose to mouse × (mouse body weight/mink body weight)$^{1/4}$

= 1.26 mg/kg mouse/day × (0.03 kg[mouse]/1 kg[mink])$^{1/4}$

= 0.52 mg[As]/kg-day

NOAEL for mink = 0.052 mg[As]/kg-day

American Merganser Arsenic (As) TRV

As described above for the Eco-SSL for birds, EPA identified an arsenic TRV for birds based on one of four toxicity studies of acceptable quality that examined growth and reproduction (U.S. EPA 2005b). Of those four studies, one reported NOAELs for both reproduction and growth at 2.24 mg/kg[body weight]-day in domestic chickens (Holcman and Stibilj 1997, cited by U.S. EPA 2005b). Although a study of Camardese et al. (1990) identified a lower LOAEL of 1.49 mg/kg-day for growth for mallard duck, EPA did not use it to estimate a TRV because a NOAEL was not determined in that study (U.S. EPA 2005b).

We are concerned that mallards might be more sensitive to arsenic than domestic chickens. Camardese et al. (1990) exposed mallard ducklings to arsenic in food from days 1 through 14 after hatching. Although four doses were administered, effects on growth were seen at the lowest dose tested (1.49 mg/kg-day). For purposes of RTR screening, we use the mallard duck LOAEL. The NOAEL is estimated as the LOAEL divided by a UF of 10 (and rounded to 2 significant digits).

LOAEL for mallard duck = 1.5 mg[As]/kg-day

NOAEL for mallard duck = 0.15 mg[As]/kg-day

Given the similarity in size between mallard (1 kg) and American merganser (1.27 kg), no dose conversions were estimated:

LOAEL for merganser = 1.5 mg[As]/kg-day

NOAEL for merganser = 0.15 mg[As]/kg-day

A.3.2.2 Cadmium (Cd) Wildlife TRVs

Data were available to calculate a TRV for both mink and American merganser.

Mink Cadmium (Cd) TRV

Sample et al (1996) identified the study of Sutou et al. (1980), which reported a NOAEL and a LOAEL for reproduction in rats. Sutou et al. (1980) exposed female rats (mean body weight 0.30 kg) to cadmium for 6 weeks (from mating through gestation) by oral gavage adjusted to body weight to attain three doses: 0.1, 1.0, and 10.0 mg[Cd]/kg[body weight]-day. At the LOAEL of 10 mg/kg-day, fetal implantations were reduced 28 percent, fetal survivorship was reduced by 50 percent, and fetal resorptions increased by 400 percent; the NOAEL in the experiment was 0.1 mg/kg-day.

NOAEL for mink = daily dose to rat × (rat body weight / mink body weight)$^{1/4}$

= 1 mg/kg[rat]-day × (0.30 kg[rat] /1 kg[mink])$^{1/4}$

= 0.74 mg/kg-day

LOAEL for mink = 10 × NOAEL

= 7.4 mg/kg-day

Common Merganser Cadmium (Cd) TRV

For the Eco-SSLs to protect ground-feeding birds, EPA calculated the geometric mean of all bounded NOAELs for reproduction and growth across several species of birds to estimate a TRV of 1.47 mg/kg-day (U.S. EPA 2005d). In 2009, the EPA Region 9 BTAG reevaluated the Eco-SSL TRV for cadmium considering data published after 2004 and using revised allometric equations (Nagy 2001) to estimate FIRs rather than the earlier equations (Nagy 1987) used for the Eco-SSL TRVs.

Based on kidney toxicity in mallards (Cain et al. 1983), EPA Region 9 and the California Department of Toxic Substances Control, Human and Ecological Risk Division recommended an avian LOAEL of 1.0 mg/kg-day (U.S. EPA 2009b; CA DTSC HERD 2009). Cain et al. (1983) reported mild-to-severe kidney degeneration in four growing mallard ducklings fed 14.6 ppm cadmium in their diet for 12 weeks, which they calculated to equal an ingested dose of 1.0 mg/kg-day. Other studies also identified potential reproductive effects near that dose (White et al. 1978; Leach et al. 1979). EPA Region 9 and CA DTSC HERD identified a NOAEL of

0.7 mg/kg-day from a study by Mayack et al. (1981) that identified kidney damage in wood ducks after 3 months of exposure to 7 mg/kg-day, but not in wood ducks exposed at 0.68 mg/kg-day (which we round to 0.7 mg/kg-day).

We follow EPA Region 9 and use the lowest LOAEL of 1 mg/kg-day from Cain et al. (1993) and the highest NOAEL of 0.7 mg/kg-day from Mayack et al. (1981) for merganser TRVs for cadmium (Table 5-1 in the main report). Given the similarity in size between mallard (1 kg) and American merganser (1.27 kg), no dose conversions were estimated.

A.3.2.3 Divalent Mercury (Hg^{++}) Wildlife TRVs

We did not calculate TRVs for mink and American merganser for divalent mercury because it is not bioaccumulative. Instead, we focused on methyl mercury for fish-eating wildlife (Section A.3.2.4).

A.3.2.4 Methyl Mercury (MeHg) Wildlife TRVs

Data were available to calculate a TRV for both (1) mink and (2) American merganser.

Mink Methyl Mercury (MeHg) TRV

Verschuuren et al. (1976) exposed rats to methyl mercury chloride (which is 79.89% Hg by weight) at doses of 0.1, 0.5, and 2.5 ppm in the diet for three generations. Reduced pup survival was observed at 2.5 ppm MeHgCl, but not at the lower dietary concentrations. The exposure level of 0.5 ppm MeHgCl, or 0.4 mg[Hg]/kg[diet], is considered the NOAEL. Sample et al. (1996) calculated the doses for the rat assuming a rat body weight of 0.35 kg and FIR of 28 g/day (U.S. EPA 1988a). The calculations below are based on doses expressed in mg Hg, not MeHg or MeHgCl, per kg body weight per day.

NOAEL for rat	=	(concentration in food × FIR/day) / body weight
	=	(0.4 mg[Hg]/kg[food] × 28 g[food]/day) / 0.35 kg[rat body weight]
	=	0.032 mg[Hg]/kg-day
LOAEL for rat	=	(2.0 mg[Hg]/kg[food] × 28 g[food]/day) / 0.35 kg[rat body weight]
	=	0.16 mg[Hg]/kg-day

NOAEL for mink = daily dose to rat × (rat body weight / mink body weight)$^{1/4}$

= 0.032 mg[Hg]/kg[rat]-day × (0.35 kg[rat] / 1 kg[mink])$^{1/4}$

= 0.0246 mg[Hg]/kg-day

LOAEL for mink = 0.16 mg[Hg]/kg[rat]-day × (0.35 kg[rat] / 1 kg[mink])$^{1/4}$

= 0.123 mg[Hg]/kg-day

American Merganser Methyl Mercury (MeHg) TRV

Heinz (1974, 1975, 1976a,b, and 1979) identified a LOAEL for reduced production of eggs and ducklings for mallards exposed for up to three generations to MeHg added to the diet as methyl mercury dicyandiamide at 0.5 ppm Hg. EPA estimated the average daily dose to the mallards at that dietary concentration to be 0.078 mg[Hg]/kg bw-day (U.S. EPA 1995b). Sample et al. (1996) also calculated a dose to mallards from 0.5 ppm Hg in the diet to be 0.064 mg[Hg]/kg bw-day based on slightly different assumptions about body weight and FIR than used by EPA. To estimate a NOAEL, EPA used a compound UF of 6: a UF of 2 for the LOAEL-to-NOAEL extrapolation, because the effect level at the LOAEL was slight, and a UF of 3 for interspecies extrapolation (U.S. EPA 1995b).

LOAEL for mallard = 0.078 mg[Hg]/kg-day

NOAEL for mallard = 0.078 mg[Hg]/kg-day / 6 (UF)

= 0.013 mg[Hg]/kg-day

LOAEL for merganser = 0.078 mg[Hg]/kg-day

NOAEL for merganser = 0.013 mg[Hg]/kg-day

A.3.2.5 POM Index Chemical—Benzo[a]pyrene (BaP) Wildlife TRV

Data were available to calculate a TRV only for mink; insufficient data were identified to estimate a TRV for birds.

Mink Benzo[a]pyrene (BaP) TRV

Mackenzie and Angevine (1981) exposed female mice to BaP during days 7 to 16 of gestation via oral intubation. Exposure doses were 10, 40, and 160 mg/kg-day. Total sterility occurred in 97 percent of offspring in the 40- and 160-mg/kg-day groups; fertility was impaired in offspring at the lowest exposure dose tested of 10 mg/kg-day (Sample et al. 1996). To estimate a NOAEL for the mouse from the unbounded LOAEL, the LOAEL is divided by a UF of 10 (U.S. EPA 1995b).

LOAEL for mouse	=	10 mg/kg-day
NOAEL for mouse	=	1 mg/kg-day
LOAEL for mink	=	daily dose to mouse × (mouse body weight/mink body weight)$^{1/4}$
	=	10 mg/kg[mouse]/day × (0.03 kg[mouse] / 1 kg[mink])$^{1/4}$
	=	4.17 mg/kg-day
NOAEL for mink	=	0.417 mg/kg-day

Common Merganser Benzo[a]pyrene (BaP) TRV

In the absence of data for BaP toxicity to birds, we did not estimate TRVs for birds for BaP.

A.3.2.6 Dioxins Index Chemical—2,3,7,8-TCDD Wildlife TRV

Data were available to calculate a TRV for both (1) mink and (2) American merganser.

Mink 2,3,7,8-TCDD TRV

Using a three-generation study design, Murray et al. (1979) identified a NOAEL and LOAEL for reproductive effects of 2,3,5,8-TCDD in rats of 0.001 and 0.01 µg/kg-day (U.S. EPA 1995b, Sample et al. 1996):

NOAEL for rat	=	0.001 µg/kg-day
LOAEL for rat	=	0.01 µg/kg-day
NOAEL for mink	=	daily dose to rat × (rat body weight / mink body weight)$^{1/4}$
	=	0.000001 mg/kg[rat]/day × (0.35 kg[rat] / 1 kg[mink])$^{1/4}$
	=	7.71E-07 mg/kg-day
LOAEL for mink	=	7.71E-06 mg/kg-day

American Merganser 2,3,7,8-TCDD TRV

No avian toxicity studies are available for TCDD administered orally. We did identify an intraperitoneal (i.p.) injection study. Nosek et al. (1992, 1993) dosed female ring-necked pheasants (*Phasianus colchicus*) one time per week for 10 weeks by i.p. injection. The equivalent average daily doses were 0.14, 0.014, and 0.0014 µg/kg-day. This route of administration ensures "uptake" of the complete dose and avoids the "first pass through the liver." We investigated, however, the possibility that oral administration can result in lower uptake.

Available data indicate no differences in gastrointestinal tract absorption of dioxins across taxonomic groups of mammals and some birds (van den Berg et al. 1984). Moreover, uptake of 2,3,7,8-TCDD by mammals following oral administration appears high, ranging from 75 percent (hamster) to >86 percent (humans), with absorption depending on the oil content of the vehicle (van den Berg et al. 1984). In mammals, the tissue distribution of administered 2,3,7,8-substituted polychlorinated dibenzo-p-dioxins (PCDDs) and 2,3,7,8-substituted polychlorinated dibenzofurans (PCDFs) following i.p. and subcutaneous administration is similar to that following oral administration, with the highest proportion of the dose retained in the liver and in adipose tissues (van den Berg et al. 1984). Based on that information, we conclude that i.p. administration can represent ingestion toxicity.

The NOAEL and LOAEL values for egg production by pheasants administered 2,3,7,8-TCDD were 0.014 and 0.14 µg/kg-day, respectively (Nosek et al. 1992, 1993, U.S. EPA 1995b). Given the similarity in size between female ring-necked pheasant (0.9 to 1.1 kg) and American merganser (1.3 kg), no dose conversions were estimated:

NOAEL for merganser = 1.4E-05 mg/kg-day

LOAEL for merganser = 1.40E-04 mg/kg-day

A.4 Derivation of Ecological TEFs for POM and Dioxin Benchmarks

Section A.4.1 covers the derivation of TEFs for POM relative to BaP for the benchmarks for surface waters, sediments, and soils. If the POM is more toxic than BaP, the POM's benchmark would be lower than the benchmark for BaP, and the TEF would be greater than 1.0. If the POM is less toxic than BaP, the POM's benchmark would be higher than the benchmark for BaP, and the TEF would be less than 1.0

Section A.4.2 covers the derivation of TEFs for dioxins relative to 2,3,7,8-TCDD. If the congener is less toxic than TCDD, the TEF would be less than 1. All TEFs for dioxins are less than or equal to 1.

A.4.1 TEFs for POM for Surface Water, Sediments, and Soils

Table A-12 lists the TEFs for POM compounds relative to BaP. Physical and chemical properties of the unsubstituted PAHs, and their toxic mode of action (MOA), tend to be similar, with values for some parameters, including toxic potency, changing predictably with the number of aromatic

rings and the configuration of those rings (e.g., compact, elongated). With any substitutions (e.g., alkyl groups, alcohol groups, chlorine or bromine atoms) or with noncarbon atoms (e.g., nitrogen) included in five-carbon non-aromatic rings, the MOA can change for some groups of organisms (e.g., crustaceans, insects, algae, terrestrial plants) in ways that would not be predicted on the basis of other groups (e.g., aquatic or terrestrial vertebrates).

Table A-12. Toxicity Equivalency Factors (TEFs) for Surface Waters, Soils, Sediments, and Mammalian Wildlife—POM Compounds Relative to BaP

Polyaromatic Organic Matter	CAS RN	Surface Water TEF	Soil TEF	Sediment TEF	Mammalian TEF
1-Methylnaphthalene	90-12-0	0.000042	0.47	7.4	0.014
2-Acetylaminofluorene	53-96-3	0.000026	2.55	9.79	1
2-Chloronaphthalene [beta]	91-58-7	0.0354	125	0.36	1
2-Methylnaphthalene	91-57-6	0.000042	0.470	7.4	0.014
3-Methylcholanthrene	56-49-5	0.16	19.5	0.000018	1
7,12-Dimethylbenz[a]anthracene	57-97-6	0.026	0.093	0.002	1
Acenaphthene	83-32-9	0.00037	0.002	22.4	0.0057
Acenaphthylene	208-96-8	0.000003	0.002	25.6	0.056
Anthracene	120-12-7	0.40	0.001	2.62	0.001
Benz[a]anthracene	56-55-3	0.56	0.292	1.39	1
Benz[a]anthracene/Chrysene	NA	0.1	0.1	0.1	0.1
Benzo[a]fluoranthene	203-33-8	0.0015	0.025	0.014	7.5
Benzo[c]phenanthrene	195-19-7	0.42	0.292	1.4	1
Benzo[g,h,i]fluoranthene	203-12-3	0.71	0.292	1.39	1
Benzo(g,h,i)perylene	191-24-2	0.0018	0.013	0.882	1
Benzo[a]pyrene	50-32-8	1	1	1	1
Benzo[b]fluoranthene	205-99-2	0.0015	0.025	0.014	4.4
Benzo[b+k]fluoranthene	NA	0.0015	0.010	0.625	7.5
Benzo[e]Pyrene	192-97-2	3	1	1	1
Benzo[j]fluoranthene	205-82-3	0.0015	0.01	0.625	7.5

Polyaromatic Organic Matter	CAS RN	Surface Water TEF	Soil TEF	Sediment TEF	Mammalian TEF
Benzo[k]fluoranthene	207-08-9	0.0015	0.010	0.625	7.5
Carbazole	86-74-8	0.0054	0.2	0.36	0.008
Chrysene	218-01-9	0.0020	0.321	0.904	6
Dibenzo[a,h]anthracene	53-70-3	0.0028	0.083	4.54	2
Dibenzo[a,i]pyrene	189-55-9	0.003	0.14	0.8	1
Dibenzo[a,j]acridine	224-42-0	0.43	0.29	1.4	0.1
Fluoranthene	206-44-0	0.0074	0.012	0.355	0.0067
Fluorene	86-73-7	0.00074	0.012	1.94	0.008
Indeno[1,2,3-c,d]pyrene	193-39-5	0.0032	0.014	0.75	1
PAH, Total	NA	0.1	0.1	0.1	0.1
Perylene	198-55-0	2	1	1	1
Phenanthrene	85-01-8	0.0039	0.033	0.735	0.001
Polycyclic organic matter	NA	0.1	0.1	0.1	0.1
Pyrene	129-00-0	0.047	0.019	0.77	0.013
Retene	483-65-8	0.015	0.093	0.002	1

Note: If the TEF is less than 1, the chemical is not as toxic to organisms in that medium as is BaP (in bold). If the TEF is equal to or greater than 1, the chemical is as toxic or more toxic to organisms in that medium as BaP.

Most research on toxic effects of PAHs and similar POM in the United States has focused on their mutagenic and carcinogenic potential in mammals, with several being known human carcinogens. As stated in the introduction, cancer is not an endpoint pursued for wildlife and nonvertebrate animal risk assessment (CCME 2010); thus, TEFs based on carcinogenic potency of POM relative to BaP are not applicable to ecological risk assessments.

General considerations for deriving TEFs for POM for surface water, soil, and sediment in Table A-12 are described in Sections A.4.1.1 through A.4.1.5. Chemical-specific derivations are described in Section A.4.1.6.

A.4.1.1 Data Retrieval and Comparison to Estimate TEFs for POM

To develop TEFs for POM relative to BaP, we first consulted ORNL RAIS to identify any benchmarks available for POM compounds. We used the same hierarchy of preferred data

sources as described in Section A.2.1.4. When available, if the source was the same source that we used for the BaP benchmark for the particular environmental medium (e.g., surface water, sediment, soil), we could compare the POM benchmark to the BaP benchmark to calculate a TEF.

For some POM chemicals, EPA OW or an EPA region had calculated sediment quality criteria using the using the equilibrium partitioning (EqP) approach. That approach includes several assumptions about environmental characteristics, as explained below.

The EqP approach estimates pore-water concentrations of a chemical assuming an equilibrium between the chemical adsorbed to organic carbon in the sediments and the chemical freely dissolved in the pore water. The model uses a chemical-specific surface water quality benchmark (WQB), such as an NAWQC-ALC, and an organic carbon partitioning factor (K_{oc}), which is based on experimentally measured value(s) or can be estimated using Equation A-3 from an experimentally measured octanol-water partitioning coefficient(s) (K_{ow}) for the specified chemical (provided suitable empirical data are available):

$$SQB = f_{oc} \times K_{oc} \times WQB \qquad \text{Eq. A-3}$$

where

SQB	=	chemical-specific sediment quality benchmark
f_{oc}	=	fraction total organic carbon (TOC) in sediments
K_{oc}	=	chemical-specific organic carbon/water partition coefficient
WQB	=	chemical-specific water-quality benchmark for the protection of water-column biota

The EqP model requires the risk assessor to assign a total organic carbon (TOC) concentration in sediments using site-specific measurements or to use values typical of certain types of water bodies [e.g., data presented in U.S. EPA (2003d) are from sediments with 0.201 to 15.2 percent organic carbon]. For a regional or nationwide environmental screen, however, a common approach is to assume a relatively low TOC value to maximize the chemical's bioavailability. Several EPA regions and Environment Canada have adopted the 1-percent TOC value used by Jones et al. (1997) for DOE. Using that assumption, one can calculate a chemical-specific SQB

as the total concentration of the chemical in sediment (on a dry-weight basis) that would produce a sediment pore-water concentration equal to the WQB.

TRIM.FaTE assumes a TOC for sediments of 2 percent. Had a TOC of 2 percent been used instead of 1 percent with the EqP model to estimate SQBs for the non-ionic organic chemicals, the EPA-calculated SQB values would have been higher (i.e., less conservative) by a factor of 2. In that case, the ratio of TRIM.FaTE-predicted sediment concentrations to the sediment benchmarks would have been lower, meaning that more facilities might have passed the tiered screening (i.e., be removed from further consideration).

For many POM chemicals, however, no benchmarks are included in RAIS. We next consulted the EPA NAWQC, the Eco-SSLs, and the EPA regional benchmark compilations to identify benchmarks for POM that might not have been included in ORNL RAIS. When those efforts failed, we finally sought individual toxicity study entries in EPA's ECOTOX database. Data presentation in ECOTOX is difficult to interpret because the results from one experiment are presented as separate records depending on the endpoint (EC_{10}, EC_{50}, NOEC, LOEC, LC_{50}) and sometimes experimental conditions that are not coded into ECOTOX (e.g., presence or absence of UV radiation). We therefore used ECOTOX primarily to identify original publications with titles indicating a focus on endpoints and chemicals of interest. Finally, for chemicals not included in ECOTOX, we conducted web searches by chemical name or CAS Registry Number for ecotoxicity tests, revealing additional original study reports for several POM.

A.4.1.2 General Characteristics of RTR POM

Most POM included for RTR multipathway assessment are PAHs. Excluding naphthalene, PAHs have relatively high melting and boiling points and low water solubility. Their water solubility increases with decreasing molecular weight. Most PAHs are highly lipophilic, with lipophilicity (i.e., K_{ow}) increasing with increasing molecular weight. Overall, aquatic invertebrates (e.g., annelids, insect larvae, daphnids) are more sensitive than fish, and benthic fish are among the least sensitive species to PAHs (Wang et al. 2013, 2014, as cited by Wu et al. 2016). Four characteristics of POM challenge attempts to develop new TEFs as discussed in the paragraphs that follow:

- Aquatic toxicity testing is complicated by low solubility of high-K_{ow} chemicals.
- Quantitative structure-activity relationships (QSARs) are not available for PAH modes of action.
- Sunlight can modify the toxicity of PAHs to plants and invertebrates.
- Bioaccumulation depends on taxonomy.

A.4.1.3 Aquatic Toxicity Testing Limited by Low Solubility of High-K_{ow} Chemicals

Aquatic toxicity testing for PAHs, including BaP, and other POM has been limited by the low solubility of high-K_{ow} chemicals. Typical endpoints often are not reached at the limit of solubility for the high-molecular-weight (HMW) polycyclics (e.g., four or more aromatic rings, 225 grams per mole or higher). In an early-lifestage study of BaP toxicity to rainbow trout (*Oncorhynchus mykiss*), a NOEC of 1.5 µg/L and an EC_{10} of 2.9 µg/L were identified for developmental abnormalities (Hannah et al. 1982, ECHA 2016). *Danio rerio* (zebra danio) exposed for 28 days, on the other hand, showed no abnormalities at 4 µg/L (Hooftman and Evers-de Ruiter 1992, as cited by ECHA 2016), which is at or above the water solubility of BaP (approximately 3.8 µg/L). As a consequence, many of the high K_{ow} POM have not been tested for aquatic toxicity. Fish in bodily contact with sediments (e.g., salmon eggs, other fish eggs and fry, flatfish) can be exposed both dermally and via desorption from sediment particles; thus embryo toxicity tests with spiked sediments are available for some HMW POM.

A.4.1.4 QSAR Applicability is Limited for High K_{ow} Chemicals

QSAR models that predict the aquatic toxicity of HMW non-ionic organic chemicals (e.g., ECOSAR in EPA EPI-Suite, U.S. EPA 2012b) are generally not valid for high-K_{ow} chemicals because HMW chemicals with logK_{ow} values over 6 are too large to readily penetrate membranes despite their lipophilicity. For example, the maximum logK_{ow} for which ECOSAR estimates of a fish 96-hour LC_{50}, a daphnid 48-hour LC_{50}, or a mysid 96-hour LC_{50} are valid is 5.0, and the maximum for an ECOSAR prediction of an earthworm LC_{50} is 6.0.

The maximum logK_{ow} for which chronic values for fish and invertebrates might be valid is 8.0. Droge et al. (2006) demonstrated, however, that PAHs can induce mortality via narcosis (corresponding to K_{ow} at concentrations for which the PAH is soluble), whereas reproductive effects did not follow K_{ow}. We do not recommend a QSAR based on narcosis as a valid predictor of POM-induced endocrine disruption or developmental abnormalities. Sverdrup et al. (2001) demonstrated that some PAHs are more or less toxic than predicted on the basis of neutral

organic QSAR models. For example, carbazole and acridine are more toxic to springtail reproduction than predicted on the basis of K_{ow} for nonpolar organics, while fluoranthene is less toxic than predicted on the basis of K_{ow} values and estimated pore-water EC_{10} values (Sverdrup et al. 2001, 2002a).

We conclude that the QSAR models available for aquatic or earthworm toxicity, such as those included in EPA's ECOSAR model, are unlikely to provide reasonable predictions of POM toxicity to aquatic organisms, acute or chronic. Instead, for POM chemicals without toxicity data, we used chemical structure (e.g., elongated versus compact, locations of substitutions) in addition to K_{ow} and number of benzene rings to identify which parameterized POM appeared most similar, as described in Section A.4.1.6.

A.4.1.5 Photomodification of PAH Toxicity

Another attribute of PAHs that complicates interpretation of aquatic and surface soil toxicity testing is that sunlight can increase the toxicity of many congeners. For PAHs in surface waters and surface soils, two or more conjugated benzene rings facilitate absorption of UV-A and UV-B, and, in some instances, visible light (i.e., wavelengths of 400–700 nm) (Lampi et al. 2006). PAHs strongly absorb photons in the UV-B (290–320 nm) and UV-A (320–400 nm) wavelength regions (both regions are in sunlight). The toxicity of PAHs can be enhanced in the presence of UV radiation; however, lighting in laboratory settings usually is in the visible range only. Photosensitization (PSC) reactions result from generation of singlet-state oxygen (Krylov et al. 1997). Photomodification (PMC) results from photooxidation or photolysis (Huang et al. 1997).

Krylov et al. (1997) examined the phytotoxicity of 16 PAHs to *Lemna gibba* (duckweed). All 16 PAHs exhibited half-lives in simulated sunlight including UV-A and UV-B of 100 hours or less. Anthracene was by far the most toxic of the PAHs examined. Intact anthracene is not a strong photosensitizer; perhaps its degradation products cause its toxicity. K_{ow} values do not predict photoinduced toxicity for PAHs. Krylov et al. (1997) provided a QSAR model based on the 16 PAHs that includes both PSC and PMC. The predictive model indicates that PSC and PMC contribute additively to toxicity. We consider attempts to use this model beyond the scope of identifying TEFs for aquatic plants for PAHs. Moreover, because plants generally are less

sensitive to PAHs than are invertebrates and fish, we did not use aquatic plant toxicity tests to estimate TEFs for surface water.

PSC and PMC reactions can affect PAH toxicity to aquatic invertebrates, such as *Daphnia magna.* Lampi et al. (2006) found that toxicity increased (EC_{50} decreased) approximately threefold in simulated sunlight compared with visible-plus-UVA (no UV-B). Again, K_{ow} does not predict PSC or PMC reactions. Lampi et al. (2006) demonstrated that some PAHs absorb UV-A poorly (e.g., chrysene, fluorene), while decreases in EC_{50} values were substantial for PAHs that strongly absorb in the UV-B region (benzo[e]pyrene, benzo[g]pyrene, dibenzo[a,i]pyrene). Accounting for PSC and PMC reactions and toxicity to invertebrates is beyond the scope of this work. To estimate TEFs by matching individual toxicity tests of a congener with BaP, however, we did attempt to match the lighting conditions (e.g., simulated sunlight or visible light only).

A.4.1.6 TEFs for POM for Surface Waters, Soils, and Sediments

For POM chemicals for which we identified appropriate benchmarks to compare with the BaP benchmarks for surface water, sediment, or soil, we used the ratio of the benchmarks to calculate a TEF appropriate for the ecological assessment endpoint and environmental medium. Benchmarks were not available, however, for many individual POM chemicals. We therefore needed to use original toxicity tests to estimate TEFs.

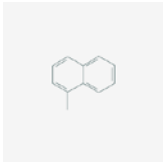
To calculate TEFs from individual toxicity tests requires a different approach than estimating TEFs from community benchmarks already calculated on the basis of many different species' tests and possibly some "uncertainty" factors based on available data. We compared individual ecotoxicity tests for POM *only to the same type of study* for BaP. We therefore also used ECOTOX and web searches to compile individual toxicity test data for BaP to enable those comparisons. Study types and endpoints include 96-hour algal EC_{50} values, 48-hour daphnid LC_{50} values, 96-hour fish LC_{50} values, 2- to 4-week early-lifestage studies with fish (considered chronic), and 4-week earthworm or springtail survival and reproduction tests (soil toxicity tests), although only one or two study types/endpoints were available for most of the POM not included in RAIS. A compilation of those data is available on request. We document below the comparisons on which we based individual POM TEFs, presented in Table A-12, where

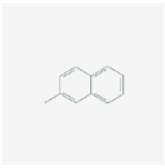
benchmarks were not already established. For each POM, we include its CAS Registry Number, its logK_{ow}, and a schematic of its chemical structure to help the reader follow our rationale.

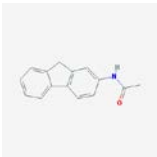
Benzo[a]pyrene (BaP; CAS # 50-32-8): LogK_{ow} = 5.97. Index chemical for POM TEFs. The average of six 4-day and one 2-day daphnid LC_{50} values is 4.0 µg/L (range 1.0–6.1 µg/L) (Lampi et al. 2006; Wu et al. 2016; Trucco et al. 1983; Ikenaka et al. 2013). With sunlight (which includes UV-A and UV-B), BaP is more toxic to daphnids than under visible light plus UV-A, which is more toxic than under laboratory lighting with no UV-A or UV-B. When fed to *Chlorella* sp., BaP is less toxic than it is to daphnids not fed during the exposure period (Ikenaka et al. 2013). Nonetheless, the range of LC_{50} values across those conditions is limited (i.e., 1.3–6.1 µg/L). For bluegill fish (*Lepomis macrochirus*), the 4-day LC_{50} is 5 µg/L (Wu et al. 2016). For rainbow trout (*Oncorhynchus mykiss*, formerly *Salmo gairdneri*), a 36-day EC_{10} for abnormalities in development of early lifestages is 2.9 µg/L (NOEC is 1.5 µg/L) (ECHA 2016 calculated from exposure-response data reported by Hannah et al. 1982). Algae are not as sensitive as daphnids or fish by two or more orders of magnitude (Warshawsky et al. 1995). The aquatic toxicity benchmarks border on the limit of solubility of 0.0063 µmol/L (Pearlman et al. 1984) or about 1.6 µg/L (BaP molecular weight = 252.3 g/mole). These toxicity data are compared to available toxicity test data for other POM chemicals below.



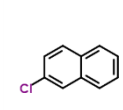
1-Methylnaphthalene (CAS # 90-12-0): LogK_{ow} = 3.87. In freshwater, the 4-day LC_{50} for fathead minnow (static test) is 9,000 µg/L (Mattson et al. 1976), compared with the BaP-exposed bluegill (*Lepomis macrochirus*) 4-day LC_{50} of 5 µg/L (Wu et al. 2016). The ratio of those values yields an acute TEF of 0.00056 µg/L. Those studies, however, do not predict chronic toxicity of 1-methylnaphthalene to aquatic organisms or to soil communities and birds and mammals. Therefore, we assigned the same TEFs as for 2-methylnaphthalene (CAS # 91-57-6) (figure above right) to 1-methylnaphthalene.







2-Acetylaminofluorene (CAS # 53-96-3): An aromatic amine with LogK_{ow} = 3.28. TEFs for surface water, soil, and sediment are from Region 5 ESL benchmarks relative to Region 5 ESLs for BaP for each medium, respectively (U.S. EPA 2003c).



2-Chloronaphthalene (CAS # 91-48-7): $LogK_{ow}$ = 4.14. TEFs for surface water, soil, and sediment are from Region 5 ESLs relative to Region 5 ESLs for BaP for each medium, respectively (U.S. EPA 2003c).

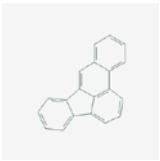


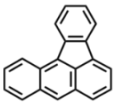
3-Methylcholanthrene (CAS # 56-49-5): $LogK_{ow}$ = 6.42. TEFs for surface water, soil, and sediment are from Region 5 ESLs relative to Region 5 ESLs for BaP for each medium, respectively (U.S. EPA 2003c).

Benzo[e]pyrene (BeP, CAS # 192-97-2): More compact and with a higher $logK_{ow}$ (6.44) than BaP. Under visible light plus UV-A, the 48-hour daphnid immobility EC_{50} (= LC_{50}) = 1.43 µg/L, whereas under simulated sunlight spectrum, the EC_{50} = 0.325 µg/L (Lampi et al. 2006). For BaP, under visible light plus UV-A, the 48-hour daphnid immobility EC_{50} = 1.62 µg/L, whereas under simulated sunlight spectrum, the EC_{50} = 0.98 µg/L (Lampi et al. 2006). No other data on freshwater organisms was identified. Freshwater TEF for BeP = under simulated sunlight = 0.98/0.325 = 3.0. No ecotoxicity data were identified for sediments or soils; therefore, we set the remaining TEFs to 1.0 assuming similarity to BaP.

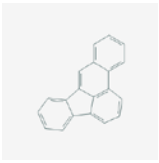


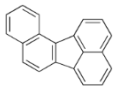
Benzo[a]fluoranthene (BaF, CAS # 203-33-8): $LogK_{ow}$ = 6.11, which is higher than the $logK_{ow}$ for benzo[b]fluoranthene (BbF, CAS # 205-99-2, $logK_{ow}$ = 5.78, figure to the right). BaF also has a higher $logK_{ow}$ than benzo[k]fluoranthene (BkF, $logK_{ow}$ 5.94) and a lower $logK_{ow}$ than benzo[j]fluoranthene (BjF, $logK_{ow}$ 6.4). No ecotoxicity data were identified for BaF for water, sediment, or soils. TEFs were set equal to those for BbF (figure right).



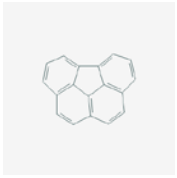


Benzo[j]fluoranthene (BjF, CAS # 205-82-3): $LogK_{ow}$ = 6.4, which is higher than for the other benzofluorenes. All benzo[x]fluorenes have the same molecular weight (252.3 g/mole). No toxicity data were identified for BjF for water, sediment, or soils. TEFs were set equal to those for BbF (figure right).

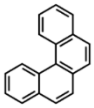




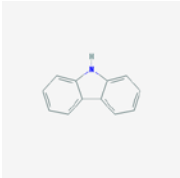
Benzo[g,h,i]fluoranthene (CAS # 203-12-3): $LogK_{ow}$ = 5.52 (molecular weight 226.3 g/mole). With $logK_{ow}$ close to 5.5, we used EPA EPI-Suite ECOSAR Version 4.11 (U.S. EPA 2012b) to estimate four freshwater TEFs: 48-hour LC_{50} for daphnids, 96-hour LC_{50} value for fish, chronic value for daphnids, and chronic value for fish. Toxicity to algae is not evaluated because algae are generally less sensitive than invertebrates to PAHs and because less sensitive species can replace more sensitive species in the water column community. The four toxicity values were compared with the same values from EPI-Suite ECOSAR for BaP to calculate the four corresponding TEFs. The highest of the four corresponding TEFs (chronic fish TEF of 0.71) represents the surface water TEF (range of four TEFs from 0.60 to 0.71). For the remaining TEFs for sediment, soil, birds, and mammals, for which no toxicity data were identified, we set TEFs equal to those for benz[a]anthracene which has a $logK_{ow}$ of 5.79, a similar molecular weight (226.3 g/mole), and a similar TEF (0.56) for surface waters.

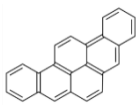


Benzo[c]phenanthrene (CAS # 195-19-7): $LogK_{ow}$ = 5.52. With $logK_{ow}$ close to 5.5, we used EPA's EPI-Suite ECOSAR Version 4.11 to estimate four freshwater TEFs as explained for benzo[g,h,i]fluoranthene above. The highest of the four corresponding TEFs, chronic fish TEF of 0.42, represents the surface water TEF (range of four TEFs from 0.32 to 0.42). For the remaining TEFs for sediment and soil, for which no toxicity data were identified, we set TEFs equal to those for benz[a]anthracene, which has a $logK_{ow}$ of 5.79, the same molecular weight (228.29 g/mole), and a similar TEF (0.56) for surface waters.

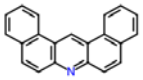


Carbazole (CAS # 86-74-8): $LogK_{ow}$ = 3.72. In freshwater, fathead minnow 4-day LC_{50} (flow-through design) = 930 µg/L (Brooke 1991) compared with BaP exposed bluegill (*Lepomis macrochirus*) 4-day LC_{50} of 5 µg/L (Wu et al. 2016) for a TEF of 0.0054. For soils, earthworm 28-day LC_{50} = 106/2 (division by 2 because endpoint is 50 percent lethality) = 53 mg/kg dry soil and EC_{50} for growth = 54 mg/kg dry soil (Sverdrup et al. 2002b) compared with a BaP 28-day LOEC for earthworm survival of 10 mg/kg dry soil (Achazi et al. 1995 as cited by Sverdrup et al. 2007) for TEF of 0.20. For sediments, no data were found; therefore, we assign a sediment TEF of 7.5, which is similar to the other PAHs with $logK_{ow}$ values between 3.28 and 3.87. Carbazole is structurally similar to fluorene, except the nitrogen atom is at the apex of a five-member nonaromatic ring in center.





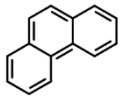
Dibenzo[a,i]pyrene (DaP, CAS #189-55-9): $LogK_{ow}$ = 7.28. Extremely low solubility. We assigned the same TEFs as indeno[1,2,3-c,d]pyrene, which has a K_{ow} value of 6.72. Aquatic toxicity benchmarks are higher than the limit of solubility.



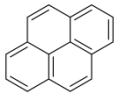
Dibenzo[a,j]acridine (CAS # 224-42-0): $LogK_{ow}$ = 5.63. With $logK_{ow}$ under 6.0, we used EPA EPI-Suite ECOSAR Version 4.11 to estimate four freshwater TEFs as explained for benzo[c]phenanthrene (above). The highest of the four corresponding TEFs (chronic fish TEF of 0.43) represents the surface water TEF (range of four TEFs from 0.35 to 0.43). That aquatic toxicity benchmarks are not reached at the limit of solubility is possible.

Perylene (CAS # 198-55-0): $LogK_{ow}$ = 5.82 (although the MSDS, http://datasheets.scbt.com/sc-206007.pdf, states 6.25 for $logK_{ow}$). Fish 4-day LC_{50} values range from 1.1 to 5.0 (MSDS). In a daphnid acute test, 0.6 µg/L kills 50% of individuals in 0.764 days (LT_{50} in renewal system). Thus, perylene appears to be more toxic in the water column to both fish and daphnids than is BaP. Based on those data, we set the TEF for freshwater to 2.0 compared with BaP. We set the remaining TEFs for perylene for soil and sediments to 1.0 compared with BaP.

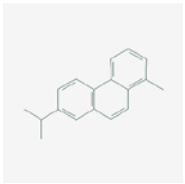




Phenanthrene (PHE, CAS # 85-01-8): $LogK_{ow}$ = 4.46. TEFs for surface water, soil, and sediment are from Region 5 ESLs relative to Region 5 ESLs for BaP for each medium, respectively (U.S. EPA 2003c).



Pyrene (PYR, CAS # 129-00-0): $LogK_{ow}$ = 4.88. TEFs for surface water, soil, and sediment are from Region 5 ESLs relative to Region 5 ESLs for BaP for each medium, respectively (U.S. EPA 2003c).



Retene (CAS # 483-65-8): $LogK_{ow}$ = 6.35. In freshwater, the 14-day LC_{50} (flow-through design) for zebra Danio fish = 353 µg/L (Billiard et al. 1999) compared with bluegill 4-day LC_{50} of 5 µg/L for BaP (Wu et al. 2016). Comparing those two studies as "acute" values, the TEF equals 0.014. A zebra Danio early lifestage 42-day LOEC (flow-through) = 180 µg/L (NOEC = 100 µg/L; Billiard et al. 1999). Compared with the LOEC of 2.9 µg/L for developmental abnormalities in a 36-day BaP exposure of early lifestage *Oncorhynchus mykiss* (rainbow trout, ECHA 2016, exposure-response calculation from

data in Hannah et al. 1982), the chronic TEF equals 0.016. Surface water TEFs based on both acute and chronic exposures of fish equal 0.015. No relevant data were found for sediments or soils; therefore, those TEFs for retene were assigned based on 7,12-dimethylbenz[a]anthracene, which has a similar freshwater TEF of 0.026 µg/L, a similar $\log K_{ow}$ of 5.8, and two alkyl groups attached to its rings (figure above right).

A.4.2 TEFs for Dioxins for Surface Water, Sediments, and Soils

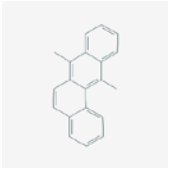


Table A-13 presents the TEFs for dioxins for surface water, soils, and sediments. Surface water TEFs are based on 1998 World Health Organization (WHO) TEFs for fish (from Van den Berg et al. (1998) as presented in Table 4 of *Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated Dioxins, Furans, and Biphenyls in Ecological Risk Assessment* (U.S. EPA (2008). For sediments, we set the TEF for each congener to the TEF for surface water based on the concept of equilibrium partitioning as per the Canadian ISQG for the protection of aquatic life (CCME 2001).

For soils, we adopted a different approach. Plants and most invertebrate groups are not adversely affected by dioxins, because they lack the aryl hydrocarbon (Ah) receptor that mediates the adverse effects in vertebrates, including birds and mammals (UKDTER 1999).We concluded that the soil TEFs should be based on relative toxicity to birds or to mammals to reflect possible toxicity to ground-feeding birds and mammals. We therefore set the soil TEFs for dioxins to the TEF for mammalian or avian wildlife, whichever of the two was higher.

A.5 TEFs for Wildlife TRVs

This section describes the derivation of TEFs for the wildlife TRVs for POM and for dioxins.

A.5.1 TEFs for Wildlife for POM

Most POM in the RTR multipathway list have been screened in vitro (cell cultures) for carcinogenic potential; however, cancer is not an endpoint evaluated for wildlife risk assessments (CCME 2010). Most animals die from starvation, disease, extreme weather, or predation before tumors can develop. Disruption of vertebrate endocrine systems, immune effects, and fetal abnormalities are endpoints of concern for wildlife (CCME 2010).

Table A-13. Toxicity Equivalency Factors (TEFs) for Surface Waters, Soils, Sediments, and Mammalian and Avian Wildlife—Dioxins Relative to 2,3,7,8-TCDD

Congener	CAS RN	Surface Water TEF	Soil TEF	Sediment TEF	Mammalian TEF[a]	Avian TEF
1,2,3,4,6,7,8,9-OCDD	3268-87-9	0.0001	0.0003	0.0001	0.0003	0.0001
1,2,3,4,6,7,8,9-OCDF	39001-02-0	0.0001	0.0003	0.0001	0.0003	0.0001
1,2,3,4,6,7,8-HpCDD	35822-46-9	0.001	0.01	0.001	0.01	0.0005
1,2,3,4,6,7,8-HpCDF	67562-39-4	0.01	0.01	0.01	0.01	0.01
1,2,3,4,7,8,9-HpCDF	55673-89-7	0.01	0.01	0.01	0.01	0.01
1,2,3,4,7,8-HxCDD	39227-28-6	0.5	0.1	0.5	0.1	0.05
1,2,3,4,7,8-HxCDF	70648-26-9	0.1	0.1	0.1	0.1	0.1
1,2,3,6,7,8-HxCDD	57653-85-7	0.01	0.1	0.01	0.1	0.01
1,2,3,6,7,8-HxCDF	57117-44-9	0.1	0.1	0.1	0.1	0.1
1,2,3,7,8,9-HxCDD	19408-74-3	0.01	0.1	0.01	0.1	0.1
1,2,3,7,8,9-HxCDF	72918-21-9	0.1	0.1	0.1	0.1	0.1
2,3,4,6,7,8-HxCDF	60851-34-5	0.1	0.1	0.1	0.1	0.1
1,2,3,7,8-PeCDD	40321-76-4	1	1	1	1	1
1,2,3,7,8-PeCDF	57117-41-6	0.05	0.1	0.05	0.03	0.1
2,3,4,7,8-PeCDF	57117-31-4	0.5	1	0.5	0.3	1
2,3,7,8-TCDD	**1746-01-6**	**1**	**1**	**1**	**1**	**1**
2,3,7,8-TCDF	51207-31-9	0.05	1	0.05	0.1	1

Abbreviations: CDD = chlorinated dibenzo-p-dioxins; CDF = chlorinated dibenzo-p-furans. Hp = hepta (seven); Hx = hexa (six); O = octa (eight), Pe = penta (five); T = tetra (four)
Note: If the TEF is less than 1, the chemical is not as toxic to organisms as is 2,3,7,8-TCDD (in bold). If the TEF is equal to or greater than 1, the chemical is as toxic or more toxic to organisms as 2,3,7,8-TCDD.
[a] Source: Van den Berg et al. (2006).

As was the case for the benchmarks for surface water, soils, and sediments, we found no additional avian or mammalian toxicity data for many of the recently added POM. Although EPA's ECOTOX database does include avian and toxicity data in addition to aquatic toxicity information, we found that few of the new POM are included in ECOTOX.

We found avian embryo toxicity data for several POM chemicals based on egg injection studies. Brunstrom et al. (1990) reported the proportion of eider duck embryos that died or were malformed after a single injected dose (2 mg/kg egg) for several POM (BaP, 30% mortality; BkF, 100%; fluoranthene, 20%; benzo[g,h,i]perylene, 15%; and indeno[1,2,3-c,d]pyrene, 85% mortality). Many of the POM chemicals were not toxic to eggs at 2 mg/kg egg (i.e., anthracene,

fluorene, pyrene, BeP, perylene, benzo[g,h,i]perylene). These data are insufficient to estimate avian toxicity TEFs for most RTR POM; we therefore did not estimate avian TEFs.

For mammals, we checked the Canadian TRV data for wildlife in CCME (2010) for unsubstituted PAHs to identify original toxicity study data that focused on endpoints other than carcinogenicity and mutagenicity. For other POM, we checked EPA's Integrated Risk Information System. We considered subchronic or chronic oral administration studies for which the concentration of chemical in the diet had been converted to a dose in mg/kg body weight per day. In addition, for several POM, we compared the immunocompetence of mice following a single intraperitoneal dose of the POM with the immunocompetence of mice following a single dose of BaP (Silkworth et al. 1995). We did not consider LD_{50} data for mammals to be appropriate for estimating chronic TEFs because the MOA of acute lethality and chronic effects on immunity, reproduction, or development likely differs. The variety of chemical compounds included in the 2016 list of POM also suggests that several different MOAs might be relevant.

Table A-14 presents the data used to estimate TEFs for the RTR POM for mammalian wildlife. Insufficient data were available to calculate TEFs for POM for birds. The mammalian TEFs also are included in Table A-12 for comparison with other benchmark TEFs.

Table A-14. TEFs for Oral Exposures of Mammalian Wildlife—POM Congeners Relative to BaP

Compound (surrogate PAH)	CAS RN	NOAEL/LOAEL[a] (mg/kg-d)	Species/Effect (exposure regimen) [notes]	NOAEL-based TEF	Reference
1-Methylnaphthalene (Naphthalene surrogate)	90-12-0 (91-20-3)	71/143	Rats/decreased body wt (5 d/wk for 13 wk) [dose is time adjusted]	0.014	BCL (1980) in CCME (2010)
2-Acetylaminofluorene	53-96-3	ND	Set = BaP reproduction	1	none identified
2-Chloronaphthalene	91-58-7	ND	Set = BaP reproduction	1	none identified
2-Methylnaphthalene (Naphthalene surrogate)	91-57-6 (91-20-3)	71/143	Rats/decreased body wt (5 d/wk for 13 wk) [dose is time adjusted]	0.014	BCL (1980) in CCME (2010)
3-Methylcholanthrene	56-49-5	ND	Set = BaP reproduction	1	none identified
7,12-Dimethylbenz[a] anthracene	57-97-6	ND	Set = BaP reproduction	1	none identified

Compound (surrogate PAH)	CAS RN	NOAEL/LOAEL[a] (mg/kg-d)	Species/Effect (exposure regimen) [notes]	NOAEL-based TEF	Reference
Acenaphthene	83-32-9	175/350	Mouse/liver wt (13 wk)	0.0057	U.S. EPA (1989a) in CCME (2010) and ATSDR (1995)
Acenaphthylene	208-96-8	18/180	Mouse/immunocompetence (12 d) TDLo (NEL = LEL/10)	0.056	RTECS (1999 Toxicologist 48:13, in CCME 2010)
Anthracene	120-12-7	1000/ >1000	Mouse/multiple systems examined (13 wk)	0.001	U.S. EPA (1989b) in CCME (2010) and ATSDR (1995)
Benz[a]anthracene[a]	56-55-3	−8% at 1	Mouse/immunocompetence (1 dose)	1	Silkworth et al. (1995)
Benzo[a]fluoranthene	203-33-8	ND	Set = BkF immune	7.5	none identified
Benzo[c]phenanthrene	195-19-7	ND	Set = BaP reproduction	1	none identified
Benzo[g,h,i] fluoranthene	203-12-3	ND	Set = BaP reproduction	1	none identified
Benzo[g,h,i]perylene	191-24-2	ND	Set = BaP reproduction	1	none identified
Benzo[a]pyrene[a]	50-32-8	−8% at 1	By definition of index chemical	1	Silkworth et al. (1995)
Benzo[a]pyrene	50-32-8	1/10	Mouse/reduced fertility of progeny of exposed animals (gd 7–16)	1	Mackenzie & Angevine (1981)
Benzo[b]fluoranthene[a]	205-99-2	−35% at 1	Mouse/immuno-competence (1 dose)	4.4	Silkworth et al. (1995)
Benzo[e]pyrene	192-97-2	ND	Set = BaP reproduction	1	none identified
Benzo[j]fluoranthene	205-82-3	ND	Set = BkF immune	7.5	none identified
Benzo[k]fluoranthene[a]	207-08-9	−60% at 1	Mouse/immunocompetence (1 dose)	7.5	Silkworth et al. (1995)
Carbazole	86-74-8	ND	Set = fluorene	0.008	none identified
Chrysene[a]	218-01-9	−48% at 1	Mouse/immunocompetence (1 dose)	6	Silkworth et al. (1995)
Dibenzo[a,h] anthracene[a]	53-70-3	−15% at 1	Mouse/immunocompetence (1 dose)	2	Silkworth et al. (1995)
Dibenzo[a,i]pyrene	189-55-9	ND	Set = BaP reproduction	1	none identified
Dibenzo[a,j]acridine	224-42-0	ND	Less toxic than BaP	0.1	none identified

Compound (surrogate PAH)	CAS RN	NOAEL/LOAEL[a] (mg/kg-d)	Species/Effect (exposure regimen) [notes]	NOAEL-based TEF	Reference
Fluoranthene	206-44-0	150/250	Rat/increased liver wt (13 wk)	0.0067	U.S. EPA (1988b); Knuckles et al. (2004)
Fluorene	86-73-7	125/250	Mouse/liver wt hemato-logical effects (13 wk)	0.008	U.S. EPA (1989c) in CCME (2010)
Indeno[1,2,3-c,d] pyrene	193-39-5	ND	Set = BaP reproduction	1	none identified
Perylene	198-55-0	ND	Set = BaP reproduction	1	none identified
Phenanthrene	85-01-8	1000/ >1000	Set = anthracene	0.001	none identified
Pyrene	129-00-0	75/125	Mouse/nephrotoxicity (90 d, gavage)	0.013	U.S. EPA (1989d) in CCME (2010)
Retene	483-65-8	ND	Set = BaP reproduction	1	none identified

Abbreviations: Ah = aromatic hydrocarbon; d = day; gd = gestation day; LEL = lowest-effect level; NEL = no effect level; ND = no data found; TDLo = threshold dose–lowest observed effect level; wk = week

[a]For Silkworth et al. (1995), data in NOAEL/LOAEL column are the percent decrease in ability to suppress the antibody response in $Ah^{+/+}$ mice immunized 12 hours after administration of one dose of chemical at 1 mg/kg bw.

A.5.2 TEFs for Wildlife for Dioxins

To estimate TEFs for dioxins mammals and birds (listed in Table A-13 along with TEFs for soils, sediments, and surface waters), we used the 1998 and 2005 WHO TEFs for dioxins and furans as presented in EPA's *Framework for Application of Toxicity Equivalency Methodology* (U.S. EPA 2008; Van den Berg et al. 1998; Van den Berg et al. 2006). The dioxin TEFs apply to both cancer and noncancer (e.g., reproductive) endpoints, and therefore we did not need to look for noncancer toxicity tests for individual dioxin congeners.

A.6 Piscivorous Wildlife Exposure Factors

To calculate wildlife exposures via fish ingestion, a series of exposure factor values and an assumed diet are required for the representative species: mink and American merganser. Those values are then used with the TRIM.FaTE estimates of chemical concentrations in fish in the most contaminated lake to estimate mink and merganser chemical intake, in mg/kg-day, via fish ingestion.

Although conceptually considered part of the "exposure assessment" described in the main report, the values selected to parameterize the wildlife exposures via consumption of aquatic

prey are used to backcalculate facility emission screening threshold rates that correspond to the TRVs. Therefore, the input data used for the piscivorous wildlife exposure assessments, calculated outside of TRIM.FaTE, are described in this section.

For the RTR environmental screen, the wildlife are assumed to consume their entire diet from the lake located near the emissions source in the screening scenarios. To calculate the wildlife exposure for each TRIM.FaTE screening scenario, the TRIM.FaTE estimates of chemical concentrations in various compartments of the aquatic biota were calculated first. Then, wildlife exposure based on those data and values for the wildlife exposure factors were calculated. The wildlife exposure factors include an estimated FIR, the caloric energy of the food ingested, the ability of the wildlife species to assimilate calories from the food, and the proportion of the animal's diet consisting of each food type. Food ingestion rates were either obtained from measured values in the open literature or calculated from estimates of free-living metabolic rate (FMR) using allometric equations developed by Nagy (1987). Measured data were selected from the information presented in EPA's (1993c,d) *Wildlife Exposure Factors Handbook* (WEFH) to be "representative" of the data available for the species across its range.

Estimates of FMRs across animals of varying body size within numerous taxa have become available with modern techniques using labeled oxygen measurements. Nagy and his colleagues used the empirical data to develop allometric equations relating FMR to body weight for numerous taxonomic groups (Nagy 1987; Nagy 2001). Estimates of FMR with body weight within a taxon allows estimates of the required daily caloric intake from food. As described in EPA's WEFH (U.S. EPA 1993c,d), with additional information on the caloric content of different types of food and the food habits of a wildlife species, one can estimate the total weight of different foods (e.g., different trophic levels of prey) ingested.

Information on the diets of wildlife species are obtained from field studies in which animals are captured and their gut contents removed or from studies of animals found dead in the field. In general, even the most specialized of feeders must adjust its food sources based on circumstance. For piscivorous wildlife, consumption of fish and invertebrate species varies with availability according to location and season. Nonetheless, comparisons of studies of the same species across years and locations have revealed some consistent patterns that can be used as default assumptions in an ecological risk screening scenario.

The next two subsections describe the exposure parameter values and assumed diets used to estimate consumption of fish for both mink (Section A.6.1) and American merganser (Section A.6.2).

A.6.1 Mink Exposure Factor Values and Assumed Diet

For mink (*Mustela vison or Neovison vison*), none of the measured FIRs available for captive animals were considered representative of free-living animals. Caged animals might not be as active as free-ranging animals that must catch their prey in cold waters and escape predators. We therefore used Nagy's (1987) allometric model for nonherbivorous mammals to estimate an FMR first, which was converted to units of kcal/day as recommended by EPA (U.S. EPA 1993c). The FMR then was normalized to body weight (Table A-15). The estimate of a FMR of 245 kcal/kg bw/day in Table A-12 is similar to the estimated metabolic rate of 258 kcal/kg bw/day for farm-raised (ranch cage) female mink as estimated by Farrell and Wood (1968).

Table A-15. Mink Exposure Factor Values

Parameter	Value	Comments/References
Body weight (kg)	0.8	Average of male and female body weights in summer in Montana (Mitchell 1961)
Free-living metabolic rate (FMR):		Estimated for 1.0-kg mink using Nagy's (1987) allometric equation for nonherbivorous mammals
FMR (kJoules/day)	821	FMR (kJoules/day) = 2.582 × BW (g)$^{0.862}$ (Nagy 1987)
FMR (kcal/day)	196	FRM (kcal/day = 0.6167 × BW (g)$^{0.862}$ (U.S. EPA 1993c)
FMR normalized to BW (kcal/kg-day)	245	FMR normalized to body weight (kcal/kg-day) = FMR (kcal/day) / BW (kg)
Gross energy (GE) of fish (kcal/g ww)	1.20	Table 4-1 of U.S. EPA (1993c)
Food assimilation efficiency (AE) for mammal consuming fish	0.91	U.S. EPA (1993c), Table 3-1
Metabolizable energy (ME) in fish (kcal/g ww)	1.09	ME (kcal/g ww) = GE (kcal/g wet weight) × AE
Normalized food ingestion rate (FIR) (g/g-day)	0.225	FIR (g/g-day) = FMR (kcal/kg-day) × 0.001 kg/gram / ME (kcal/g wet weight)
FIR (percent total body weight)	22.5%	(see previous cell)
FIR per animal (g/d)	180	assuming an 800-g mink

Acronyms: BW = body weight

The gross energy (GE) content for fish and a caloric assimilation efficiency (AE) for a mammal consuming fish were obtained from the WEFH to estimate the metabolizable energy (ME) for the diet on a wet-weight basis (Table A-15). Based on the energy requirements (FMR) of mink and

the ME per unit wet-weight prey, an FIR then could be calculated as the FMR/ME (with units corrected), which in this case equals 22.5 percent of the adult mink's body weight daily. For an individual mink weighing 800 grams, that would be 180 grams of fish, wet weight, ingested per day. To determine chemical ingestion rates, the proportion of the diet obtained by mink from each aquatic biotic compartment in TRIM.FaTE must be specified. All data summarized in EPA's 1993 WEFH, Volume 2, Appendices (U.S. EPA 1993d) were consulted to generalize the dietary assumptions for the RTR environmental screen and to maximize the higher trophic level components of the diet. Those assumptions are listed in Table A-16 (Diet Composition column). The total daily FIR of 180 grams of fish could then be divided among the TRIM.FaTE aquatic biota compartments. Table A-16 shows the resulting FIR in three different units.

Table A-16. Mink Diet Assumptions[a]

Food Type	Percent Diet Composition	Food Ingestion Rate (g/day)	Food Ingestion Rate (kg/day)	Food Ingestion Rate (kg/kg bw-day)
Benthic invertebrates[b]	25	44.9	0.0449	0.0561
Benthivorous fish (consuming benthic invertebrates only)	25	44.9	0.0449	0.0561
Bottom-feeding carnivores)	0	0.0	0.0000	0.0000
Water-column herbivore (planktivore)	25	44.9	0.0449	0.0561
Water-column omnivore	25	44.9	0.0449	0.0561
Water-column carnivore	0	0.0	0.0000	0.0000
TOTAL	100	179.6	0.1796	0.2245

[a]Dietary studies provided in U.S. EPA (1993d) were reviewed to develop assumptions in this table.
[b]The gross energy (GE) and assimilation efficiency (AE) for invertebrates are not identical to the GE and AE for fish; however, assuming that they are the same should have negligible effects on the overall results of the screen.

To evaluate the spatial extent of chemical contamination above a level that would be toxic to mink consuming fish from a water body, the home range of a mink or a mink family is important. Home range size depends on the location, type of habitat, season, and type of water body. In the prairie potholes region of the United States, mink home ranges of 259–380 hectares have been reported (U.S. EPA 1993d). In the pothole region of Manitoba, Canada, Arnold and Fritzell (1987) reported breeding home ranges of 770 hectares per mink or mink family. Along rivers and very large lakes, home ranges generally are expressed as length of river or shoreline. In Sweden, Gerell (1970) reported home ranges between 1.0 and 5.0 km in length depending on age and sex.

A.6.2 Merganser Exposure Factor Values and Assumed Diet

For American merganser (*Mergus merganser americanus*), a few measured FIRs were available from the literature (Salyer and Lagler 1940; Gooders and Boyer 1986; Alexander 1977) that suggested FIR values between 33 and 50 percent of the bird's body weight daily. Using Nagy's (1987) allometric equation for nonpasserine birds,[16] we estimated a FIR of 20 percent daily for a 1.27-kg American merganser. Based on that broad range of possible values, we selected 33 percent as the normalized FIR to use in the RTR screening scenarios. Assuming the body weight of mergansers in Michigan, the FIR equals 419 grams of fish, wet weight, per day per merganser (Table A-17).

Table A-17. Common Merganser Exposure Factor Values

Parameter	Value	References, Comments
Body weight (kg)	1.27	Salyer and Lagler (1940), Michigan
Normalized food ingestion rate (FIR) (g/g-day)	0.33	Salyer and Lagler (1940), Alexander (1977), Gooders and Boyer (1986), and estimated from Nagy (1987)
FIR (percent total body weight)	33%	(see previous cell)
FIR per animal (g/d)	419	Assuming a 1.27-kg American merganser

Estimates of the diet of American merganser, shown in Table A-18, are based on the reported lengths of fish caught in Michigan (Alexander 1977), with some consideration of studies from other locations (e.g., White 1936, 1937; Huntington and Roberts 1959) and experimental choice studies (Latta and Sharkey 1966).

Table A-18. Common Merganser Diet Assumptions[a]

Food Type	Diet Composition (%)	Food Ingestion Rate (g/day)	Food Ingestion Rate (kg/day)	Food Ingestion Rate (kg/kg bw-day)
Benthic invertebrates	0	0.0	0.000	0.00
Benthivorous fish (consuming benthic invertebrates only)	35	146.7	0.147	0.1155
Bottom-feeding carnivores (e.g., eel)	0	0.0	0.000	0.00
Water-column planktivore (YOY fish, shiners, 1–5 inches)	35	146.7	0.147	0.1155
Water-column omnivore (perch, young trout; 6–10 inches)	25	104.8	0.105	0.0825

[16]Groups of birds that generally are larger with slower metabolic rates per unit body weight than are birds in the Order Passeriformes, which includes the song birds such as warblers, robins, thrushes.

Food Type	Diet Composition (%)	Food Ingestion Rate (g/day)	Food Ingestion Rate (kg/day)	Food Ingestion Rate (kg/kg bw-day)
Water-column piscivore (e.g., largemouth bass >12 inches)	5	21.0	0.021	0.0165
TOTAL	100	419.1	0.419	0.33

Acronyms: YOY = young of the year
[a]Diet consumption compartmentalized into TRIM.FaTE biotic compartments is based on the lengths of fish reported caught in Michigan by Alexander (1977), with some consideration of studies from other locations (e.g., White 1936,1937; Huntington and Roberts 1959) and experimental choice studies (Latta and Sharkey 1966)

Most fish consumed are 10–30 cm long, although American merganser will choose larger fish in higher proportion than their availability relative to smaller fish (Mallory and Metz 1999). Fish up to 36 cm long are commonly consumed; mergansers have been reported to eat eels up to 55 cm long. The size of fish consumed apparently is determined by fish girth not length.

American merganser is not territorial. Groups of several females might nest together near productive water bodies during the breeding season, while in winter, large flocks often travel together from one body of water to another. In the Canadian Clay Belt Region (north of the Great Lakes), breeding densities of 7.2 pairs/100 km^2 (7.2 pairs /10,000 hectares) have been reported. Overall breeding densities in Atlantic Canada range from 0 to 81 pairs/10,000 hectares, with densities of 9–10 pairs/10,000 hectares typical of Newfoundland and Nova Scotia (Mallory and Mertz 1999). Along California rivers, 0.5–4.7 birds per linear km have been reported throughout the year (Mallory and Mertz 1999).

A.7 Derivation of Bioaccumulation Factors for Arsenic

Use of BAFs or biota-sediment accumulation factors (BSAFs) "*depends on the assumption that the concentration of chemicals in organisms is a linear no threshold function of the concentration in sediment. This will not be the case if uptake or depuration of the chemical in question is well-regulated by the organism, either because it is an essential nutrient or because it is a toxicant for which the organism has inducible mechanisms for metabolism or excretion*" (BJC 1998). Thus, for several metals, aqueous concentrations are not good predictors of concentrations in fish (BJC 1998; Chen and Folt 2000; Williams et al. 2006).

In addition, bioaccumulation of ionic inorganic chemicals that dissolve in water is different in marine vs. freshwater ecosystems. Because cations and anions are abundant in marine waters, they compete with chemical contaminant ions for transport through gills, although the overall

concentration of "salts" in fish blood and tissues is similar to that in ocean water. In freshwaters, aquatic organisms must osmoregulate, retaining cations and anions at higher concentrations in blood and tissues than in the surrounding water. Physiological mechanisms, therefore, differ between saltwater and freshwater fish and among species that can tolerate excess salinity or that live in estuarine environments.

We therefore conducted a literature search for studies of arsenic bioaccumulation in freshwater fish only, looking for field-measured BAFs for both pelagic and benthic feeding fish (many freshwater species feed in both habitats). Of particular concern was the possibility that bottom-feeding carnivorous fish might accumulate more arsenic than pelagic carnivorous fish. The bottom-feeding fish could ingest arsenic from both their prey and from sediment particles. We first present BAFs that relate dissolved arsenic concentrations in the water column to arsenic concentrations in top trophic-level fish. We then present data for BSAFs for bottom-dwelling freshwater fish.

The next three subsections discuss differences between freshwater and marine fish-tissue arsenic concentrations (Section A.7.1), BAFs (Section A.7.2), and biota-sediment accumulation factors (Section A.7.3).

A.7.1 Differences between Freshwater and Marine Fish

Differences between marine and freshwater organisms are evident from the concentrations of inorganic arsenic in water that produce acute lethality. For As(III) in saltwater, acute toxicity ranges from 250 µg/L for invertebrates (crabs and copepods) to more than 1,500 µg/L for filter-feeding mollusks and for fish (U.S. EPA 1985, 2003e). For As(III) in freshwaters, however, acute toxicity values range from 1,000 to 3,000 µg/L for invertebrates (amphipods and cladocerans) to more than 10,000 µg/L for most freshwater fish.

Marine fish usually contain more arsenic (0.19–65 mg[As]/kg[fish dry weight]) than freshwater fish (0.007–1.46 mg[As]/kg[fish dw]) (Donohue and Abernathy 1999). Table A-19 summarizes arsenic concentration data for marine and freshwater fish. As reported by ATSDR (2007), Hellou et al. (1996) measured 8–37 mg[As]/kg[fish fillet dw] in yellowtail flounder from the Northwest Atlantic in 1993. Assuming fish to be 75-percent water, the tissue concentration on a wet-weight (ww) basis would be approximately 2–9.3 mg/kg ww. Buchet and Lison (1998) measured total

arsenic concentrations in several fish species in Belgian markets; they found total arsenic at concentrations from 2.4 to 19.8 mg[total As]/kg[fish dw], which would equal approximately 0.6–5.0 mg/kg ww. They also found that inorganic arsenic contributed only a small fraction (0.003–0.2 mg[As]/kg[fish dw]) to the total arsenic.

Table A-19. Marine and Freshwater Fish Tissue Concentrations

Habitat	mg[As]/kg[fish dry weight]	mg[As]/kg[fish wet weight]	Species/location	Reference
Marine	0.190–65[a]	0.048–16	fish, marine	Donohue and Abernathy (1999)
Marine	8–37[a]	2–9.3	yellowtail flounder, Northwest Atlantic Ocean	Hellou et al. (1996)
Marine	2.4–19.8 (inorganic: 0.003–0.2)[a]	0.6–5	several species in Belgian fish market	Buchet and Lison (1998)
Freshwater	0.007–1.46[a]	0.028–5.8	fish, freshwater	Donohue and Abernathy (1999)
Freshwater	6.4	0.16 ± 0.23[a]	bottom feeding	Kidwell et al. (1995)
Freshwater	6.4	0.16 ± 0.14[a]	predatory fish	Kidwell et al. (1995)
Freshwater	<0.4	<0.1[a]	several, Savanna River	Burger et al. (2002)
Freshwater	1.3	0.32 ± 0.040[a]	bowfin, Savanna River	Burger et al. (2002)
Freshwater	NC	0.01–0.03	bluegill, yellow perch, largemouth bass	Chen and Folt (2000)
Freshwater	NC	0.017 (0.012.5–0.028)	6 fish species, California	CA OEHHA (2012)
Freshwater	<0.005–0.2[a]	<0.001–0.05	mixed, Candamo River, Peru	Gutleb et al. (2002)

Acronyms: NC = not calculated

[a]Fish tissue concentrations reported as wet weight were converted to dry weight (and the reverse) assuming 75% moisture content in fresh fish.

Arsenic concentrations in freshwater fish are much lower. As reported in ATSDR (2007), Kidwell et al. (1995) analyzed data from the National Contaminant Biomonitoring Program (1984–1985, 112 stations) and found similar concentrations in bottom-feeding fish (0.16 ± 0.23 mg[As]/kg[fish ww]; n = 2,020) and in "predatory" fish (0.16 ± 0.14 mg[As]/kg[fish ww]; n = 12). In fish from the Savannah River below DOE's Savanna River Site, Burger et al. (2002) found concentrations less than 0.1 mg[As]/kg[fish fillet ww] for bass, channel catfish, pickerel, yellow perch, black crappie, American eel, bluegill, and other fish, with only the bowfin showing higher concentrations—0.32 ± 0.04 mg[As]/kg[fish fillet ww]. Similarly, Gutleb

et al. (2002) found concentrations in freshwater fish from the unpolluted Candamo River in Peru from <0.005 to 0.2 mg[As]/kg[fish fillet dw], which would approximate <0.001–0.05 mg/kg ww.

In marine, estuarine, and freshwater bodies, inorganic arsenic (As), predominates (U.S. EPA 2003e). In fish, however, organoarsenical compounds predominate, with arsenobetaine, arsenocholine, monomethylarsonic acid (MMA), dimethylarsenic acid (DMA), and trimethylarsenic (TMA) identified in various species (U.S. EPA 2003e). In marine fish and shellfish, only 10–15 percent of arsenic is inorganic (U.S. EPA 2003e). In freshwater fish, limited field data suggest that organoarsenical compounds might predominate, but laboratory data indicate a wide range of organic-to-total arsenic ratios (U.S. EPA 2003e). Kaise et al. (1997) reported 88–99 percent organic arsenic in six freshwater fish species caught in a river, with more than half as TMA and most of the remainder as DMA. On the other hand, in laboratory studies in which fish were exposed to As(III) or As(V) in water, the fraction of total arsenic comprising organic arsenic compounds varied substantially from 0 to 94 percent (U.S. EPA 2003e).

Laboratory data on measured bioconcentration factor (BCF) values in saltwater and freshwater fish species are too sparse to allow comparison. For selecting BAFs and BSAFs, preference is given to field studies that are adequately conducted, with concentrations measured in water, sediments, and fish that are sampled at the same locations on the same dates. Those data for freshwater fish are described in the next section.

A.7.2 BAFs for Arsenic in Freshwater Fish

For the RTR Tiers 1 and 2 environmental screens for arsenic, the screening scenario assumes that people catch and consume fish from an onsite pond and that they eat 50-percent trophic level 4 (TL4) fish from the water column and 50-percent trophic level 3 (TL3) fish from the benthic environment. For arsenic modeling, EPA chose not to use the biokinetic model of aquatic food chain bioaccumulation (or trophic transfers) included in TRIM.FaTE. Instead, EPA uses arsenic-specific BAFs and BSAFs applied to TRIM.FaTE-estimated water and sediment concentrations, respectively. The BAF/BSAF approach should require fewer empirical data to estimate values for fewer model parameters than the biokinetic approach, which requires values for parameters

related to uptake and elimination via gills and ingestion with food for six components of an aquatic food web.

As of February 2016, EPA has not published BAFs for arsenic in fish that could be used to estimate bioaccumulation and risk at a national level. The current EPA NAWQC for arsenic are based on a BAF of 44, with the value for fish 1.0 and the value for oysters 350 (U.S. EPA 1985; Williams et al. 2006). Recently, EPA published BAF values and other data related to arsenic in organisms in marine and freshwaters (U.S. EPA 2003e). For BAFs, EPA separated data by habitat (marine, freshwater) and by trophic level (i.e., TLs 2, 3, and 4). The water-column fish consumed by people in the screening scenario for RTR assessments is assumed TL4. We therefore recommended using the highest BAF reported, 46.1 L/kg, for a freshwater carnivorous fish, largemouth bass, in the compilation for freshwater lentic ecosystems (see Table 3-4 in U.S. EPA 2003e). That value rounds to 46 L/kg for the arsenic BAF for the water-column carnivore for use in RTR environmental screens.

More recently, the State of California Office of Environmental Health Hazard Assessment (CA OEHHA 2012) derived a freshwater fish BAF of 17 L/kg[fish ww], calculated as the arithmetic mean arsenic BAF from six species of freshwater fish (based on Baker and King 1994, Huang et al. 2003, Lin et al. 2001, Liao et al. 2003, and Skinner 1985) (range of field-measured BAFs in natural lakes 12.5–28). California OEHHA concluded that a BAF of 44 is too high for its freshwater fish risk assessments and now uses the calculated value of 17 L/kg[fish ww] instead.

Given the variation in arsenic BAFs (and BSAFs) in the data presented by EPA (2003e), we decided to investigate arsenic bioaccumulation in more detail to provide additional information for consideration by EPA's Office of Air Quality Planning and Standards. In its 2003 technical review, EPA concluded that arsenic BAF values were too variable to allow the Agency to recommend a single BAF that would apply nationwide (U.S. EPA 2003e). Arsenic concentrations tend to be higher in estuarine and marine fish than in freshwater fish (Table A-19) and higher in filter-feeding invertebrates, including oysters and mussels, than in fish. Arsenic does not bioaccumulate in food chains (U.S. EPA 2003e, Section 1.2). In its grouping of BAF data in 2003, EPA calculated BAFs for animals in trophic level 2 (TL2), TL3, and TL4 for lakes, rivers, and estuaries separately. Thus, BAFs can potentially differ for TL2 lakes, TL2 rivers, and TL2 estuaries; TL3 lakes, TL3 rivers, and TL3 estuaries; and TL4 lakes, TL4 rivers, and TL4

estuaries (U.S. EPA 2003e). The Agency grouped organisms from different phyla (e.g., fish, insect larvae, mussels) if their food habits indicated the same or similar trophic level in the same habitat (e.g., TL3 lakes). We believe that including BAFs for species from different phyla for a specified habitat and trophic level contributed to the variation among BAFs within each habitat/trophic level group.

We found one study that appears to have identified a parameter that explains much of the variation in the freshwater BAF data reviewed by EPA (U.S. EPA 2003e). Williams et al. (2006) focused on field and lab studies of arsenic bioaccumulation and bioconcentration in freshwater *fish* only. They found an inverse relationship between field BAFs and arsenic concentrations in water, a trend observed for other metals (McGeer et al. 2003). Overall, measured concentrations of arsenic in the fillet or in the whole body of fish collected in the field were relatively constant (i.e., 51–370 µg[As]/kg[fish ww]),[17] although most freshwater fish contained less than 200 µg[As]/kg[fish ww] across fish species, trophic levels, and sizes (Table A-19).

In contrast, measured arsenic concentrations in the water ranged over roughly 3.5 orders of magnitude (0.02–56 µg[As]/L[freshwater]) (Williams et al. 2006). The measured BAFs ranged from 0.5 to 1,600 L/kg. Measured BAFs in waters with the highest concentrations (56 µg[As]/L) were 6.1 L/kg ww or less (one exception), while waters with the lowest arsenic concentration (0.085 µg[As]/L) yielded the highest BAF (1,600 L/kg, bluegill) as shown in Table A-20. The inverse correlation between the magnitude of field-measured BAFs and arsenic concentrations in water suggests some degree of internal regulation of arsenic by the fish at typical environmental concentrations (Williams et al. 2006).

BCFs measured in the laboratory, with higher arsenic concentrations in water than in the field studies, ranged from 0.1 to 15 L/kg at water concentrations ranging from 10 to 18,100 µg/L; whole-fish concentrations ranged from 100 to 11,700 µg[As]/kg[fish ww]. The laboratory BCF values are presented after the BAF values in Table A-20.

[17]From over 50 separate fish species/sizes sampled over 6 field studies, Table 1, in Williams et al. (2006). Four unidentified composite samples and one measurement from creek chub of 2,360 µg[As]/kg[fish ww] excluded.

Table A-20. BAF/BCF Values for Freshwater Fish Exposed to Different Water Concentrations of Arsenic

Fish species, condition	Study Type	µg[As]/ L[water]	BAF or BCF (L/kg)	Location	Reference
Bluegill	Field BAF	0.085	1600	20 lakes in northeastern United States for U.S. EPA EMAP	Chen et al. (2000)
Mixed salmonids		0.022	3091		
Smallmouth bass		0.107	542		
Smallmouth bass		0.107	533		
White perch		0.367	322		
Pumpkinseed		0.113	265		
Largemouth bass		0.409	46		
Mottled sculpin	Field BAF	0.37	811	Blacklick Run, MD	Mason et al. (2000), as cited in Williams et al. (2006) [incorrectly cited as 2002 in Table 1]
Blacknose dace			541		
Brook trout, small			541		
Brook trout, large			270		
White sucker	Field BAF	0.67	448	Harrington Creek, MD	Mason et al. (2000), as cited in Williams et al. (2006) [incorrectly cited as 2002 in Table 1]
Brook trout, large			299		
Brook trout, small			299		
Creek chub			299		
Alewife	Field BAF	0.78	46	Upper Mystic Lake, MA	Chen and Folt (2000)
Killifish			41		
Yellow perch			28		
Largemouth bass			23		
Bluegill			22		
Black crappie			19		
Miscellaneous “omnivores”	Field BAF	5.1	5.1	Moon Lake, MS	Cooper and Gillespie (2001)
Carp (n = 5)	Field BAF	12	12	Upper Gila River, AZ	Baker and King (1994)
Channel catfish (n = 4)		20	9.7		
Flathead catfish		20	6.3		
Amphidormous goby	Field BAF	30	12	Haya-kawa River, Japan	Kaise et al. (1997)
Goby			11		
Fatminnow			8.9		
Japanese dace			3.3		
Sweet fish			1.7		
Creek chub	Field BAF	56	42*	Moira Lake, Ontario Canada	Azcue and Dixon (1994) *considered an outlier
Pumpkinseed			6.1		
Golden shiner			3.0		
White sucker			2.4		
Rock bass			2.3		
Banded killifish			1.8		
Largemouth bass			1.5		
Yellow perch			1.4		
Walleye			1.4		
Bluntnose minnow			0.9		
Longnose gar			0.9		
Emerald shiner			0.6		
Spottail shiner			0.5		
Northern pike			0.4		

Fish species, condition	Study Type	µg[As]/ L[water]	BAF or BCF (L/kg)	Location	Reference
Bluegill juvenile	Lab BCF	10	12	Laboratory mesocosm, 16 weeks	Gilderhus (1966)
Bluegill adult		10	14		
juvenile		50	10		
adult		50	7.8		
juvenile		260	2.5		
adult		260	2.0		
juvenile		610	2.5		
adult		610	1.9		
Rainbow trout 5 °C	Lab BCF	10	15	Lab, 11-week exposure, Ontario groundwater, at 5 and 15 °C	McGeachy and Dixon (1990)
Rainbow trout 15 °C		10	15		
5 °C		1,400	0.2		
15 °C		1,400	0.2		
15 °C		8,400	0.2		
5 °C		16,300	0.1		
15 °C		18,100	0.2		
Rainbow trout	Lab BCF	<20	15	Lab, 181-day exposure	Rankin and Dixon (1994)
		760	0.3		
		2,480	0.2		

Acronyms and abbreviations: BAF = bioaccumulation factor (i.e., arsenic accumulation from both water and food); BCF = bioconcentration factor (i.e., arsenic accumulation from water via the gills); EMAP = U.S. EPA Environmental Monitoring and Assessment Program; Lab = laboratory
Source: Williams et al. (2006).

Williams et al. (2006) demonstrated an inverse relationship between arsenic concentrations in water and in fish for low, environmentally common arsenic concentrations in surface waters (i.e., 0.02–56 µg/L). The relationship (Equation A-4) is close, with an r^2 of 0.82.

$$\textit{Field\_BAF}\ (\text{L/kg}) = 87.4 * \textit{Water\_Concentration}\ (\mu\text{g/L})^{-0.925} \qquad \text{Eq. A-4}$$

At higher arsenic concentrations in water (e.g., 10–12,000 µg/L), the laboratory BCFs were still inversely related to water concentration; however, the exponent was smaller (Equation A-5; Williams et al. 2006). The relationship is close ($r^2 = 0.79$).

$$\textit{Lab\_BCF}\ (\text{L/kg}) = 78.7 * \textit{Water\_Concentration}\ (\mu\text{g/L})^{-0.669} \qquad \text{Eq. A-5}$$

The smaller exponent suggests that internal arsenic regulation might be impaired at higher water concentrations.

The trends shown in Table A-20 are apparent despite grouping fish that feed at different trophic levels. In fact, some evidence indicates that arsenic concentrations in fish decrease slightly with increasing trophic level. For example, Chen and Folt (2000) measured arsenic and lead concentrations in Upper Mystic Lake, Massachusetts, in small and large zooplankton and in six

species of fish in three different seasons. The lake, designated as a Superfund site, had been contaminated by past leather and chemical manufacturing upstream. Arsenic was elevated in the zooplankton relative to zooplankton in uncontaminated lakes. Arsenic decreased, however, with increasing trophic level. Fish from Mystic Lake contained the same arsenic concentrations as fish from uncontaminated lakes in the northeastern United States. The highest arsenic concentrations were in planktivorous fish that consumed zooplankton that were high in arsenic. Subsequent consumers in the food chain had lower tissue concentrations of arsenic, leading to the idea that arsenic "biodiminishes" with increasing trophic level in fish. Chen and Folt (2000) found that arsenic concentrations in fish were 10–20 times lower than in the zooplankton, and concentrations in larger plankton (>202 µm) were less than in smaller plankton (45–202 µm). Arsenic concentrations in all fish sampled (planktivores—alewife and killifish; omnivores—black crappie, bluegill sunfish, and yellow perch; and piscivores—largemouth bass) were between 0.01 and 0.03 µg/g wet weight.

Based on the analysis of Williams et al. (2006), for refined site-specific RTR assessments, we recommend using the two equations above (Equation A-4 and Equation A-5) to estimate bioaccumulation of arsenic in water-column fish (water-column carnivore). Application of the equations would be conditional on the TRIM.FaTE-estimated arsenic concentration in the water column being less than or more than 10 µg/L. A warning flag should alert the user if the estimated arsenic concentrations in water are less than 0.01 µg/L or more than 20,000 µg/L, which are concentrations beyond the observed data upon which the empirical models are based.

For simplicity, however, we applied a BAF for the water-column carnivore of 46 L[water]/kg[fish wet weight] (USEPA 2003e, Tables 3.4 and 3.9, highest value for TL4 fish, largemouth bass). That BAF is below 1,000 L/kg, which is a typical criterion for a chemical to be considered bioaccumulative. The BAF values for TL3 fish (alewife) and TL2 fish (carp) were 95 L/kg and 71 L/kg, respectively (USEPA 2003e).

A.7.3 BSAFs for Arsenic in Freshwater Benthic Invertebrates and Fish

As discussed in detail in Appendix 6 of the Risk Report,[18] for the RTR Tiers 1, 2, and 3 human health screens for arsenic, EPA relies on the BSAF/BAF approach rather than biokinetic modeling of aquatic food chain bioaccumulation (or trophic transfers). Predicting bioaccumulation of metals and transition elements requires chemical-specific empirical data; no chemical property, such as K_{ow}, predicts bioaccumulation of these elements across organisms in aquatic food chains.

Bechtel Jacobs Company (BJC 1998) assembled data to estimate freshwater BSAFs for benthic invertebrates (predominantly the aquatic larval stage of several groups of insects) for use in risk assessments on DOE properties. As for most estimates of BSAFs for metals published in the literature, BJC (1998) reported BSAFs as the ratio of dry-weight biota concentration to dry-weight sediment concentration (i.e., kg[dry weight sediments]/kg[dry weight biota]. For a dataset of 55 sediment-invertebrate BSAFs, BJC (1998) found a mean value of 0.329 kg[dw]/kg[dw]. For 49 of those studies for which the organisms had not been depurated (i.e., moved to clean sediments and allowed to eliminate the chemical), the mean BSAF was 0.240 kg[dw]/kg[dw]. TRIM.FaTE calculates both invertebrate and fish concentrations on a wet-weight basis. For the benthic invertebrates reviewed by BJC, typically 70-percent water, the fresh-weight BSAF would be lower. The BSAF multiplied by 0.30 (fraction dry weight) yields BSAFs of 0.1 and 0.07 kg[dry sediment]/kg[wet weight invertebrates] for the set of 55 and set of 49 studies, respectively.

The data described above could be used to parameterize the beginning of the benthic food chain in TRIM.FaTE for arsenic. For the RTR human health and environmental screens, however, we are not employing the TRIM.FaTE biokinetic food-web model to estimate bioaccumulation. Thus, we needed to find a BSAF value for freshwater fish that consume benthic invertebrates and small bottom fish to calculate their tissue concentrations relative to sediment concentrations.

We found a single study that measured a BSAF for freshwater fish in the field. Davis et al. (1996) measured arsenic concentrations in fish and sediments in a holding pond at the

[18] Appendix 6 to the Risk Report is the *Technical Support Document for the TRIM-Based Multipathway Tiered Screening Methodology for RTR*.

Industriplex Superfund Site north of Boston, Massachusetts, that had been contaminated with arsenic in the 1970s. At a depth of 45 cm in the sediments, they measured approximately 500 µg[As]/L[pore water] and 1,000 mg[As]/kg[dry weight sediment]. They found increasing arsenic concentrations with decreasing depth in the sediment column: 1,700 µg[As]/L[pore water] and 1,200 mg[As]/kg[sediment dw] at a depth of 30 cm; and 5,500 µg[As]/L[pore water] and 3,000 mg[As]/kg[sediment dw] at the surface (the top few cm). They calculated a sediment-water Kd for arsenic of 560 L/kg. Arsenic near the surficial sediments was 1,700 µg[total As]/L, with <1.0 µg[As]/L MMA, <1.9 µg[As]/L DMA, 1,100 µg/L as As(III), and 610 µg/L as As(V).

Davis et al. (1996) measured arsenic in the fillet portion of bottom-feeding fish (brown bullhead and white sucker) and in nearby surficial sediments. Although they did not describe their methods for estimating arsenic concentrations in the fish or in bulk sediments in detail, their goal was to report a BSAF that could predict wet-weight fish concentrations of arsenic. They reported 1.19 mg[As]/kg[ww fish fillet] and a surficial sediment concentration of 1,830 mg[As]/kg[sediment]. Those data indicate a BSAF of 6.5×10^{-4} kg[bulk sediment]/kg[ww fish fillet], which we have adopted for RTR analyses.

We have only a single estimate of a BSAF for freshwater fish. This BSAF might be lower than is typical in most surface water bodies for two reasons. First, the exposure concentration is relatively high. Based on the findings of Williams et al. (2006), high exposure concentrations would likely result in low bioaccumulation for arsenic. Second, Davis et al. (1996) measured a relatively high sediment-water Kd for arsenic of 560 L/kg, which is higher than the median value of 316 L/kg (logKd of 2.5 L/kg, range of logKd 1.6–4.3 L/kg) reported by EPA for a sediment-water Kd (U.S. EPA 2005b). Thus, the bioavailability of arsenic in sediments at the Superfund site investigated by Davis et al. (1996) might have been lower than at most locations.

A.8 Environmental Screening Threshold Emission Rates

As described in the main report, the Tier 1 environmental screening thresholds are expressed as chemical- and assessment-endpoint-specific emission rates (in tons per year). They are backcalculated from media-specific benchmarks or TRVs for fish-eating birds and mammals using TRIM.FaTE. Those screening emission thresholds are listed in Table A-21. The methods of changing thresholds for Tiers 2 and 3 also are described in the main report.

Table A-21. Tier 1 Environmental Screening Threshold Emission Rates (ESTER) for each PB-HAP and each Benchmark Assessed in the Environmental Risk Screen

PB-HAP	Assessment Endpoint	Benchmark and Effect Level[a]	Tier 1 ESTER (TPY)
Arsenic	Fish-consuming birds	NOAEL (American merganser)	6.20E+00
		LOAEL (American merganser)	6.20E+01
	Fish-consuming mammals	NOAEL (mink)	6.57E-01
		LOAEL (mink)	6.57E+00
	Sediment Community	Threshold Level	5.97E-01
		Probable-effect Level	2.40E+00
	Surface Soil – Dist. 1 – 312 m	Threshold – Mammalian Insectivores (shrew)	1.92E+00
		Threshold – Avian Insectivores (woodcock)	1.80E+00
		Threshold Level – Plant Community	7.53E-01
	Surface Soil – Dist. 2 – 850 m	Threshold – Mammalian Insectivores (shrew)	3.63E-01
		Threshold – Avian Insectivores (woodcock)	3.39E-01
		Threshold Level – Plant Community	1.42E-01
	Surface Soil – Dist. 3 – 1,500 m	Threshold – Mammalian Insectivores (shrew)	7.25E-01
		Threshold – Avian Insectivores (woodcock)	6.77E-01
		Threshold Level – Plant Community	2.84E-01
	Surface Soil – Dist. 4 – 3,500 m	Threshold – Mammalian Insectivores (shrew)	3.35E+00
		Threshold – Avian Insectivores (woodcock)	3.13E+00
		Threshold Level – Plant Community	1.31E+00
	Surface Soil – Dist. 5 – 7,500 m	Threshold – Mammalian Insectivores (shrew)	1.55E+01
		Threshold – Avian Insectivores (woodcock)	1.45E+01
		Threshold Level – Plant Community	6.06E+00
	Water-column Community	Threshold Level (chronic)	7.24E+01
		Frank-effect Level (acute)	1.64E+02
Cadmium	Fish-consuming birds	NOAEL (American merganser)	2.22E-02
		LOAEL (American merganser)	3.17E-02
	Fish-consuming mammals	NOAEL (mink)	4.43E-02
		LOAEL (mink)	4.44E-01
	Sediment Community	No-effect Level	1.04E-01
		Threshold Level	3.77E-01
		Probable-effect Level	1.10E+00
	Surface Soil – Dist. 1 – 312 m	Threshold – Mammalian Insectivores (shrew)	3.28E-02
		Threshold – Avian Insectivores (woodcock)	7.01E-02
		Threshold Level – Plant Community	2.91E+00
		Threshold Level – Invertebrate Community	1.27E+01

PB-HAP	Assessment Endpoint	Benchmark and Effect Level[a]	Tier 1 ESTER (TPY)
	Surface Soil – Dist. 2 – 850 m	Threshold – Mammalian Insectivores (shrew)	7.46E-03
		Threshold – Avian Insectivores (woodcock)	1.60E-02
		Threshold Level – Plant Community	6.64E-01
		Threshold Level – Invertebrate Community	2.90E+00
	Surface Soil – Dist. 3 – 1,500 m	Threshold – Mammalian Insectivores (shrew)	1.51E-02
		Threshold – Avian Insectivores (woodcock)	3.23E-02
		Threshold Level – Plant Community	1.34E+00
		Threshold Level – Invertebrate Community	5.88E+00
	Surface Soil – Dist. 4 – 3,500 m	Threshold – Mammalian Insectivores (shrew)	8.52E-02
		Threshold – Avian Insectivores (woodcock)	1.82E-01
		Threshold Level – Plant Community	7.58E+00
		Threshold Level – Invertebrate Community	3.31E+01
	Surface Soil – Dist. 5 – 7,500 m	Threshold – Mammalian Insectivores (shrew)	3.99E-01
		Threshold – Avian Insectivores (woodcock)	8.53E-01
		Threshold Level – Plant Community	3.54E+01
		Threshold Level – Invertebrate Community	1.55E+02
	Water-column Community	Threshold Level (chronic)	2.41E-01
		Frank-effect Level (acute)	6.02E-01
Mercury – divalent mercury (Hg++) emissions and exposures	Sediment Community	Threshold Level	3.64E-03
		Probable-effect Level	1.91E-02
	Surface Soil – Dist. 1 – 312 m	Threshold Level – Plant Community	1.96E-03
		Threshold Level – Invertebrate Community	6.54E-04
	Surface Soil – Dist. 2 – 850 m	Threshold Level – Plant Community	9.15E-04
		Threshold Level – Invertebrate Community	3.05E-04
	Surface Soil – Dist. 3 – 1,500 m	Threshold Level – Plant Community	2.20E-03
		Threshold Level – Invertebrate Community	7.35E-04
	Surface Soil – Dist. 4 – 3,500 m	Threshold Level – Plant Community	1.15E-02
		Threshold Level – Invertebrate Community	3.83E-03
	Surface Soil – Dist. 5 – 7,500 m	Threshold Level – Plant Community	7.23E-02
		Threshold Level – Invertebrate Community	2.41E-02
	Water-column Community	Threshold Level (chronic)	2.91E-01
		Frank-effect Level (acute)	5.30E-01
Mercury – Hg++ emissions, but exposure to MeHg	Fish-consuming birds	NOAEL (American merganser)	3.37E-03
		LOAEL (American merganser)	2.02E-02
	Fish-consuming mammals	NOAEL (mink)	1.79E-02
		LOAEL (mink)	8.89E-02

PB-HAP	Assessment Endpoint	Benchmark and Effect Level[a]	Tier 1 ESTER (TPY)
	Sediment Community	Threshold Level	2.08E+00
		Probable-effect Level	1.04E+01
	Surface Soil – Dist. 1 – 312 m	Threshold Level – Invertebrate Community	3.94E-02
	Surface Soil – Dist. 2 – 850 m	Threshold Level – Invertebrate Community	1.84E-02
	Surface Soil – Dist. 3 – 1,500 m	Threshold Level – Invertebrate Community	4.42E-02
	Surface Soil – Dist. 4 – 3,500 m	Threshold Level – Invertebrate Community	2.32E-01
	Surface Soil – Dist. 5 – 7,500 m	Threshold Level – Invertebrate Community	1.45E+00
	Water-column Community	Threshold Level (chronic)	1.48E-01
		Frank-effect Level (acute)	5.23E+00
BaP-equivalents	Fish-consuming mammals	NOAEL (mink)	1.33E+02
		LOAEL (mink)	1.33E+03
	Sediment Community	No-effect Level	1.32E+00
		Threshold Level	6.20E+00
		Probable-effect Level	5.99E+01
	Surface Soil – Dist. 1 – 312 m	Threshold – Mammalian Insectivores (shrew)	6.56E-01
	Surface Soil – Dist. 2 – 850 m	Threshold – Mammalian Insectivores (shrew)	8.17E-01
	Surface Soil – Dist. 3 – 1,500 m	Threshold – Mammalian Insectivores (shrew)	1.43E+00
	Surface Soil – Dist. 4 – 3,500 m	Threshold – Mammalian Insectivores (shrew)	5.06E+00
	Surface Soil – Dist. 5 – 7,500 m	Threshold – Mammalian Insectivores (shrew)	1.83E+01
	Water-column Community	Threshold Level (chronic)	5.16E+00
		Frank-effect Level (acute)	8.84E+01
2,3,7,8-TCDD equivalents	Fish-consuming birds	NOAEL (American merganser)	6.61E-06
		LOAEL (American merganser)	6.61E-05
	Fish-consuming mammals	NOAEL (mink)	8.58E-06
		LOAEL (mink)	8.58E-05
	Sediment Community	Threshold Level	6.68E-06
	Surface Soil – Dist. 1 – 312 m	Threshold – Mammalian Insectivores (shrew)	1.17E-07
	Surface Soil – Dist. 2 – 850 m	Threshold – Mammalian Insectivores (shrew)	5.04E-08
	Surface Soil – Dist. 3 – 1,500 m	Threshold – Mammalian Insectivores (shrew)	8.33E-08
	Surface Soil – Dist. 4 – 3,500 m	Threshold – Mammalian Insectivores (shrew)	2.80E-07
	Surface Soil – Dist. 5 – 7,500 m	Threshold – Mammalian Insectivores (shrew)	8.78E-07
	Water-column Community	Threshold Level	6.67E-04
		Frank-effect Level	6.67E+00
Lead	Ambient Air	NAAQS Secondary Standard	NA

Acronyms and abbreviations: BaP = benzo[a]pyrene; Dist. = distance; TCDD = tetrachlorodibenzo-p-dioxin; Hg = mercury; Hg++ = divalent mercury; MeHg = methyl mercury; NA = not applicable; NAAQS = National Ambient Air

Quality Standards; NOAEL = no-observed-adverse-effect level; LOAEL = lowest-observed-adverse-effect level; TPY = tons per year

[a]Insectivore means diet of insects; however, here insectivore means specifically feeding on both insects (larvae and adults) and other invertebrates (e.g., earthworms) that dwell in surface soil, as the named species (shrew and woodcock) suggest.

A.9 References

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Appendix 10
Detailed Risk Modeling Results

Table 1 - Facility Identification Information

Facility NEI ID	Facility Name	Address	City	State	County
01033110005979824	Constellium	3509 E 2Nd St	Muscle Shoals	AL	Colbert County
01073110006548566	Precoat Metals Division Sequa Corporation	3399 Davey Allison Blvd	Hueytown	AL	Jefferson County
05093110028304957	Precoat Metals	5888 East Couny Road 180	Blytheville	AR	Mississippi County
06071110002647743	Nci Group Inc., Dba, Metal Coaters Of Ca	9133 Center Ave	Rancho Cucamonga	CA	San Bernardino County
12069110001969576	Wolverine Advanced Materials	10825 County Road 44	Leesburg	FL	Lake County
13045110035892371	Southwire Company -Carrollton	One Southwire Drive	Carrollton	GA	Carroll County
17031110000429000	Material Sciences Corp	2300 E Pratt Blvd	Elk Grove Village	IL	Cook County
17031110000433102	Signode Corp	7701 W 71St St	Bridgeview	IL	Cook County
17031110067426137	Wismarq Corp	11440 Addison St	Franklin Park	IL	Cook County
17119110003064567	Precoat Metals	25 Northgate Industrial Dr	Granite City	IL	Madison County
18059110000396535	Precoat Metals A Division Of Sequa Corp	1950 E Main St	Greenfield	IN	Hancock County
18091110000592136	Precoat Metals Corp.	2Nd Rd & Hupp Rd	La Porte	IN	La Porte County
18127110000747862	Precoat Metals Division Sequa Coatings	6300 Us Hwy 12	Portage	IN	Porter County
18141110000399453	Edcoat Limited Partnership	30350 Edison Road	New Carlisle	IN	St. Joseph County
18145110021290413	Jupiter Aluminum Corporation Jupiter C	205 E Carey St	Fairland	IN	Shelby County
18173110000602045	Alcoa Inc. - Warrick Operations	State Road 66 & State Road 61	Newburgh	IN	Warrick County
19163110000416005	Alcoa Inc	4879 State St	Riverdale	IA	Scott County
19163110000416014	Nichols Aluminum - Davenport - Rockingham Rd	1725 Rockingham Rd	Davenport	IA	Scott County
21091110000380935	Precoat Metals	2604 River Rd	Hawesville	KY	Hancock County
21091110058164187	Aleris Rolled Products Inc	1372 Ky 1957	Lewisport	KY	Hancock County
21141110000605364	Logan Aluminum Inc	6920 Lewisburg Rd	Russellville	KY	Logan County
27053110022448092	Edco Products Inc	8700 Excelsior Blvd	Hopkins	MN	Hennepin County
28049110000376815	Metal Coaters	951 Prisock Road	Byram	MS	Hinds County
28049110000376879	Precoat Metals	1095 Mendell Davis Drive	Jackson	MS	Hinds County
29510110000441255	Precoat Metals-Precoat Metals	4301 S Spring	St. Louis	MO	St. Louis City
34035110000858670	Agfa Corporation	50 Meister Ave	Branchburg	NJ	Somerset County
36015110046535344	Vulcraft Of New York Inc	5362 Railroad St	Chemung	NY	Chemung County
39017110000392575	Metal Coaters (1409000037)	2400 Yankee Road	Middletown	OH	Butler County
39035110012520744	Usg Interiors, Inc., American Metals Corp., Westlake (1318617346)	1000 Crocker Rd.	Westlake	OH	Cuyahoga County
39059110000384548	Centria (0630010006)	530 North Second Street	Cambridge	OH	Guernsey County
39095110000384218	Crown Cork & Seal (0448010466)	5201 Enterprise Blvd	Toledo	OH	Lucas County
39119110000384487	Ak Steel - Zanesville Works (0660010006)	1724 Linden Avenue	Zanesville	OH	Muskingum County
39129110000381836	Aleris Rolled Products, Inc. (0165000045)	1 Reynolds Road	Ashville	OH	Pickaway County
39149110000394314	Ply Gem (0575010103)	2615 Campbell Road	Sidney	OH	Shelby County
39155110000389963	Novelis Corporation (0278080136)	390 Griswold Street, N.E.	Warren	OH	Trumbull County

Table 1 - Facility Identification Information

Facility NEI ID	Facility Name	Address	City	State	County
39173110000383353	Msc Walbridge Coatings Inc (0387000046)	30610 East Broadway	Walbridge	OH	Wood County
42003110008470724	Us Steel Corp/Irvin Plt	Camp Hollow Rd	Dravosburg	PA	Allegheny County
42017110000336495	Brightsmith Llc/Falls Twp	120 Enterprise Ave	Morrisville	PA	Bucks County
42071110000334200	Alumax Mill Prod Inc/Mill Prod	1480 Manheim Pike	Lancaster	PA	Lancaster County
42081110000584163	Springs Window Fashions/Montgomery Plt	8601 State Route 405	Montgomery	PA	Lycoming County
45047110016994118	Fuji Film Inc P Plant	211 Puckett Ferry Rd	Greenwood	SC	Greenwood County
45079110000352494	Consolidated Systems, Inc	650 Rosewood Dr	Columbia	SC	Richland County
48201110000500583	Precoat Metals Houston	16402 Jacinto Port	Houston	TX	Harris County
51121110000343003	Wolverine Advanced Materials - Blacksburg	201 Industrial Park Rd	Blacksburg	VA	Montgomery County
51121110000343012	Wolverine Gasket Division - Eagle-Picher Ind	3175 State Street	Blacksburg	VA	Montgomery County
54009110000344716	Precoat Metals	4502 Freedom Way	Weirton	WV	Brooke County
54009110067212475	Wheeling Corrugating Company	West Virginia Route 2	Beech Bottom	WV	Brooke County
55133110000858652	Wismarq Corporation	930 Armour Rd	Oconomowoc	WI	Waukesha County

Table 2a – Maximum Predicted HEM-3 Chronic Risks
Actual Emissions

Facility NEI ID	Category Chronic Risk [1]				Facility Chronic Risk [1]			SC % of Facility-wide Cancer Risk
	Cancer MIR	Cancer Incidence	Noncancer Max HI	Target Organ	Cancer MIR	Noncancer Max HI	Target Organ	
01033110005979824	1.29E-08	6.26E-06	2.41E-04	neurological	1.29E-08	2.41E-04	neurological	100%
01073110006548566	1.57E-07	5.39E-05	1.58E-03	respiratory	**4.62E-06**	1.17E-02	kidney	3%
05093110028304957	9.44E-08	4.30E-06	8.98E-04	respiratory	4.62E-07	3.85E-04	respiratory	20%
06071110002647743	1.49E-08	3.37E-06	6.63E-04	neurological	2.53E-07	2.39E-03	respiratory	6%
12069110001969576	0.00E+00	0.00E+00	1.25E-05	developmental	0.00E+00	1.22E-05	developmental	NA
13045110035892371	3.40E-09	7.81E-08	3.23E-05	respiratory	3.40E-09	3.23E-05	respiratory	100%
17031110000429000	5.16E-10	3.48E-06	1.42E-05	neurological	5.16E-10	1.42E-05	neurological	100%
17031110000433102	1.40E-06	8.19E-04	1.33E-02	respiratory	1.40E-06	1.33E-02	respiratory	100%
17031110067426137	1.20E-07	5.26E-04	1.12E-03	respiratory	1.20E-07	1.12E-03	respiratory	100%
17119110003064567	1.07E-07	9.83E-05	7.83E-03	respiratory	1.07E-07	8.26E-03	respiratory	100%
18059110000396535	5.94E-07	1.27E-04	5.82E-03	respiratory	**2.78E-05**	2.65E-01	respiratory	2%
18091110000592136	1.01E-07	7.54E-05	9.93E-04	respiratory	1.52E-07	2.14E-03	respiratory	67%
18127110000747862	0.00E+00	0.00E+00	1.07E-02	respiratory	**1.80E-06**	1.52E-02	respiratory	0%
18141110000399453	1.51E-07	1.63E-05	1.40E-03	respiratory	5.50E-07	5.00E-03	respiratory	27%
18145110021290413	3.39E-08	9.84E-06	3.22E-04	respiratory	3.39E-08	3.22E-04	respiratory	100%
18173110000602045	4.38E-09	2.83E-06	7.25E-04	reproductive	8.01E-07	7.17E-02	reproductive	1%
19163110000416005	6.07E-08	5.37E-06	5.77E-04	respiratory	**2.22E-06**	**1.52E+00**	respiratory	3%
19163110000416014	**1.05E-05**	2.13E-04	9.86E-02	respiratory	**1.05E-05**	9.86E-02	respiratory	100%
21091110000380935	1.68E-07	4.08E-05	1.56E-03	respiratory	4.00E-07	1.91E-03	respiratory	42%
21091110058164187	3.94E-07	7.61E-06	3.86E-03	respiratory	1.08E-06	2.11E-02	respiratory	36%
21141110000605364	2.93E-07	2.09E-06	2.78E-03	respiratory	**2.48E-05**	3.71E-01	developmental	1%
27053110022448092	7.14E-08	2.80E-05	6.02E-04	respiratory	7.82E-08	6.29E-04	respiratory	91%
28049110000376815	6.78E-07	2.79E-04	6.46E-03	respiratory	6.78E-07	6.46E-03	respiratory	100%
28049110000376879	**3.76E-06**	4.67E-04	4.40E-02	respiratory	**1.63E-05**	4.31E-02	respiratory	23%
29510110000441255	1.33E-06	5.99E-05	1.11E-01	reproductive	**1.78E-06**	1.14E-01	reproductive	75%
34035110000858670	**6.85E-06**	6.12E-04	6.51E-02	respiratory	**6.85E-06**	6.51E-02	respiratory	100%
36015110046535344	1.17E-07	3.14E-06	1.12E-03	respiratory	1.17E-07	1.12E-03	respiratory	100%
39017110000392575	1.36E-07	4.77E-05	1.30E-03	respiratory	1.36E-07	1.30E-03	respiratory	100%
39035110012520744	5.04E-07	3.62E-05	4.67E-03	respiratory	6.18E-07	5.70E-03	respiratory	81%
39059110000384548	0.00E+00	0.00E+00	4.68E-03	neurological	1.62E-09	4.40E-03	neurological	0%
39095110000384218	3.40E-07	5.71E-05	6.48E-03	neurological	3.40E-07	6.48E-03	neurological	100%
39119110000384487	1.72E-07	9.29E-06	1.64E-03	respiratory	1.93E-07	1.21E-02	skeletal	89%
39129110000381836	5.04E-07	9.43E-05	4.83E-03	respiratory	6.46E-07	6.09E-03	respiratory	78%
39149110000394314	**5.57E-06**	2.95E-04	5.58E-02	respiratory	**3.54E-05**	3.53E-01	respiratory	16%
39155110000389963	**1.10E-05**	3.07E-04	1.05E-01	respiratory	**1.10E-05**	1.05E-01	respiratory	100%
39173110000383353	0.00E+00	0.00E+00	6.07E-05	neurological	2.09E-08	1.10E-03	respiratory	0%
42003110008470724	3.46E-09	6.61E-07	4.80E-04	reproductive	3.41E-07	4.96E-02	respiratory	1%
42017110000336495	0.00E+00	0.00E+00	1.34E-06	neurological	0.00E+00	3.77E-05	neurological	NA
42071110000334200	5.45E-09	2.25E-06	4.95E-05	respiratory	8.88E-08	**4.92E+00**	respiratory	6%
42081110000584163	1.14E-06	7.38E-06	1.12E-02	respiratory	1.14E-06	1.12E-02	respiratory	100%
45047110016994118	0.00E+00	0.00E+00	3.52E-05	developmental	4.44E-07	6.34E-02	reproductive	0%
45079110000352494	5.70E-08	2.37E-05	5.30E-04	respiratory	1.01E-07	1.19E-03	neurological	56%
48201110000500583	7.15E-07	5.46E-04	9.36E-03	respiratory	**1.10E-05**	2.03E-02	respiratory	6%
51121110000343003	0.00E+00	0.00E+00	8.92E-06	neurological	0.00E+00	8.94E-06	neurological	NA
51121110000343012	0.00E+00	0.00E+00	6.79E-05	neurological	0.00E+00	4.73E-05	neurological	NA
54009110000344716	**6.51E-06**	3.47E-04	6.34E-02	respiratory	**6.51E-06**	6.34E-02	respiratory	100%
54009110067212475	8.60E-07	1.05E-05	8.06E-03	respiratory	8.60E-07	8.06E-03	respiratory	100%
55133110000858652	2.22E-07	3.17E-05	2.11E-03	respiratory	2.22E-07	2.11E-03	respiratory	100%

1 BOLD indicates a cancer risk great than 1 in a million or a noncancer risk greater than 1

Table 2b – Maximum Predicted HEM-3 Chronic Risks
Allowable Emissions

	Category Chronic Risk [1]			
Facility NEI ID	Cancer MIR	Cancer Incidence	Noncancer Max HI	Target Organ
01033110005979824	1.42E-08	6.88E-06	2.65E-04	neurological
01073110006548566	1.73E-07	5.93E-05	1.74E-03	respiratory
05093110028304957	1.04E-07	4.73E-06	9.87E-04	respiratory
06071110002647743	1.64E-08	3.67E-06	7.29E-04	neurological
12069110001969576	0.00E+00	0.00E+00	1.38E-05	developmental
13045110035892371	3.74E-09	8.59E-08	3.55E-05	respiratory
17031110000429000	5.68E-10	3.81E-06	1.57E-05	neurological
17031110000433102	**1.54E-06**	9.01E-04	1.46E-02	respiratory
17031110067426137	1.32E-07	5.79E-04	1.23E-03	respiratory
17119110003064567	1.18E-07	1.08E-04	8.62E-03	respiratory
18059110000396535	6.53E-07	1.40E-04	6.40E-03	respiratory
18091110000592136	1.11E-07	8.29E-05	1.09E-03	respiratory
18127110000747862	0.00E+00	0.00E+00	1.18E-02	respiratory
18141110000399453	1.66E-07	1.79E-05	1.54E-03	respiratory
18145110021290413	3.73E-08	1.08E-05	3.54E-04	respiratory
18173110000602045	4.82E-09	3.11E-06	7.98E-04	reproductive
19163110000416005	6.68E-08	5.91E-06	6.34E-04	respiratory
19163110000416014	**1.16E-05**	2.34E-04	1.08E-01	respiratory
21091110000380935	1.85E-07	4.49E-05	1.72E-03	respiratory
21091110058164187	4.33E-07	8.37E-06	4.25E-03	respiratory
21141110000605364	3.22E-07	2.30E-06	3.06E-03	respiratory
27053110022448092	7.85E-08	3.08E-05	6.62E-04	respiratory
28049110000376815	7.46E-07	3.07E-04	7.10E-03	respiratory
28049110000376879	**4.13E-06**	5.14E-04	4.83E-02	respiratory
29510110000441255	1.47E-06	6.59E-05	1.22E-01	reproductive
34035110000858670	**7.54E-06**	6.74E-04	7.16E-02	respiratory
36015110046535344	1.29E-07	3.46E-06	1.23E-03	respiratory
39017110000392575	1.50E-07	5.24E-05	1.43E-03	respiratory
39035110012520744	5.54E-07	3.98E-05	5.14E-03	respiratory
39059110000384548	0.00E+00	0.00E+00	5.15E-03	neurological
39095110000384218	3.74E-07	6.28E-05	7.12E-03	neurological
39119110000384487	1.90E-07	1.02E-05	1.80E-03	respiratory
39129110000381836	5.55E-07	1.04E-04	5.32E-03	respiratory
39149110000394314	**6.12E-06**	3.24E-04	6.14E-02	respiratory
39155110000389963	**1.21E-05**	3.37E-04	1.15E-01	respiratory
39173110000383353	0.00E+00	0.00E+00	6.68E-05	neurological
42003110008470724	3.81E-09	7.25E-07	5.28E-04	reproductive
42017110000336495	0.00E+00	0.00E+00	1.47E-06	neurological
42071110000334200	6.00E-09	2.47E-06	5.45E-05	respiratory
42081110000584163	1.25E-06	8.12E-06	1.23E-02	respiratory
45047110016994118	0.00E+00	0.00E+00	3.87E-05	developmental
45079110000352494	6.27E-08	2.60E-05	5.83E-04	respiratory
48201110000500583	7.86E-07	6.00E-04	1.03E-02	respiratory
51121110000343003	0.00E+00	0.00E+00	9.81E-06	neurological
51121110000343012	0.00E+00	0.00E+00	7.47E-05	neurological
54009110000344716	**7.16E-06**	3.81E-04	6.97E-02	respiratory
54009110067212475	9.45E-07	1.15E-05	8.87E-03	respiratory
55133110000858652	2.44E-07	3.49E-05	2.32E-03	respiratory

1 BOLD indicates a cancer risk great than 1 in a million or a noncancer risk greater than 1

Table 3 – Maximum Predicted Acute Risks
Actual Emissions

Facility NEI ID	Pollutant	Maximum Hazard Quotient[1]				
		REL	AEGL1	AEGL2	ERPG1	ERPG2
01033110005979824	Ethyl benzene	0.00E+00	5.30E-06	1.55E-07	0.00E+00	0.00E+00
01033110005979824	Formaldehyde	2.52E-04	1.26E-05	8.15E-07	1.15E-05	1.15E-06
01033110005979824	Xylenes (mixed)	1.73E-04	6.81E-06	9.53E-07	0.00E+00	0.00E+00
01073110006548566	Ethyl benzene	0.00E+00	2.06E-06	6.00E-08	0.00E+00	0.00E+00
01073110006548566	Toluene	2.78E-06	4.11E-07	4.90E-08	5.41E-07	9.35E-08
01073110006548566	Xylenes (mixed)	6.08E-05	2.39E-06	3.34E-07	0.00E+00	0.00E+00
05093110028304957	Cumene	0.00E+00	1.04E-07	1.74E-08	0.00E+00	0.00E+00
05093110028304957	Ethyl benzene	0.00E+00	1.89E-06	5.53E-08	0.00E+00	0.00E+00
05093110028304957	Formaldehyde	2.09E-03	1.05E-04	6.77E-06	9.59E-05	9.59E-06
05093110028304957	Glycol Ethers	2.39E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
05093110028304957	Hydrofluoric acid	1.12E-03	3.27E-04	1.34E-05	1.67E-04	1.67E-05
05093110028304957	Methanol	1.00E-05	4.08E-07	1.04E-07	1.08E-06	2.16E-07
05093110028304957	Toluene	5.68E-05	8.40E-06	1.00E-06	1.11E-05	1.91E-06
05093110028304957	Xylenes (mixed)	6.18E-05	2.43E-06	3.40E-07	0.00E+00	0.00E+00
06071110002647743	Formaldehyde	5.06E-03	2.53E-04	1.64E-05	2.32E-04	2.32E-05
06071110002647743	Tetrachloroethene	3.23E-04	2.69E-05	4.03E-06	9.49E-06	4.61E-06
06071110002647743	Xylenes (mixed)	1.94E-04	7.62E-06	1.07E-06	0.00E+00	0.00E+00
12069110001969576	Toluene	3.50E-05	5.17E-06	6.16E-07	6.81E-06	1.18E-06
13045110035892371	Cumene	0.00E+00	1.27E-09	2.11E-10	0.00E+00	0.00E+00
13045110035892371	Ethyl benzene	0.00E+00	2.30E-08	6.71E-10	0.00E+00	0.00E+00
13045110035892371	Formaldehyde	2.54E-05	1.27E-06	8.22E-08	1.16E-06	1.16E-07
13045110035892371	Glycol Ethers	2.90E-04	0.00E+00	0.00E+00	0.00E+00	0.00E+00
13045110035892371	Methanol	1.22E-07	4.95E-09	1.27E-09	1.31E-08	2.63E-09
13045110035892371	Toluene	6.89E-07	1.02E-07	1.21E-08	1.34E-07	2.32E-08
13045110035892371	Xylenes (mixed)	7.51E-07	2.95E-08	4.13E-09	0.00E+00	0.00E+00
17031110000429000	Ethyl benzene	0.00E+00	2.98E-07	8.68E-09	0.00E+00	0.00E+00
17031110000429000	Toluene	3.75E-06	5.55E-07	6.61E-08	7.31E-07	1.26E-07
17031110000429000	Xylenes (mixed)	1.29E-05	5.08E-07	7.12E-08	0.00E+00	0.00E+00
17031110000433102	Cumene	0.00E+00	2.45E-07	4.09E-08	0.00E+00	0.00E+00
17031110000433102	Ethyl benzene	0.00E+00	4.45E-06	1.30E-07	0.00E+00	0.00E+00
17031110000433102	Formaldehyde	4.92E-03	2.46E-04	1.59E-05	2.25E-04	2.25E-05
17031110000433102	Glycol Ethers	5.61E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
17031110000433102	Methanol	2.36E-05	9.58E-07	2.45E-07	2.54E-06	5.09E-07
17031110000433102	Toluene	1.33E-04	1.97E-05	2.35E-06	2.60E-05	4.49E-06
17031110000433102	Xylenes (mixed)	1.45E-04	5.71E-06	7.99E-07	0.00E+00	0.00E+00
17031110067426137	Cumene	0.00E+00	5.52E-07	9.19E-08	0.00E+00	0.00E+00
17031110067426137	Dimethyl formamide	0.00E+00	0.00E+00	3.52E-08	1.59E-06	3.17E-08
17031110067426137	Ethyl benzene	0.00E+00	3.77E-06	1.10E-07	0.00E+00	0.00E+00
17031110067426137	Formaldehyde	1.56E-03	7.78E-05	5.04E-06	7.13E-05	7.13E-06
17031110067426137	Glycol Ethers	1.92E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
17031110067426137	Toluene	2.83E-06	4.18E-07	4.98E-08	5.51E-07	9.51E-08
17031110067426137	Xylenes (mixed)	7.13E-05	2.80E-06	3.92E-07	0.00E+00	0.00E+00
17119110003064567	Ethyl benzene	0.00E+00	1.45E-06	4.24E-08	0.00E+00	0.00E+00
17119110003064567	Glycol Ethers	3.04E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
17119110003064567	Toluene	6.52E-06	9.64E-07	1.15E-07	1.27E-06	2.19E-07
17119110003064567	Xylenes (mixed)	4.85E-05	1.90E-06	2.67E-07	0.00E+00	0.00E+00
18059110000396535	Xylenes (mixed)	2.81E-04	1.10E-05	1.55E-06	0.00E+00	0.00E+00
18091110000592136	Xylenes (mixed)	1.32E-04	5.17E-06	7.24E-07	0.00E+00	0.00E+00
18127110000747862	Xylenes (mixed)	3.36E-04	1.32E-05	1.85E-06	0.00E+00	0.00E+00
18141110000399453	Ethyl benzene	0.00E+00	1.01E-06	2.95E-08	0.00E+00	0.00E+00
18141110000399453	Xylenes (mixed)	2.44E-05	9.58E-07	1.34E-07	0.00E+00	0.00E+00
18145110021290413	Cumene	0.00E+00	1.45E-08	2.41E-09	0.00E+00	0.00E+00

Table 3 – Maximum Predicted Acute Risks
Actual Emissions

Facility NEI ID	Pollutant	Maximum Hazard Quotient[1]				
		REL	AEGL1	AEGL2	ERPG1	ERPG2
18145110021290413	Ethyl benzene	0.00E+00	2.63E-07	7.66E-09	0.00E+00	0.00E+00
18145110021290413	Formaldehyde	2.90E-04	1.45E-05	9.39E-07	1.33E-05	1.33E-06
18145110021290413	Glycol Ethers	3.31E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00
18145110021290413	Methanol	1.39E-06	5.65E-08	1.44E-08	1.50E-07	3.00E-08
18145110021290413	Toluene	7.87E-06	1.17E-06	1.39E-07	1.53E-06	2.65E-07
18145110021290413	Xylenes (mixed)	8.57E-06	3.37E-07	4.71E-08	0.00E+00	0.00E+00
18173110000602045	Cumene	0.00E+00	2.94E-06	4.89E-07	0.00E+00	0.00E+00
18173110000602045	Ethyl benzene	0.00E+00	1.86E-05	5.42E-07	0.00E+00	0.00E+00
18173110000602045	Formaldehyde	2.57E-06	1.28E-07	8.30E-09	1.18E-07	1.18E-08
18173110000602045	Glycol Ethers	9.33E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
18173110000602045	Toluene	2.17E-04	3.21E-05	3.82E-06	4.22E-05	7.29E-06
18173110000602045	Triethylamine	6.66E-05	0.00E+00	0.00E+00	0.00E+00	0.00E+00
18173110000602045	Xylenes (mixed)	6.97E-04	2.74E-05	3.84E-06	0.00E+00	0.00E+00
19163110000416005	Cumene	0.00E+00	1.02E-07	1.70E-08	0.00E+00	0.00E+00
19163110000416005	Ethyl benzene	0.00E+00	1.85E-06	5.41E-08	0.00E+00	0.00E+00
19163110000416005	Formaldehyde	2.05E-03	1.02E-04	6.62E-06	9.38E-05	9.38E-06
19163110000416005	Glycol Ethers	2.34E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
19163110000416005	Methanol	9.83E-06	3.99E-07	1.02E-07	1.06E-06	2.12E-07
19163110000416005	Toluene	5.56E-05	8.22E-06	9.79E-07	1.08E-05	1.87E-06
19163110000416005	Xylenes (mixed)	6.05E-05	2.38E-06	3.33E-07	0.00E+00	0.00E+00
19163110000416014	Ethyl benzene	0.00E+00	4.78E-05	1.39E-06	0.00E+00	0.00E+00
19163110000416014	Formaldehyde	3.61E-02	1.80E-03	1.17E-04	1.65E-03	1.65E-04
19163110000416014	Toluene	5.86E-05	8.67E-06	1.03E-06	1.14E-05	1.97E-06
19163110000416014	Xylenes (mixed)	6.90E-04	2.71E-05	3.79E-06	0.00E+00	0.00E+00
21091110000380935	Cumene	0.00E+00	3.41E-10	5.68E-11	0.00E+00	0.00E+00
21091110000380935	Ethyl benzene	0.00E+00	7.52E-09	2.19E-10	0.00E+00	0.00E+00
21091110000380935	Formaldehyde	9.20E-03	4.60E-04	2.98E-05	4.22E-04	4.22E-05
21091110000380935	Glycol Ethers	7.50E-04	0.00E+00	0.00E+00	0.00E+00	0.00E+00
21091110000380935	Toluene	1.96E-05	2.90E-06	3.46E-07	3.82E-06	6.60E-07
21091110000380935	Xylenes (mixed)	6.89E-07	2.71E-08	3.79E-09	0.00E+00	0.00E+00
21141110000605364	Cumene	0.00E+00	7.18E-07	1.20E-07	0.00E+00	0.00E+00
21141110000605364	Ethyl benzene	0.00E+00	1.30E-05	3.80E-07	0.00E+00	0.00E+00
21141110000605364	Formaldehyde	1.44E-02	7.20E-04	4.66E-05	6.60E-04	6.60E-05
21141110000605364	Glycol Ethers	1.64E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00
21141110000605364	Methanol	6.91E-05	2.80E-06	7.17E-07	7.44E-06	1.49E-06
21141110000605364	Toluene	3.91E-04	5.78E-05	6.88E-06	7.61E-05	1.31E-05
21141110000605364	Xylenes (mixed)	4.25E-04	1.67E-05	2.34E-06	0.00E+00	0.00E+00
27053110022448092	Cumene	0.00E+00	3.79E-09	6.31E-10	0.00E+00	0.00E+00
27053110022448092	Ethyl benzene	0.00E+00	1.77E-06	5.17E-08	0.00E+00	0.00E+00
27053110022448092	Toluene	4.12E-06	6.09E-07	7.25E-08	8.02E-07	1.38E-07
27053110022448092	Xylenes (mixed)	4.62E-05	1.81E-06	2.54E-07	0.00E+00	0.00E+00
28049110000376815	Butyl carbitol acetate	1.30E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00
28049110000376815	Diethylene glycol monobutyl ether	1.17E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
28049110000376815	Ethyl benzene	0.00E+00	4.67E-06	1.36E-07	0.00E+00	0.00E+00
28049110000376815	Formaldehyde	7.27E-04	3.64E-05	2.35E-06	3.33E-05	3.33E-06
28049110000376815	Methanol	4.76E-07	1.93E-08	4.94E-09	5.13E-08	1.03E-08
28049110000376815	Toluene	3.37E-06	4.99E-07	5.95E-08	6.57E-07	1.14E-07
28049110000376815	Xylenes (mixed)	1.69E-04	6.65E-06	9.32E-07	0.00E+00	0.00E+00
28049110000376879	Cumene	0.00E+00	4.40E-07	7.34E-08	0.00E+00	0.00E+00
28049110000376879	Ethyl benzene	0.00E+00	3.26E-05	9.50E-07	0.00E+00	0.00E+00
28049110000376879	Formaldehyde	2.44E-02	1.22E-03	7.89E-05	1.12E-03	1.12E-04

Table 3 – Maximum Predicted Acute Risks
Actual Emissions

Facility NEI ID	Pollutant	Maximum Hazard Quotient[1]				
		REL	AEGL1	AEGL2	ERPG1	ERPG2
28049110000376879	Glycol Ethers	8.89E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00
28049110000376879	Methanol	5.00E-06	2.03E-07	5.19E-08	5.39E-07	1.08E-07
28049110000376879	Toluene	1.43E-04	2.12E-05	2.52E-06	2.78E-05	4.81E-06
28049110000376879	Xylenes (mixed)	8.79E-04	3.45E-05	4.83E-06	0.00E+00	0.00E+00
29510110000441255	Ethyl benzene	0.00E+00	1.76E-04	5.12E-06	0.00E+00	0.00E+00
29510110000441255	Formaldehyde	2.17E-02	1.09E-03	7.02E-05	9.95E-04	9.95E-05
29510110000441255	Glycol Ethers	**1.80E+00**	0.00E+00	0.00E+00	0.00E+00	0.00E+00
29510110000441255	Toluene	1.00E-03	1.48E-04	1.76E-05	1.95E-04	3.37E-05
29510110000441255	Xylenes (mixed)	8.29E-03	3.26E-04	4.56E-05	0.00E+00	0.00E+00
34035110000858670	Cumene	0.00E+00	1.11E-05	1.85E-06	0.00E+00	0.00E+00
34035110000858670	Ethyl benzene	0.00E+00	2.02E-04	5.89E-06	0.00E+00	0.00E+00
34035110000858670	Formaldehyde	2.23E-01	1.11E-02	7.21E-04	1.02E-02	1.02E-03
34035110000858670	Glycol Ethers	**2.54E+00**	0.00E+00	0.00E+00	0.00E+00	0.00E+00
34035110000858670	Methanol	1.07E-03	4.34E-05	1.11E-05	1.15E-04	2.31E-05
34035110000858670	Toluene	6.05E-03	8.95E-04	1.07E-04	1.18E-03	2.04E-04
34035110000858670	Xylenes (mixed)	6.59E-03	2.59E-04	3.62E-05	0.00E+00	0.00E+00
36015110046535344	Cumene	0.00E+00	2.57E-08	4.28E-09	0.00E+00	0.00E+00
36015110046535344	Ethyl benzene	0.00E+00	4.66E-07	1.36E-08	0.00E+00	0.00E+00
36015110046535344	Formaldehyde	5.15E-04	2.58E-05	1.67E-06	2.36E-05	2.36E-06
36015110046535344	Glycol Ethers	5.88E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00
36015110046535344	Methanol	2.47E-06	1.00E-07	2.57E-08	2.66E-07	5.33E-08
36015110046535344	Toluene	1.40E-05	2.07E-06	2.46E-07	2.72E-06	4.70E-07
36015110046535344	Xylenes (mixed)	1.52E-05	5.98E-07	8.37E-08	0.00E+00	0.00E+00
39017110000392575	Cumene	0.00E+00	2.07E-08	3.44E-09	0.00E+00	0.00E+00
39017110000392575	Ethyl benzene	0.00E+00	3.75E-07	1.09E-08	0.00E+00	0.00E+00
39017110000392575	Formaldehyde	4.14E-04	2.07E-05	1.34E-06	1.90E-05	1.90E-06
39017110000392575	Glycol Ethers	4.72E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00
39017110000392575	Methanol	1.99E-06	8.07E-08	2.06E-08	2.14E-07	4.28E-08
39017110000392575	Toluene	1.12E-05	1.66E-06	1.98E-07	2.19E-06	3.78E-07
39017110000392575	Xylenes (mixed)	1.22E-05	4.80E-07	6.73E-08	0.00E+00	0.00E+00
39035110012520744	Ethyl benzene	0.00E+00	1.91E-06	5.58E-08	0.00E+00	0.00E+00
39035110012520744	Toluene	1.49E-05	2.21E-06	2.63E-07	2.90E-06	5.01E-07
39035110012520744	Xylenes (mixed)	4.64E-05	1.82E-06	2.55E-07	0.00E+00	0.00E+00
39059110000384548	Methanol	8.94E-04	3.63E-05	9.28E-06	9.63E-05	1.93E-05
39059110000384548	Xylenes (mixed)	3.23E-03	1.27E-04	1.78E-05	0.00E+00	0.00E+00
39095110000384218	Ethyl benzene	0.00E+00	5.38E-05	1.57E-06	0.00E+00	0.00E+00
39095110000384218	Xylenes (mixed)	1.63E-03	6.40E-05	8.96E-06	0.00E+00	0.00E+00
39119110000384487	Cumene	0.00E+00	6.33E-08	1.05E-08	0.00E+00	0.00E+00
39119110000384487	Ethyl benzene	0.00E+00	1.15E-06	3.35E-08	0.00E+00	0.00E+00
39119110000384487	Formaldehyde	1.27E-03	6.34E-05	4.11E-06	5.82E-05	5.82E-06
39119110000384487	Glycol Ethers	1.45E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
39119110000384487	Methanol	6.09E-06	2.47E-07	6.32E-08	6.56E-07	1.31E-07
39119110000384487	Toluene	3.44E-05	5.10E-06	6.07E-07	6.70E-06	1.16E-06
39119110000384487	Xylenes (mixed)	3.75E-05	1.47E-06	2.06E-07	0.00E+00	0.00E+00
39129110000381836	Ethyl benzene	0.00E+00	6.81E-06	1.99E-07	0.00E+00	0.00E+00
39129110000381836	Xylenes (mixed)	9.52E-05	3.74E-06	5.23E-07	0.00E+00	0.00E+00
39149110000394314	Diethylene glycol monobutyl ether	**3.93E+00**	0.00E+00	0.00E+00	0.00E+00	0.00E+00
39149110000394314	Toluene	3.78E-03	5.60E-04	6.66E-05	7.37E-04	1.27E-04
39149110000394314	Triethylamine	1.12E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00
39155110000389963	Cumene	0.00E+00	2.24E-06	3.74E-07	0.00E+00	0.00E+00
39155110000389963	Ethyl benzene	0.00E+00	4.07E-05	1.19E-06	0.00E+00	0.00E+00

Table 3 – Maximum Predicted Acute Risks
Actual Emissions

Facility NEI ID	Pollutant	Maximum Hazard Quotient[1]				
		REL	AEGL1	AEGL2	ERPG1	ERPG2
39155110000389963	Formaldehyde	4.50E-02	2.25E-03	1.46E-04	2.06E-03	2.06E-04
39155110000389963	Glycol Ethers	5.13E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00
39155110000389963	Methanol	2.16E-04	8.76E-06	2.24E-06	2.33E-05	4.65E-06
39155110000389963	Toluene	1.22E-03	1.81E-04	2.15E-05	2.38E-04	4.10E-05
39155110000389963	Xylenes (mixed)	1.33E-03	5.22E-05	7.31E-06	0.00E+00	0.00E+00
39173110000383353	Xylenes (mixed)	2.53E-05	9.94E-07	1.39E-07	0.00E+00	0.00E+00
42003110008470724	Ethyl benzene	0.00E+00	8.60E-06	2.51E-07	0.00E+00	0.00E+00
42003110008470724	Glycol Ethers	8.97E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
42003110008470724	Toluene	1.38E-04	2.05E-05	2.44E-06	2.70E-05	4.66E-06
42003110008470724	Xylenes (mixed)	5.92E-05	2.32E-06	3.25E-07	0.00E+00	0.00E+00
42017110000336495	Toluene	8.31E-04	1.23E-04	1.46E-05	1.62E-04	2.79E-05
42071110000334200	Ethyl benzene	0.00E+00	8.94E-08	2.61E-09	0.00E+00	0.00E+00
42071110000334200	Formaldehyde	2.07E-05	1.03E-06	6.69E-08	9.48E-07	9.48E-08
42071110000334200	Toluene	1.54E-07	2.27E-08	2.71E-09	2.99E-08	5.17E-09
42071110000334200	Xylenes (mixed)	2.43E-06	9.54E-08	1.34E-08	0.00E+00	0.00E+00
42081110000584163	Glycol Ethers	2.45E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00
42081110000584163	Toluene	1.54E-03	2.27E-04	2.71E-05	2.99E-04	5.17E-05
42081110000584163	Xylenes (mixed)	1.20E-04	4.70E-06	6.58E-07	0.00E+00	0.00E+00
45047110016994118	Methanol	2.87E-02	1.17E-03	2.98E-04	3.09E-03	6.19E-04
45079110000352494	Cumene	0.00E+00	4.87E-07	8.12E-08	0.00E+00	0.00E+00
45079110000352494	Ethyl benzene	0.00E+00	2.47E-07	7.20E-09	0.00E+00	0.00E+00
45079110000352494	Formaldehyde	1.26E-04	6.30E-06	4.08E-07	5.78E-06	5.78E-07
45079110000352494	Glycol Ethers	1.69E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00
45079110000352494	Methanol	1.57E-07	6.36E-09	1.62E-09	1.69E-08	3.37E-09
45079110000352494	Toluene	6.82E-07	1.01E-07	1.20E-08	1.33E-07	2.29E-08
45079110000352494	Xylenes (mixed)	1.78E-05	7.00E-07	9.80E-08	0.00E+00	0.00E+00
48201110000500583	Ethyl benzene	0.00E+00	1.29E-05	3.75E-07	0.00E+00	0.00E+00
48201110000500583	Formaldehyde	4.49E-03	2.24E-04	1.45E-05	2.06E-04	2.06E-05
48201110000500583	Glycol Ethers	2.84E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00
48201110000500583	Toluene	3.92E-05	5.79E-06	6.90E-07	7.62E-06	1.32E-06
48201110000500583	Xylenes (mixed)	4.15E-04	1.63E-05	2.28E-06	0.00E+00	0.00E+00
51121110000343003	Toluene	1.09E-04	1.62E-05	1.92E-06	2.13E-05	3.67E-06
51121110000343012	Toluene	1.16E-03	1.71E-04	2.04E-05	2.25E-04	3.89E-05
54009110000344716	Cumene	0.00E+00	2.03E-06	3.38E-07	0.00E+00	0.00E+00
54009110000344716	Ethyl benzene	0.00E+00	1.59E-05	4.64E-07	0.00E+00	0.00E+00
54009110000344716	Formaldehyde	3.60E-03	1.80E-04	1.16E-05	1.65E-04	1.65E-05
54009110000344716	Hydrofluoric acid	3.20E-02	9.35E-03	3.83E-04	4.79E-03	4.79E-04
54009110000344716	Toluene	1.89E-05	2.80E-06	3.34E-07	3.69E-06	6.37E-07
54009110000344716	Xylenes (mixed)	6.42E-04	2.52E-05	3.53E-06	0.00E+00	0.00E+00
54009110067212475	Cumene	0.00E+00	3.66E-08	6.09E-09	0.00E+00	0.00E+00
54009110067212475	Ethyl benzene	0.00E+00	1.71E-06	4.98E-08	0.00E+00	0.00E+00
54009110067212475	Formaldehyde	1.10E-03	5.52E-05	3.57E-06	5.06E-05	5.06E-06
54009110067212475	Xylenes (mixed)	1.55E-05	6.09E-07	8.53E-08	0.00E+00	0.00E+00
55133110000858652	Cumene	0.00E+00	3.99E-08	6.65E-09	0.00E+00	0.00E+00
55133110000858652	Ethyl benzene	0.00E+00	7.24E-07	2.11E-08	0.00E+00	0.00E+00
55133110000858652	Formaldehyde	8.00E-04	4.00E-05	2.59E-06	3.67E-05	3.67E-06
55133110000858652	Glycol Ethers	9.13E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00
55133110000858652	Methanol	3.84E-06	1.56E-07	3.98E-08	4.14E-07	8.28E-08
55133110000858652	Toluene	2.17E-05	3.21E-06	3.83E-07	4.23E-06	7.30E-07
55133110000858652	Xylenes (mixed)	2.36E-05	9.29E-07	1.30E-07	0.00E+00	0.00E+00

Table 3 – Maximum Predicted Acute Risks
Actual Emissions

Facility NEI ID	Pollutant	Maximum Hazard Quotient[1]				
		REL	AEGL1	AEGL2	ERPG1	ERPG2

[1] Note: **BOLD** indicates acute HQ greater than 1

Refined Acute Modeling Approach

Initial acute screening risk calculations were performed with the HEM-3 model. HEM-3 estimates acute (1-hour) impacts at both polar and census block receptors. It is assumed for this short period of time that an exposed individual could be located at any off-site location. The lack of readily available detailed property boundary information for many of the facilities evaluated made it difficult to determine whether receptors were on- or off-site. In the absence of such information, the first ring of polar receptors was placed 100 meters from the plant center for many facilities. However, these polar rings often transected on-site locations, restricting public access to exposures at these levels and thereby overestimating exposures.

The screening approach used by HEM-3 to estimate maximum 1-hour exposures also likely overestimates. To estimate maximum 1-hour concentrations at each receptor, HEM-3 sums the maximum concentrations attributed to each source, regardless of whether those maximum concentrations occurred during the same hour. In other words, HEM-3 assumes that the maximum impact from each source at each receptor occurs at the same time. In actuality, maximum impacts from different sources may occur at different times.

This appendix includes the results of refined acute assessments performed for facilities that exceeded short-term health benchmarks using the HEM-3 approach. The refinements can address both areas of conservatism described above. The simplest refinement is to plot the HEM-3 polar grid results on aerial photographs of the facilities; This allows the assessment of off-site locations that may be accessible to the public (*e.g.,* roadways and public buildings.). The attached figure present the estimated hazard quotient values (the modeled 1-hour concentration of a pollutant divided by its short-term health benchmark) at the polar receptors near the facilities.

Table 4 – Maximum Predicted Acute Risks Greater than 1 (Refined Approach)

Facility NEI ID	Pollutant	Criteria	Baseline - Actuals		Refined Modeling Approach [1]
			HEM-3 (Screening)	Refined Results	
39149110000394314	Diethylene glycol monobutyl ether	REL	4	3	Max revised to off-site plocation (see figure)
34035110000858670	Glycol Ethers	REL	3	<1	Max revised to off-site plocation (see figure)
29510110000441255	Glycol Ethers	REL	2	2	Max at residential location; no redinment needed

[1] Indicates modeling technique used to refined estimates; see figures depicting off-site impacts

Figure 1 – 39149110000394314
Acute Diethylene glycol monobutyl ether HQ (REL)



Figure 2 – 34035110000858670
Acute Glycol Ethers HQ (REL)



Appendix 11 Site-Specific Human Health Multipathway Residual Risk Assessment Report

A site-specific multipathway analysis was not performed for this source category.