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Screening Methodologies to Support Risk and Technology Reviews (RTR): A Case Study Analysis

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ACRONYMS/ABBREVIATIONS

ADD	average daily dose
AERMOD	American Meteorological Society/EPA Regulatory Model
As	arsenic
ASOS	Automated Surface Observing Systems
AWQB	ambient water quality benchmarks
AWQC	ambient water quality criteria
BC	benthic carnivores (fish)
BaP	benzo[a]pyrene
CAA	Clean Air Act
Cd	cadmium
CSF	cancer slope factor
D/F	dioxins/furans
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DEHP	bis-(2-ethylhexyl)phthalate
DEM	digital elevation model
DOE	Department of Energy
EcoEEFs	ecological exposure equivalency factors
Eco-SSLs	ecological soil screening levels (Superfund)
EcoTEF	ecological toxic equivalency factors
EEFs	exposure equivalency factors
EFH	Exposure Factors Handbook
EPA	Environmental Protection Agency
EqP	equilibrium partitioning
ESRI	Environmental Systems Research Institute
GEAE	generic ecological assessment endpoints
GIS	Geographic Information Systems
GLWQI	Great Lakes Water Quality Initiative
HAP	hazardous air pollutant
HCl	hydrogen chloride
HEM	Human Exposure Model
HF	hydrogen fluoride
Hg	mercury
HHRAP	Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities
HQ	hazard quotient
HxCDD	hexachlorodibenzo-p-dioxin
IR	ingestion rate
IRIS	Integrated Risk Information System
K_{ow}	octanol-water partition coefficient
LADD	lifetime average daily dose
LC	land cover
LOAEL	lowest observed adverse effect level
LOEL	lowest observed effect level
MACT	maximum achievable control technology
MIR	maximum individual risk
MIRC	Multimedia Ingestion Risk Calculator
MVP	minimum viable population

NAAQS	national ambient air quality standards
NAWQC-ALC	national ambient water quality criteria–aquatic life criteria
NEI	National Emissions Inventory
NEL	no effect level
NESHAP	national emission standards for hazard air pollutants
NOAEL	no observed adverse effect level
OAQPS	Office of Air Quality Planning and Standards (U.S. EPA)
ORD	Office of Research and Development (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
OW	Office of Water (U.S. EPA)
PAH	polycyclic aromatic hydrocarbon
Pb	lead
PB	persistent and bioaccumulative
PB-HAP	persistent and bioaccumulative HAP
PCB	polychlorinated biphenyl
PEL	probable effect level
POM	polycyclic organic matter
RAIS	Risk Assessment Information System (ORNL)
RCRA	Resource Conservation and Recovery Act
REFs	risk equivalency factors
RfD	reference dose
RTR	Risk and Technology Review
SAB	Science Advisory Board
SCV	secondary chronic value
SEB	soil ecotoxicity benchmark
SQB	sediment quality benchmark
SQC	sediment quality criteria
SV	screening value
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin, termed “dioxin” in this report
TCE	trichloroethylene
TEFs	toxic equivalency factors
TEL	threshold effect level
TL	trophic level
TL2	trophic level 2
TL3	trophic level 3
TL3.5	between tropic 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
TRV	toxicity reference value
UF	uncertainty factor
USGS	U.S. Geological Survey
WCC	water-column carnivores (fish)
WW	wet weight

1. INTRODUCTION

This report was prepared to facilitate the U.S. Environmental Protection Agency (EPA, or the Agency) Science Advisory Board (SAB) peer review of recently enhanced methods used for Risk and Technology Review (RTR) risk assessments. The enhancements are the latest in a series of improvements and methodological advances that EPA developed to evaluate the risks that hazardous air pollutants (HAPs) emitted from numerous categories of industrial sources pose to human health and the environment. The SAB has advised the Agency on several occasions regarding RTR risk assessment methods. We consider the enhancements described in this report to be incremental improvements to our risk assessment methods, consistent with SAB recommendations made during RTR methods reviews (see Table 1-1).

This report describes enhanced screening methods used to estimate potential risks from industrial sources of HAPs. EPA uses these screens to quickly identify those facilities in particular stationary source categories that have little potential for human health or environmental risk, while also identifying those facilities where a refined risk assessment might be needed. We consider these screens an important step in the RTR risk assessment process because refined risk assessments: (1) are completed only for one facility at a time; (2) are very costly for each facility analyzed; and (3) can take several months to complete.

In this report, EPA presents information on enhancements to two specific risk-assessment screens: (1) the screen for human health multipathway risk, which looks at human health exposures to HAPs through ingestion (via air, soil, water pathways); and (2) the screen for environmental risk, which evaluates ecological exposures to HAPs through air, soil, and water pathways. This report describes the potential addition of a new multipathway exposure scenario to estimate ingestion risk for members of urban or rural households who consume contaminated homegrown fruits and vegetables. Also described are several relatively recent improvements to EPA's chronic inhalation risk assessment methodology. We highlight application of these updated methodologies by presenting screens of hypothetical facilities that emit HAPs.[1]

[1]Some chapters use graphical depictions of actual locations. Emissions from example facilities have been either increased or decreased to illustrate the technical approach more clearly and how EPA risk managers interpret the results. Thus, the estimates of risk presented here might not reflect actual risks in the locations chosen as examples.

In an accompanying memorandum, charge questions are provided to the SAB panel. These questions are intended to elicit comment on whether the enhancements to our multipathway and environmental risk screening methodologies and chronic inhalation assessment enhancements are reasonable, and if not, how they could be improved.

This report is organized into six chapters. Chapter 1 discusses the nature of RTR risk management decisions, including a discussion of the statutory authority and EPA's risk management framework for the RTR program. It also states the goals for this SAB review. Chapter 2 presents an overview and the methodology of the scenarios analyzed for tiered multipathway screens. Chapters 3 and 4 describe in detail EPA's multipathway and environmental screening methodologies, respectively. These chapters include examples of applications of each screening method, for illustrative purposes, and discussion of the appropriate use of the results (e.g., do screening results suggest that further, more-refined analysis is needed?). Chapter 5 presents several enhancements to the RTR chronic inhalation risk assessment, including a new procedure that determines the urban/rural designation of the census block closest to each evaluated facility, as well as a census block receptor tool to locate (and modify where appropriate) census block centroids that are on facility properties and also identify blocks where the centroid location might not adequately represent the locations of residences in the block. Complete reference citations are provided in Chapter 6, and additional information that could be helpful to the SAB in their review of these methodologies is presented in two appendices.

1.1 Nature of RTR Risk Management Decisions

Section 112 of the Clean Air Act (CAA) establishes a two-stage regulatory process for addressing emissions of HAPs from stationary sources. In the first stage, section 112(d) requires EPA to develop technology-based standards for categories of sources (e.g., petroleum refineries, pulp and paper mills). EPA has issued national emission standards for HAPs (NESHAPs) for more than 170 source categories and has largely completed the initial maximum achievable control technology (MACT) standards. Under section 112(d)(6), EPA must review each technology-based standard at least every 8 years and revise the standard, as necessary, "taking into account developments in practices, processes and control technologies." In the second stage of the regulatory process, the residual risk 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 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, EPA must complete the residual risk review within 8 years of promulgation of the initial technology-based standards.[2]

Residual risks are assessed separately for each source category. The CAA requires EPA to promulgate additional standards for a source category "if promulgation of such standards is required to provide an ample margin of safety to protect public health" or "to prevent, taking into consideration costs, energy, safety, and other relevant factors, an adverse environmental effect." A key factor in this risk management decision is the determination of the "lifetime excess cancer risk to the individual most exposed to emissions from a source in the category," or the maximum individual risk (MIR).

In 1989, EPA established the risk management decision framework for residual risk rulemakings in the Benzene National Emission Standards for Hazardous Air Pollutants, or Benzene NESHAP (see 54 *Federal Register* 38044). This framework implements the determination of an "ample margin of safety" in two steps. In the first step, EPA determines "acceptable risk." EPA generally presumes that if the MIR is no higher than approximately 100-in-one million, the risk is acceptable. In determining acceptability, EPA considers other risk metrics (e.g., the total estimated cancer incidence due to emissions from the source category) and other health factors, including chronic or acute noncancer human health risks, multipathway risks, and uncertainties in the risk estimates. EPA may not consider costs of regulatory actions in this step. In the second step, EPA determines the "ample margin of safety." Here, EPA factors in the costs and feasibility of controlling emissions from the source category as it evaluates further risk reductions across the source category with the goal of minimizing the number of persons whose lifetime cancer risks due to emissions from the source category is greater than 1-in-one million. EPA also assesses whether remaining emissions might cause adverse environmental effects, and if so whether feasible and cost-effective control technologies are available.

[2]Note that by combining the 112(d)(6) and 112(f)(2) rulemaking into a single rulemaking package, EPA created the term "Risk and Technology Review" or RTR.

1.2 Scope of this Review

In this review, we are asking the SAB to evaluate whether the enhanced screening methods are scientifically reasonable for identifying facilities that have little potential for human or environmental risk, while also identifying those facilities where additional analysis is needed.[3] The tiered screening approach is designed to narrow the scope of analysis to potentially high-risk facilities through a series of progressively more detailed steps. At each tier, the approach "screens out" low-risk facilities for which no additional analysis is needed, so that only facilities with potentially higher risk remain in the pool for further analysis. The Tier 1 screen has less detail and more health-protective assumptions than Tier 2, and the same is true of Tier 2 relative to Tier 3. This approach ensures that the screening process does not generate false negatives, that is, rule out a facility that might actually be a source of appreciable multipathway or environmental risk. Each tier of these screens is more labor intensive than the previous, so having a stepwise approach enables us to identify efficiently which facilities might actually pose higher risks. Even if just a few facilities in a source category screen out at the most health-protective tier (i.e., Tier 1), not having to do subsequent, more labor intensive tiers of the multipathway or environmental risk screen (i.e., Tier 2 or Tier 3) saves time and resources. Even if a facility does not screen out until the final screening tier (i.e., Tier 3), the tiered screening approach is still much less costly and time consuming than performing a refined risk assessment.

We are specifically requesting SAB review on the following methods described in this document:

- Use of risk equivalency factors in the Tier 1 multipathway screen to account for differences in the environmental fate and transport among polycyclic organic matter (POM) chemicals and dioxin congeners (Section 3.1.2).
- The Tier 2 multipathway screen that replaces some of the health-protective assumptions in the Tier 1 screen with more site-specific information, such as, site-specific annual average values for meteorological parameters and actual locations of fishable lakes near the facility (Section 3.2).

[3]New and updated risk screening methodologies described in this document evolved from methods the SAB previously reviewed (see Table 1-1).

- The Tier 3 multipathway screen that further refines the Tier 2 assessment for a facility with review of aerial photography of surrounding lakes, a plume-rise assessment, and use of an hourly time-series of meteorological data and effective release heights (Section 3.3).
- The proposed gardener scenario designed to characterize ingestion risks for members of urban and rural households who consume contaminated homegrown fruits and vegetables (Section 3.4).
- The environmental risk screening approach (Section 4).
- The urban/rural enhancement to the inhalation risk assessment that allows us to better characterize dispersion of chemicals in air near facilities (Section 5.1).
- The census block receptor enhancement that allows us to model air concentrations more accurately where populations actually reside (Section 5.2).

Among the methods presented in this document, the following have undergone previous SAB review, and therefore, we are not requesting the SAB to review them (see Table 1-1):

- The structure of the Tier 1 multipathway screen (reviewed by SAB in 2009).
- Use of the Total Risk Integrated Methodology (TRIM), specifically its Fate, Transport, and Ecological Exposure module (TRIM.FaTE), to estimate media concentrations for the multipathway screen and the environmental risk screen (reviewed by SAB in 2009).
- The overall approach for conducting chronic inhalation risk assessments for RTR (reviewed by SAB in 2009).
- Use of the American Meteorology Society/EPA Regulatory Model (AERMOD) air dispersion model to estimate air concentrations of acid gases for the environmental risk screen (reviewed by SAB in 2009).

To guide the SAB in this effort, we have included a series of charge questions that indicate which specific science issues EPA would like the SAB Panel to consider. These charge questions are organized by chapter and can be found in the memorandum accompanying this technical document.

Table 1-1. Previous Reviews or Consultations on RTR Methodologies[a]

Year	Review Focus
1998	SAB reviewed the Draft Residual Risk Report to Congress, which described the analytical and policy approach for assessing residual risk from hazardous air pollutants emitted from stationary sources.
2000	SAB reviewed whether EPA's overall approach to assessing residual risk in the Secondary Lead Source Category was consistent with the methods described in the Residual Risk Report to Congress.
2006	EPA consulted with the SAB on development of emissions inventories for source categories and updated methods for characterizing human exposure and risks.
2009/ 2010	SAB reviewed updated and expanded air toxics risk assessment methodologies—including updated techniques for multipathway assessments using the Total Risk Integration Model (TRIM), refined screening methods for acute risk, and methods for assessing potential environmental risk.

[a]This list is not exhaustive for all components of RTR risk assessments. For instance, the health benchmark values used in RTR assessments have been the subject of peer reviews through the agencies that developed them (including EPA, through its Integrated Risk Information System, or IRIS; the California Environmental Protection Agency; and the Agency for Toxic Substances and Disease Registry).

2. SCENARIOS FOR TIERED MULTIPATHWAY SCREENS

2.1 Introduction

In RTR assessments, EPA considers risk from both inhalation and ingestion pathways. Non-inhalation risks are assessed using a "multipathway" approach, which includes ingestion exposures through contaminated food products such as vegetables, fruit, meat, and fish. Multipathway risk assessments can be extremely complex, due to the wide array of ecological and exposure factors that must be considered. A full-scale multipathway assessment—when warranted—must be conducted on a facility-by-facility basis, which is both time consuming and expensive. To ensure such analyses target facilities that actually pose higher potential risk, based on emissions and surrounding site characteristics, EPA has developed screening tools that can help reduce the number of facilities needing detailed analysis. These screening tools model the dispersion, transport, and fate of HAPs and human uptake of HAPs from food products and other media contaminated by emissions from a facility. Using specified exposure assumptions, the tools estimate risks that could result from these emissions.

In 2009, the SAB reviewed a multipathway screening scenario based on a combined subsistence fisher and farmer that had the goal of quickly and cost effectively identifying facilities that might need a complete multipathway risk assessment using the TRIM model (which can take up to 3 months to complete and cost more than 20 thousand dollars per site). In practice, we found that the assumptions in this original screening scenario were so health protective[4] that the potential risk for individual facilities was greatly overestimated, indicating the approach was not effectively screening out low-risk facilities.[5] Since the most recent SAB review of RTR methods, we refined our original multipathway screen. The refinements we made include a three-tiered multipathway screening approach that progressively replaces health-protective default assumptions with location-specific data for some inputs that had the most influence on potential

[4]We use "health-protective" rather than "conservative" in this document. These terms refer to characteristics of the approach that will often lead to overestimates in health risks or effects, rather than underestimates.

[5]Health-protective model inputs included those for wind speed, precipitation rate, and the amount of time the wind was blowing in the direction of the receptor of interest (i.e., the lake, farm, or garden). The original screening scenario combined these health-protective meteorological inputs with the assumption that the receptor of interest was near the source. It also did not take into account that different congeners of POM and dioxins can move through the environment at different rates. That is, all POM congeners initially were assumed to move, partition, and degrade in the environment as benzo[a]pyrene (BaP) does, and all dioxins were assumed to exhibit the same fate and transport as 2,3,7,8-TCDD (see Section 3.1.2).

risk, such as meteorological and locational data for water bodies. The result is a tiered screening approach that provides a more realistic, but still health-protective, estimate of potential multipathway risk. This approach, including technical detail on each tier of the multipathway screen, is laid out in Chapter 3. In addition, EPA is considering expanding its multipathway screening capabilities by adding a gardening exposure scenario that could better characterize multipathway risk in some instances, especially in locations where the presence of a subsistence farm is either unlikely (e.g., in urban areas) or difficult to confirm based on the characterization of land use surrounding a facility. This proposed gardener scenario is described in detail in Chapter 3, section 3.4.

Section 2.2 of this document provides an overview of the tiered multipathway screening methodology, including a brief description of each multipathway screening tier.

2.2 Overview of Tiered Multipathway Screen

In general, EPA uses the following approach and tools for the three tiers of the multipathway screen:

1. We use TRIM.FaTE[6] to model fate and transport of air emissions of persistent and bioaccumulative HAPs (PB-HAPs). This modeling includes chemical partitioning into soil, water, and other environmental media (including fish). Outputs include chemical concentrations in fish (mg/kg wet weight), soil (µg/g dry weight), and water (mg/L).

2. We use TRIM.FaTE outputs (e.g., chemical air deposition or environmental media concentrations) as inputs to the Multimedia Ingestion Risk Calculator (MIRC) to model the transfer and uptake of PB-HAPs into farm-raised food products (e.g., produce, livestock, and dairy products) from soil and air. MIRC outputs include chemical concentrations in the farm food products (and in fish for arsenic) in mg/kg wet weight.

3. MIRC uses these calculated chemical concentrations, along with food ingestion rates and other human exposure factors, to estimate ingestion exposures from the selected media for hypothetical human receptors. Specifically, MIRC calculates average daily doses (ADDs,

[6]The SAB reviewed the TRIM.FaTE and MIRC models, which are not part of the current RTR consultation; refer to appendices for more information on the set-up of the model scenario design used for the multipathway screens.

used for cadmium and mercury) and lifetime ADDs (LADDs, used for arsenic, dioxins/furans [abbreviated in this document as D/F or dioxins], and POM).

4. MIRC also uses the modeled chemical-specific ingestion exposures to calculate screening-level estimates of lifetime cancer risk or chronic noncancer hazard (expressed as a hazard quotient [HQ]) (as appropriate) for each PB-HAP, at the modeled emission rate of 1 g/day.

5. For each PB-HAP, based on the estimated cancer risk or HQ at the 1 g/day emission rate, we determine the emission rate at which the excess lifetime cancer risk of 1-in-one million, or the chronic noncancer HQ of 1, is reached. *These are 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 is compared with the screening threshold emission rate for cadmium). *The resulting ratio of a facility's actual emission rate to a screening threshold emission rate is termed a "screening value" (SV).*

These six steps are repeated for each tier of the multipathway screen. The primary difference between the Tier 1 and Tier 2 screens is the amount of site-specific data input to TRIM.FaTE in step 1 of this process. The Tier 3 screen incorporates additional site-specific data to refine estimates of plume rise and height of the air mixing layer, and thus loss of PB-HAPs to the upper atmosphere. For some facilities, emissions are low enough that risk is minimal, regardless of prevailing meteorological conditions or the proximity of farms or lakes to the source. For other facilities, the site-specific meteorological conditions and location of the nearest lake or farm will strongly affect risk estimates.

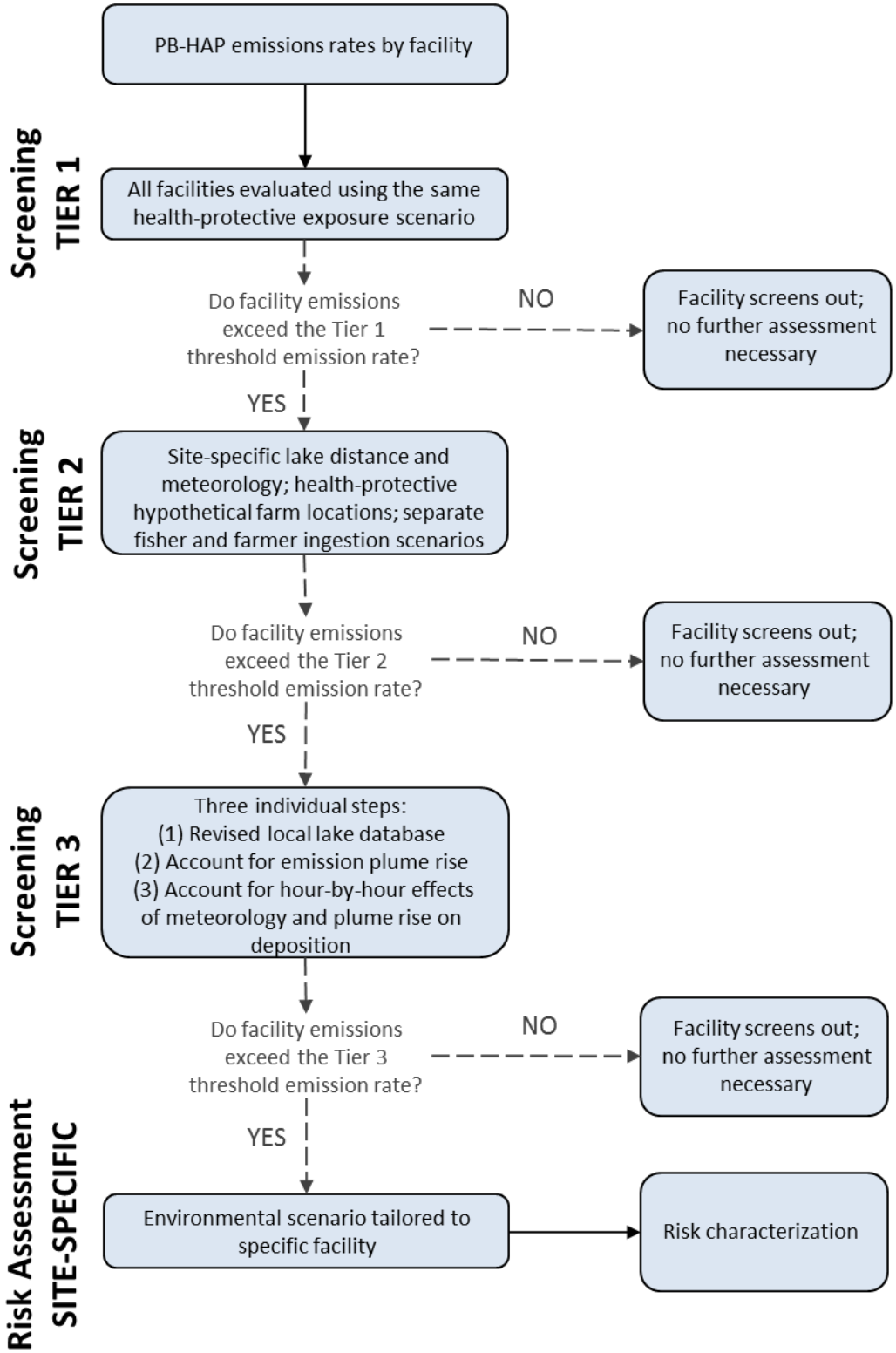
The Tier 1 screening scenario is the least labor intensive but also the most health protective of the screening tiers. Tier 1 is based on a hypothetical facility for which the surrounding environment was designed to encompass a health-protective environmental layout that would maximize PB-HAP concentrations in fish and in terrestrial environments in the immediate vicinity of a facility. It assumes a fishable lake is close to the facility in question and incorporates health-protective meteorological conditions. Moreover, the Tier 1 screen combines, for a single individual, the highest fishing and farming risks. If a facility's Tier 1 SV for a PB-HAP is less than or equal to 1, we are confident that potential multipathway risks are low and no additional

multipathway screening is needed for the facility's emissions of that PB-HAP. If the Tier 1 SV is greater than 1, however, the facility's emissions of that PB-HAP can be evaluated using the Tier 2 multipathway screen (see Figure 2-1).

The Tier 2 multipathway screen is more labor intensive because it replaces the health-protective lake distance and many of the health-protective meteorological assumptions used in the Tier 1 screen with more site-specific data. The Tier 2 screen also evaluates PB-HAP exposure and risk from the fishing scenario separately from the farming scenario (i.e., risks from the fishing and farming scenario are not combined for a single individual). If a facility's Tier 2 SVs imply a cancer risk less than or equal to 1-in-one million or an HQ less than or equal to 1 for a PB-HAP, we are confident that potential multipathway risks are low and no additional multipathway screening is needed for the facility's emissions of that PB-HAP. For facilities for which the Tier 2 SV(s) indicate a potential health risk to the public, we can conduct a Tier 3 multipathway screen (Figure 2-1).

The Tier 3 multipathway screen has three progressive steps that include additional evaluations of (1) lake data, (2) plume rise, and (3) time-series meteorological and plume-rise data. If the resulting Tier 3 SVs for a facility indicate a potential health risk, a final step could be a complete, site-specific multipathway assessment using TRIM.FaTE and MIRC (Figure 2-1).

Figure 2-1. Decision Tree for Evaluating Non-inhalation Exposures for PB-HAPs



3. DESCRIPTION OF THREE TIERS OF MULTIPATHWAY SCREENS

This chapter describes in detail each of the three tiers of the RTR multipathway human health risk screen for PB-HAPs. Section 3.1 presents the Tier 1 multipathway screen and includes an example application of the screen. The Tier 2 and Tier 3 multipathway screens are described, along with example applications, in Sections 3.2 and 3.3, respectively. The proposed gardener scenario is presented in Section 3.4, including discussion of its potential applicability in both urban and rural settings.

3.1 Tier 1 Multipathway Screen

The Tier 1 multipathway screen is described in detail in this section. Section 3.1.1 presents an overview of the Tier 1 exposure pathway and scenario development, while Section 3.1.2 introduces and discusses the concept of risk-equivalency factors (REFs) for dioxins and POM. Section 3.1.3 provides an example of how the Tier 1 multipathway screen is used. The Tier 1 multipathway screen is the tier most similar to the multipathway screening scenario previously reviewed by the SAB in 2009 (see Table 1-1). The substantive difference between the previously reviewed multipathway screen and the current Tier 1 multipathway screen is use of REFs, as described in Section 3.1.2.

3.1.1 Overview of Exposure Pathway and Scenario Development

The Tier 1 multipathway screen begins by identifying the facility-level emission rates of PB-HAPs within a source category and comparing them to risk-based screening threshold emission rates for those same PB-HAPs. The risk-based screening threshold emission rates are derived using TRIM.FaTE and MIRC (which, as previously noted, have been reviewed by SAB) applied to a hypothetical environmental and exposure scenario. The assumptions for this application are: fish are caught at nearby lakes, locally raised produce and animal products are ingested, and contaminated soil is ingested incidentally (with ingestion of breast milk included when evaluating risks from dioxins). Table 3-1 summarizes these ingestion pathways for Tier 1 (and Tier 2). The TRIM.FaTE and MIRC model runs supporting the Tier 1 multipathway screens do not simulate specific facilities; instead, they use a combination of key model inputs (e.g., distance of lake and farm from emission source; wind speed and direction; rainfall; mixing height) designed to represent a health-protective exposure and to estimate upper-bound risks.

This hypothetical screening scenario represents a high-end, combined ingestion exposure of fish and farm products unlikely to be exceeded at any actual facility evaluated through the RTR program.

The modeling parcels and runoff/erosion patterns for Tier 1 are shown in Figure 3-1. The areas above and below each two-dimensional parcel contain one or more modeling compartments where chemical mass is calculated in soil and various other abiotic and biotic media.[7] The locations of ingestion exposure are a lake and a farm that are within 0.5 km of the emission source and which receive most (in the case of the farm) or all (in the case of the lake) of the available chemical runoff and erosion from the watershed.

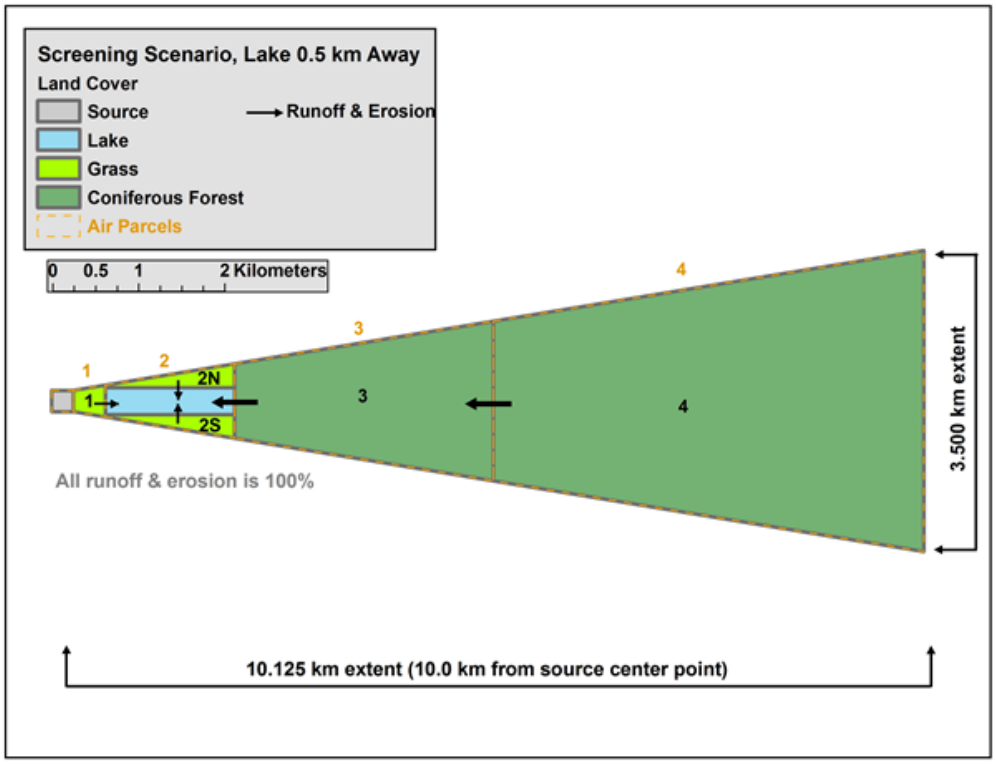
Table 3-1. Summary of Ingestion Pathways for Tier 1 and Tier 2 Screening Scenario

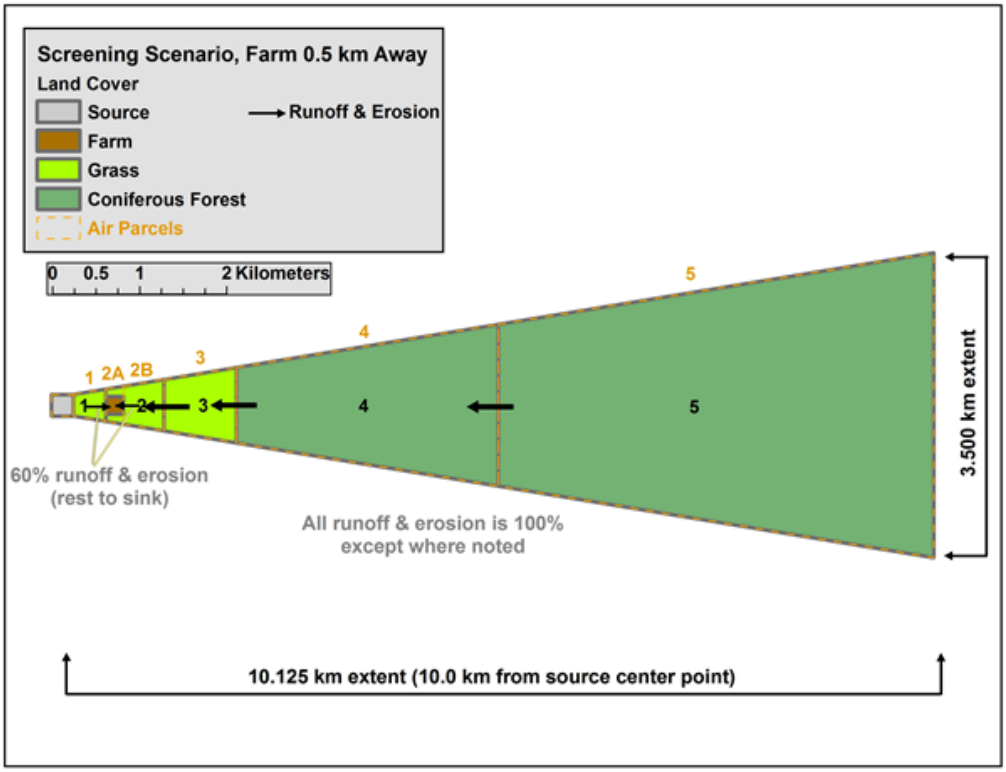
Scenario	Soil	Protected Fruit	Exposed Fruit	Protected Vegetable	Exposed Vegetable	Root Vegetable	Dairy	Beef	Pork	Chicken	Eggs	Breast Milk[a]	Fish
Farmer / Fisher (Tier 1)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Fisher (Tier 2)												✓	✓
Farmer (Tier 2)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	
Pollutants of Concern	As[b] D/F POM	As D/F POM	As D/F POM	As D/F POM	As D/F POM	As D/F POM	As D/F POM	As D/F POM	As D/F POM	As D/F POM	As D/F POM	D/F	As[b] D/F POM Cd[c] Hg

[a]Ingestion of breast milk (for children less than 1 year old and for D/F (dioxins/furans) only.
[b]The health endpoint for exposure to As (as inorganic arsenic), D/F, and POM is cancer.
[c]The health endpoint for exposure to Hg (divalent mercury emissions, exposure to methyl mercury) and Cd (cadmium) is noncancer.

[7]TRIM.FaTE modeling compartments are where chemical mass in TRIM.FaTE is stored, transformed, and transported. Each compartment is defined by its location, size, and physical composition (e.g., environmental media) and how it receives and sends chemicals from/to other compartments.

Figure 3-1. TRIM.FaTE Lake-centric (Top) and Farm-centric (Bottom) Surface Layouts for the Tier 1 Multipathway Screen





We parameterized the physical/chemical environment represented in the screening scenario by using health-protective values for certain parameters to which chemical exposures are especially sensitive.[8] This approach tends to overestimate concentrations in the media that drive ingestion exposures for humans. Some of the more health-protective assumptions in the Tier 1 screening scenario include:

- Low wind speeds (more chemical deposition to the nearby lake and farm relative to higher wind speeds).
- Low mixing heights (higher chemical concentrations in the mixed layer of air next to the ground relative to higher mixing heights).
- High precipitation rates (higher chemical deposition relative to lower precipitation rates).
- Winds frequently blowing chemical emissions toward the lake and farm, and, as previously mentioned, watershed characteristics that enhance chemical loading to the lake and farm via erosion and runoff.

In reality, the probability of such risk-maximizing environmental conditions prevailing across multiple meteorological parameters over the course of the entire year is very low. Also, we assess Tier 1 screening-level risks for a single hypothetical person consuming all food groups and soil at high-end rates (for 350 days per year, and for 70 years for carcinogenic PB-HAPs), from a lake and farm located less than 0.5 km from the facility, which received high-end PB-HAP loading rates resulting from health-protective meteorological and watershed conditions. In reality, such chronic high-end ingestion rates across all media by the same person are highly unlikely. Also highly unlikely is that the same person would consume all food categories from media located so close to the facility, and that those media would receive high-end chemical loading rates from health-protective meteorological conditions day after day.

For ingestion, all consumption of produce and animal products occurs from the modeled locations of the farming area and lake (i.e., ingestion of food grown or bought off site is not considered). The selected food ingestion rates are from the upper ends of representative

[8]The tiered screen was designed so that, even with some national average values as inputs, the exposure/ingestion scenarios for the Tier 1 multipathway screen generally overestimate exposure, and thus, favor health-protective risk estimates that minimize the chance a facility with appreciable risk screens out and is removed from further evaluation.

distributions—at the 90th percentile for each category of farmed foods (U.S. EPA 2011a) and the 99th percentile for fish (373 g/day wet weight fish fillet, Burger 2002, U.S. EPA 2011b). We do not include the drinking water or dermal-contact exposure pathways in the assessments. We assume that humans will not use untreated lake water as a drinking water source. Also, through numerous modeling exercises, ingestion of groundwater contaminated solely from air deposition for the RTR-assessed PB-HAPs contributes negligibly to total exposure. Finally, dermal absorption of originally airborne chemicals similarly has been shown to be a relatively minor exposure pathway compared with other pathways (U.S. EPA 2006a, Cal/EPA OEHHA 2000).[9]

A screening threshold emission rate that denotes negligible risk in the Tier 1 screening scenario is derived by estimating the emission rate corresponding to either a lifetime cancer risk of 1-in-one million or a chronic noncancer HQ of 1 for an individual that consumes all commodities noted above (i.e., the farmer/fisher screening scenario). Table 3-2 provides the oral-pathway dose-response values used in these risk calculations (cancer slope factor [CSF] for cancer risk, reference dose [RfD] for chronic noncancer effects).[10] For benzo(a)pyrene (BaP, the index chemical for POM), which has a mutagenic mode of action, MIRC includes age-specific factors to account for the mutagen's higher potency during childhood, as EPA specifies in supplemental guidance for cancer risk assessments (U.S. EPA 2005a). Specifically, cancer potency for BaP is assumed 10-fold greater for the first 2 years of life and 3-fold greater for the next 14 years. We incorporated these factors into a time-weighted total increase in potency over a lifetime of 70 years.

Table 3-3 provides the corresponding Tier 1 screening threshold emission rates. Tier 1 screening threshold emission rates for cadmium and mercury are based on noncancer effects from exposures to children ages 1–2 years, which is the age group with the highest ADD for these chemicals. Moreover, for cadmium and mercury, the screening threshold emission rates are

[9]Arsenic was added only recently as a chemical that is assessed for RTR ingestion exposures and has not been screened as robustly as the other PB-HAPs to determine its potential for exposure through groundwater ingestion or dermal exposure. EPA plans to investigate the potential for arsenic risks through these pathways in the near future.
[10]If a PB-HAP is carcinogenic and has chronic noncancer effects, the effect resulting in the more stringent screening threshold emission rate is used.

based on an HQ of 1. With respect to arsenic, dioxin, and POM exposures, screening threshold emission rates are based on a potential cancer risk of 1-in-one million.

Table 3-2. Dose-Response Values for PB-HAPs Considered in the RTR Multipathway Assessments

PB-HAP	CSF ([mg/kg-day]$^{-1}$)	Source	RfD (mg/kg-day)	Source
Inorganics				
Cadmium compounds[a,b]	not available		1E−3	IRIS
Arsenic compounds[a]	1.5	IRIS	not critical health endpoint	
Elemental mercury[c]	not available		not available	
Divalent mercury[c]	not available		3E−4	IRIS
Methyl mercury[a]	not available		1E−4	IRIS
Organics				
BaP (index chemical for POM)[a,d]	1	IRIS	not critical health endpoint and no RfD	
2,3,7,8-TCDD (index chemical for dioxin)[a]	1.5E+5	EPA ORD	not critical health endpoint	

Notes: CSF = cancer slope factor; RfD = reference dose; IRIS = EPA's Integrated Risk Information System; EPA 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 (2017a).
[b]RfD for cadmium in food (not water).
[c]Exposure to elemental mercury is not assessed in the multipathway screen due to limited information on oral dose-response. Exposure to divalent mercury is not assessed in the multipathway screen due to its higher (i.e., less stringent) RfD and lower bioaccumulation potential in the ingested food products in the screen, relative to methyl mercury.

For a given facility, if the emission rate of a PB-HAP is less than the Tier 1 screening threshold emission rate, the SV (i.e., ratio of emission rate to threshold emission rate) is less than 1 and risks are assumed to be low. No additional multipathway screen is done on those emissions. If, however, the emission rate of a PB-HAP exceeds the Tier 1 screening threshold emission rate, the SV is greater than 1, and the potential multipathway risk from those emissions can be evaluated further (using the Tier 2 screening scenario).

Table 3-3. Tier 1 Screening Threshold Emission Rates for Multipathway Exposures

PB-HAP	Screening Threshold Emission Rate (TPY)	Basis of Threshold (Type of Health Endpoint)
POM (as BaP equivalents)	3.72E−03	Cancer
Dioxin (as 2,3,7,8-TCDD equivalents)	6.02E−10	Cancer
Cadmium compounds	2.38E−03	Noncancer

PB-HAP	Screening Threshold Emission Rate (TPY)	Basis of Threshold (Type of Health Endpoint)
Arsenic compounds	2.02E−04	Cancer
Mercury (as divalent mercury emissions, but exposure to methyl mercury)	1.46E−04	Noncancer

Notes: PB-HAP = persistent and bioaccumulative hazardous air pollutant; POM = polycyclic organic matter; BaP = benzo(a)pyrene; TCDD = tetrachlorodibenzo-p-dioxin; TPY = short tons per year

For POM and dioxin, we developed a "cancer risk-equivalency" approach to convert congener emissions to equivalent emissions of BaP and 2,3,7,8-TCDD, which are the index chemicals for which screening threshold emission rates were derived for POM and dioxin, respectively. A risk-equivalency factor (REF) is a function of the congener's carcinogenic potency *and* its exposure potential, relative to the index chemical for the group (i.e., relative to BaP for POM and relative to 2,3,7,8-TCDD for dioxin) at equivalent emission rates in a given scenario. The carcinogenic potency of a chemical does not depend on the design of the multipathway assessment. Changes in exposure location and meteorological conditions from one screening tier to another, however, can affect different chemicals in different ways in terms of chemical deposition and subsequent uptake and bioaccumulation in the ingested media, which in turn affects a chemical's exposure potential. Therefore, REFs are specific to the tier at which the screen is conducted (i.e., Tier 1 REFs can be different from Tier 2 REFs). The use of REFs allows a facility's emissions of all evaluated POM congeners (or dioxin congeners) to be summed to equivalent emissions of BaP (or 2,3,7,8-TCDD) for comparison to the screening threshold emission rates. The development of these REFs is described in detail in the next section (Section 3.1.2).

3.1.2 Risk Equivalency Factors for POM and Dioxin Emissions

The multipathway screen the SAB previously reviewed did not account for differences in environmental fate and transport among POM or dioxin congeners (i.e., all POM congeners were assumed to move, partition, and degrade in the environment as BaP does, and all dioxins were assumed to exhibit the same fate and transport as 2,3,7,8-TCDD). The new REF approach includes an exposure-equivalency factor (EEF) that reflects an individual chemical's fate and transport relative to the index chemical for each group (BaP for POM and 2,3,7,8-TCDD for dioxin). The REF equals the product of the chemical-specific EEF and the chemical-specific toxic equivalency factor (TEF), the ratio of the congener's oral toxicity to that of the index chemical, as shown in Equation 1 below.

$$REF_{congener} = EEF_{congener} \times TEF_{congener} \quad \text{Eq. 1}$$

EEFs represent the ratio of the exposure to a particular congener to the exposure to the index chemical at equivalent emission rates. EEF varies across chemicals, depending on the values for various properties that influence environmental fate and transport (e.g., K_{ow} [octanol-water partition coefficient], intermedia partition coefficients, molecular weight, half-life, potential for biodegradation). EEFs for a given congener change as environmental conditions change and vary with distance from the source. That means EEFs in the multipathway screens differ across the tiers because they depend on lake location, farm location, and meteorological conditions used.

The TEF represents the ratio of the oral toxicity (i.e., carcinogenic potency) of a particular congener to that of its index chemical, and the TEF remains the same across all tiers. When available, the oral toxicity values for POM congeners are equal to their CSF (i.e., the TEF is $CSF_{congener}/CSF_{BaP}$). When POM CSFs are not available, the congener toxicity is estimated from other POM congeners with similar molecular structures and characteristics. The TEFs for dioxin congeners were developed by Van den Berg et al. (2006), with the exception of 1,2,3,7,8,9- and 1,2,3,6,7,8-hexachlorodibenzo-p-dioxin (an IRIS CSF is available for a mixture of these two congeners). The EEFs and TEFs for POM congeners currently assessed are shown in Table 3-4, and those for dioxin are listed in Table 3-5.

Twenty-two POM congeners are not fully characterized for modeling in TRIM.FaTE due to lack of chemical-specific values for certain environmental parameters (see second column of Table 3-4). For most of these congeners, we used K_{ow}-based regression estimates to develop EEFs (refer to Figure 3-2, which illustrates a strong and direct correlation between K_{ow} and exposure).[11] For unspecified POM compounds (i.e., reported as "PAH, total", "polycyclic organic matter", and "benz[a]anthracene/chrysene"), we assumed K_{ow} values near the upper end of the range across all congeners (a health-protective assumption). In addition, due to lack of information on the K_{ow} for benzo[b+k]fluoranthene, we made a health-protective assumption that

[11]We evaluated the relationships between chemical-specific properties (e.g., K_{ow}, Henry's Law Constant) and intermediate modeled values (e.g., deposition rate, soil concentration) and exposure. The correlation between K_{ow} and exposure is stronger than for any other chemical-specific property. For POM congeners not characterized in TRIM.FaTE and MIRC, we interpolated age-adjusted LADDs—an estimate of exposure—using the chemical's K_{ow} value in a power regression based on the modeled POMs.

the K_{ow} equaled that of benzo[k]fluoranthene, which has the larger K_{ow} value of the two. Like the EEFs derived from modeling, these regression-derived EEFs also differ across the screening tiers. All dioxin congeners are fully characterized and modeled in TRIM.FaTE.

Table 3-4. Toxic Equivalency Factors and Tier 1 Exposure and Risk Equivalency Factors, Relative to BaP for POM Congeners Currently Evaluated in Multipathway 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	0.01	0.003
2-Acetylaminofluorene	No	0.003	1	0.003
Fluoranthene	Yes	0.03	0.05	0.002
Acenaphthylene	Yes	0.02	0.05	0.001

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)
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
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 (2017b); professional judgment.

Table 3-5. Toxic Equivalency Factors, and Tier 1 Exposure and Risk Equivalency Factors, Relative to 2,3,7,8-TCDD for Modeled Dioxin Congeners Currently Evaluated in Multipathway Screens

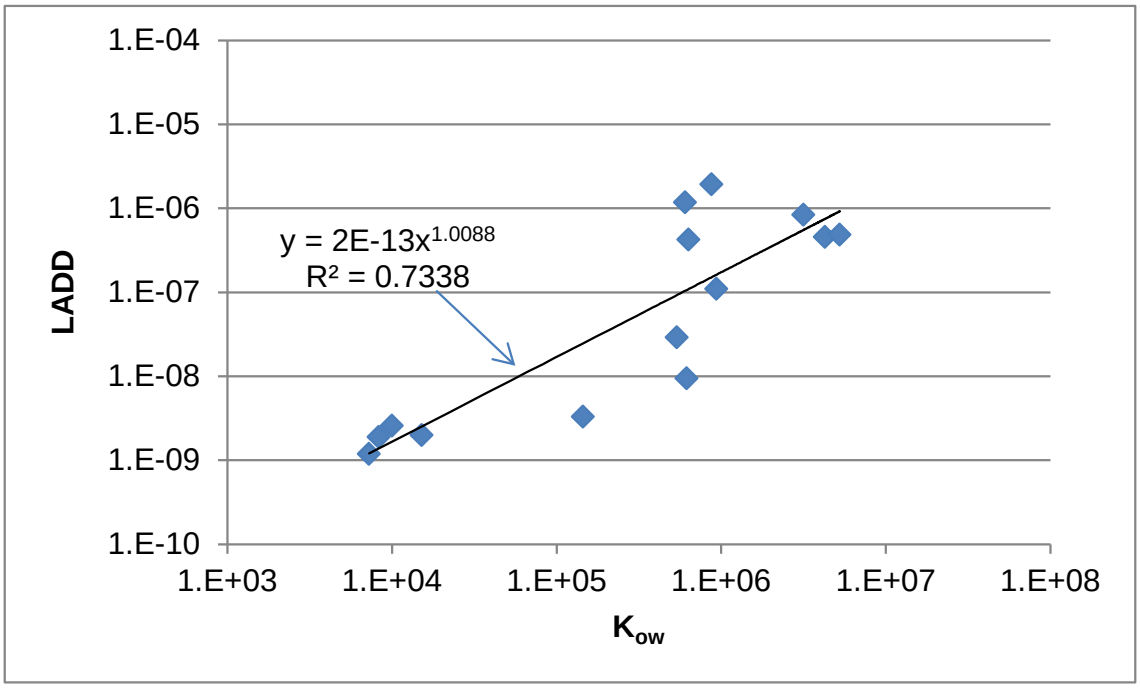
PB-HAP	Tier 1 Exposure-equivalency Factor (EEF)	Toxic-equivalency Factor (TEF)[a]	Tier 1 Risk-equivalency Factor (REF)
1,2,3,4,6,7,8,9-Octochlorodibenzofuran	0.2	0.0003	0.00007
1,2,3,4,6,7,8,9-Octochlorodibenzo-p-dioxin	0.3	0.0003	0.00008
1,2,3,4,6,7,8-Heptachlorodibenzofuran	0.2	0.01	0.002
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	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

PB-HAP	Tier 1 Exposure-equivalency Factor (EEF)	Toxic-equivalency Factor (TEF)[a]	Tier 1 Risk-equivalency Factor (REF)
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	1.0	0.04	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	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

[a]Sources: Van den Berg et al. (2006), except for 1,2,3,7,8,9- and 1,2,3,6,7,8-hexachlorodibenzo-p-dioxin, which are calculated based on the ratio of the IRIS-based CSF for a mixture of 1,2,3,6,7,8-HxCDD and 1,2,3,7,8,9-HxCDD to the IRIS-based CSF for 2,3,7,8-TCDD.

Figure 3-2. Relationship between Exposure (LADD) and K_{ow} for POM Congeners Characterized for TRIM.FaTE Modeling



Notes: LADD = age-adjusted lifetime average daily dose; K_{ow} = octanol-water partition coefficient; POM = polycyclic organic matter.

3.1.3 Example of a Tier 1 Multipathway Screen

This section presents an example Tier 1 multipathway for hypothetical Facility Y (see Table 3-6). An analysis of the stack test data identified emissions exceeding the detection limit for mercury (as divalent mercury), 1,2,3,7,8-pentaCDD, and 2,3,7,8-TCDD.

Table 3-6. Emissions and Stack Release Parameters for Facility Y

PB-HAP	Emissions (TPY)	Stack Release Parameters			
		Height (m)	Diameter (m)	Exit Gas Velocity (m/sec)	Temperature (K)
Divalent mercury	0.042	34	3	11.5	350
1,2,3,7,8-PentaCDD	2.28E−07	34	3	11.5	350
2,3,7,8-TCDD	3.00E−07	34	3	11.5	350

Notes: PB-HAP = persistent and bioaccumulative hazardous air pollutant; TPY = short tons per year; m = meters; m/sec = meters per second; K = Kelvin; PentaCDD = pentachlorodibenzo-p-dioxin; TCDD = tetrachlorodibenzo-p-dioxin.

Mercury does not require using REFs, as only one chemical is evaluated. We calculated the Tier 1 SV for mercury as

$$\textit{Tier 1 SV} = \frac{Facility\ Emission\ Rate}{Tier\ 1\ Screening\ Threshold\ Emission\ Rate} \qquad \text{Eq. 2}$$

$$\textit{Tier 1 Mercury SV} = \frac{0.042\ TPY}{1.46E{-}04\ TPY} = \boldsymbol{288}.$$

Dioxin assessment requires utilization of REFs. We calculated the Tier 1 SV for dioxin as

$$\textit{Tier 1 SV} = \frac{\sum^{n} Facility\ Emission\ Rate_n \times REF_n}{Tier\ 1\ Screening\ Threshold\ Emission\ Rate} \qquad \text{Eq. 3}$$

where n = each congener

$$\textit{Tier 1 Dioxin SV} = \frac{\sum(2.28E{-}07\times 3.6) + (\ 3.00E{-}07\times 1)}{6.02E{-}10} = \boldsymbol{1{,}861}.$$

These example Tier 1 screening results for Facility Y are also provided in Table 3-7. The emissions of mercury and dioxins greatly exceeded the respective Tier 1 screening threshold emission rates—mercury emissions were a factor of 300 above the noncancer screening threshold emission rate that reflects a potential HQ of 1.0. Dioxin emissions were a factor of 2,000 above the cancer screening threshold emission rate that reflects a potential cancer risk of 1-in-one million (and a factor of 20 above a level reflecting a potential cancer risk of 100-in-one

million). Potential hazards and risks for both pollutants cannot be ruled out, and a Tier 2 assessment would be conducted.

Table 3-7. Tier 1 Screening Results for Example Facility Y

PB-HAP Group	PB-HAP Chem.	Emiss. (TPY)	Index Chem.	TEF	EEF	REF	Equiv. Emiss. of Index Chem. (TPY)	Screening Threshold Emiss. Rate (TPY)	Screening Value (Rounded)
Mercury	Divalent mercury	0.042	Divalent mercury	1	1	1	0.042	1.46E−04	288 (300)
Dioxin	1,2,3,7,8-PentaCDD	2.28E−07	2,3,7,8-TCDD	1	3.6	3.6	8.21E−07	6.02E−10	1,862 (2,000)
	2,3,7,8-TCDD	3.00E−07		1	1	1	3.00E−07		

Notes: PB-HAP = persistent and bioaccumulative hazardous air pollutant; TPY = short tons per year; TEF = toxic equivalency factor; EEF = exposure equivalency factor; REF = risk equivalency factor; chem. = chemical; equiv. = equivalent; emiss. = emission; pentaCDD = pentachlorodibenzo-p-dioxin; TCDD = tetrachlorodibenzo-p-dioxin

3.2 Tier 2 Multipathway Screen

In the Tier 2 multipathway screening scenario, some of the health-protective assumptions in the Tier 1 screen are replaced with more site-specific information. That is, the Tier 2 screen incorporates some site-specific meteorology data and actual locations of fishable lakes near the facility. Tier 2 also evaluates farm-level exposures at various distances from the facility and identifies the location of maximum estimated risk. The food-specific ingestion rates used in Tier 1 are used in Tier 2, but total exposures are estimated separately for the fisher and farmer scenarios used in Tier 2. The Tier 2 screen does not combine fish and farm exposures as in Tier 1, resulting in separate Tier 2 screening threshold emission rates for the farmer and fisher scenarios. Also in contrast to the Tier 1 screen, screening threshold emission rates can vary across specific locations based on site-specific meteorology and the distance a facility is from a fishable lake or a farm. 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 might require additional analysis.

In the Tier 2 assessment, the key model inputs that are refined from the Tier 1 screen with more site-appropriate information include:

- Locations of potentially fishable lakes
- Annual characteristics of wind speed, wind direction, precipitation rate, and mixing height.

These inputs were selected for site-specific modifications based on:

- Degree of influence on the SV
- Ease of implementation in TRIM.FaTE
- Ease of obtaining parameter values more representative of specific locations.

The Tier 2 assessment also contains other modifications to the Tier 1 exposure scenario, including:

- A screening configuration that assesses the fisher and farmer exposure scenarios separately (i.e., assumes that exposed individuals consume either a subsistence-level amount of fish or a subsistence-level amount of farm foods, but not both; see Sections 3.2.1.2 and 3.2.1.3).
- An estimation of lake productivity—the estimated daily quantity of fish that a fisher could catch for 50 years before that lake could no longer support the types of fish typically consumed by humans (see Section 3.2.2.2).
- The consideration that a fisher is catching and consuming fish from more than one nearby contaminated lake, because more than one lake might be needed to catch enough fish for subsistence living (see Section 3.2.2.3).
- Accounting for PB-HAP deposition into a lake from multiple facilities in the same RTR source category (see Section 3.2.2.3).

The overall implementation of the Tier 2 multipathway screen is illustrated in Figure 3-3. The steps on the left in Figure 3-3, which are discussed in Section 3.2.1, need only be conducted once. These “one-time” steps are where more site-specific values are determined for meteorological, lake, and farm parameters, and those values then were used in a large set of TRIM.FaTE and MIRC model runs to create Tier 2 screening threshold emission rates and REFs.

The steps on the right in Figure 3-3, which we discuss later in Section 3.2.2, are conducted for each facility. As discussed in that section, a facility being screened is matched to real lakes in its vicinity and to its closest meteorological station. The Tier 2 screening threshold emission rates and REFs are used from the corresponding modeling scenarios that most closely match those lake locations and meteorological conditions (which is the third step on the right in Figure 3-3).

As with Tier 1, the facility screens out if all Tier 2 SVs do not exceed 1 (emissions are below the Tier 2 screening threshold emissions rate); otherwise, additional refinement might be needed (i.e., Tier 3; see Section 3.3).

3.2.1 Development of Scenario for Tier 2 Multipathway Screen

The "one-time" steps presented on the left side of Figure 3-3 are discussed in this section. Section 3.2.1.1 discusses the use of site-specific meteorological data. Section 3.2.1.2 discusses the potential locations of lakes and farms. Finally, Section 3.2.1.3 discusses the creation of a library of Tier 2 screening emission threshold rates, REFs, and mixing height refinements.

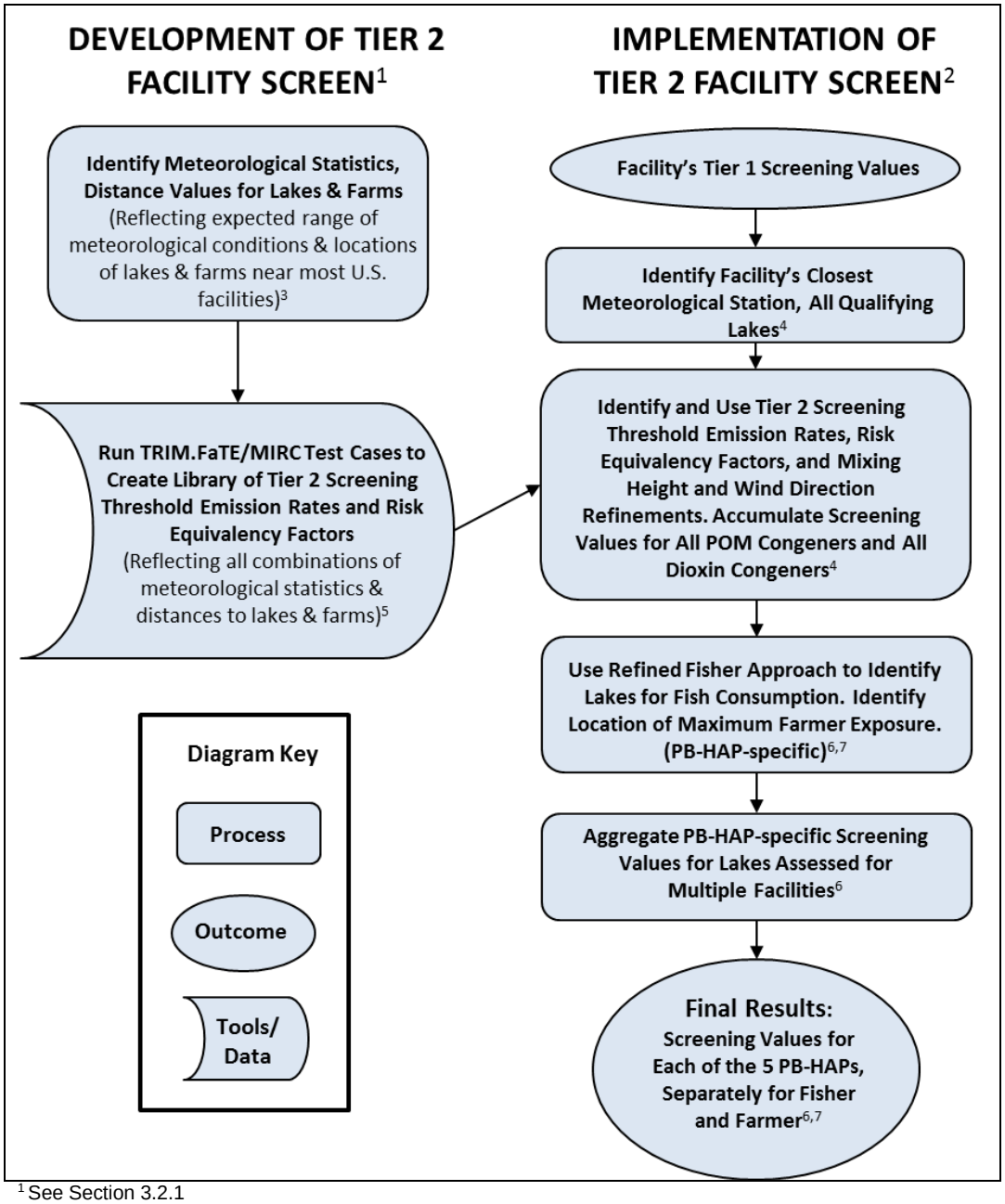
3.2.1.1 Meteorology Data

We created a database of the relevant U.S. meteorology data for 824 surface stations paired with their closest upper-air stations located throughout the country. The hourly surface data cover 2014 and are the same AERMOD-ready data EPA's Office of Air Quality Planning and Standards (OAQPS) uses for RTR inhalation modeling.

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 hypothetical locations of farm food and soil exposures during the screening of a facility (facility screening is discussed in Section 3.2.2). The area around a facility is divided into the eight octants representing the direction toward which the wind was blowing. N is north, NE is northeast, and so on:

N:	>337.5 to 360 or >0 to 22.5 degrees	NE: >22.5 to 67.5 degrees
E:	>67.5 to 112.5 degrees	SE: >112.5 to 157.5 degrees
S:	>157.5 to 202.5 degrees	SW: >202.5 to 247.5 degrees
W:	>247.5 to 292.5 degrees	NW: >292.5 to 337.5 degrees

Figure 3-3. Basic Process for Implementing the Tier 2 Multipathway Screen



[1] See Section 3.2.1
[2] See Section 3.2.2
[3] See Section 3.2.1.1 on meteorology data and Section 3.2.1.2 and locations of lakes and farms
[4] See Section 3.2.2.1
[5] See Section 3.2.1.3
[6] See Section 3.2.2.3
[7] See Section 3.2.2.4

From the hourly meteorology 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 missing winds
- 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 preferring 30-year normal data[12] if available, to avoid biasing the screening results in favor of any precipitation anomalies that existed in 2014).

We selected the median values because they were usually smaller than mean values, and smaller values of wind speed and mixing height are more health protective (i.e., typically lead to higher chemical deposition in areas near the emission source). We examined the distributions of the median wind speeds, median mixing heights, and average precipitation amounts (for the 823 stations together) and identified a range of values to represent conditions that reasonably would be encountered at RTR facilities across the United States. These values represent lower-bound, middle, and upper-bound distributions across all sites (i.e., the 5th, 35th, 65th, and 95th percentile values). We used these values, shown in Table 3-8, 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 3.2.1.3).

Table 3-8. 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 leads to more chemical transfer out of the model 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, more wet deposition occurs over the modeled domain.
	706	
	1069	
	1474	
Mixing Height (m)	226	

[12]We obtained 30-year-average annual precipitation, for 1981–2010, from the National Oceanic and Atmospheric Administration. http://www.ncdc.noaa.gov/oa/climate/normals/usnormals.html.

Parameter	Value	Risk Direction
	351	As mixing heights increase, pollutants in the air can become more diluted, resulting in lower modeled deposition and other transfer rates from air to surfaces and, consequently, lower ingestion exposures.
	454	
	674	

Notes: Bold font indicates the value is equal to 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 (Equation 5) is discussed in Section 3.2.2.1.

3.2.1.2 Locations of Lakes and Farms

The database of lakes used in the Tier 2 screen is based on U.S. Geological Survey (USGS) data and includes information on the location, surface area, use or type designation, and name (if available) of all lakes in the United States (see Section 3.2.2 for more information). We include lakes (or farms) within a 50-km radius around a facility. We believe that a 50-km domain reasonably restricts how far a nearby resident will routinely travel to catch and consume fish from area lakes. Extending the modeling domain beyond 50 km 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.[13]

As indicated in Table 3-9, within this radius we evaluate effects on lakes and farms at five distances in Tier 2: three distances within a 10-km radius where most chemical deposition occurs (at 0.5, 5, and 10 km from the facility) and two distances beyond 10 km (at 20 and 40 km from the facility). All farm locations are hypothetical, so potential exposure is evaluated in Tier 2 at each possible distance and octant. Note that if a lake's actual distance is between two of the distances in Table 3-9, the more health-protective distance (i.e., the distance expected to yield the greatest risk) is considered. For example, if an actual lake is 7 km from the facility being analyzed, that lake will be placed 5 km from the facility for screening purposes.

[13]Mass deposited at the outer edge of the watershed is expected to result in a negligible increase in estimated exposure via fish- or farm-food-chain consumption by increasing the chemical mass transported to the lake and farm through erosion and runoff. The distance from these more distant locations to the lake and farm would attenuate transport of chemical mass by erosion and runoff, dampening the effect of including additional deposition beyond the parameterized watershed. Wind speeds of 13 m/s (approximately 29 mph) or greater must be sustained for an 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.

Figure 3-4 depicts the spatial layouts of each lake- and farm-distance scenario in Tier 2 for a single octant.[14] The 0.5-km-distance scenario is the same layout as the Tier 1 scenario and is shown in Figure 3-1 and not repeated in Figure 3-4. The runoff and erosion characteristics are unchanged from the Tier 1 screen.

Table 3-9. Values for Lake and Farm Distances Used to Develop the Tier 2 Screening Threshold Emission Rates and REFs

Parameter	Value	Risk Direction
Lake and Farm Distances, measured at the inside geographic centroid of the feature (km)	0.5	As the lake or farm and its watershed are moved farther from the facility, they tend to receive reduced chemical deposition and, consequently, exposures from the fish-consumption pathway or farm-food pathway are reduced.
	5	
	10	
	20	
	40	

Note: Bold font indicates the value is equal to the value used in Tier 1.

3.2.1.3 Development of Tier 2 Screening Threshold Emission Rates, REFs, and Mixing Height Refinements

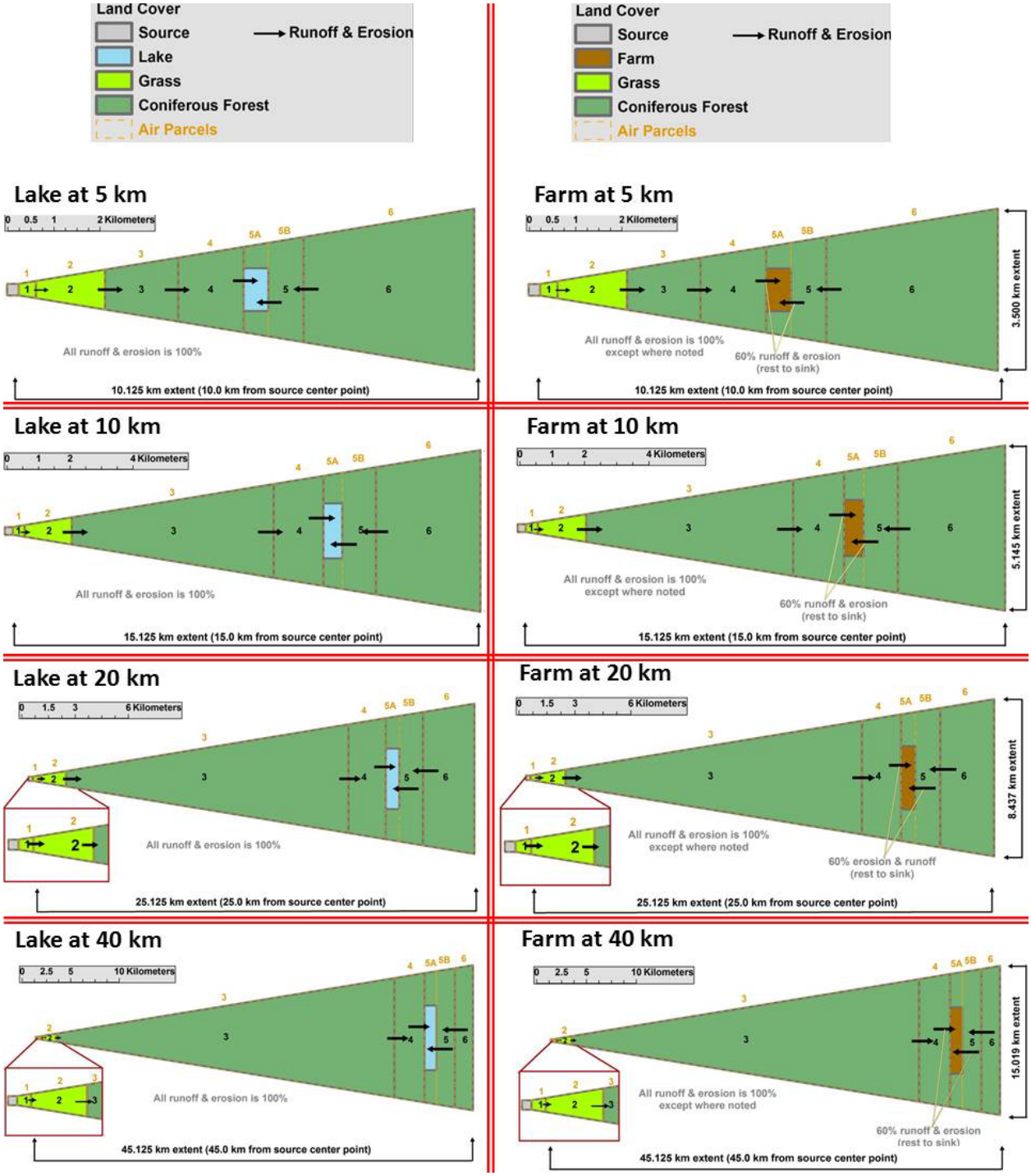
Based on unit emissions of 1 g/day and taking into account: (1) wind speed, mixing height, and precipitation rate values shown in Table 3-8; (2) lake- and farm-distance values shown in Table 3-9; and (3) spatial layouts shown in Figure 3-1 and Figure 3-4, we conducted a large set of TRIM.FaTE and MIRC modeling runs. These runs systematically varied each parameter so that all possible combinations were evaluated.[15] The result of this analysis was a matrix of screening level *risk estimates* based on 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.

We know that mixing height directly affects chemical concentrations—lower mixing heights increase chemical air concentrations, increase deposition, and increase chemical concentrations in environmental media across a range of distances from the emission source. Precipitation, on the other hand, causes competing effects on chemical concentrations—more precipitation causes more chemical deposition, but it also dilutes the deposited chemical. Further, the effects of wind speed on chemical air concentrations change based on distance from the emission source—lower

[14]The lake and farming 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 farming areas.
[15]For each PB-HAP chemical, (e.g., cadmium, each dioxin/furan congener), 640 independent modeling runs were completed using TRIM.FaTE and MIRC.

wind speeds increase chemical air concentrations and deposition in the near field but decrease them in the far field, and vice versa for higher wind speeds.

Figure 3-4. TRIM.FaTE Surface Layouts for the Tier 2 Multipathway Screen, Using Alternative Distances Between the Facility and the Fishable Lake or Farm



Notes: Heavy, black arrows depict the direction of chemical runoff and erosion.
Base layout is the same as Tier 1 shown in Figure 3-1 and is not shown here.

Because of (1) the more direct and predictable effects of mixing heights on concentrations of chemical in the air and environmental media and (2) the wide range of median mixing heights across the 823 meteorological stations (i.e., less than 200 m to more than 2,000 m, we used the matrix of risk estimates just described above to further quantify the relationship between mixing height and risk. We found that mixing heights share an exponential relationship with the screening-level risk estimates described above. That is, decreases in mixing height lead to exponential increases in risk. These exponential relationships are specific to each combination of PB-HAP, distance from facility to lake or farm, wind speed, and precipitation amount. Therefore, for each of these combinations, we used their corresponding screening-level risk estimates to derive a power-curve regression equation that relates changes in mixing height (i.e., the four values used in the modeling) to changes in risk. From this curve, we then were able to derive a matrix of power-curve regression coefficients based on each combination of values. These power-curve regression coefficients ultimately are used to adjust the ratio between a facility's actual emissions and a Tier 2 screening threshold emission rate (i.e., the Tier 2 SV) to reflect the effects on risk of the actual mixing height near a facility more accurately.

After developing the mixing height regression coefficients, we created the Tier 2 library of screening threshold emission rates and REFs. Each library entry is a unique combination of PB-HAP, wind speed, precipitation rate, and distance from the facility to a lake or farm. For mixing height, the screening threshold emission rates and REFs are derived using a mixing height value of 226 m (i.e., the Tier 1 mixing height).[16] Moreover, as in the Tier 1 screen, the Tier 2 screening threshold emission rate is defined as the emission rate necessary to reach a 1-in-one million excess lifetime cancer risk or an HQ of 1 for a given PB-HAP. Also similar to the Tier 1 screen, 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 3.1.2). We finally note that the Tier 2 screen assesses potential risk from fish ingestion separately from farm-food and soil ingestion, and therefore,

[16]All Tier 2 screening threshold emission rates are based on a mixing height of 226 m. The SVs calculated from these screening threshold emission rates, however, 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), the refined Tier 2 SV for this facility would be 2.4 (3 × 0.8). This calculation is described further in Section 3.2.2.

there is one library of screening threshold emission rates for fish ingestion and a separate library for farm-food and soil ingestion.

The rest of this chapter discusses how the screening threshold emission rates, REFs, and power regression coefficients discussed in this section are used to estimate potential multipathway risk.

3.2.2 Implementing the Tier 2 Multipathway Screen

To implement the Tier 2 multipathway screen, we developed a Microsoft® Access™ tool that is preloaded with (1) U.S. lake location data; (2) the meteorology database; and (3) libraries of Tier 2 screening threshold emission rates, REFs, and mixing height regression coefficients described in Section 3.2.1, above.

As noted above, the database of lakes used in the Tier 2 screen is available from ESRI® and based on USGS data. This 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 the catching of upper-trophic-level fish, we excluded lakes used for disposal, evaporation, or treatment, and we included only lakes greater than 25 acres (see Section 3.2.2.2 for more detail). We did not include lakes larger than 100,000 acres because they are not readily modeled in TRIM.FaTE due to the complexity of nearshore and offshore hydrodynamics and size of their watersheds. The database of lakes contains approximately 433,000 fishable lakes for evaluating Tier 2 effects.

3.2.2.1 Calculations for Tier 2 Screen

The Microsoft Access tool identifies all qualifying lakes in the area surrounding a screened facility and determines their distances and directions relative to the facility; each distance subsequently is matched to the closest (i.e., more health-protective) lake distance in the Tier 2 library (lakes closer to the emission source receive more chemical deposition). The user may alter 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 also may review matching lakes and exclude lakes considered unsuitable for modeling (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 locations are not site specific, so we calculate the Tier 2 farmer SV at all distances available in the Tier 2 library and at all directions relative to the facility.

Each facility being screened then is matched with the same surface meteorological station used in the RTR inhalation risk assessment (i.e., typically the closest station). The tool enables the user to provide a custom match of facility to meteorological station. We match the meteorological station's annual precipitation amount to the closest, larger precipitation amount in the Tier 2 library (higher values of precipitation lead to greater chemical deposition). We identify the annual median wind speed blowing toward each lake or farm location at the facility and match it to the closest, smaller wind speed value in the Tier 2 library (smaller wind speeds increase chemical deposition near the facility).

Given the matching meteorology (wind speed and precipitation) and distance (for farms 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 value for the facility's matching meteorological station then is used with the mixing height regression coefficients from the Tier 2 library to develop a site-specific refinement factor incorporated into the calculation of the facility's Tier 2 SV, as discussed later in this section. This mixing height refinement factor is calculated as

$$RefMix_{T2} = I \times (M^S) \qquad \text{Eq. 4}$$

where:

$RefMix_{T2}$	=	mixing height refinement factor (multiplier) for Tier 2
I	=	intercept coefficient of the power-curve regression
M	=	median mixing height (in meters) associated with the facility
S	=	slope coefficient of the power-curve regression.

With the identification of the appropriate Tier 2 screening threshold emission rate, REF, and mixing height refinement factor now complete, one final site-specific factor is considered: the frequency that winds blow toward an evaluated lake or hypothetical farm location. In the Tier 2 TRIM.FaTE and MIRC runs, as in the Tier 1 runs, winds were modeled as blowing toward the lake and farm 43 percent of the time (i.e., 3 days per week—an unusually consistent long-term

wind pattern but not unrealistic; e.g., similar to wind direction patterns in Yakima, Washington). The screening threshold emission rates in the Tier 2 library correspond to this wind direction frequency. Using the Tier 2 database of meteorology data, the screening implementation in the Tier 2 Microsoft Access tool accounts for the percentage of time that the wind actually blows in the direction of the lake or farm being evaluated in the Tier 2 screen. This wind direction refinement factor is calculated as

$$RefWD_{T2} = \frac{FreqWD_{T2}}{FreqWD_{T1}} \quad \text{Eq. 5}$$

where:

$RefWD_{T2}$	=	Tier 2 wind direction refinement factor (multiplier)
$FreqWD_{T2}$	=	percentage of time winds blow toward the Tier 2 lake or farm
$FreqWD_{T1}$	=	percentage of time winds blow toward the Tier 1 lake and farm (i.e., 43%).

Finally, for each chemical emitted by a facility, we calculate the Tier 2 SV for each lake and farm as

$$SV_{T2} = \left(\frac{ER \times REF_{T2}}{Th_{T2}}\right) \times RefMix_{T2} \times RefWD_{T2} \quad \text{Eq. 6}$$

where:

SV_{T2}	=	Tier 2 SV
ER	=	facility emission rate
REF_{T2}	=	Tier 2 REF
Th_{T2}	=	Tier 2 screening threshold emission rate.

As is done in the Tier 1 screen, the Tier 2 SVs for all emitted POM chemicals are summed to a total SV of POM as BaP, and the Tier 2 SVs for all emitted dioxin/furan chemicals are summed to a total SV of dioxin/furan as 2,3,7,8-TCDD.

At this stage of the Tier 2 screen, the Tier 2 fisher-scenario SVs reflect subsistence fishing at each individual lake, regardless of whether the lake can sustain that level of fishing. As discussed in Section 3.2.2.2, we further refine the Tier 2 fisher-scenario SVs to better reflect sustainable fish withdrawals. The Tier 2 farmer-scenario SVs reflect subsistence farming at each hypothetical farm. As discussed in Section 3.2.2.4, we identify the location with the largest Tier 2 farmer-scenario SV for a facility and PB-HAP.

3.2.2.2 Accounting for Sustainable Fishing

Early in the process of compiling the Tier 2 lake database, we encountered the question: "What size water body qualifies as a 'lake' for the purposes of this assessment?" The Tier 2 screen must focus on lakes large enough to support relatively intensive fishing pressure to be compatible with the high-end fish ingestion rates assumed for the exposure scenario (i.e., 373 grams/day).

Note that, in the TRIM.FaTE model screening scenario, water-column carnivores (WCCs) are modeled as trophic level 4 (TL4) fish (e.g., pickerel, largemouth bass), with their diet consisting of smaller "prey" or "pan" fish in the water column that are simulated as trophic level 3 (TL3). The benthic carnivores (BC) in TRIM.FaTE are modeled to represent an intermediate trophic level between 3 and 4 (i.e., TL3.5). Benthic carnivores (e.g., catfish) obtain half their diet from TL2 (trophic level 2, benthic invertebrates that feed on detritus at the sediment surface) and half from TL3 fish in the benthic environment. Together, we refer to these two fish compartments (WCC and BC) as piscivorous fish.

To estimate the relationship between high human fish-consumption rates, harvest rates, and lake area, we made eight key assumptions, listed below. Appendix A provides information and citations to peer-reviewed literature supporting these assumptions.

1. Very small ponds/lakes (e.g., <1 acre) might not support more than three trophic levels, given the size constraints on total lake productivity (e.g., algal and invertebrate productivity) (Brönmark and Weisner 1996).

2. In larger lakes, piscivorous fish (i.e., WCC TL4 and BC TL3.5) might comprise approximately 20–22 percent of the total fish biomass (Appendix A, Sections A.1.3 and A.5.3).

3. Productivity in most lakes of small to moderate size depends substantially on the benthos, with benthic invertebrates consuming detritus. We expect more biomass in the BC than in the WCC compartment. Assuming 21 percent of the standing biomass of fish is 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 (see Appendix A, Sections A.1.3 and A.5.3). The remaining approximately 80 percent would include "pan" fish (e.g., sunfish, perch),

minnows, and young-of-the-year of piscivorous fish. This set of assumptions represents a "point estimate" of fish biomass in 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 availability of fish in the BC and WCC compartments (which would result in a ratio closer to 80:20 of BC to WCC consumption). 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 fish compartment than in the WCC compartment. Because we could not predict, a priori, which fish compartment—WCC or BC—would have higher chemical concentrations for any of the chemicals, we assumed the 50:50 split in fish harvested from the WCC and BC compartments.

5. The total fish standing biomass is assumed 40 g wet weight/m^2, which might represent relatively high productivity for natural ponds and lakes across much of the United States. Leidy and Jenkins (1977) found the mean total fish biomass of 61 reservoirs to be 41.3 g[fish ww]/m^2 (± 30.4 g/m^2), with median 30.9 g[fish ww]/m^2. We assume that reservoirs tend to be larger and shallower than natural lakes and therefore might have higher primary production than natural lakes. Additional data on total fish biomass measured in several lakes that suggest lower productivities in general are provided in Appendix A, Section A.5. Overestimates of lake productivity would bias results to be more health protective, because more fish could be harvested from the more 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 the short term (e.g., more than a decade) (Shaffer 1981, 1987; Appendix A, Section A.3.3). Interbreeding populations of 500 breeding adults (with 50:50 male:female) or more 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 (Appendix A, Section A.3.3).

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 (see Section A.3.4).

8. Only 33 percent of the fish caught for consumption is edible fillet muscle (Ebert et al. 1993). 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 the standing biomass of BC fish, we focus on lakes that can provide the MVP of 50 breeding individuals for the WCC fish compartment. Appendix A, Section A.6, presents the calculations and steps required to estimate which combinations of lake size and productivity could sustain at least 50 individual WCC fish, and then the human fish ingestion rates that could be supported for those combinations.

The gray shading in Table 3-10 indicates combinations of lake size and lake productivity that would not support an MVP of 50 individual adult WCC fish (Appendix A, Section A.6, Table A-15). The white, or unshaded, cells in Table 3-10 indicate combinations of lake area and productivity that could sustain the listed fish-ingestion rates for WCC plus BC fish over some time (e.g., perhaps several decades) but might not be sufficient to prevent inbreeding depression. Finally, the yellow shading in Table 3-10 indicates combinations of lake productivity and lake size likely to provide long-term sustainability of WCC fish in the lake.

Once we had established which cells of Table 3-10 were in the gray, white, and yellow zones, we calculated the fish ingestion rates associated with each cell (Appendix A, Section A.6, Tables A-16 and A-17).

Table 3-10. Estimated Maximum Fish-ingestion Rate (g/day) Associated with Sustainable Fishing[a]

Total Fish Biomass (g ww/m²)[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 Appendix A.

[a]Dark gray shading indicates that we calculated the number of adult WCC fish to be less than 50, the minimum viable population size; yellow-shaded cells indicate that we assume that 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 assumed 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 upper-limit total fish standing biomass of 40 g ww/m^2 estimated for natural lakes over much of the United States, a 25-acre lake could support a water-column WCC fish population but would provide at most 26 grams of fillet 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/day of fish fillet. Reading straight across the row associated with 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 could quickly estimate lake productivity in grams of fish fillet [WCC & BC]/day/person as equal to the lake surface area in acres.

At the assumed upper-limit 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 minimum size for an actual lake near a facility to be included in the Tier 2 and Tier 3 assessments. 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 nearshore and offshore hydrodynamics and size of their watersheds.

As shown in Table 3-10, the fish-ingestion rate associated with a 25-acre lake and the assumed upper-limit total standing fish biomass of 40 g ww/m^2 is 26 g/day, or approximately 1 g-fish/acre/day. Thus, a 25-acre lake alone cannot 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. A fisher could fish multiple lakes, totaling 370 to 380 acres, however, to achieve the adult ingestion rate. In Section 3.2.2.3, we discuss the refined-fisher scenario, whereby a fisher withdraws and consumes fish at an assumed sustainable rate of 1 g-fish/acre/day from as many acres of lake(s) as necessary to achieve a 373-g-fish/day rate. The refined-fisher scenario also aggregates SVs at lakes influenced 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. We assume that when introduced to the lake, however, 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 roughly 3 to 6 months (i.e., the fishing season) for most of the fish stocked as large juveniles or adults, instead of several years for fish hatched or born in 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, which fish compartment might have higher chemical concentrations is unclear. Alternatively, fishers could consume “pan” fish like sunfish and small perch and perhaps meet their daily fish ingestion rates fishing smaller lakes than predicted in Table 3-10. Pan fish, however, represent TL3 fish in the water column that feed on zooplankton and fish fry (which are TL2). Therefore, chemical

concentrations in the tissues of pan fish likely would be less than in the TL3.5 BC or the TL4 WCC fish compartments.

3.2.2.3 Methodology for the 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 3.2.2.1) does not provide an adequate catch to satisfy the assumed ingestion rate (i.e., 373 g-fish/day for adults). We assume that the biological productivity of each lake is limited to 1 g-fish/acre; that is, to fulfill the adult ingestion rate, the fisher will need to fish from 373 total acres of lakes (see also Appendix A for further discussion on lake productivity). If less than 373 total acres of fishable lakes (i.e., lakes of at least 25 acres) are present within the modeling radius of 50 km from the facility, the ingestion rate will be less than 373 g-fish/day (i.e., it will be based on the number of acres of fishable lake).

For the Tier 2 screening to remain health protective, we assumed that the fisher visits lakes in order of highest to lowest Tier 2 SV for a PB-HAP (as calculated in Section 3.2.2.1), until either the rate of 373 total g-fish/day is achieved or until all lakes have been fished. This ordering of fished lakes can be different for different PB-HAPs emitted by a facility, as fate-and-transport characteristics vary by chemical. That is, because fate-and-transport characteristics vary by chemical, the most contaminated lake of at least 25 acres for one PB-HAP might differ from the most contaminated 25-acre or larger lake for another PB-HAP. In this scenario, the order of lakes fished would differ for these two PB-HAPs in the Tier 2 analysis, consistent with a health-protective approach to estimating risk. 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. Three possible lake fishing scenarios, as discussed below, are (1) the first lake is 373 acres or larger, (2) multiple lakes total 373 acres or larger, and (3) the lakes do not total 373 acres or larger.

1. If the first lake fished is 373 acres or larger, the fisher is assumed to catch all fish from that lake (i.e., 373 g-fish/day are caught and consumed from the first lake). After evaluating the first facility (Facility A), we identify any other nearby facilities in the source category that are also depositing PB-HAP to the lake. If another nearby facility

(Facility B) in the same source category also influences the lake (i.e., if the lake is within 50 km of Facility A and Facility B is in the source category), 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.

If the first lake fished is smaller than 373 acres, however, multiple lakes must be fished. If *n* lakes are fished, and 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 7 is calculated for each contributing facility and all resulting values are summed:

$$\left[\sum_{All\ Contributing\ Facilities}\left(SV_{T2RefFish,Lake}\right)\right],$$

which incorporates deposition from multiple source category facilities.

For lakes 1 to *n*−1, which total less than 373 acres:

$$SV_{T2RefFish,Lake} = SV_{T2Fish,Lake} \times \left(\frac{A_{Lake}}{373\ acres}\right) \qquad \text{Eq. 7}$$

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).

2. 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 influencing 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:

$$SV_{T2RefFish,Lake\ n} = SV_{T2Fish,Lake\ n} \times \left(\frac{373\ acres - \sum(A_{Lakes\ 1\ to\ n-1})}{373\ acres}\right) \quad \text{Eq. 8}$$

Finally, the cumulative Tier 2 SV for the refined fisher is calculated as:

$$SV_{T2RefFish,Total} = (Eq.8) + \sum(Eq.7) \quad \text{Eq. 9}$$

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 the modeling domain includes *n* total lakes to assess, and their total surface area is smaller than 373 acres, we use Equation 7 above to calculate the refined Tier 2 fisher SV for each lake (summing for other contributing source category facilities as necessary, see above). Equation 10, below, then calculates the final fisher SV for all fished lakes combined.

$$SV_{T2RefFish,Total} = \sum(Eq.7) \quad \text{Eq. 10}$$

or

$$SV_{T2RefFish,Total} = \left[\sum_{All\ Contributing\ Facilities} \left(SV_{T2RefFish,Lake\ 1\ to\ n}\right)\right]$$

3.2.2.4 Identifying the Location of the Largest Tier 2 Farmer-scenario Screening Value

For a facility screened in Tier 2, farmer-scenario SVs are calculated for each PB-HAP at each hypothetical farm location (at five distances in eight directions). To be health protective, we use the highest of these SVs as the final Tier 2 farmer-scenario SV for that PB-HAP.

3.2.3 Example of Tier 2 Multipathway Screen

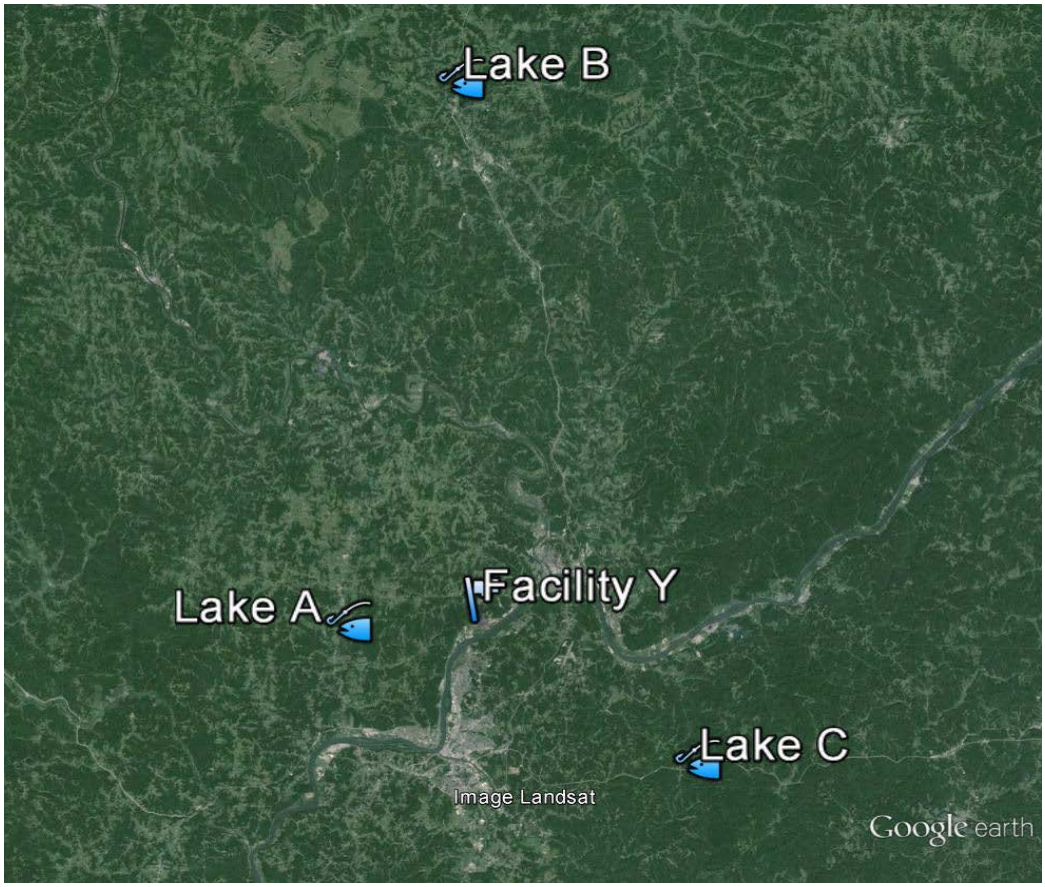
The Tier 1 screening results for Facility Y for divalent mercury and dioxins (see Section 3.1.3) indicated SVs larger than 1; thus some of the health-protective assumptions of the Tier 1 screen are replaced with more site-specific values in Tier 2.

For Facility Y, the selected meteorological station is 5 miles away (identified using the proximity matching step in the Tier 2 implementation tool; this station is the same station used in the RTR inhalation risk modeling). Three nearby lakes (see Figure 3-5) were identified for assessment (using the proximity and surface area assessments in the Tier 2 implementation tool). Based on lake size, these lakes cumulatively could sustain an annual fish ingestion rate of 373 g/day. Lake A is 16 km to the west (100 acres in area), Lake B is 42 km to the north (200 acres), and Lake C is 30 km to the southeast (120 acres). These distances respectively match the10-km, 40-km, and 20-km lake distances in the Tier 2 library. For this facility, the site-wide typical annual precipitation amount is 1,187 mm, and the site-wide typical mixing height is 710 m; wind speeds are 2.8 m/s in the directions of each lake and 4 m/s in all other directions (each of which does not contain a suitable lake). No other source category facility is nearby to provide additional chemical deposition to this area.

Equation 6 (see Section 3.2.2.1) is used to calculate the Tier 2 SV at each lake, prior to any adjustments based on lake area, and at each hypothetical farming location (i.e., at each of the five farm distances evaluated in the Tier 2 library, in each of eight directions relative to the facility).

Once the Tier 2 SV is calculated for each relevant lake and PB-HAP, refinements based on sustainable fish withdrawals (shown in Equations 7–9 in Section 3.2.2.3) are applied. Table 3-11 illustrates the steps of the Tier 2 screening process for Facility Y for the fisher scenario. Based on the Tier 2 SVs at each lake (i.e., values of $SV_{T2Fish,Lake}$), Lake A is fished first, Lake C is second, and Lake B is third. The $SV_{T2RefFish}$ calculations in Table 3-11 reflect 100 g-fish/day ingested from Lake A, 200 g-fish/day from Lake C, and 73 g-fish/day from Lake B, totaling the target 373 g-fish/d. The lakes are not influenced by another facility in the source category.

Figure 3-5. Location of the Assessed Facility (Facility Y) and Impacted Lakes



Once the Tier 2 SV (i.e., values of SV_{T2Farm}) is calculated at all hypothetical farm locations, the largest value of SV_{T2Farm} is selected. Table 3-12 illustrates the steps of the Tier 2 screening process for Facility Y for the farmer scenario. This table contains results only for the 0.5-km hypothetical farm distance, which is the distance with the highest SVs in this example assessment. For the fisher scenario, the Tier 2 mercury noncancer SVs at the individual lakes range between 2 and 4 with a total noncancer SV of 9 (summed across all target lakes). The Tier 2 cancer (i.e., dioxin) SV for the fisher scenario at the individual lakes ranges between 1 and 5 for individual dioxin congeners with a total cancer SV of 18 (summed across all dioxin congeners and all target lakes; rounding to 20 at 1 significant figure).

For the farmer scenario, the Tier 2 cancer SV is 7 (dioxin) based on the most highly contaminated hypothetical farm, which is located 0.5 km north of the facility. The Tier 2 noncancer SV for the farmer scenario is minimal at 0.05, with the most highly contaminated

hypothetical farm located 0.5 km north of the facility. Although each lake location is paired with an SV for the fisher scenario and then summed across all lakes, the farm SVs are not summed and, instead, the largest value is selected as the final Tier 2 SV for the farm scenario.

Table 3-11. Tier 2 Screening Results for Example Facility Y, for Fisher Only

PB-HAP Group	PB-HAP Chem.	Target Chem. for Equivalency	Lake	ER	REF_{T2}	Th_{T2}	$FreqWD_{T2}$	$RefMix_{T2}$	SV_{T2Fish}	A_{Lake}	$SV_{T2RefFish}$	Cumulative $SV_{T2RefFish}$ (Rounded)
Mercury	Divalent mercury	Divalent mercury	A	0.042	1	3.81E−04	8%	0.72	14.8	100	4.0	9 (9)
			B			3.12E−03	20%	0.89	5.6	200	3.0	
			C			9.88E−04	12%	0.81	9.6	120	1.9	
Dioxin	1,2,3,7,8-PentaCDD	2,3,7,8-TCDD	A	2.28E-07	2.43	2.23E−09	8%	0.39	18.0	100	4.8	18 (20)
			B		5.51	1.32E−08	20%	0.50	9.8	200	5.3	
			C		4.72	4.95E−09	12%	0.43	13.4	120	2.6	
	2,3,7,8-TCDD		A	3.00E-07	1	2.23E−09	8%	0.34	8.5	100	2.3	
			B			1.32E−08	20%	0.37	3.9	200	2.1	
			C			4.95E−09	12%	0.35	5.9	120	1.2	

PB-HAP = persistent and bioaccumulative hazardous air pollutant; chem. = chemical; ER = emission rate in tons per year; REF_{T2} = Tier 2 risk equivalency factor; Th_{T2} = Tier 2 screening threshold emission rate in tons per year; $FreqWD_{T2}$ = frequency of winds blowing toward the lake; $RefMix_{T2}$ = Tier 2 mixing height refinement factor; SV_{T2Fish} = Tier 2 fisher-scenario SV for the lake; A_{Lake} = area of the lake in acres; $SV_{T2RefFish}$ = Tier 2 refined fisher SV for the lake; PentaCDD = pentachlorodibenzo-p-dioxin; TCDD = tetrachlorodibenzo-p-dioxin

Table 3-12. Tier 2 Screening Results for Example Facility Y, for Farmer Only

PB-HAP Group	PB-HAP Chem.	Target Chem. for Equivalency	Octant[a]	ER	REF_{T2}	Th_{T2}	$FreqWD_{T2}$	$RefMix_{T2}$	SV_{T2Farm}	Cumulative SV_{T2Farm}[b] for Octant (Rounded)
Mercury	Divalent mercury	Divalent mercury	N	0.042	1	1.86E−01	20%	0.49	0.052	0.052 (0.05)
			NE			2.16E−01	20%	0.46	0.042	0.042 (0.04)
			E			2.16E−01	7%	0.46	0.015	0.015 (0.02)
			SE			1.86E−01	12%	0.49	0.031	0.031 (0.03)
			S			2.16E−01	8%	0.46	0.017	0.017 (0.02)
			SW			2.16E−01	10%	0.46	0.021	0.021 (0.02)
			W			1.86E−01	8%	0.49	0.021	0.021 (0.02)
			NW			2.16E−01	15%	0.46	0.031	0.031 (0.03)
Dioxin	1,2,3,7,8-PentaCDD	2,3,7,8-TCDD	N	2.28E−07	2.16	1.82E−08	20%	0.33	4.1	6.5 (7)
			NE		2.16	2.40E−08	20%	0.32	3.1	5.0 (5)
			E		2.16	2.40E−08	7%	0.32	1.1	1.8 (2)
			SE		2.16	1.82E−08	12%	0.33	2.5	4.0 (4)
			S		2.16	2.40E−08	8%	0.32	1.2	1.9 (2)
			SW		2.16	2.40E−08	10%	0.32	1.5	2.4 (2)
			W		2.16	1.82E−08	8%	0.33	1.7	2.7 (3)
			NW		2.16	2.40E−08	15%	0.32	2.3	3.7 (4)
	2,3,7,8-TCDD	2,3,7,8-TCDD	N	3.00E−07	1	1.82E−08	20%	0.32	2.4	6.5 (7)
			NE		1	2.40E−08	20%	0.32	1.9	5.0 (5)
			E		1	2.40E−08	7%	0.32	0.7	1.8 (2)
			SE		1	1.82E−08	12%	0.32	1.5	4.0 (4)
			S		1	2.40E−08	8%	0.32	0.7	1.9 (2)
			SW		1	2.40E−08	10%	0.32	0.9	2.4 (2)
			W		1	1.82E−08	8%	0.32	1.0	2.7 (3)
			NW		1	2.40E−08	15%	0.32	1.4	3.7 (4)

[a]The screen generates Tier 2 SVs for the farmer scenario at all five hypothetical facility-to-farm distances in all eight directions (totaling 40 SVs). In this example assessment, the 0.5-km distance contained the largest SVs, so only the 0.5-km results are shown in this table.

[b]In this example, for each octant, the cumulative SV for dioxins is the sum of the SV for 1,2,3,7,8-pentaCDD and the SV for 2,3,7,8-TCDD. The way the table is formatted, the values in this column are identical for the 1,2,3,7,8-pentaCDD rows and the 2,3,7,8-TCDD rows. The bolded values are the largest values per PB-HAP group—these are the final SV_{T2Farm} values.

PB-HAP = persistent and bioaccumulative hazardous air pollutant; chem. = chemical; ER = emission rate in tons per year; REF_{T2} = Tier 2 risk equivalency factor; Th_{T2} = Tier 2 screening threshold emission rate in tons per year; $FreqWD_{T2}$ = frequency of winds blowing toward the farm; $RefMix_{T2}$ = Tier 2 mixing height refinement factor; SV_{T2Farm} = Tier 2 farmer-scenario SV for the farm; PentaCDD = pentachlorodibenzo-p-dioxin; TCDD = tetrachlorodibenzo-p-dioxin

Because the Tier 2 cancer SVs were below 100 (i.e., a screening risk level of 100-in-a million) for both the farmer and fisher scenarios, we do not conduct a Tier 3 assessment for dioxins for this example. The Tier 2 mercury noncancer HQ SV for the fisher scenario, however, is 9 (i.e., representing an exceedance of an HQ of 1). Therefore, additional refinement is used to assess the potential for excess mercury hazard from fish ingestion (see Section 3.3).

3.3 Tier 3 Multipathway Screen

The Tier 3 multipathway screen can be conducted on facilities that still have potential for appreciable multipathway risk following the Tier 2 screen. The Tier 3 screening approach consists of three individual refinements (described in more detail below) based on additional site-specific data. These refinements are conducted in a step-wise fashion, and all three might not always be needed, as potential multipathway risk is evaluated at the end of each refinement. These refinements include further analysis of the affected lakes identified in the Tier 2 screen (Section 3.3.1), analysis of plume rise resulting in PB-HAPs lost to the upper atmosphere (Section 3.3.2), and the use of time-series meteorology and effective release heights (Section 3.3.3). An example Tier 3 application is provided in Section 3.3.4, followed by a summary of the example screen applications for Tiers 1–3 (Section 3.3.5).

3.3.1 Lake Evaluation

A Tier 3 lake evaluation is conducted when the Tier 2 SV for the fisher scenario indicates the potential for appreciable multipathway risk. During this evaluation, we examine (1) whether a given lake used in the Tier 2 screen truly exists; (2) the intended purpose of the lake (e.g., recreation); (3) lake accessibility; and (4) whether the lake is likely fishable. This evaluation is conducted because the USGS data set of lakes and reservoirs used in the Tier 2 multipathway screen does not contain information on lake accessibility and whether the lake is likely fishable. In addition, the data set 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. We use aerial and street-view imagery and internet searches to ascertain if an assessed lake actually exists and whether it can be fished. Lakes that appear swampy or covered in algae, or are used for industrial or waste disposal/treatment purposes, and/or adjacent/connected to a river or saltwater body (estuaries and rivers) are not ideal for the RTR assessment models and methods.

Based on the evaluation described above, we can remove any lakes that are unsuitable for multipathway screening. For example, the area outlined in blue in the first (top) aerial image in Figure 3-6 identifies a lake from the USGS data set that originally qualified for Tier 2 screening based on the information provided in that data set. Aerial imagery, however, shows the lake is adjacent to an industrial facility and likely used only for on-site industrial purposes. The area outlined in blue in the second (bottom) aerial image in Figure 3-6 identifies an area the USGS data set originally identified as a lake. Aerial imagery (current and historical), however, shows the lake is mostly or entirely dried up and not suitable for fishing. At this point in the human health assessment, both lakes would be removed from consideration and the Tier 2 screen would be rerun. Waterbodies that need more investigation are evaluated by conducting geographic information system (GIS) searches and web searches to determine if the target waterbodies are open to the public and if they have any fishing history before they are removed from the USGS data set.

For facilities undergoing Tier 3 screening, we assess all lakes from which the fisher catches and consumes fish according to the Tier 2 fisher methods discussed in Section 3.2.2.3. We permanently remove all lakes that we determine do not qualify for screening (based on the criteria discussed in the previous paragraphs) from the human health screen and no longer use them in any human health screen. If we remove a lake from the Tier 3 screen, the hypothetical fisher might need to catch and consume fish from new or additional lake(s) to fish a total of 373 acres. We assess these new or additional lakes using the methods discussed in this section. After all lakes that are fished in the scenario (for the facilities not screening out in Tier 2) have been evaluated, we rerun the Tier 2 screen with the revised lake database, producing revised SVs.

Although we conduct this additional lake assessment based on the Tier 2 screening results of a particular set of facilities, lakes removed during this assessment could affect the screening results of other facilities in the source category beyond that original facility set. For example, removing a lake could affect the screening results of both facilities if an assessed facility is within 100 km of another assessed facility. For this reason, we rerun the Tier 2 screen with the revised lake data set, for all facilities in the source category, including lakes influenced by multiple source category facilities. This lake assessment does not affect SVs for the farmer scenario.

Figure 3-6. Examples of Lakes Removed from Screen During Tier 3 Lake Assessment

3.3.2 Plume-rise Assessment

A Tier 3 plume-rise assessment is conducted when the Tier 2 farmer-scenario SVs or the Tier 3 lake-assessment fisher-scenario SVs, or both, indicate the potential for appreciable multipathway risk. The Tier 3 plume-rise assessment estimates the amount of emitted PB-HAP that remains in the mixing layer after considering plume rise. In Tiers 1 and 2, we model all PB-HAPs as though they are emitted inside the mixing layer and are available for ground-level exposure. In reality, the physical height of an emission source, in combination with ambient conditions and the temperature and velocity of the chemical plume as it leaves the source, can cause some of the chemical plume to extend above the mixing layer. In TRIM.FaTE modeling, the chemical mass ejected 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). Many emission sources in RTR source categories are physically taller than the mixing height during some hours, and hot exit gas temperatures (i.e., buoyancy) or high exit gas velocities (i.e., momentum), or both, can further elevate the chemical plume well above the source height and mixing height.

The Tier 3 plume-rise assessment uses methods summarized by Seinfeld and Pandis (1998) to estimate how often a facility's emissions reach the upper-air sink. The methods to estimate the amount of chemical lost to the upper-air sink require

- Hourly meteorological data (e.g., air temperature, wind speed)
- 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, exit gas temperature and velocity)
- An estimate of facility size (to estimate the plume height at the estimated edge of the facility).

We use EPA guidance (U.S. EPA 2000) to calculate the wind speed at the stack height. With these data and the assumed average vertical gradients of temperature and potential temperature corresponding to the stability class, we use equations reproduced in Seinfeld and Pandis (1998) to calculate plume rise.

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 blow toward the lake or farm 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 refinement factor. We then multiply this factor by the SV for the Tier 2 farmer scenario or the Tier 3 lake-assessment fisher scenario. That is:

$$SV_{T3PR} = SV_X \times \left[\frac{Hrs(W \text{ and } E<M)}{Hrs(W)}\right], \quad \text{Eq. 11}$$

where:

SV_{T3PR}	=	SV for the Tier 3 plume-rise assessment
SV_X	=	SV for either the Tier 3 fisher scenario from the Tier 3 lake assessment or the Tier 2 farmer scenario
Hrs(W and E<M)	=	number of hours when winds are blowing toward the lake or farm and 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 of interest.

If lakes are in the modeling domain, after the SVs at each lake are adjusted based on plume rise, we reapply the refined fisher SV calculation discussed in Section 3.2.2.3. The order in which lakes are fished, however, does not change.

For example, two lakes are being assessed, having surface areas of 273 and 100 acres and having Tier 3 lake-assessment SVs for mercury of 5 and 3, respectively, before the refined-fisher calculations; after refined-fisher calculations, the site-wide fisher-scenario SV for mercury is [5 × (273/373)] + [3 × (100/373)] = 4.5 (rounds to 5). 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 for 1,000 hours; its plume-rise refinement factor is 1,000/1,800 = 0.56. 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 refinement factor is 400/500 = 0.8. The Tier 3 plume-rise fisher-scenario SVs for mercury are

$5 \times 0.56 = 2.8$ and $3 \times 0.8 = 2.4$. With the refined-fisher calculations, the site-wide fisher-scenario SV for mercury is $[2.8 \times (273/373)] + [2.4 \times (100/373)] = 2.7$ (rounds to 3).

3.3.3 Assessment of Time-series Meteorology and Effective Release Heights

If a Tier 3 plume-rise refinement indicates that appreciable multipathway risks remain, we conduct a Tier 3 time-series meteorology refinement. This refinement uses hourly, site-specific meteorological and plume-rise values in new TRIM.FaTE modeling runs to refine potential multipathway risk.[17] The use of time-series meteorology data, which captures hour-by-hour changes in each of the assessed meteorological parameters (as opposed to constant average values as used in the Tier 1 and Tier 2 screens), increases the accuracy of the assessment by accounting for potential statistical interactions between the meteorological parameters.

For a facility undergoing a Tier 3 time-series assessment, we use the facility's emissions; a time series of hourly effective release heights; a time series of hourly meteorology data (i.e., winds, mixing height, temperature, precipitation); and the Tier 2 spatial scenario that best matches each lake fished by the hypothetical fisher (or the relevant farm location if farm exposure is the concern). We use these site-specific data to model PB-HAP fate and transport using TRIM.FaTE and then estimate exposure and risk using MIRC, which provides a screening-level risk or HQ value for each lake and farm. Notably, if multiple lakes are fished to achieve the desired fish ingestion rate, the percentage of daily-ingested fish caught at each lake would be multiplied by the screening level risk or HQ value for that lake. Those products then would be summed across all lakes (i.e., the refined-fisher calculations discussed in Section 3.2.2.3 would be applied to the modeling results).

3.3.4 Example of a Tier 3 Multipathway Screen

As discussed in Section 3.2.3, for hypothetical Facility Y, the Tier 2 screen suggests appreciable multipathway risk from divalent mercury. As a result, in a stepwise fashion, we apply the three Tier 3 multipathway refinements described above.

[17] As discussed in Section 3.2, the Tier 2 multipathway screening results are based on typical meteorological conditions prevailing at the assessed facility. This is in contrast to the modeling with TRIM.FaTE for the Tier 3 multipathway screen, which incorporates hour-by-hour site-specific meteorological and plume-rise data into new modeling runs.

We start with the Tier 3 lake evaluation. Visual inspection of aerial photographs indicates that all three lakes evaluated in Tier 2 are likely fishable and potentially have public access. Therefore, the first step of the Tier 3 screen for Facility Y does not change the results of the Tier 2 screen.

We next evaluate plume rise for Tier 3 using hourly data from the same meteorological station used in the Tier 2 multipathway screen, along with the effective release height for Facility Y. As shown in Table 3-13, although winds blew toward Lake A (i.e., toward the west) 8 percent of the time, the mixing heights during those times were frequently low enough that the buoyancy and momentum of the chemical plume ejected it above the mixing layer. That is, the chemical plume extended above the mixing layer 43 percent of the approximately 700 hours when winds were blowing toward Lake A (i.e., approximately 300 hours of the year). The resulting Tier 3 plume-rise refinement factor for Lake A, as used in the screening tool, is $1 - 0.43 = 0.57$, which is the fraction of divalent mercury emitted during these 700 hours that remains in the mixing layer and is available for deposition to Lake A. Thus, the previously calculated SV for fish ingested from Lake A is reduced by 43 percent—from 4.0 to 2.3. For the other two lakes, the Tier 3 changes to plume rise decrease SVs less than at Lake A. The SV at Lake B (where 26 percent of emitted chemical was lost to the upper-air sink) decreases from 3.0 to 2.2. The SV at Lake C (where 20 percent of emitted chemical was lost to the upper-air sink) decreases from 1.9 to 1.5. Because multiple lakes are needed to accumulate 373 g/day of fish ingestion, the SVs are summed, as described earlier in Section 3.2.2.3. The cumulative lake SV decreases from 9 to 6.

Table 3-13. Screening Values from the Tier 3 Plume-rise Assessment for the Fisher Scenario for Emissions of Divalent Mercury from Facility Y

Target Lakes in Fishing Order	Frequency of Winds Blowing Toward Lake	Frequency of Plume Elevated Above Mixing Layer	Frequency of Plume Remaining in Mixing Layer (i.e., the refinement factor to multiply previous SV by)	SV[a]
A (100 ac; 16 km W of plant)	8%	43%	57%	2.3
C (120 ac; 30 km SE of plant)	12%	20%	80%	2.2
B (200 ac; 42 km N of plant)	20%	26%	74%	1.5
			Total Noncancer SV	6

[a]The SVs shown for the fish scenario have been scaled based on the ingestion fraction, which is based on lake surface area.
ac = acres; km = kilometers; W = west; SE = southeast; N = north

Based on those results, divalent mercury emissions from Facility Y do not screen out. Therefore, a Tier 3 time-series assessment is conducted.

Whereas the Tier 3 lake and plume-rise assessments discussed above did not require new modeling, new TRIM.FaTE runs are needed for the Tier 3 time-series assessment using the facility's emissions of divalent mercury and the same hourly meteorological and hourly plume-rise values used in the plume-rise assessment above. The runs use the spatial layout appropriate for the lake (i.e., the "lake at 10 km" scenario for Lake A), and final processing with MIRC yields screening-level HQs for divalent mercury. As shown in Table 3-14, the Lake A SV dropped slightly (from 2.3 to 1.9), while the SVs at Lakes B and C decreased from 1.5 to 1.1 and from 2.2 to 1.4, respectively. The cumulative SV decreased from 6 to 4. Based on these results, additional refinement (i.e., a site-specific assessment) could be conducted to assess effects from mercury emissions from this facility. That site-specific assessment is not discussed in this report, but its results are incorporated into the discussion in Section 3.3.5 below. The SAB has reviewed TRIM site-specific analysis in the past.

Table 3-14. Screening Values from the Tier 3 Time-series Meteorology Assessment for the Fisher Scenario for Emissions of Divalent Mercury from Facility Y

Target Lakes in Fishing Order	SV[a]
A (100 ac; 16 km W of plant)	1.9
C (120 ac; 30 km SE of plant)	1.4
B (200 ac; 42 km N of plant)	1.1
Total Noncancer SV	4

[a]The SVs shown for the fish scenario have been scaled based on the ingestion fraction, which is based on lake surface area.
SV = screening value; ac = acres; km = kilometers; W = west; SE = southeast; N = north

3.3.5 Discussion of Results for Multipathway Screen

As discussed above and illustrated in the summary tables below (Table 3-15 for dioxins and Table 3-16 for mercury), refining the input data used in the multipathway assessments (i.e., replacing health-protective assumptions with site-specific values) leads to decreasing estimates of potential health risks, as expected.

For dioxins, as shown in Table 3-15, the SV in the most health-protective assessment, the Tier 1 screen, was 2,000 (see the first column). In Tier 2, better accounting for local meteorology and lake locations led to an SV of 20 for fish ingestion (no more than 7 at each lake; see the Tier 2 columns for the fish scenario, "SV Itemized" and "SV Total"). In Tier 2, the farm-scenario SV

was 7 (see the final column). These Tier 2 SVs were below acceptable levels, so the dioxin emissions were not assessed further.

For mercury emitted by Facility Y, as shown in Table 3-16, the Tier 1 SV was 300 (see the first column) indicating that the emissions exceeded the Tier 1 threshold emission rate by a factor of 300. When meteorology data and lake locations were refined in Tier 2 to better represent the assessed site, the mercury SV was 4 or less at each lake, totaling 9 across the three lakes (see the Tier 2 "SV Itemized" and "SV Total" columns, which show the SV at each lake and then at all three lakes cumulatively). The farm-scenario Tier 2 SV for mercury was 0.05, which is far below the level of concern (not shown in Table 3-16). Accounting for plume-rise effects decreased the mercury SV to 2 or less at each lake, totaling 6 across the three lakes (see the "Tier 3 Plume Rise" columns of the table, showing the plume-rise adjustment factors applied to the Tier 2 SVs and the resultant new SVs, both itemized and total). Further accounting for the effects of using hourly meteorology again produced mercury SVs of 2 or less at each lake, and the total across the three lakes decreased to 4 (see the "Tier 3 Time-series Meteorology" columns of the table); the SV for each lake did not exceed 2.

After the most refined screen (i.e., the Tier 3 time-series-meteorology assessment), SVs for mercury still exceeded the level of concern, so the most-refined assessment (i.e., the site-specific assessment, not presented in this report) was conducted. The site-specific analysis indicated that mercury HQs did not exceed the level of concern at any lake (see the final column of Table 3-13), totaling 0.6 across all three lakes. 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.

As discussed here, the screening-assessment process (with relatively small effort) determined that dioxin emissions from Facility Y were below the level of concern and that no further, or more complex, assessments were necessary beyond the Tier 2 assessment. For mercury, although each refinement step in the screening process notably lowered the SVs (i.e., from 300 in the Tier 1 assessment to 4 in Tier 3 time-series-meteorology assessment), SVs remained above the level of concern, necessitating a complex site-specific assessment that resulted in HQs below the level of concern (i.e., 0.6).

Table 3-15. Screening Values for Ingestion of Fish and Farm-food-chain Products Exposed to Dioxin Emissions from Facility Y

Tier 1	Tier 2				
	Fish Scenario[a]				Farm Scenario[b]
SV (Fish + Farm)	Target Lakes in Fishing Order	Ingestion Fraction	SV (Itemized)	SV (Total)	SV (Largest of All Hypothetical Farms)
2,000	A (100 ac; 16 km W of plant)	0.27	7.1	20	7
	C (120 ac; 30 km SE of plant)	0.32	3.8		
	B (200 ac; 42 km N of plant)	0.41	7.4		

All SVs are rounded to one significant figure.
SV = screening value; ac = acres; W = west; SE = southeast; N = north
[a]The SVs shown for the fisher scenario have been scaled based on the ingestion fraction, which is based on lake surface area and fishing order.
[b]For the farm scenario, the largest single farm value is used as the final Tier 2 SV. The farm-scenario SVs range from 2 to 7 across all hypothetical farm locations (not shown here), and the largest value (7) is the selected value shown here.

Table 3-16. Screening Values and Site-specific Hazard Quotients for Ingestion of Fish Exposed to Mercury Emissions from Facility Y

Tier 1	Tier 2, Tier 3, and Full Site-specific[a,b]										
	Lake Information		Tier 2		Tier 3 Plume Rise			Tier 3 Time-series Meteorology		Full, Site-specific	
SV (Fish + Farm)	Target Lakes in Fishing Order	Fish Ingestion Fraction	SV (Itemized)	SV (Total)	Refinement Factor	SV (Itemized)	SV (Total)	SV (Itemized)	SV (Total)	HQ (Itemized)	HQ (total)
300	A (100 ac; 16 km W of plant)	0.27	4.0	9	0.57	2.3	6	1.9	4	0.05	0.6
	C (120 ac; 30 km SE of plant)	0.32	1.9		0.80	2.2		1.4		0.004	
	B (200 ac; 42 km N of plant)	0.41	3.0		0.74	1.5		1.1		0.5	

Total screening values (SVs) and hazard quotients (HQs) are rounded to one significant figure.
ac = acres; W = west; SE = southeast; N = north
[a]The SVs shown for the fish scenario have been scaled based on the ingestion fraction, which is based on lake surface area.
[b]Farm-scenario SVs for mercury are not shown here for the more-refined assessments because they were orders of magnitude below acceptable levels. The farm-scenario SVs are shown for Tier 2 in Section 3.2.3.

3.4 Proposed Addition of a Gardener Exposure Scenario to RTR Screens

EPA is considering expanding its multipathway screening capabilities by adding a gardening exposure scenario to the Tier 2 and Tier 3 multipathway screens. This scenario could be applied in locations where we know at least some home-grown produce ingestion risk is likely, but the presence of a subsistence farm is either unlikely (e.g., in urban areas) or difficult to confirm based on the characterization of land use surrounding a facility. Moreover, an individual's exposure to PB-HAP emissions from a facility through ingestion of home-garden or community-garden produce is a realistic possibility for almost any land use encountered around facilities assessed in the RTR program. In addition, we are proposing to refine the gardener scenario further by characterizing an individual as an urban gardener or rural gardener, thus recognizing that individuals who live in rural areas tend to have more land available to grow fruits and vegetables and therefore likely have higher ingestion rates of these goods (see Section 3.4.2).

3.4.1 Ingested Media for the Gardener

If implemented, a gardener scenario would comprise the exposure pathways through which individuals might be exposed in an urban or nonfarm rural setting. Notably, the gardener exposure scenario is analogous to the "Resident" exposure scenario provided in *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (HHRAP) (U.S. EPA 2005b).

Similar to the Resident scenario in HHRAP, we propose that the gardener ingests a subset of the media that a subsistence farmer would ingest, namely:

- Soil
- Exposed fruits and vegetables
- Protected fruits and vegetables
- Root vegetables
- Breastmilk (as an infant).

Table 3-17 compares the ingested media that would be considered in the gardener scenario to those of the farmer scenario.

Table 3-17. Ingested Media for Farmer and Gardener Scenarios

Scenario	Soil	Protected Fruit	Exposed Fruit	Protected Vegetable	Exposed Vegetable	Root Vegetable	Dairy	Beef	Pork	Chicken	Eggs	Breast Milk
Farmer	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Gardener	✓	✓	✓	✓	✓	✓						✓

3.4.2 Ingestion Rates for a Rural versus Urban Gardener

For the RTR inhalation risk assessment, receptor locations are designated as rural or urban using a method described in Section 5.1. Using designations determined for the inhalation assessment, we propose the gardener could be classified as either urban or rural. For a gardener in a rural environment, we would use the same ingestion rates (IRs) as used for the farmer for those media the gardener ingests. As discussed above, this is a subset of the media a farmer would ingest (see Table 3-17). A reasonable assumption is that a gardener in a rural setting would be more likely to have sufficient land to support a sustainable garden large enough to provide for the assumed 90th-percentile IRs and would have an increased tendency to consume larger amounts of home-produced foods than would gardeners in urban settings. Gardeners in an urban setting likely would grow produce on smaller plots and, in general, would be less likely to consume homegrown produce at farmer or rural-gardener rates. We therefore believe using a central-tendency value from the *Exposure Factors Handbook: 2011 Edition* (EFH) IR distributions for gardeners in urban settings would be reasonable—specifically, the mean value, which is larger (i.e., more health protective) than the median value. Soil ingestion for the urban and rural gardeners would be at the same high-end rate as for the farmer, and the farmer and gardener (both rural and urban) would have increased soil IRs compared with the general population. We base this premise on our belief that using a central-tendency soil IR could underestimate soil ingestion for gardeners.

The age-specific food ingestion rates (IRs) for the farmer scenario currently used in the RTR screening scenario were obtained from Chapter 13 of the EFH (Tables 13-40, 13-58, 13-59, 13-

60, 13-61, and 13-62) (U.S. EPA 2011a). The farmer scenario uses 90th-percentile IRs for all farm-food-chain products, considered high-end, health-protective values.[18]

The consumer-only IR values for fruits and vegetables from the EFH are not specific to a farmer; rather, they represent *both* gardener and farmer respondents. We propose the data source used for the farmer be used for the gardener. We propose to use different IRs from the EFH IR distribution, however, depending on whether the modeled gardener is located in a rural setting or an urban setting.

Table 3-18 provides the IRs for the farmer scenario and for the proposed gardener scenarios, while Table 3-19 provides the ratio of the proposed urban gardener IRs to IRs for the farmer and rural gardener. As can be seen, the IRs for the urban gardener generally are between one-third to one-half those for the farmer and rural gardener.

Selection of IRs does not affect TRIM.FaTE modeling because IRs are not used in TRIM.FaTE, which only produces concentrations in air, soil, water, sediment, and aquatic biota. MIRC uses TRIM.FaTE outputs and biotransfer factors to calculate uptake of chemicals into the farm food chain, adjusts for cooking and preparation losses, and uses IRs and other exposure-related parameters to calculate ingested dose.

As a health-protective setting, the gardener scenarios in the screens would use the same assumptions regarding transfer of chemical from adjacent modeling areas through runoff and erosion mechanisms as the farming scenario uses. In a site-specific assessment, this approach could be refined to assume that a gardener (perhaps especially one in an urban setting) would use a raised garden bed or garden box that does not receive chemical input through these mechanisms.

[18]Worthwhile to note is that using 90th-percentile IRs for all food categories might exceed the total food IRs expected for the general population.

Table 3-18. Summary of Age Group-Specific Ingestion Rates for Farm Food Items

Product	Child (age in years)				Adult (20–70 years)
	1–2	3–5	6–11	12–19	
Mean ingestion rates (g/kg-day)					
Exposed Fruit	6.14	2.60	2.52	1.33	1.19
Exposed Vegetable	3.48	1.74	1.39	1.07	1.38
Protected Fruit	16.6	12.4	8.50	2.96	5.19
Protected Vegetable	2.46	1.30	1.10	0.78	0.86
Root Vegetable	2.52	1.28	1.32	0.94	1.03
Soil (mg/day)[a]	50.0	50.0	50.0	50.0	20.0
90th percentile ingestion rates (g/kg-day)					
Exposed Fruit	12.7	5.41	6.98	3.41	2.37
Exposed Vegetable	10.7	3.47	3.22	2.35	3.09
Protected Fruit	44.8	32.0	23.3	7.44	15.1
Protected Vegetable	3.88	2.51	2.14	1.85	1.81
Root Vegetable	7.25	4.26	3.83	2.26	2.49
Soil (mg/day)	200.0[b]	200.0[b]	201.0[c]	201.0[c]	201.0[c]

Primary source was the 1987–1988 Nationwide Food Consumption Survey; compiled results are presented in Chapter 13 of 2011 Exposure Factors Handbook (EPA, 2011a). When data were unavailable for a particular age group, ingestion rate for all age groups was used, multiplied by the age-specific ratio of intake based on national population intake rates from Continuing Survey of Food Intakes by Individuals. Children less than 1 year old assumed only to ingest breast milk. g = grams; kg = kilograms.

[a]For mean 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]Recommended "upper percentile" value for children from EPA's 2011 EFH, Chapter 4, Table 4-23.

[c]Values are 90th percentile adult ingestion rates calculated in Stanek et al. 1997; used to represent older children and adults.

Table 3-19. Ratio of Proposed Urban Gardener IRs to Farmer/Rural Gardener IRs

Product	Mean ÷ 90th-percentile Ingestion Rate					
	Age Group (years)					Range
	1–2	3–5	6–11	12–19	20–70	
Exposed Fruit	0.48	0.48	0.36	0.39	0.50	0.36–0.50
Exposed Vegetable	0.33	0.50	0.43	0.46	0.45	0.33–0.50
Protected Fruit	0.37	0.39	0.37	0.40	0.34	0.34–0.40
Protected Vegetable	0.63	0.52	0.51	0.42	0.48	0.42–0.63
Root Vegetable	0.31	0.30	0.35	0.42	0.41	0.30–0.42
Soil[a]	1.00	1.00	1.00	1.00	1.00	1.00

[a]The soil ingestion rates used for the Urban Gardener ingestion scenario are reflective of the 90th percentile ingestion rate to be consistent with high-end soil ingestion.

4. ENVIRONMENTAL RISK SCREEN

4.1 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 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 to provide a systematic, scientifically defensible, and efficient approach that EPA can use to screen for potential adverse environmental effects associated with emissions of HAPs from facilities in 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 multipathway modeling and 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. Some aspects of the screen design presented in the human multipathway discussion (Section 3) are repeated in this environmental risk screen discussion for continuity and readability.

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 OAQPS Workshop on Ecological Risk Assessment of Air Toxics, June 2006 (ICF 2006).

The RTR ecological risk assessment methodology, reviewed by the SAB in 2009, was presented in two case studies—one for Petroleum Refining and one for Portland Cement Manufacturing. In these case studies, an ecological risk screen was conducted by comparing HAP concentrations (in media for PB-HAPs, in ambient air for non PB-HAPs) to human health thresholds. The assumption was that, if the human health thresholds were not exceeded, there was no potential for adverse environmental effects. For those HAPs exceeding human health thresholds, a refined ecological risk assessment was conducted that compared HAP concentrations to ecological benchmarks. These refined ecological risk assessments were conducted for dioxin, mercury, and hydrogen chloride (HCl) emissions for the Portland Cement case study, and for POM emissions for the Petroleum Refining case study. In addition, a proximity and vulnerability screen was conducted for Portland Cement facilities that investigated the indirect effects of HCl deposition on ecologically sensitive environments. In this analysis, Portland Cement facilities were ranked according to emission rates, the pH of regional rainfall, surface water alkalinity, and proximity to sensitive environments (e.g., wildlife refuges, state and federal parks).

The revised environmental risk screen presented in this document builds on and enhances the methods the SAB reviewed in 2009. For example:

- We no longer use human health thresholds to screen for environmental health. In the previous review, the SAB stated, "the assumption that ecological receptors will be protected if human health is protected is incorrect." In the revised environmental risk screen, we compare modeled environmental concentrations to ecological benchmarks for all pollutants included in the screen.
- As the SAB recommended, we have systematically evaluated and ranked the environmental HAPs for potential inclusion in the screen.

- We have expanded the environmental risk screen to include the following additional environmental HAPs: cadmium, hydrogen fluoride, lead, arsenic, and additional POMs.
- We have expanded the number of ecological endpoints and effect levels that we evaluate.
- We conducted a comprehensive literature review to identify the most up-to-date ecological benchmarks.
- The soil and water acidification screen that was conducted for HCl emissions from Portland Cement facilities was not retained in the current environmental risk screen for the following reasons:
  - A screen that investigates pH data for lakes and soils near facilities could indicate where acidification is not an issue. If lakes and soils near a facility were found to be acidic, attributing that acidification to any specific facility, as would be required for an RTR assessment, would be difficult.
  - The TRIM.FaTE multipathway model is not parameterized for HCl nor for modeling changes in soil or lake pH.
  - The identification of soil and water pH data around each facility would be time-consuming and such data are not readily available in all areas. For example, in the Portland Cement assessment, soil pH data were either unavailable or inconclusive for three of the four facilities investigated.

Below, we summarize the design and key features of the environmental risk screen. In Section 4.2, we present the environmental risk screen conceptual model, the HAPs included in the screen, and the endpoints for which environmental risk are screened. Section 4.3 presents the approach used to identify ecological benchmarks for each HAP for each assessment endpoint. Section 4.4 describes the methods used to estimate HAP exposures in the environment. This section also describes how we used the benchmarks identified in Section 4.3 to calculate "screening threshold emission rates" and how we compared those thresholds to exposure estimates to screen for adverse environmental effects. Section 4.5 presents the outputs and analyses of the risk screening results for an example source category.

4.2 Key Components of the Environmental Risk Screen

4.2.1 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 4-1) 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 4-1:

- Persistence and bioaccumulation (PB-HAP) potential
- Inclusion in the TRIM.FaTE multipathway model
- Magnitude of emissions
- Relative environmental toxicity – based on toxicity to wildlife, soil communities, and aquatic biota.

The far right column of Table 4-1 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 (PB-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 4-1 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.

Table 4-1. 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	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.

Table 4-1. 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]	
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

[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 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 (2005e) "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 (2006b). Region 3 Biological Technical Assistance Group (BTAG) Freshwater Sediment Screening Benchmarks.

4.2.2 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 4-2. 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 4-2, we identified ecotoxicity benchmarks as discussed in Section 4.3.

Table 4-2. 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 4-2 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 4-2 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 4-2 represent community-based GEAEs. Specifically, these community endpoints include the following communities: soil invertebrate, terrestrial plant, benthic, and aquatic.

4.2.3 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.

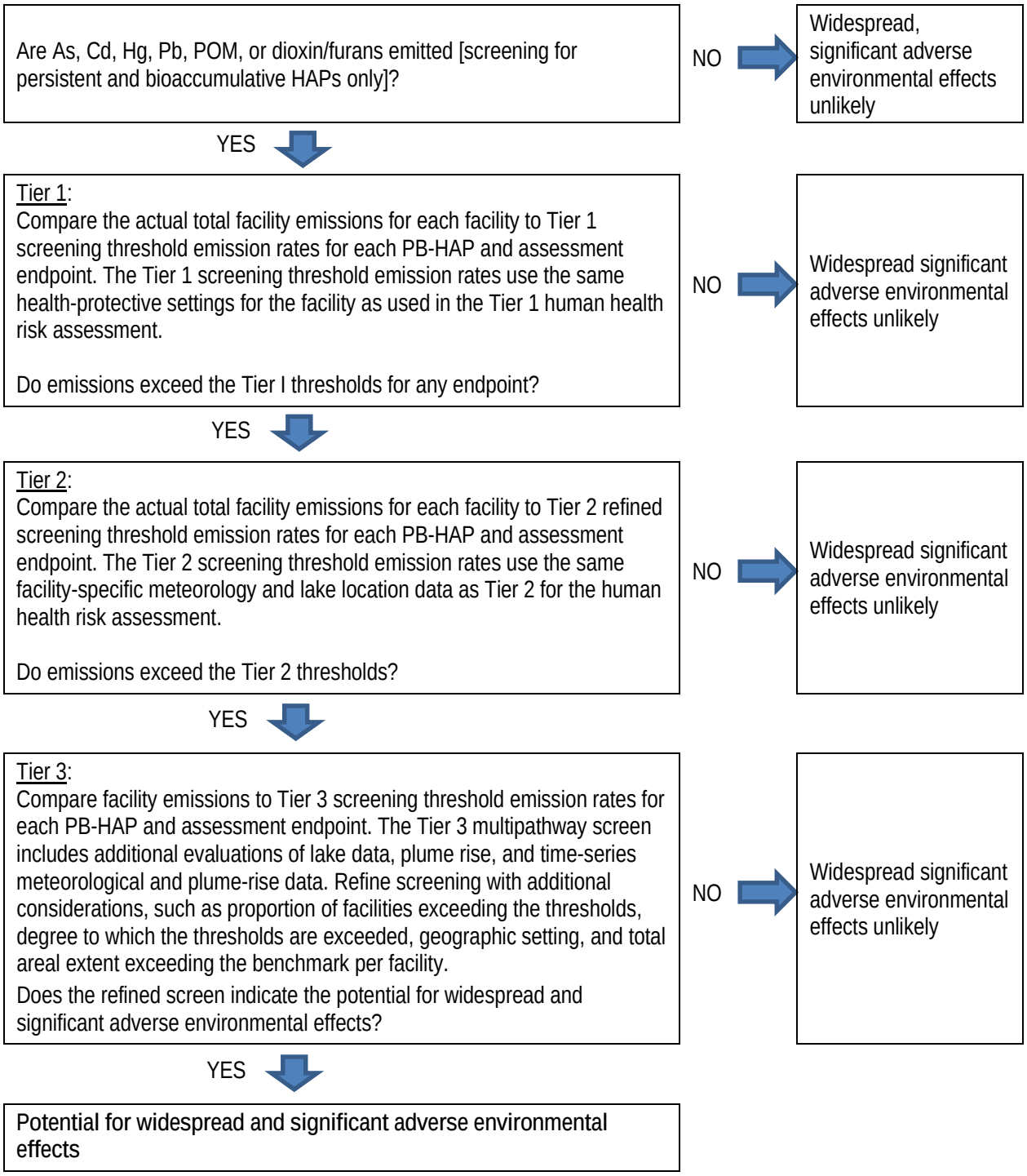
An overview of the environmental risk screen for each group is provided below.

PB-HAPs

For the five PB-HAPs—arsenic, cadmium, mercury, POM, and dioxins/furans—the environmental risk screen consists of three tiers (Figure 4-1). The tiered design used for the environmental risk screen is the same as that used for the human multipathway screen described in Section 3. 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 4.2.2).

Figure 4-1. Overview of the Environmental Risk Screen for PB-HAPs



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 4.4 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.[19]

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. If emissions from a facility exceed any of the Tier 1 screening threshold emission rates, the facility could be further evaluated in Tier 2.

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 3.3). If

[19]See Section 4.3, Effects Assessment, for discussion of the ecological benchmarks and wildlife toxicity reference values used for all three tiers of the environmental assessment.

emissions from a facility do not exceed the Tier 3 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 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.

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.[20] We consider values below the level of the secondary lead NAAQS to be unlikely to cause adverse environmental effects.[21]

Acid Gases

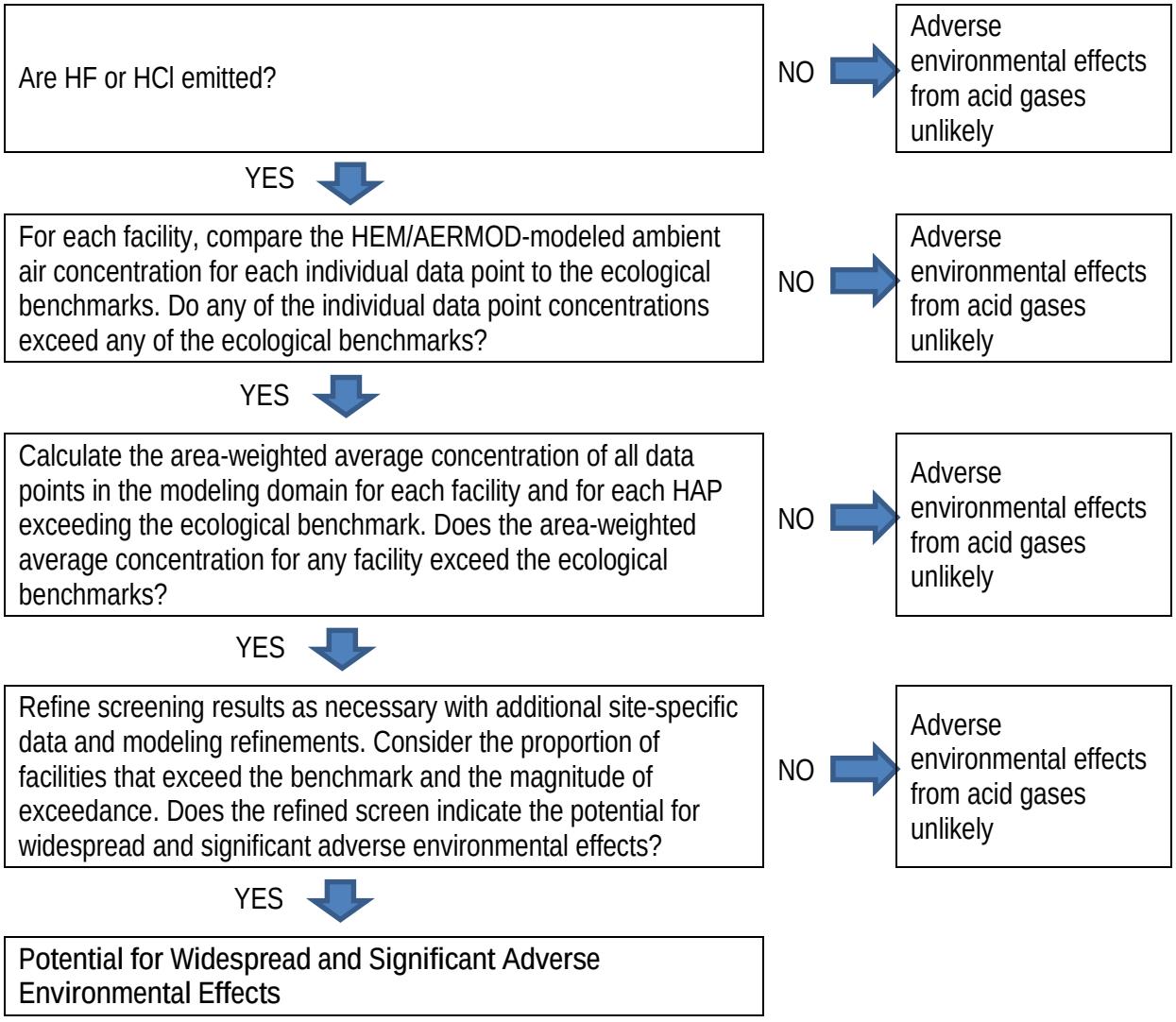
For HF and HCl, the environmental risk screen evaluates potential phytotoxicity (i.e., poisonous to plants) and reduced productivity of plants due to chronic exposure. For each acid gas, the environmental risk screen compares the HEM/AERMOD-modeled ambient air concentrations in

[20] 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."

[21] On October 18, 2016, EPA issued its final decision to retain the 2008 NAAQS for lead (U.S. EPA 2016a).

the modeling domain around each facility to ecological benchmarks (Figure 4-2). 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.

Figure 4-2. Overview of the Environmental Risk Screen for Acid Gases



4.3 Effects Assessment

To assess effects, we identified appropriate ecological benchmarks to compare to exposure concentrations. As indicated in Section 4.2.2, we searched for available ecological benchmarks for each assessment endpoint listed in Table 4-2. 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. Toxicity reference values (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.[22]
- **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[23] that assessed growth, reproduction, or survival species from the same taxonomic class to calculate assessment population benchmarks that represent NELs.[24]

[22]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.

[23]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.

[24]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.

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 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 4.3.1.2 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 4.3.1 – Benchmarks for PB-HAPs and Section 4.3.2 – Benchmarks for Acid Gases. Appendix B contains additional discussion about the ecological benchmarks, wildlife toxicity reference values, and TEFs. Appendix B also includes additional tables and citations to those presented in this section.

4.3.1 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 toxic

equivalency factors (TEFs) for the remaining chemicals in each category relative to the appropriate index chemical.

4.3.1.1 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 4-2 (Section 4.2.2), focusing on PB-HAPs. We also briefly reiterate the three generic effect levels for which we sought benchmarks.

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 (RCRA) 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.[25]

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).

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 4-2). 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

[25]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 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).

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 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 (OW) 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 (µg) per liter (L) of water.

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 4-2, 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 (Table 4-2, Endpoint 8) are piscivores. Both river otters and mink commonly are assessed for risks from persistent and bioaccumulative (PB) chemicals (e.g., DDT, DDE, PCBs, 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 TL3 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.

4.3.1.2 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 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.

Soil Ecotoxicity Benchmarks (SEB)

For soils, EPA's national Superfund Program Eco-Soil Screening Levels (Eco-SSLs, U.S. EPA 2005c) 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 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.

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 (SQC) 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 (OW), 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]).

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.

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 4.3.1.1). 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 Appendix B, Section B.4.

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 2011c). The derivation of TRVs for PB-HAPs for piscivorous wildlife are presented in Appendix B, Section B.4.

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 (Appendix B, Section B.3.2.4). 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 2011c).

4.3.1.3 Selected PB-HAP Benchmarks

Table 4-3 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 Appendix B.

Table 4-3. 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
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 2005g, 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 2005f, 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, 2006b (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

Eco-HAP	Assessment Endpoint	Benchmark Effects Level	Benchmark Value	Benchmark Source
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 2015a, Region 4
Mercuric chloride	Water-column community	Threshold level (chronic)	0.77 (µg/L)	U.S. EPA 1993c, 1995a, 2015a, 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)	
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 2015a, 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 2015a, Region 4
Mercury (methyl)	Water-column community	Threshold level (chronic)	0.0028 (µg/L)	U.S. EPA 2015a, 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, 2006b
		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; 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 (2015a) 0.13 mg/kg; U.S. EPA (2006b) 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.

4.3.2 Benchmarks for Acid Gases

4.3.2.1 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 Appendix B, Section B.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.

4.3.2.2 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).

Appendix B, Section B.2.3 contains the following background sections.

- Appendix B.2.3.1: Methods for Establishing HF Benchmarks – presents three bases that can be used to establish HF regulatory standards.
- Appendix B.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.
- Appendix B.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 (Appendix B, Section B.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).

4.4 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 4-1 and Figure 4-2 in Section 4.2.3). 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 4.4.1 details the exposure assessment methods for PB-HAPs. Section 4.4.2 details the exposure assessment methods for lead and the acid gases.

4.4.1 Environmental Risk Screen for PB-HAPs

Figure 4-1 in Section 4.2.3 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 4.4.1.1, 4.4.1.2, and 4.4.1.3. See Section 4.5 for further discussion of outputs from an environmental risk screen and presentation of results for an example source category.

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 4-2. 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 A.

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 Appendix B, Section B.6.

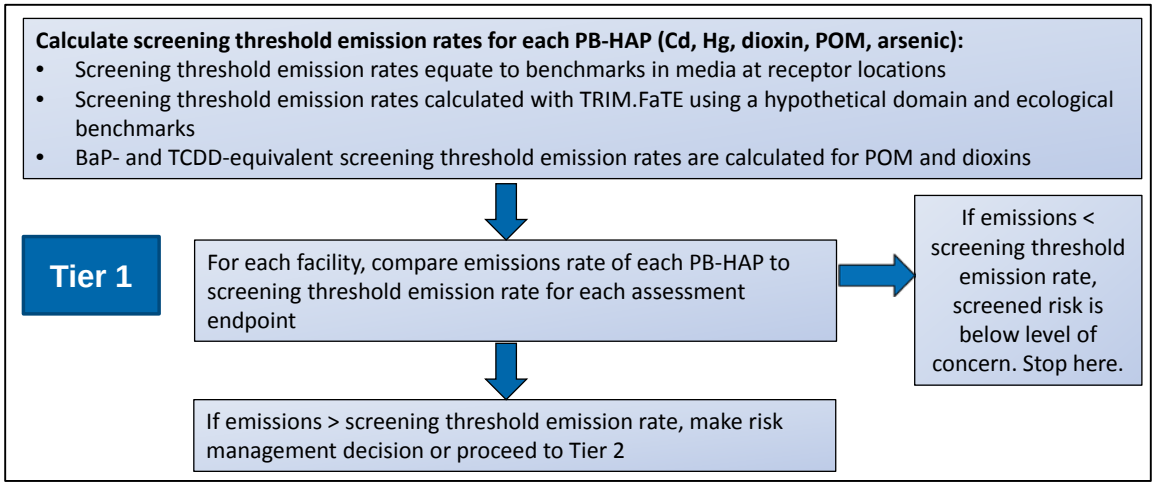
We also assumed that ground-feeding birds and mammals that consume primarily soil invertebrates (e.g., earthworms, grubs) could be exposed to PB-HAPs that have bioaccumulated in the invertebrates from ingestion of or contact with soils. We assumed that the assessment populations obtained 100 percent 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.

4.4.1.1 Tier 1 Exposure Assessment

Figure 4-3 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 4.4.1.2) might be needed.

Figure 4-3. Approach for Tier 1 Environmental Risk Screen for PB-HAPs

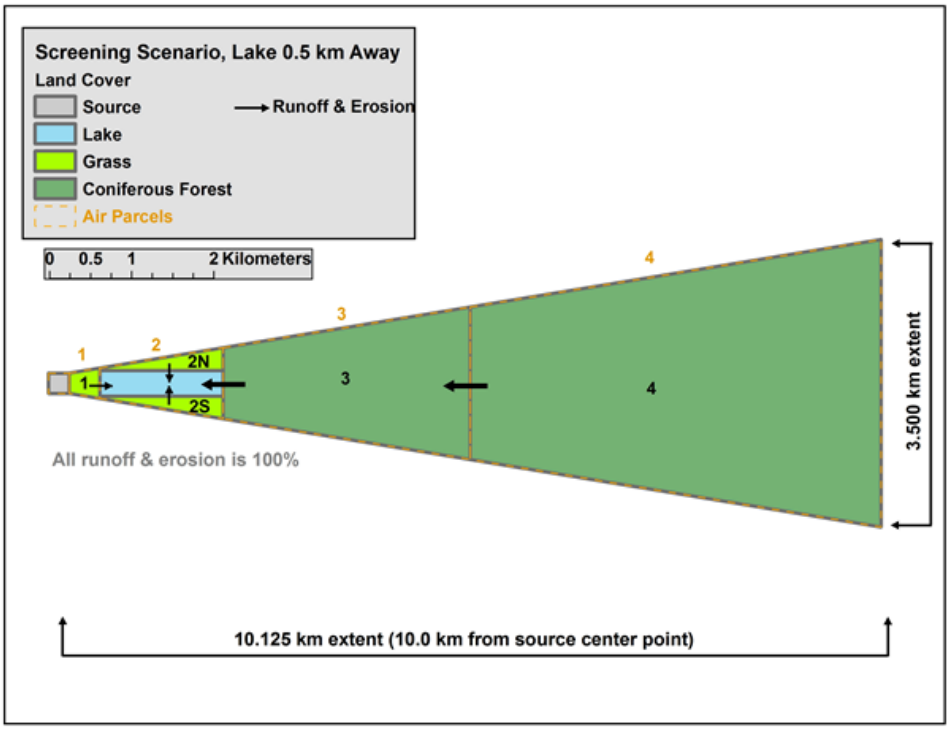


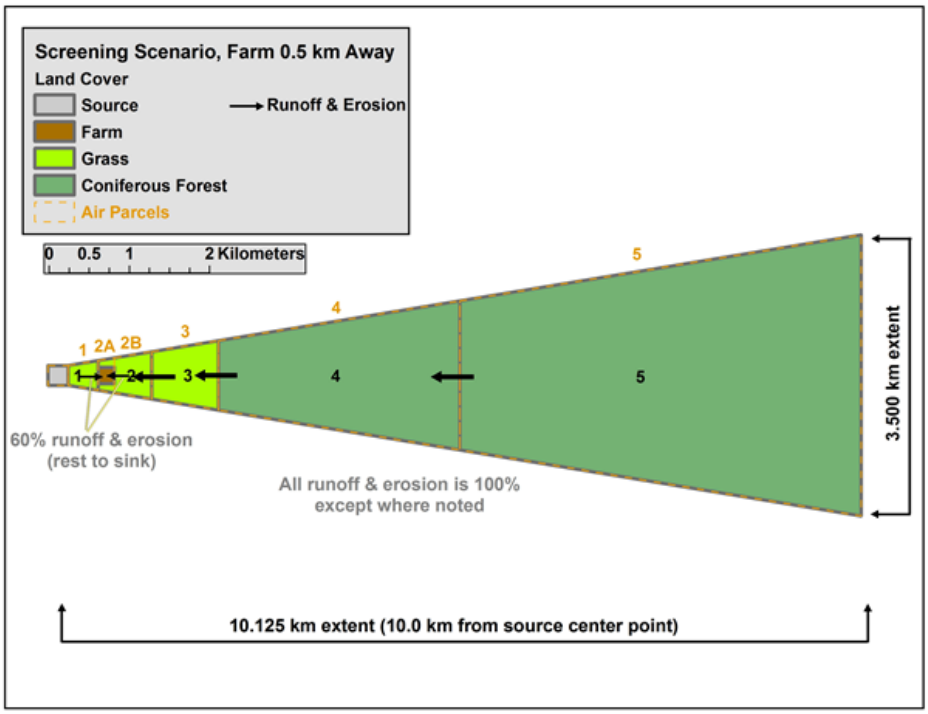
The hypothetical environmental settings are the same as used in the human health risk screen. The lake-centric setting (top panel of Figure 4-4) is used to assess fish and other biota in surface water and sediment. The nonfarm (i.e., grass and forest) parcels in the farm-centric setting (bottom panel of Figure 4-4) are used for the environmental risk screen related to soil.[26] Both 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 4-4). 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 4-4), 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 4-4 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 A. 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 (see Table 1-1).

[26]The farm itself is not used in the environmental risk screen.

Figure 4-4. TRIM.FaTE Lake-centric (Top) and Farm-centric (Bottom) Surface Layouts for the Tier 1 Multipathway 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. Appendix B 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 Section 3.1.2) 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 Appendix B, Section B.6, including the data used to assess ingestion of chemicals from each dietary component for mink and common mergansers.

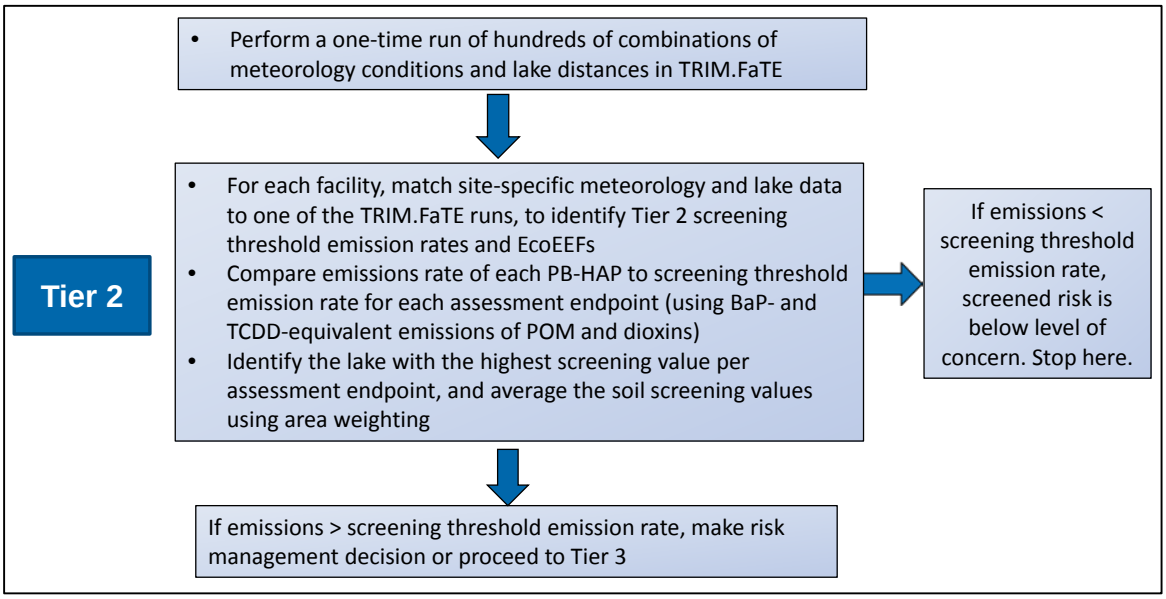
For wildlife-consuming aquatic biota, no adjustments were needed for variation in chemical assimilation efficiency among POM and dioxin/furan congeners, respectively. All toxicity data used to estimate TRVs for POM and dioxin/furan congeners for birds and mammals were based on "administered" doses (the 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).

4.4.1.2 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 4-5, consists of the following steps.

Figure 4-5. 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 3.2.1.1 for a discussion of the meteorology data, including modeled values in Table 3-8). We assess five different distances of the lake from the facility (see Section 3.2.1.2 for a discussion on modeling domain sizes, including modeled lake location values in Table 3-9). 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 the criteria discussed in Section 3.2.2 is identified by its location relative to the facility and by its surface area. Section 3.2.2 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. See Section 3.2.2.1 for further discussion of calculations specific to wind direction and 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) (see Section 3.2.2.3 for additional discussion on multifacility chemical loading to lakes). 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.

4.4.1.3 Tier 3 Exposure Assessment

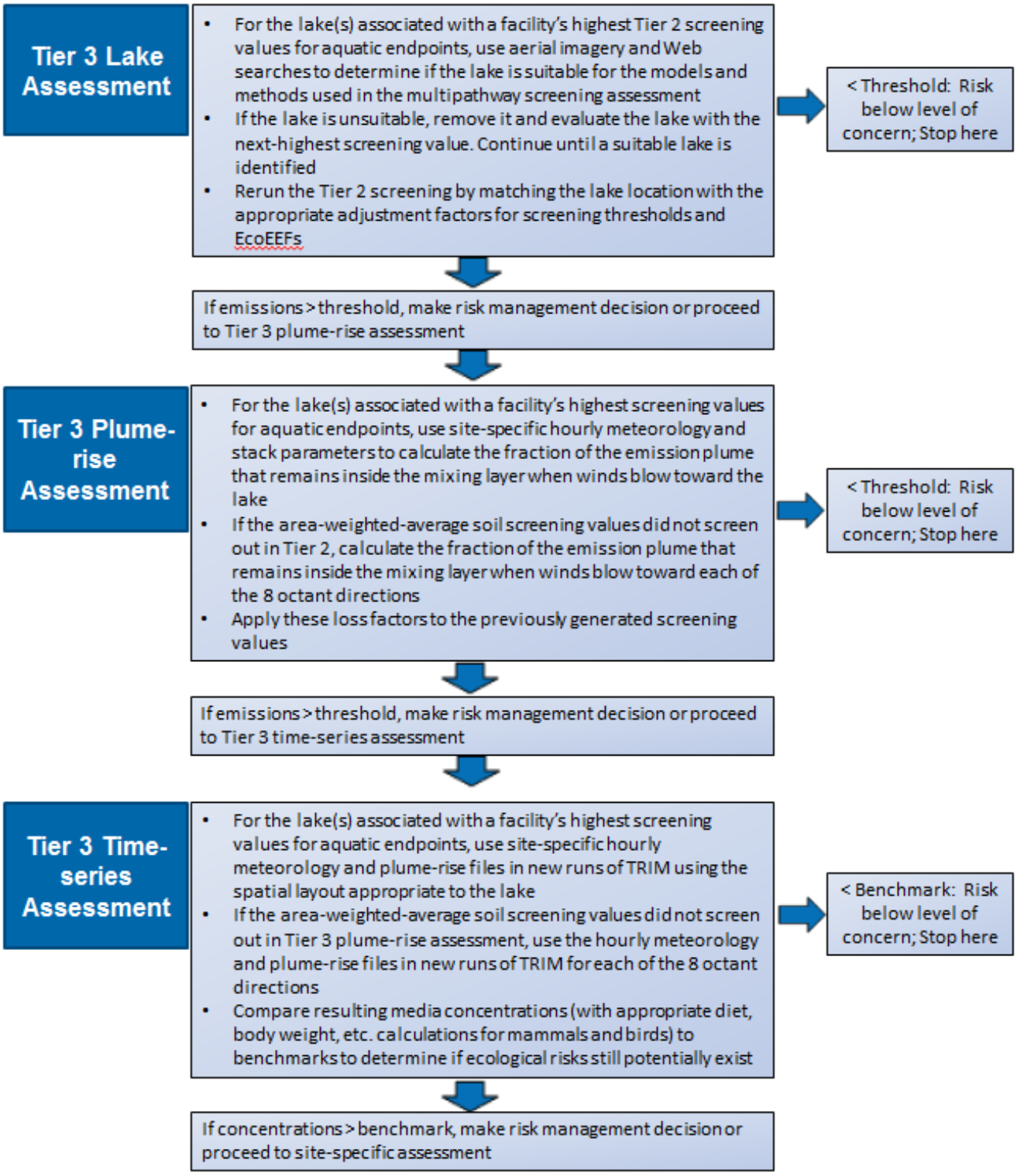
A Tier 3 screen can be conducted on facilities that do not screen out in Tier 2. The Tier 3 screening approach consists of three individual assessments (shown in Figure 4-6 and described in more detail for the human health assessment in Section 3.3) that further refine the screening scenario (beyond 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 MIRC 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), the proportion and absolute area over which soil-based screening threshold emission rates are exceeded, and the 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, 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 (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 4-6. Approach for Tier 3 Environmental Risk Screen for PB-HAPs



4.4.2 Environmental Risk Screen for Lead and Acid Gases

4.4.2.1 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).[27] 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.

4.4.2.2 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)

[27] 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."

- Direct contact of the chemical in air with plant foliage, which eliminates the need for multimedia modeling of chemical transfers
- 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 4.3.2) 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).

4.5 Environmental Risk Characterization/Screening Results

In this section, we discuss the outputs and analyses generated as part of the environmental risk screen. We also present example results for the environmental risk screen for a model category.

4.5.1 Environmental Risk Screen Metrics for PB-HAPs

4.5.1.1 Tier 1

The modeling domain for Tier 1 consists of a health-protective set of conditions (see Section 4.4.1.1). 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 4-4 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.

4.5.1.2 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 4-4 summarizes the PB-HAP environmental risk screen metrics.

Table 4-4. 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 4.4.1.3).

The Tier 3 screening approach consists of three individual assessments (shown in Figure 4-6 and described in more detail for the human health assessment in Section 3.3) 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. As shown in Table 4-8, 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.

4.5.2 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 4-5 to identify effects that are significant and widespread (covering broad areas).

If the area-weighted average SV for the facility is less than 2, 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 4-5. 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.

4.5.3 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-$\mu g/m^3$ 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.

4.5.4 Environmental Risk Screen Example: Risk Results for the Widget Manufacturing Source Category

4.5.4.1 Source Category Description and Results

The Widget Manufacturing source category contains three manufacturing facilities as shown in Table 4-6. The Widget Manufacturing source category emits five environmental HAPs (Table 4-7): mercury compounds (methyl mercury and elemental mercury), cadmium, dioxins/furans, lead, and hydrochloric acid.

Table 4-6. Widget Manufacturing Facility List

Facility Name	Address	State	Latitude	Longitude
Widget 1	1 Main Street	PA	41.802264	−71.787472
Widget 2	4500 First Street	HI	42.001586	−76.717205
Widget 3	1188 Broad Street	ND	38.8085235	−76.303631

Table 4-7. Environmental HAP Emissions from Widget Manufacturing Facilities

Facility	POLLUTANT_DESCRIPTION	EMISSIONS_TPY
1	Dioxins	6.41E−06
1	Cadmium	3.47E−03
1	Mercury (methyl)	2.73E−03
1	Elemental gaseous mercury	7.69E−04
1	Hydrochloric acid	1.42E+02
1	Lead	7.80E−02
2	Dioxins	8.17E−07
2	Cadmium	1.96E−03
2	Elemental gaseous mercury	2.39E−04
2	Mercury (methyl)	8.48E−04
2	Hydrochloric acid	1.69E+01
2	Lead	1.09E−02
3	Dioxins	8.16E−06
3	Cadmium	2.85E−03
3	Elemental gaseous mercury	1.89E−03
3	Gaseous divalent mercury	6.68E−03
3	Hydrochloric acid	4.45E+01
3	Lead	5.11E−02

4.5.4.2 Risk Characterization

Environmental Screening Results for PB-HAP

Tier 1 for PB-HAP

In the Tier 1 environmental risk screen, emissions of mercury (methyl), mercuric chloride, cadmium, and dioxins exceed some Tier 1 screening thresholds as shown in Table 4-8. Specifically, Tier 1 maximum SVs are:

- Facility 1
  - **Mercuric chloride:** exceeds invertebrate surface soil Tier 1 screening threshold at 312 m and 850 m by 10 times.
  - **Methyl mercury:** exceeds avian piscivore NOAEL Tier 1 screening threshold by 10 times.
- Facility 2
  - **Mercuric chloride:** exceeds invertebrate surface soil Tier 1 screening threshold at 312 m by 4 times.
  - **Methyl mercury:** exceeds avian piscivore NOAEL Tier 1 screening threshold by 2 times.
- Facility 3
  - **Dioxins:** exceeds mammalian insectivore NOAEL Tier 1 screening threshold by 2 times.
  - **Mercuric chloride:** exceeds invertebrate surface soil Tier 1 screening threshold at 312 m by 2 times.

In the Tier 1 environmental risk screen, cadmium and lead did not exceed the screening thresholds for any ecological benchmark for any facility in the source category.

Table 4-8. PB-HAPs: Tier 1 Environmental Risk Screen Exceedances for Widget Manufacturing

Facility	PB-HAP	Assessment Endpoint	Benchmark Effects Level	Benchmark Value	Tier 1 Screening Value
1	Mercuric chloride	Sediment Community	Threshold Level	0.15 (mg/kg dry wt)	5.E+00
1	Mercuric chloride	Surface Soil – Dist. 1 – 312 m	Threshold Level − Plant Community	0.3 (mg/kg dry wt)	5.E+00
1	Mercuric chloride	Surface Soil – Dist. 1 − 312 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	1.E+01
1	Mercuric chloride	Surface Soil − Dist. 2 − 850 m	Threshold Level − Plant Community	0.3 (mg/kg dry wt)	3.E+00
1	Mercuric chloride	Surface Soil − Dist. 2 − 850 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	1.E+01
1	Mercuric chloride	Surface Soil − Dist. 3 − 1,500 m	Threshold Level − Plant Community	0.3 (mg/kg dry wt)	2.E+00

Facility	PB-HAP	Assessment Endpoint	Benchmark Effects Level	Benchmark Value	Tier 1 Screening Value
1	Mercuric chloride	Surface Soil − Dist. 3 − 1,500 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	7.E+00
1	Mercuric chloride	Surface Soil − Dist. 4 − 3,500 m	Threshold Level − Plant Community	0.3 (mg/kg dry wt)	2.E+00
1	Mercuric chloride	Surface Soil − Dist. 4 − 3,500 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	5.E+00
1	Mercuric chloride	Surface Soil − Dist. 5 − 7,500 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	2.E+00
1	Mercury (methyl)	Fish − Avian Piscivores	NOAEL (merganser)	0.013 (mg/kg BW/day)	1.E+01
1	Mercury (methyl)	Fish − Avian Piscivores	GMATL (merganser)	0.032 (mg/kg BW/day)	6.E+00
2	Mercuric chloride	Surface Soil − Dist. 1 − 312 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	4.E+00
2	Mercuric chloride	Surface Soil − Dist. 2 − 850 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	3.E+00
2	Mercuric chloride	Surface Soil − Dist. 3 − 1,500 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	2.E+00
2	Mercury (methyl)	Fish − Avian Piscivores	NOAEL (merganser)	0.013 (mg/kg BW/day)	2.E+00
3	Dioxin	Surface Soil − Dist. 1 − 312 m	NOAEL − Mammalian Insectivores (shrew)	0.0000002 (mg/kg dry wt)	2.E+00
3	Dioxin	Surface Soil − Dist. 2 − 850 m	NOAEL − Mammalian Insectivores (shrew)	0.0000002 (mg/kg dry wt)	2.E+00
3	Mercuric chloride	Surface Soil − Dist. 1 − 312 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	2.E+00

Tier 2 for PB-HAP

We conducted a Tier 2 environmental screening analysis for dioxins, methyl mercury, and mercuric chloride. No exceedances of the screening thresholds for dioxins occur in the Tier 2 screen. In addition, no exceedances occur for any pollutant for Facility 2 or Facility 3. Facility 1 had the following Tier 2 maximum SVs (Table 4-9):

- Facility 1
  - **Mercuric chloride:** exceeds Tier 2 surface soil threshold benchmark for invertebrates at 312 m in the NE quadrant by 5 times
  - **Methyl mercury:** exceeds Tier 2 NOAEL for avian piscivores by 6 times

Table 4-9. PB-HAPs: Tier 2 Environmental Risk Screen Exceedances for Widget Manufacturing

Facility	PB-HAP	Assessment Endpoint	Benchmark Effects Level	Benchmark Value	Most Impacted Octant	Screening Value
1	Mercuric chloride	Surface Soil − Dist. 1 − 312 m	Threshold Level − Plant Community	0.3 (mg/kg dry wt)	NE	2.E+00
1	Mercuric chloride	Surface Soil − Dist. 1 − 312 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	NE	5.E+00
1	Mercuric chloride	Surface Soil − Dist. 2 − 850 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	NE	4.E+00
1	Mercuric chloride	Surface Soil − Dist. 3 − 1,500 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	NE	3.E+00
1	Mercuric chloride	Surface Soil − Dist. 4 − 3,500 m	Threshold Level − Invertebrate Community	0.1 (mg/kg dry wt)	NE	2.E+00
1	Mercury (methyl)	Fish − Avian Piscivores	NOAEL (merganser)	0.013 (mg/kg BW/day)	NE	6.E+00

The SVs indicated in Table 4-9 are for individual soil parcel concentrations. Eight (8) octants with 5 soil distances each (312 m, 850 m, 1500 m, 3500 m, and 7500 m), or 40 individual soil parcel concentrations, are assessed. When determining whether adverse environmental effects are "significant and widespread" for soil and sediment, we evaluate the area-weighted average concentration of the 40 soil concentrations for each facility. Table 4-10 shows the area-weighted average concentrations for the two mercuric chloride soil benchmarks for which a soil SV exceeds 1 for one or more individual soil parcels (soil thresholds for plant and invertebrate communities). For the plant community, the area-weighted average soil SV is less than 1, with only 0.1 percent of the soil area exceeding the benchmark. For the soil invertebrate community, the area-weighted average soil SV is less than 1, with 6 percent of the soil area exceeding the benchmark.

Table 4-10. Overall Tier 2 Soil Screening Values and Exceedance Areas for Widget Manufacturing

Facility	PB-HAP	Benchmark Effects Level	Benchmark Value (mg/kg dry wt)	Overall Soil SV Area-wgt. avg. SVs all soil parcels	**Area SV ≥ 2 for Soil** (acres) Based on individual soil parcels	% All Soil Areas **SV ≥ 2** Based on individual soil parcels
1	Mercuric chloride	Threshold Level − Plant Community	0.3	2.E−01	29	0.1%
1	Mercuric chloride	Threshold Level − Invertebrate Community	0.1	7.E−01	2,361	6%

Tier 3 for PB-HAP

We conducted a Tier 3 environmental screen for the methyl mercury emissions from Facility 1. The methyl mercury emissions from Facility 1 exceed the avian piscivore NOAEL Tier 2 screening threshold by 6 times. Because this exceedance is due to the modeled concentrations of methyl mercury in the closest lake to Facility 1, we further examined the lake as part of the Tier 3 screen. Table 4-11 shows information for the lake in question (Smith Lake) and the next closest lake (Jones Pond).

Table 4-11. Lakes Near Facility 1

Facility	Lake Name	Lake Object ID (USGS)	Lake Area (acres)	Lake Facility – Lake Dist. (km)	Lake Latitude	Lake Longitude
1	Smith Lake	463036	35	3.3	42.81906	−71.7546
1	Jones Pond (next closest)	234952	25	8	42.81900	−71.7530

Other than verifying a facility's emission rates, one of the most valuable assessments EPA can conduct following a Tier 2 environmental risk screen is to evaluate the existence and suitability of lakes for the models and methods used in the screens. The full USGS data set of lakes and reservoirs nationwide that EPA uses in Tier 2 occasionally identifies a lake that no longer exists (e.g., it has evaporated or been drained) or it uses a classification that might not accurately reflect the lake's purpose or type. EPA uses aerial and street-view imagery and internet searches to ascertain quickly whether an assessed lake actually exists, likely is used for industrial waste disposal/treatment, or is adjacent or connected to a river or saltwater body. (The assessment methods and models are intended for freshwater systems that are relatively closed and have watersheds that do not extend beyond about 50 km from the facility).

Figure 4-7 shows an aerial photo of Smith Lake, the lake driving the exceedance of the Tier 2 screen for Facility 1. The aerial imagery suggests that Smith Lake is likely nonexistent. As a result, we removed Smith Lake from the analysis and reran the Tier 2 screen using the next closest lake—Jones Pond, which is verified as an actual lake on aerial photos. None of the Tier 2 screening thresholds are exceeded using Jone Pond as the assessed lake.

Figure 4-7. Aerial Photo of Smith Lake

Environmental Screening Results for Acid Gases

The environmental risk screen for acid gases consists of one tier. The screen uses the AERMOD HAP concentrations developed for the human inhalation assessment. As indicated in Section 4.5.2, 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.

Table 4-12 shows the HCl results for the environmental screen for the Widget Manufacturing source category. For Facilities 2 and 3, none of the 208 point estimates exceeded the benchmark

concentrations (the percent modeled domain that exceeds the benchmark is 0). For Facility 3, some point estimates, encompassing a total area of 25 acres, exceeded the benchmark air concentration. These 25 acres represent a very small percentage of the total 2 million modeled acres. For all three facilities, the SV for HCl, which equals the area-weighted average concentration divided by the benchmark air concentration, is well below 1. HF was not emitted by any facility in the category.

Table 4-12. Acid Gases: Environmental Risk Screen Results for Widget Manufacturing

Facility	HAP	Benchmark Air Concentration (µg/m^3)	Area of Exceedance		% Modeled Domain that Exceeds	Area-wgt. Avg. Air Concentration (µg/m^3)	Screening Value (Area-wgh. Avg. Concentration ÷ Benchmark Air Concentration)
			(km^2)	(acres)			
1	HCl	50	0.1	25	0.001%	2.70E−03	5.E−05
2	HCl	50	0	0	0%	1.33E−03	3.E−05
3	HCl	50	0	0	0%	1.98E−03	4.E−05

4.5.4.3 Discussion of Environmental Risk Screen Results

For mercuric chloride, exceedances of the Tier 2 screening thresholds occurred for soil invertebrates for some individual soil parcels. The area-weighted average soil SV for each facility, however, is well below 1. Therefore, environmental effects from mercuric chloride are not expected to be widespread. For methyl mercury, one lake exceeds the Tier 2 screening threshold for the avian piscivores (NOAEL). Further analysis of the lake in Tier 3, however, indicated that the lake no longer exists. Subsequent screening on the next closest lake indicated no exceedances. Therefore, adverse environmental effects from methyl mercury are not expected. One facility exceeded the HCl benchmark air concentration for some modeled data points, encompassing a total exceedance area of 25 acres. For all three facilities, the SV for HCl, which equals the area-weighted average concentration divided by the benchmark air concentration, is well below 1. Therefore, adverse environmental effects from hydrogen chloride are not expected.

5. CHRONIC INHALATION RISK ASSESSMENT ENHANCEMENTS

Inhalation risk assessments performed for RTR rulemakings primarily use EPA's Human Exposure Model (HEM) to model sources emitting air toxics to ambient air. HEM, which addresses the inhalation pathway of exposure, is designed to predict risks associated with pollutants emitted into the ambient air near an emitting facility but beyond the facility's property boundary. HEM provides ambient air concentrations as surrogates for lifetime exposure along with cancer unit risk estimates and noncancer inhalation reference concentrations to produce estimates of cancer risk and noncancer hazard, respectively, for the pollutants modeled. HEM contains (1) an atmospheric dispersion model, AERMOD, with included meteorological data; (2) U.S. Census Bureau population data at the census block level; and (3) a database of acute and chronic health reference values.

We are requesting SAB review on the following two significant enhancements to our chronic inhalation risk assessment:

- In previous chronic inhalation risk assessments, we assumed the land surrounding each facility was rural. This assumption was health protective because concentrations are generally higher when the surrounding area is modeled as rural. The manmade surfaces in urban areas retain heat and lead to more convection of the air at night, which disperses pollutants more than over rural areas. Since the most recent SAB review in 2009, we developed an urban/rural enhancement to the chronic inhalation risk assessment that allows us to account for the urban/rural characteristics of the land surrounding each evaluated facility, and therefore, to better characterize the dispersion of pollutants near sources (Section 5.1). The new urban/rural procedure evaluates the urban/rural designation of the census block closest to each evaluated facility and uses these designations in the HEM/AERMOD assessment. In Section 5.1, we present the census block designation approach (the "HEM approach") and compare it to the existing approach specified in EPA's modeling guidance.
- In its 2009 review, the SAB noted that census block centroids might not always be an appropriate surrogate for residential locations. Since 2009, we developed the census block receptor enhancement (Section 5.2) that allows us to model air concentrations more accurately where populations actually reside. Specifically, the new enhancement

automatically identifies census block centroids that might be located on facility property ("on site"), and census blocks that are very large and for which the centroid is less likely representative of the block's residential locations. When onsite or large blocks are identified, we add new receptors, delete census block centroids, or move census block centroids to represent residential locations more accurately.

Several minor modifications and updates to the assessment methodology for chronic inhalation risk have been made since the 2009 SAB review—on which we are not seeking SAB review. These include use of more recent versions of AERMOD and its preprocessors and use of different model options, more recent and more complete model input data, and GIS tools to ensure the quality of model inputs. Eight AERMOD versions have been released since 2009, the most recent in June 2015 (version 15181). The AERMOD (and AERMET) change that likely would have the most significant effect on modeled air toxics concentrations is the addition of a non-default (beta) option in December 2012 (in versions 12345 and more recent) that adjusts the surface friction velocity (u*) under low-wind/stable conditions (Qian and Venkatram 2011). RTR assessments uses this beta option.

The census and meteorological data libraries used as HEM input data have been updated since 2009. In 2009, HEM used census block location and population data from the 2000 Census. Census block elevations were determined nationally from the U.S. Geological Survey 1-degree digital elevation model (DEM) data files, which have a spatial resolution of about 90 m. Currently, HEM uses census block location and population data from the 2010 Census and block elevations determined nationally from the U.S. Geological Survey 1/3-Arc-Second National Elevation Dataset, which has a spatial resolution of about 10 m.

The meteorological data used in 2009 were from approximately 200 surface observation stations of the National Weather Service across the continental United States, Alaska, Hawaii, and Puerto Rico for 1991. In its 2009 review, the SAB noted that using meteorological data from stations far from emissions sources introduces uncertainty into model results. We agree, and newer technologies have allowed us to expand our data set of meteorological stations. We obtained the meteorological data currently in use from the Automated Surface Observing Systems (ASOS) program, a joint effort of the National Weather Service, Federal Aviation Administration, and

Department of Defense. The ASOS, which serves as the nation's primary surface weather observing network, is designed to support weather forecast activities and aviation operations and the needs of the meteorological, hydrological, and climatological research communities. With the largest and most modern complement of weather sensors, ASOS has significantly expanded the amount of available meteorological information. ASOS updates observations every minute, 24 hours a day, every day of the year. ASOS is installed at more than 900 airports across the country, and our meteorological library for 2011 includes all sites that are without a significant number of missing hours (824 stations, or four times the number of stations used in 2009).

5.1 Urban/Rural Dispersion Selection

An additional update to the inhalation risk assessment methodology since the 2009 SAB review is the inclusion in HEM of a default procedure to identify more accurately whether facilities should be modeled using an urban or a rural designation (in the AERMOD portion of HEM). Before this update, we modeled all facilities as rural because modeled concentrations are generally higher when designated as rural, and our goal is to overestimate rather than underestimate concentrations to be health protective. The new urban/rural procedure within HEM enables the modeling domain to be characterized more accurately.

To account for dispersion in the "convective-like" boundary layer that forms during nighttime conditions due to the urban heat island effect, AERMOD enhances the turbulence for urban nighttime conditions over that expected in the adjacent rural, stable boundary layer. The urban-rural temperature difference that develops at night drives the magnitude of the urban heat island effect. In the past, RTR assessments did not include case-by-case determinations of urban or rural dispersion for the large number of facilities in the source categories because of the effort required.

EPA's *Guideline on Air Quality Models* (U.S. EPA 2005h) addresses the regulatory application of air quality models for assessing certain pollutants under the Clean Air Act. The Guideline provides the basis for determining the urban/rural classification of a source (U.S. EPA 2005h) and specifies a land use procedure (Auer 1978) for determining urban/rural status. With this procedure, if industrial, commercial, and compact residential land use types account for 50 percent or more of the 3-km-radius circular area around a facility, the facility should be modeled

as urban; otherwise, the facility should be modeled as rural. A source might be located within an urban area, but close enough to a body of water or to other nonurban land use categories to result in a predominantly rural land use classification within 3 km of a source following the land use procedure. Recognizing that the urban heat island is not a localized effect but is regional in character, the AERMOD Implementation Guide (U.S. EPA 2015b) cautions against applying the land use procedure without considering the potential for urban heat island influences across the full modeling domain. Consequently, following the AERMOD Implementation Guide could result in an urban classification, even if the land use procedure indicates rural because the modeling domain is largely within an urban area.

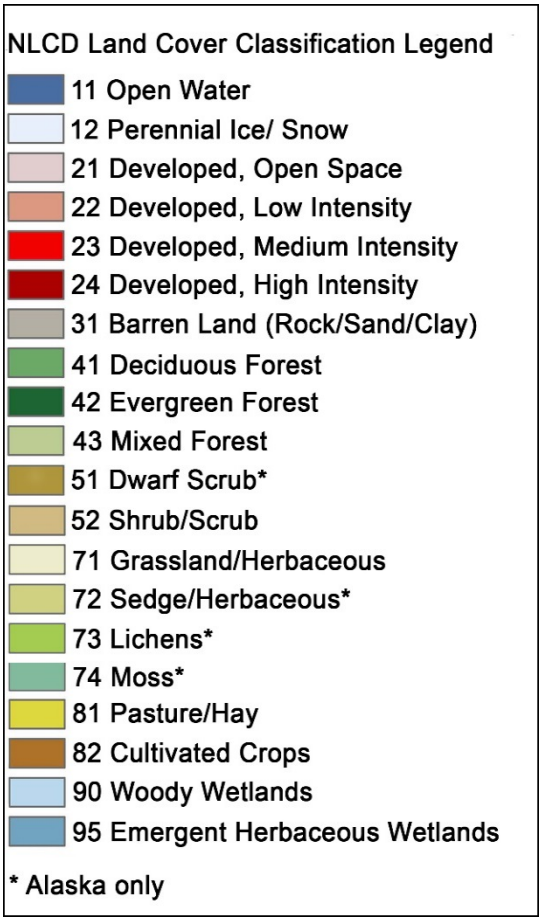
The HEM default procedure for determining urban/rural status differs from the land use procedure but is similar to the application in the AERMOD Implementation Guide. The HEM procedure identifies the census block nearest to each facility, and if that block is within an "urbanized area," the facility is modeled as urban. For the 2010 Census, an urbanized area comprises a densely settled core of census tracts or census blocks, or both, that meet minimum population density requirements (50,000 or more people), along with adjacent territory. The adjacent territory generally is also densely settled but it may also contain some lower density areas as well as nonresidential urban land uses. About 500 such areas are included in the 2010 Census. The population of the urbanized area serves as the urban population input for AERMOD in HEM.

To examine the influence of the different procedures used to make the urban/rural determination, we performed an analysis on each of the 132 facilities in the petroleum refining source category using each procedure. We performed the Guideline land use procedure for each facility using a geographic information system (GIS). Buffers of 3 km were created for each facility, and 2011 land use data (USGS 2014) were used to determine the fraction of urban land use within each buffer. Land use classes 23 (Developed, Medium Intensity) and 24 (Developed, High Intensity) were the only land use classes considered urban for this analysis because they most closely match the industrial, commercial, and compact residential land use types. To determine urban/rural classification using the HEM procedure, we used GIS to determine whether each facility was in an urbanized area (based on the 2010 Census).

Based only on the land use procedure, 26 of the 132 facilities were found to be urban. Many other facilities with less than 50 percent urban land use were actually located in urban areas but had less than 50 percent urban land use because of a nearby body of water or other non-urban land use. In these cases, we applied the AERMOD Implementation Guide, resulting in another 41 facilities classified as urban because they were located in a large urban area even though the local land use did not exceed 50 percent urban. For example, four facilities in the Philadelphia area with urban land use values ranging from 14 to 36 percent were classified as urban because they were within the large Philadelphia urban area, but were less than 50 percent urban land use because of significant water (the Delaware River) in the 3-km buffers of the facilities.

Figure 5-1 shows the key for land use categories for the 2011 data. Figure 5-2 through Figure 5-4 show several examples where facilities were determined urban (by following the AERMOD Implementation Guide), even though the urban land use (shown within the drawn circles) was less than 50 percent (urbanized areas outlined in black). Overall, 67 facilities were determined urban based on either the land use procedure or the application of the AERMOD Implementation Guide. Based on the default procedure in HEM, 82 facilities were determined urban. Fifteen facilities determined urban in HEM were not determined urban by land use or the application of the AERMOD Implementation Guide. For these facilities, HEM (Multi-HEM, version 1.3.1, U.S. EPA 2014) was run to estimate the influence on modeled concentrations based on the different determinations. HEM was used with the latest AERMOD executable file (version 15181, U.S. EPA 2015c), which includes some revised urban model algorithms compared to the previous AERMOD version. Model inputs for emissions and

Figure 5-1. 2011 National Land Cover/Land Use Classes



release parameters were those used in the risk assessment performed for the Petroleum Refinery Sector Risk and Technology Review and New Source Performance Standards final rulemaking.[28]

Table 5-1 lists each petroleum refinery, and gives the urban/rural determinations made based on the HEM procedure and based on land use. Separate columns for a facility's urban/rural determination indicate whether it was based only on the land use procedure or by applying the guidance in the AERMOD Implementation Guide. Table 5-1 also gives the percent difference in modeled concentrations. Where the percent difference is positive, the concentration resulting from the rural model run was higher than from the urban run.

In only four cases did modeled concentrations differ by more than 20 percent, and in all of these the rural model run resulted in higher concentrations. This result is expected because AERMOD enhances the turbulence for urban nighttime conditions over that expected in the adjacent rural, stable boundary layer, at least for the low-level releases typical of emissions sources in the petroleum refineries source category. The greatest difference was 100 percent. Figure 5-5 through Figure 5-8 show the four cases (urbanized areas outlined in black). As shown in the figures, each facility is located within an urbanized area, but the urban land use percentage is low, primarily because of significant water within the 3-km radius of the facility. We conclude the HEM default procedure provides a reliable and quick method of making the urban/rural determination for facilities modeled for RTR risk assessments for two key reasons. First, the urban/rural determinations using the HEM procedure matched well (about 90 percent) with those made with the land use procedure or from application of the AERMOD Implementation Guide. Second, in most cases where they did not match, the difference in modeled concentrations was small.

[28] http://www3.epa.gov/ttn/atw/petref/hem/PetRef_CurrentHEMInputs_ActualPCAllow_Thru20140131.zip

Figure 5-2. Four Facilities with Rural Land Use Fractions in Philadelphia, PA – Camden, NJ

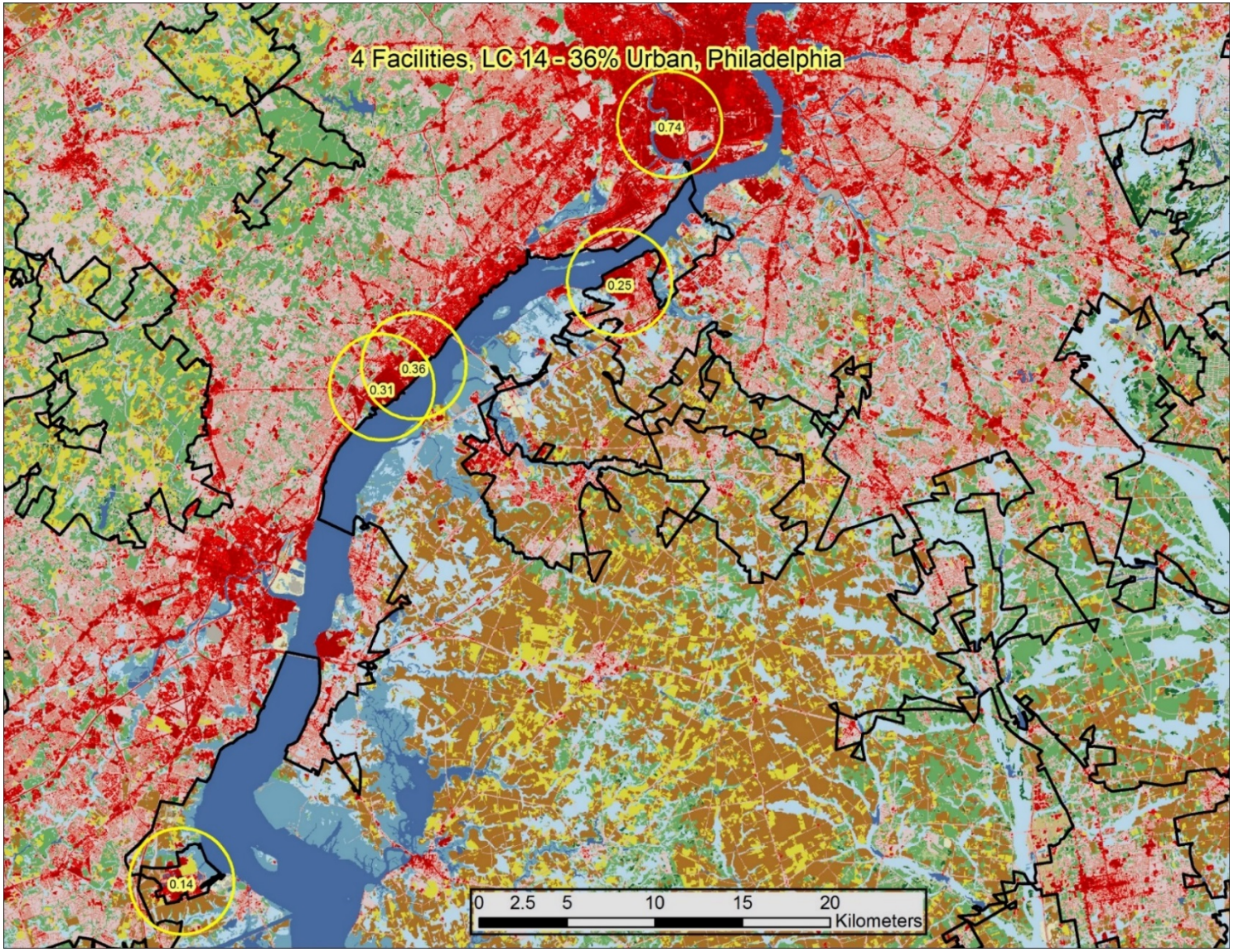


Figure 5-3. Two Facilities with Rural Land Use Fractions in Chicago, IL

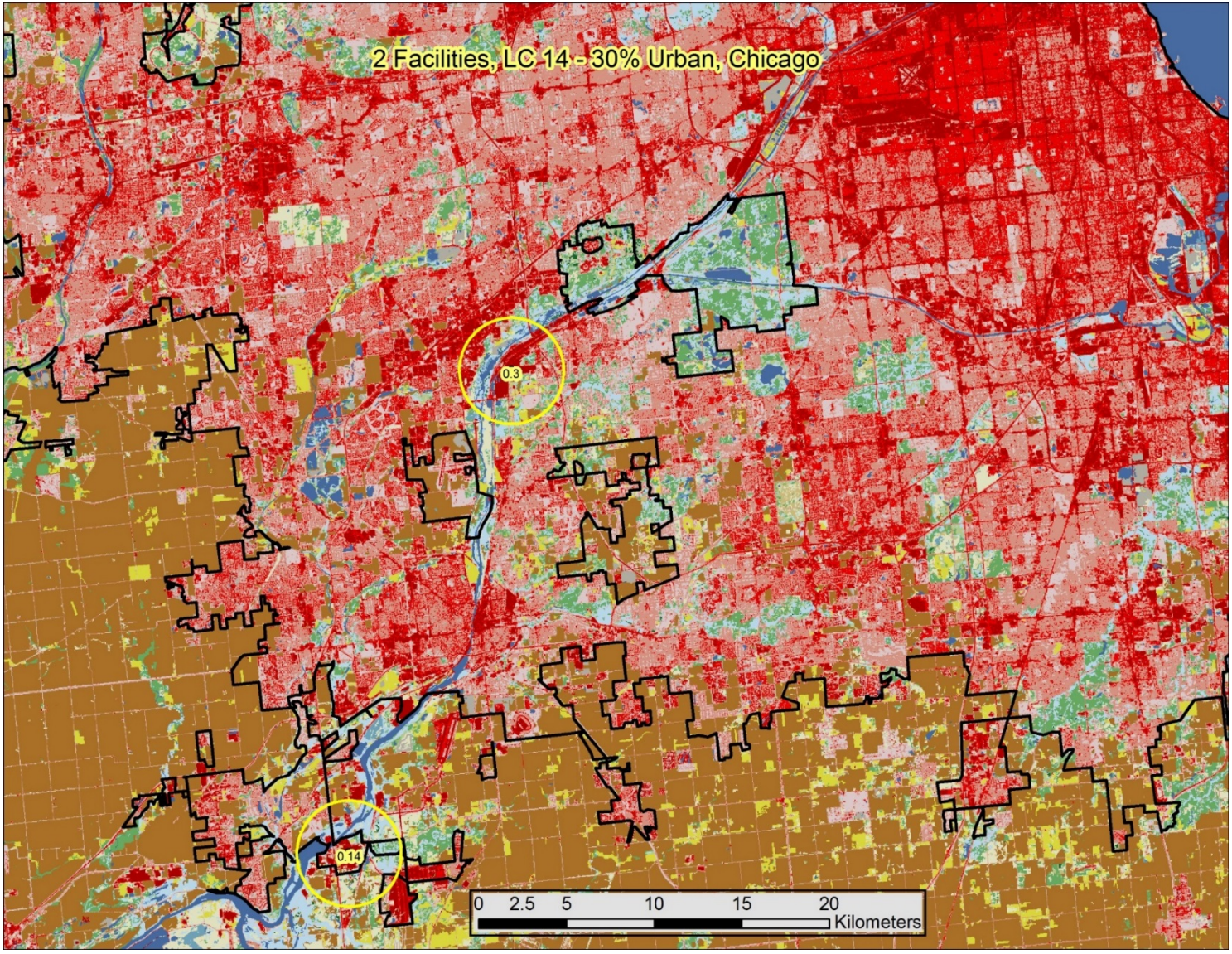


Figure 5-4. Five Facilities with Rural Land Use Fractions in San Francisco-Oakland-Vallejo-Concord, CA

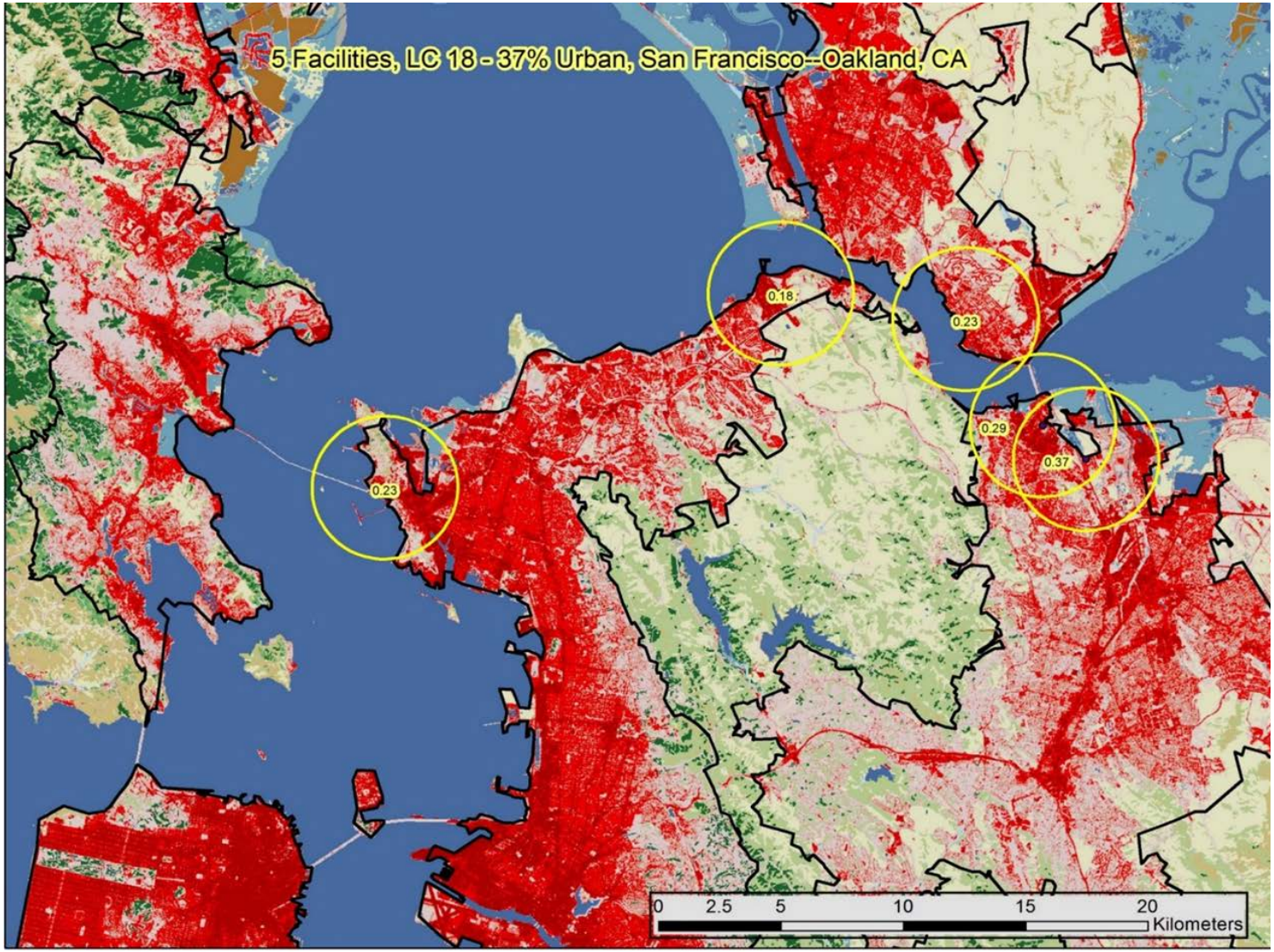


Table 5-1. Urban/Rural Classification of Petroleum Refineries Source Category.

Facility ID	Latitude	Longitude	Urban Land Use Fraction	Urban/Rural Classification			Urbanized Area	Percent Difference in Modeled Concentrations[b]
				HEM	Land Use	AERMOD Implementation Guide[a]		
32023NEI2NV110905	38.622000	−115.618000	0.00	R	R			
40039NEI40039REF1033	35.698000	−98.661000	0.00	R	R			
35025NEINMT$12478	32.879000	−103.301000	0.00	R	R			
28067NEI34061	31.817000	−89.009000	0.00	R	R			
56041NEIWYT$12156	41.261000	−110.806000	0.01	R	R			
06083NEI2CA254640	34.930000	−120.510000	0.01	R	R			
35031NEI34907	35.489000	−108.426000	0.01	R	R			
22097NEI6136	30.529000	−91.751000	0.02	R	R			
06029NEI20103	35.294000	−118.918000	0.02	R	R			
22015NEI33007	32.590000	−93.513000	0.02	R	R			
56007NEI43243	41.783000	−107.108000	0.03	R	R			
40049NEI40531	34.628000	−97.169000	0.04	R	R			
22019NEI33010	30.134000	−93.320000	0.04	R	R			
05139NEI05139REF11	33.364000	−92.713000	0.04	R	R			
06079NEI19869	35.038000	−120.588000	0.04	R	R			
28149NEI34069	32.386000	−90.907000	0.04	R	R			
48341NEI8139	35.957000	−101.883000	0.05	R	R			
48227NEI6446	32.269000	−101.414000	0.05	R	R			
51199NEI42309	37.214000	−76.448000	0.05	U	R	R	Virginia Beach, VA	50
56045NEI404	43.851000	−104.215000	0.06	R	R			
22075NEI6116	29.682000	−89.977000	0.06	R	R			
48297NEI12486	28.457000	−98.189000	0.06	R	R			

Facility ID	Latitude	Longitude	Urban Land Use Fraction	Urban/Rural Classification			Urbanized Area	Percent Difference in Modeled Concentrations[b]
				HEM	Land Use	AERMOD Implementation Guide[a]		
53073NEI42413	48.874000	−122.743000	0.07	R	R			
42123NEI40732	41.832000	−79.131000	0.07	R	R			
18129NEI32353	37.942000	−87.909000	0.07	R	R			
20113NEI32801	38.342000	−97.671000	0.07	R	R			
40019NEIOKT$11009	34.204000	−97.104000	0.08	R	R			
54029NEI46752	40.611000	−80.630000	0.08	R	R			
53073NEI42425	48.830000	−122.697000	0.08	R	R			
48039NEI6519	29.075000	−95.744000	0.08	R	R			
22093NEI6084	30.111000	−90.897000	0.09	R	R			
01097NEI18372	30.785000	−88.057000	0.09	R	R			
56025NEI371	42.861000	−106.242000	0.09	R	R			
17033NEI49781	38.998000	−87.718000	0.10	R	R			
42083NEI46764	41.971000	−78.630000	0.10	R	R			
35015NEI34898	32.848000	−104.395000	0.11	R	R			
48233NEI6963	35.710000	−101.383000	0.11	R	R			
38059NEI40371	46.852000	−100.882000	0.11	U	R	R	Bismarck, ND	0
53057NEI42381	48.494000	−122.563000	0.12	R	R			
53057NEI42382	48.489000	−122.565000	0.13	R	R			
20015NEI32762	37.800000	−96.872000	0.13	R	R			
01125NEI18394	33.201000	−87.609000	0.14	U	R	R	Tuscaloosa, AL	100
10003NEI26218	39.590000	−75.631000	0.14	U	R	U	Philadelphia, PA–NJ–DE–MD	
17197NEI53718	41.416000	−88.184000	0.14	U	R	U	Chicago, IL–IN	
27037NEI34022	44.765000	−93.044000	0.15	R	R			

Facility ID	Latitude	Longitude	Urban Land Use Fraction	Urban/Rural Classification			Urbanized Area	Percent Difference in Modeled Concentrations[b]
				HEM	Land Use	AERMOD Implementation Guide[a]		
21019NEI32864	38.374000	−82.600000	0.15	U	R	R	Huntington, WV–KY–OH	−1
40071NEI12988	36.680000	−97.093000	0.15	R	R			
20125NEI2KS125003	37.048000	−95.607000	0.16	R	R			
48245NEI41771	29.965000	−93.883000	0.17	R	R			
22089NEI33031	30.011000	−90.411000	0.17	U	R	R	New Orleans, LA	20
22095NEI6087	30.060000	−90.589000	0.17	R	R			
28059NEI34057	30.341000	−88.495000	0.18	U	R	R	Pascagoula, MS	50
06013NEI19870	38.046000	−122.250000	0.18	U	R	U	San Francisco–Oakland, CA	
22017NEI33008	32.431000	−93.817000	0.18	U	R	U	Shreveport, LA	
22089NEI6095	29.992000	−90.395000	0.19	U	R	R	New Orleans, LA	0
48355NEI41864	27.825000	−97.482000	0.20	R	R			
30111NEI12458	45.659000	−108.769000	0.20	R	R			
21199NEI32997	37.072000	−84.609000	0.20	R	R			
27163NEI34050	44.849000	−93.002000	0.21	U	R	U	Minneapolis–St. Paul, MN–WI	
48355NEI6617	27.814000	−97.496000	0.22	U	R	R	Corpus Christi, TX	5
48029NEI7130	29.348000	−98.461000	0.23	U	R	U	San Antonio, TX	
06095NEI25450	38.055000	−122.163000	0.23	U	R	U	Vallejo, CA	
06013NEI19587	37.936000	−122.403000	0.23	U	R	U	San Francisco–Oakland, CA	
30111NEI12460	45.814000	−108.430000	0.24	R	R			
22087NEI6127	29.933000	−89.941000	0.24	U	R	U	New Orleans, LA	
05139NEI876	33.200000	−92.676000	0.24	R	R			
48355NEI12084	27.816000	−97.486000	0.24	U	R	R	Corpus Christi, TX	9
22121NEI6130	30.476000	−91.205000	0.25	U	R	U	Baton Rouge, LA	

Facility ID	Latitude	Longitude	Urban Land Use Fraction	Urban/Rural Classification			Urbanized Area	Percent Difference in Modeled Concentrations[b]
				HEM	Land Use	AERMOD Implementation Guide[a]		
34015NEINJT$891	39.840000	−75.258000	0.25	U	R	U	Philadelphia, PA–NJ–DE–MD	
13051NEI26473	32.109000	−81.126000	0.27	U	R	R	Savannah, GA	90
22019NEI6166	30.176000	−93.330000	0.28	U	R	U	Lake Charles, LA	
22087NEI6123	29.932000	−89.968000	0.29	U	R	U	New Orleans, LA	
06013NEI19834	38.024000	−122.114000	0.29	U	R	U	Concord, CA	
17197NEI53702	41.654000	−88.045000	0.30	U	R	U	Chicago, IL–IN	
39003NEI11663	40.721000	−84.125000	0.30	U	R	R	Lima, OH	0
42045NEI109	39.814000	−75.429000	0.31	U	R	U	Philadelphia, PA–NJ–DE–MD	
22019NEI6062	30.236000	−93.272000	0.32	U	R	U	Lake Charles, LA	
48355NEI41863	27.821000	−97.432000	0.32	R	R			
39095NEI11449	41.677000	−83.461000	0.32	U	R	U	Toledo, OH–MI	
48245NEI7233	30.063000	−94.065000	0.33	U	R	U	Beaumont, TX	
48423NEI6475	32.362000	−95.277000	0.33	U	R	U	Tyler, TX	
49011NEI42020	40.888000	−111.905000	0.33	U	R	U	Ogden–Layton, UT	
47157NEI41591	35.086000	−90.082000	0.33	U	R	U	Memphis, TN–MS–AR	
40143NEI12968	36.119000	−95.998000	0.34	U	R	U	Tulsa, OK	
30013NEI12464	47.523000	−111.293000	0.35	U	R	R	Great Falls, MT	−3
55031NEI42583	46.710000	−92.093000	0.36	U	R	R	Duluth, MN–WI	−1
49035NEI42040	40.798000	−111.917000	0.36	U	R	U	Salt Lake City–West Valley City, UT	
48245NEI11200	29.854000	−93.974000	0.36	U	R	U	Port Arthur, TX	
42045NEI113	39.821000	−75.405000	0.36	U	R	U	Philadelphia, PA–NJ–DE–MD	

Facility ID	Latitude	Longitude	Urban Land Use Fraction	Urban/Rural Classification			Urbanized Area	Percent Difference in Modeled Concentrations[b]
				HEM	Land Use	AERMOD Implementation Guide[a]		
49035NEI42081	40.825000	−111.924000	0.36	U	R	U	Salt Lake City–West Valley City, UT	
56021NEI363	41.127000	−104.791000	0.37	U	R	R	Cheyenne, WY	−10
06013NEI2CA314628	38.016000	−122.090000	0.37	U	R	U	Concord, CA	
39095NEI11450	41.629000	−83.497000	0.37	U	R	U	Toledo, OH–MI	
48167NEI12791	29.370000	−94.901000	0.38	U	R	U	Texas City, TX	
40143NEI12969	36.139000	−96.017000	0.39	U	R	U	Tulsa, OK	
48167NEI12044	29.372000	−94.903000	0.42	U	R	U	Texas City, TX	
49011NEI42016	40.838000	−111.920000	0.42	U	R	U	Ogden–Layton, UT	
48245NEI7441	29.885000	−93.965000	0.42	U	R	U	Port Arthur, TX	
30111NEI12459	45.780000	−108.488000	0.43	U	R	R	Billings, MT	−2
22033NEI6022	30.487000	−91.182000	0.44	U	R	U	Baton Rouge, LA	
48201NEI7781	29.735000	−95.005000	0.44	U	R	U	Houston, TX	
17119NEI55835	38.843000	−90.080000	0.45	U	R	U	Alton, IL–MO	
39151NEI11574	40.773000	−81.416000	0.45	U	R	U	Canton, OH	
34023NEI34872	40.561000	−74.244000	0.46	U	R	U	New York–Newark, NY–NJ–CT	
48201NEI11119	29.720000	−95.129000	0.48	U	R	U	Houston, TX	
34023NEI34873	40.540000	−74.259000	0.48	U	R	U	New York–Newark, NY–NJ–CT	
48167NEI6436	29.368000	−94.926000	0.50	U	U		Texas City, TX	
34039NEI6375	40.620000	−74.223000	0.51	U	U		New York–Newark, NY–NJ–CT	
48201NEI12480	29.718000	−95.199000	0.53	U	U		Houston, TX	

Facility ID	Latitude	Longitude	Urban Land Use Fraction	Urban/Rural Classification			Urbanized Area	Percent Difference in Modeled Concentrations[b]
				HEM	Land Use	AERMOD Implementation Guide[a]		
48355NEI7988	27.809000	−97.424000	0.53	U	U		Corpus Christi, TX	
48355NEI7134	27.809000	−97.422000	0.54	U	U		Corpus Christi, TX	
06029NEI20174	35.382000	−119.067000	0.54	U	U		Bakersfield, CA	
18089NEI11715	41.666000	−87.484000	0.61	U	U		Chicago, IL–IN	
48201NEI11232	29.715000	−95.240000	0.61	U	U		Houston, TX	
06029NEI20154	35.397000	−119.048000	0.61	U	U		Bakersfield, CA	
53053NEI42370	47.260000	−122.398000	0.63	U	U		Seattle, WA	
08001NEI889	39.800000	−104.945000	0.63	U	U		Denver–Aurora, CO	
48141NEI11192	31.772000	−106.399000	0.64	U	U		El Paso, TX–NM	
06037NEI20467	33.909000	−118.409000	0.64	U	U		Los Angeles–Long Beach–Anaheim, CA	
48201NEI12711	29.720000	−95.254000	0.66	U	U		Houston, TX	
06037NEICA0370363	33.773000	−118.288000	0.72	U	U		Los Angeles–Long Beach–Anaheim, CA	
42101NEI40723	39.913000	−75.202000	0.74	U	U		Philadelphia, PA–NJ–DE–MD	
26163NEI11885	42.281000	−83.155000	0.75	U	U		Detroit, MI	
06037NEI20966	33.947000	−118.166000	0.81	U	U		Los Angeles–Long Beach–Anaheim, CA	
06037NEI21466	33.779000	−118.234000	0.86	U	U		Los Angeles–Long Beach–Anaheim, CA	
06037NEI21034	33.852000	−118.331000	0.88	U	U		Los Angeles–Long Beach–Anaheim, CA	
06037NEICA1910268	33.793000	−118.226000	0.89	U	U		Los Angeles–Long Beach–Anaheim, CA	

Facility ID	Latitude	Longitude	Urban Land Use Fraction	Urban/Rural Classification			Urbanized Area	Percent Difference in Modeled Concentrations[b]
				HEM	Land Use	AERMOD Implementation Guide[a]		
06037NEI20616	33.874000	−118.162000	0.89	U	U		Los Angeles–Long Beach–Anaheim, CA	
06037NEI21130	33.898000	−118.147000	0.89	U	U		Los Angeles–Long Beach–Anaheim, CA	
06037NEI2CA131003	33.810000	−118.243000	0.91	U	U		Los Angeles–Long Beach–Anaheim, CA	
06037NEICA0379991	33.805000	−118.244000	0.92	U	U		Los Angeles–Long Beach–Anaheim, CA	
06037NEI20797	33.798000	−118.239000	0.93	U	U		Los Angeles–Long Beach–Anaheim, CA	

[a]Where the percent difference is positive, the concentration resulting from the rural model run was higher.

[b]The guidance provided in the AERMOD Implementation Guide was used only in cases where the land use and HEM procedures resulted in different urban/rural determinations. An urban determination using this guidance was made only for facilities that are clearly located in large urban areas; application of the guidance could result in more facilities being classified as urban.

Figure 5-5. Facility with Rural Modeled Concentration 100 Percent Higher Than Urban – Tuscaloosa, AL

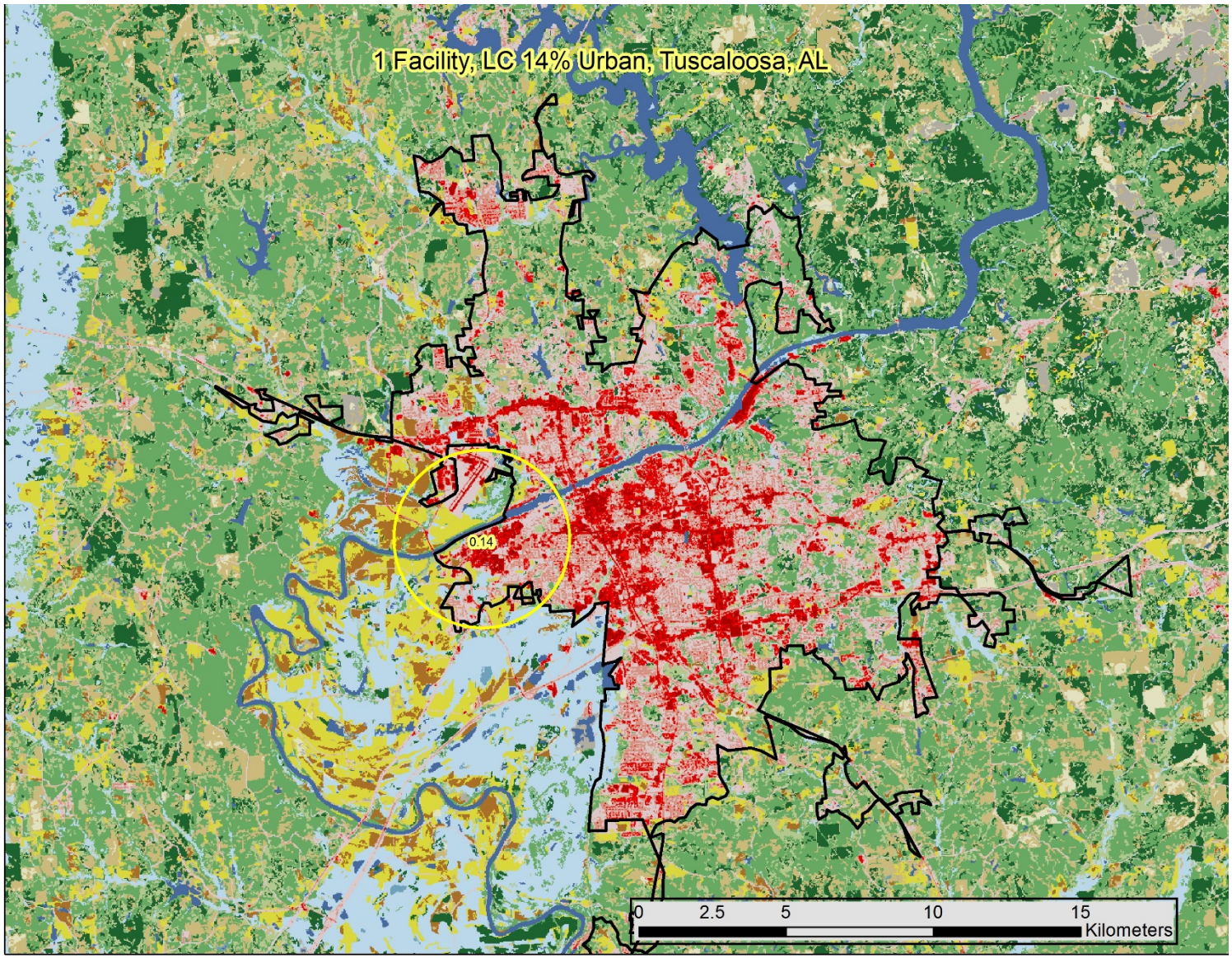


Figure 5-6. Facility with Rural Modeled Concentration 90 Percent Higher Than Urban – Savannah, GA

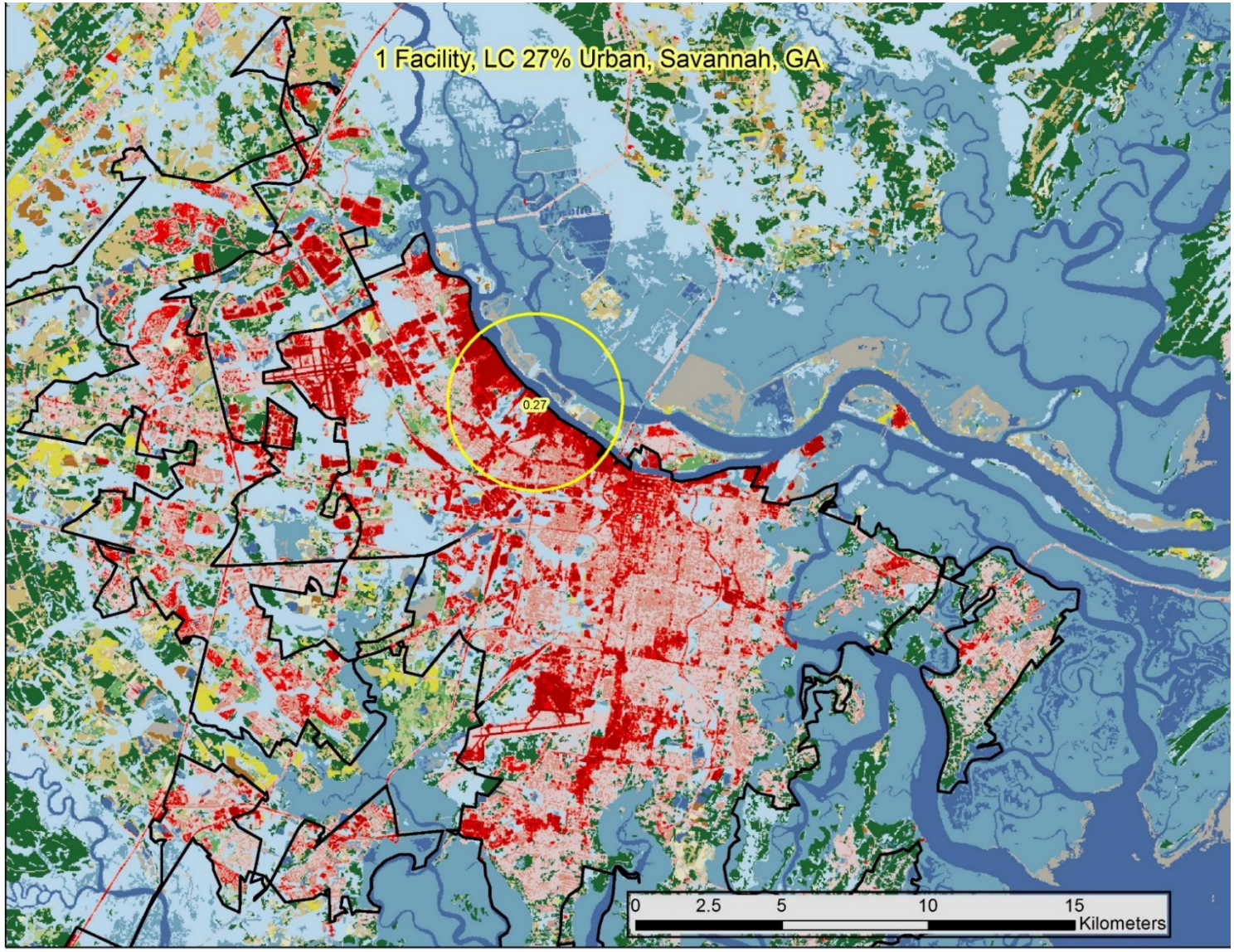


Figure 5-7. Facility with Rural Modeled Concentration 50 Percent Higher Than Urban – Pascagoula, MS

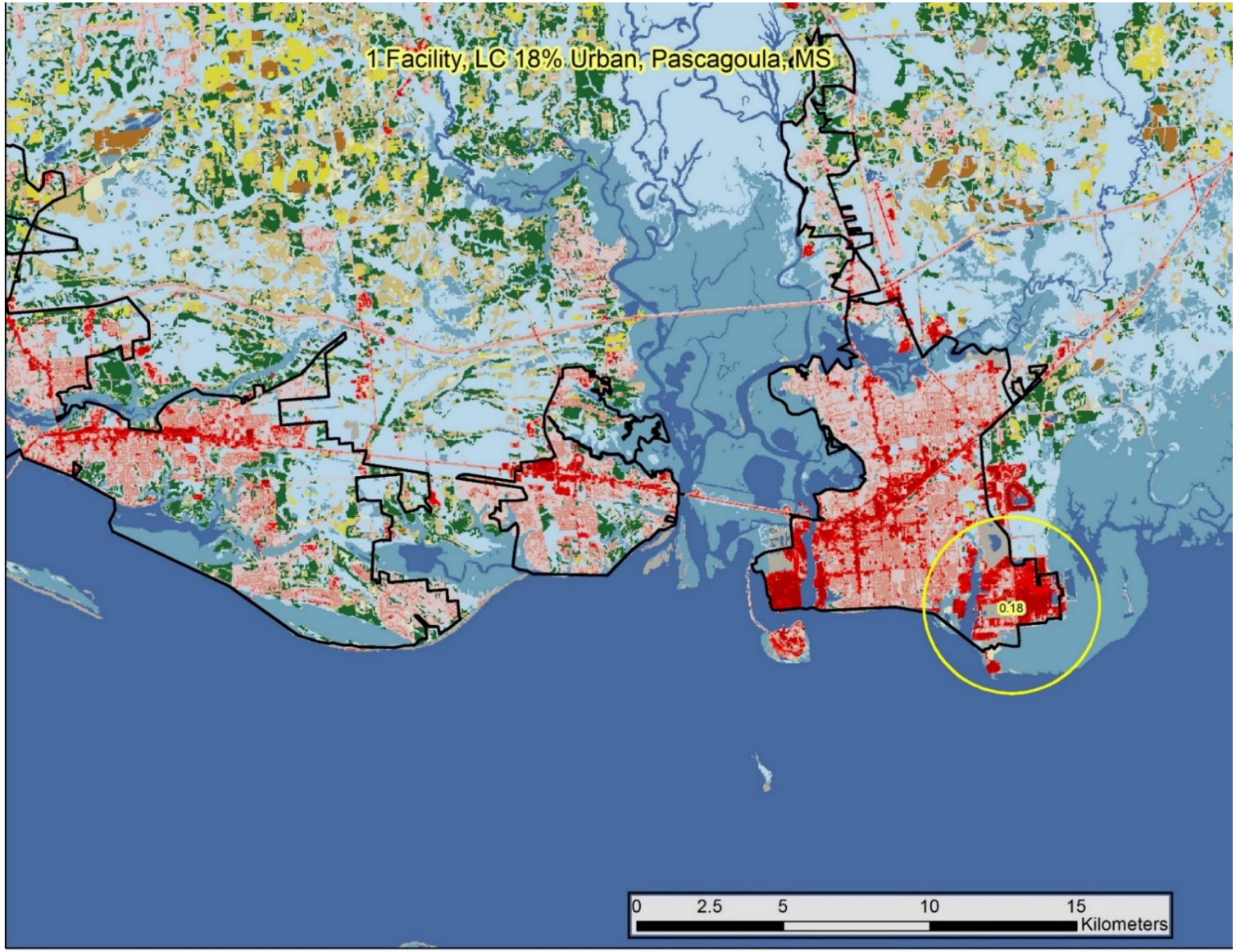
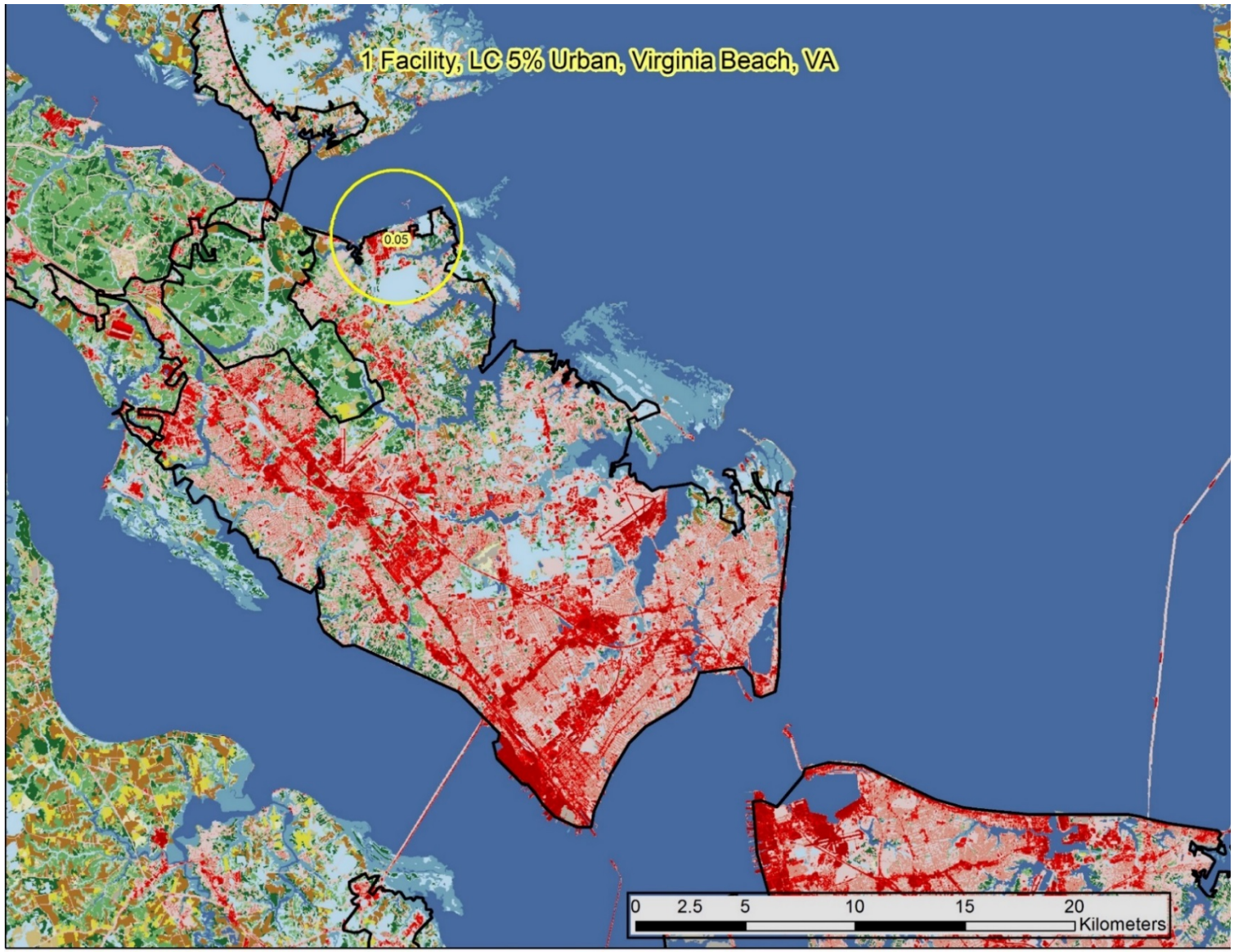


Figure 5-8. Facility with Rural Modeled Concentration 50 Percent Higher Than Urban – Virginia Beach, VA



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5.2 Census Block Receptor Check Tool

Industry sectors (e.g., pulp and paper mills) typically contain multiple facilities (sometimes more than 100), and potentially hundreds or thousands of census block receptors occur near the facilities in an industry sector. The census block centroids are generally good surrogates for where people live within a census block. A census block typically includes about 50 people or 20 households. Census blocks close to a facility are likely to have the maximum risks, and block centroids could be located on facility property.[29] When a block centroid is located on industrial property ("on site"), however, or when a census block is large and the centroid is less likely to be representative of the block's residential locations, the block centroid might not be an appropriate surrogate. In its 2009 review, the SAB also noted this issue with the use of census block centroids and suggested using individual residences as receptors. Although using individual residences is still impractical for a national-scale assessment, we have developed GIS tools to automate a process of identifying potential onsite census block centroids by their proximity to the facility's emission points. The tool also identifies nearby census blocks with large areas for which the centroid is less likely to be representative of residential locations. With this information, we modify the locations of census block centroids and include additional residential receptors where appropriate. Prior to the development of these GIS tools, examining the blocks around each facility to ensure the block centroid adequately represented the location of the population in that block was labor-intensive.

For residual risk assessments, a distance of 300 meters from the source was chosen to evaluate whether census block centroids can be considered as potentially onsite. In other words, if a census block centroid were within 300 meters of an emission source, we examined aerial images of the facility to determine whether the block centroid was likely located on facility property. Selection of the 300-meter distance reflects a compromise between too few and too many blocks identified as potentially on facility property. That is, distances less than 300 meters would identify only block centroids very near the emission sources and could exclude some block

[29]We refer to all census block internal points as centroids. The Census creates a single internal point to represent a block, and for simply shaped blocks that point is the geometric centroid of the block polygon. Where block shapes are complex, the geometric centroid can actually fall outside the block polygon, but the internal point is always located inside the polygon and, in such cases, the internal point is not the geometric centroid of the block polygon. The internal point is not determined based on the locations of actual residences in the block, nor is it located to avoid industrial or commercial activity. Consequently, the internal point can be located on industrial property.

centroids that are 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.

For large blocks, residential locations might not be represented accurately by the single block centroid. Risk estimates based on centroids in large blocks can be understated if residences are located nearer to a facility than the centroid and overstated if they are farther from the facility than the centroid. The census block receptor tool identifies all large census blocks located within 1 km of the facility boundary. Experience from previous risk characterizations shows that, in most cases, the highest modeled concentrations are generally located within 1 km of the facility boundary. A block was considered large if it had an area greater than 2.6 square kilometers, which is the area of a square with sides of about 1,600 meters. The centroid of that square area would be located 800 m from each side, and screening dispersion modeling indicated that concentrations can fall by an order of magnitude over a distance of approximately 800 meters.

Once the large blocks are identified, the aerial images of the blocks within 1 km of the facility are examined. If residential locations cannot be represented by a single receptor (that is, the residences are spread over the block), additional receptors are added for residences nearer to the facility than the centroid.

The census block receptor check tool produces a zipped keyhole markup language (kmz) file that contains multiple layers, including a point layer of the emission sources, and point and polygon layers for the potential onsite (nearby) blocks and large blocks. (The user can specify distances within which to identify nearby blocks and large blocks.) Figure 5-9 illustrates the tool's output file, showing emission points as blue dots, nearby blocks in red outline with the centroid as a red square, and large blocks in yellow outline with the centroid as a yellow square.

Using the map layers created by the tool, the user can quickly verify whether a block centroid is located on facility property or whether it represents the population of the block. When a nearby block is identified as being on facility property, the user can relocate the centroid to a location in the block that is outside facility property and is representative of the block population. In some cases, the block apparently includes no population (even though the census data indicate one or more people live in the block), and in those cases the user can remove the block as a model receptor. When a large block is identified, the user can either move the centroid to be more

representative of the population, add additional receptors (called user receptors), or both. Figure 5-10 is an example of how an onsite census block (indicated by the red square) was moved (indicated by the arrow) to a location off facility property where the population of the block resides. Figure 5-10 also shows how the centroids of the large blocks (yellow squares) were moved to locations more representative of the population. Finally, Figure 5-10 shows how a user receptor (indicated by the yellow-blue cross) was added to one block to represent a residence closer to the facility than the more centrally located centroid.

For both potentially onsite and large blocks, these changes or additions to the centroids are made before the model is run. For changes to census block centroids—either deletions or moving the centroid—the census block file in the HEM-3 model database is revised. For adding user receptors, a spreadsheet of these additions is created, which is then used as a model input file in HEM-3.

Figure 5-9. Example of Census Block Receptor Check Tool Output Layers

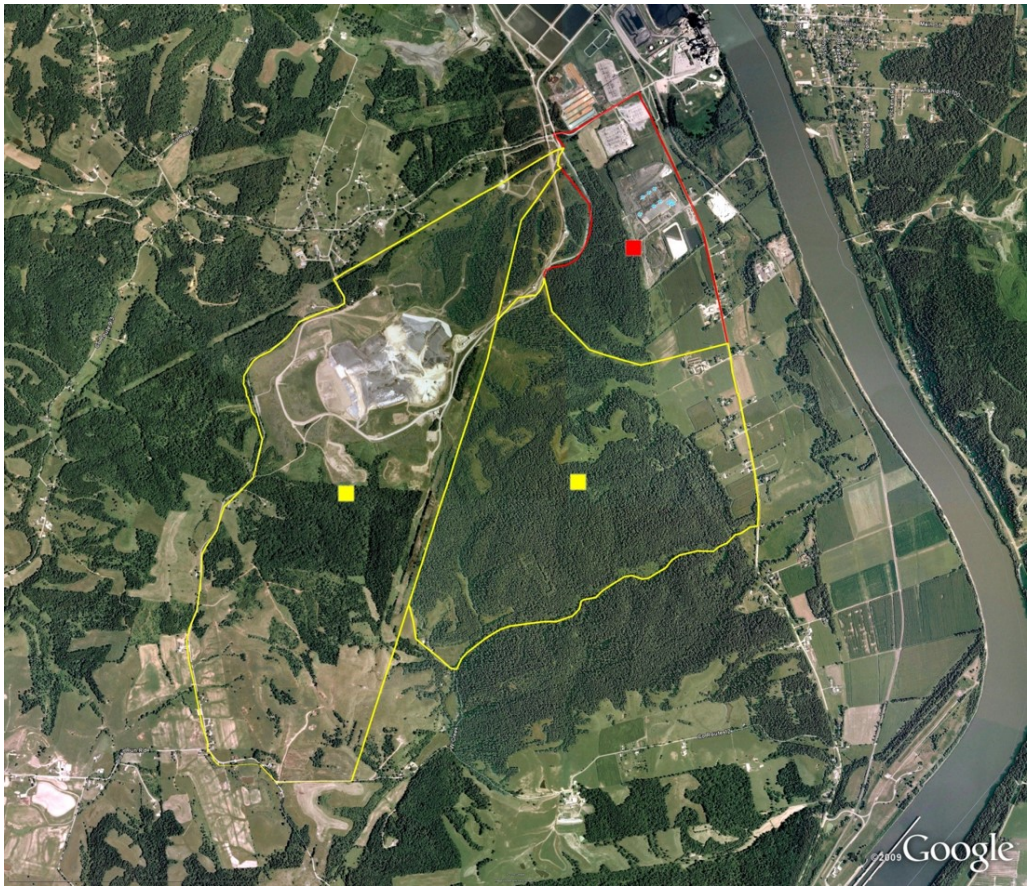
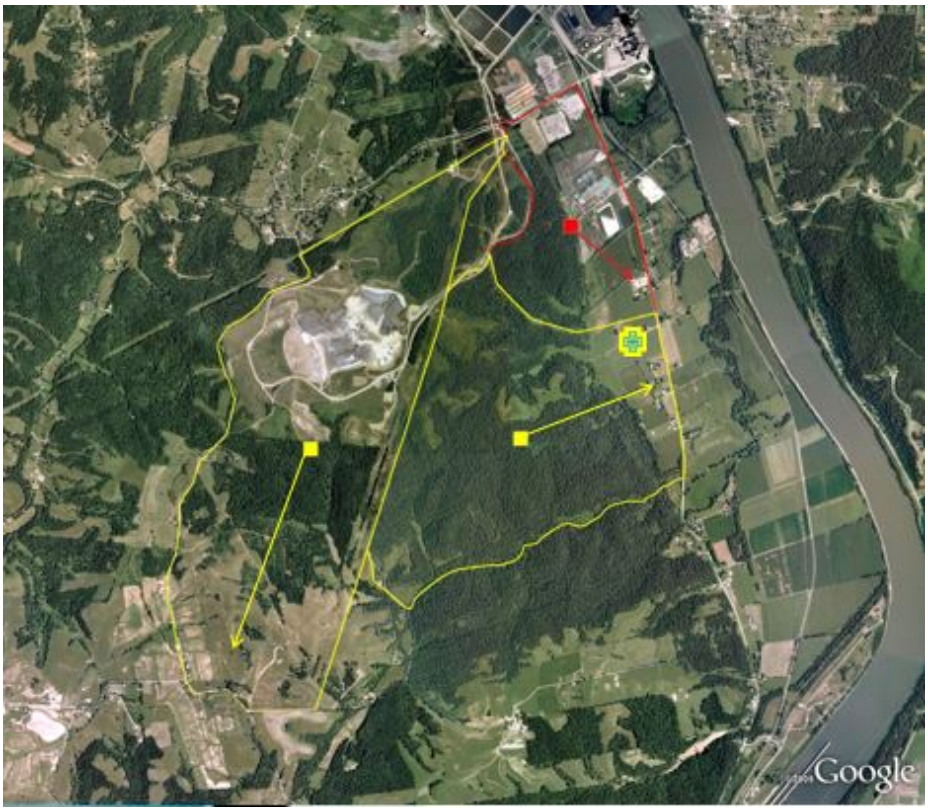


Figure 5-10. Example Illustration of Centroid Changes



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Appendix A. Analysis of Lake Size and Sustainable Fish Population for RTR Risk Screens

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Table A-17. Estimated Maximum Fish-fillet-ingestion Rate (g/day) Associated with Sustainable Fishing of WCC by Lake Size and Total Standing Fish Biomass (TFB) A-41

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
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 (TL1…TL4)
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)

A.1 OVERVIEW AND METHODS

As stated in Section 2.1 of the main report, 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 PB-HAP concentrations in the tissue than the 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 lakes at distances similar to distances for actual lakes near the facility. This appendix provides supporting information for Section 3.2.2.2 of the main report—Accounting for Sustainable Fishing. For the remainder of this appendix, the word "angler" is used to refer to the "human fisher" consuming fish at subsistence levels to distinguish it from the "fisher" (*Martes pennanti*), a small carnivore related to the marten that is used in the ecological risk assessments.

A.1.1 Purpose

To develop the screening scenarios with an angler, we needed to address two questions:

How large does a lake need to be to provide a self-sustaining population(s) of top-trophic-level fish?

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 3.1.1 of main report). 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.

A.1.2 Methods – Literature Searches

ICF searched online bibliographic databases for information on aquatic food webs and biomass distribution within those webs twice: once in 2005 and again in 2014. In 2005, using standard literature/citation databases (e.g., Elsevier BIOBASE, Enviroline), ICF's information specialist searched for articles published from 1975 to 2005, using the following search terms and logical variations:

- 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. When we use a secondary source to describe the findings of an original source, we cite both the primary and secondary sources.

The similar literature search in 2014 identified relevant references published since 2005. This search yielded an initial list of more than 200 publication titles, which 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. When we use a secondary source from our 2014 search to describe the findings of an original source, we provide both the original and secondary citations.

A.1.3 Methods – Food Web Simulated in TRIM.FaTE

The food web simulated in TRIM.FaTE is reproduced in Table A-1. Table A-2 lists values for attributes of each biotic compartment in the TRIM.FaTE-simulated lake. The total fish biomass per unit area simulated by TRIM.FaTE is 5.7 grams fish wet weight/square meter [g ww/m^2], typical of lakes in Maine and southern Ontario. The final two columns in Table A-2 show fish biomass and numbers for the purpose of evaluating fish harvesting by anglers. The total fish

standing biomass of 40 g ww/m^2 is higher than the 5.7 g ww/m^2 for TRIM.FaTE to be more representative of lakes across the United States as described in Section A.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).

Table A-1. Aquatic Food Web Simulated in TRIM.FaTE

Aquatic Biota Compartments (Consumer Groups)	Percentage of Consumer's Diet									Sum of % Total Diet
	Algae	Macrophytes	Zooplankton	Benthic Invertebrates	Water Column Herbivores (WCH)	Benthic Omnivore (BO)	Water Column Omnivore (WCO)	Benthic Carnivore (BC)	Water Column Carnivore (WCC)	
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

Table A-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

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 (K_{ow} (based) Aquatic BioAccumulation Model) for predicting pesticide bioaccumulation potential in aquatic systems).

Of the trophic compartments in Table A-1, two 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. 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 Table A-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 Table A-1). That dietary composition averages to TL2.5, which means that the BC compartment represents TL3.5. For the WCC compartment (TL4), our simplified food web assumes that 100 percent of the WCC diet consists of water-column omnivores (WCO; "pan" fish such as bluegill, other sunfish, white perch). The diet of WCO can include various types of prey, but for a simplified food web, we assume the WCO diet is 100-percent water-column "herbivores" (WCH or planktivorous fish; minnow-sized fish species and young-of-the-year fish). WCH feed in turn on zooplankton that feed on algae (TL1) in the water column. In reality, many fish species (e.g., rainbow trout) feed on smaller fish and invertebrates in both the water column and at the sediment surface, with a web-like flow of energy and contaminants.

In some aquatic models of bioaccumulation, zooplankton are combined with phytoplankton because of their high surface-to-volume ratio and similar chemical accumulation relative to the water column (e.g., Watras and Bloom 1990). Other aquatic bioaccumulation models simulate zooplankton and phytoplankton contaminant uptake separately (e.g., Arnot and Gobas 2004; U.S. EPA 2009). Including zooplankton as a separate compartment in TRIM.FaTE is health protective for the RTR multipathway screens for PB-HAPs.

A.1.4 Organization of This Report

Given the necessity of answering the two questions posed in Section A.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 represents 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 appendix is organized in six sections:

A.2 Angler Behavior

A.3 Fish Populations

A.4 Lake Fish Productivity

A.5 Proportion of Fish Biomass by Trophic Level

A.6 Derivation of Lake Sizes for Sustainable WCC Harvest

A.7 References

A.2 ANGLER BEHAVIOR

Assumptions regarding angler behavior drove some of the data selections and assumptions used to answer our two questions.

A.2.1 Consumption of Top-trophic-level Fish

As stated in Section 3.2.2 of the main report, the angler consumes only top-trophic-level fish. Although the angler might prefer to catch and consume the WCC (TL4) game fish species, individual fish in that group are the least abundant and account for the lowest group biomass of all the fish compartments (Table A-2). Fish in the BC (TL3.5) compartment are more abundant and account for more biomass than the WCC compartment (Table A-2) in most lakes of moderate size (as discussed in Section A.4.3, the Great Lakes are excluded).

We could not predict a priori whether chemical concentrations in the WCC compartment or in the BC compartment would be higher for any given PB-HAP. Depending on chemical K_{ow} (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 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.

A.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 these issues later in Section A.3–Fish Populations, Section A.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/hour 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 screening assessment, interactions among angler effort, fish population size and biomass density, and fishing success are not considered. Instead, we assume certain constants for fish harvesting.

A.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. The U.S. Environmental Protection Agency (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 A.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).

A.3 FISH POPULATIONS

Our initial question in Section A.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 A.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 A.6.

First, some basic principles of fish biology are reviewed (Section A.3.1). Next, a brief overview of fish population modeling is presented (Section A.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 A.3.3). Finally, a sustainable adult fish harvest rate is proposed (Section A.3.4).

A.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 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.

A.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 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 Leslie matrix can be specified by Equation A-1:

$$1 = \sum_{x=1}^{q} l_x m_x e^{-rx} \qquad \text{Eq. A-1}$$

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 0.9957 for lake trout to 0.99985 for walleye. Purchase et al. (2005) found that estimates of r_{max} were sensitive to estimates of early mortality, adult mortality, and growth rate. 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) that 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, approaching 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 for estimating MVP (Section A.3.3) and lake productivity (Section A.4).

A.3.3 General Estimates of MVP

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.

A.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 and 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) 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.

A.4 LAKE FISH PRODUCTIVITY

The first question in Section A.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 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 A.4.1). Empirical models of lake productivity as it relates to measurable lake attributes are presented next (Section A.4.2). Finally, some of the studies that measured fish productivity in specific locations are included to emphasize similarities and differences among lakes (Section A.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 A.5.

A.4.1 Lake Characteristics

Table A-3 provides one summary of physical and chemical characteristics of natural lakes in North America based on a sample of 72 lakes 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. Table A-3 is not meant to summarize the characteristics of lakes across all regions of the United States.

Table A-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 Table A-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[1] once per year in the autumn, whereas northeastern lakes are dimictic, also turning over in the autumn and again 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

[1]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 turnover.

eutrophic (see text box below). Values for several chemical/physical characteristics of lakes that are associated with these categories have been quantified. For example, Table A-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).

Table A-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

Note: Measurements are average, epilimnetic (describing the 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: Waters at a 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.

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 subcatchments within a 275-hectare watershed in Ontario, Canada, Tanentzap et al. (2014) found that nearshore forested and wetland subcatchment areas around Daisy Lake export more organic material to the lake than other subcatchments. 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).

A.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 A.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, also have 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). Table A-5 presents those models along with predictions of total biomass for each group for 5, 10, and 50 µg [TP]/L.

Table A-5. Predictions of Biomass (B) of Biotic Components of Lakes with Different Total Phosphorus (TP) Concentrations

		Biomass (B) (g wet weight/m^2)		
Group	**Equation**	**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).

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 A-2 from Peters (1986):

$$Depth\_of\_Euphotic\_Zone\ (m) = 24 \times TP\ (mg/m^3)^{-0.28} \qquad \text{Eq. A-2}$$

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).

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 A-3):

$$FY = 0.792 + 0.072\ (TP) \qquad \text{Eq. A-3}$$

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 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 A-4 and between standing biomass of benthic invertebrates and fish biomass is shown in Equation A-5.

$$log_{10}(FSB) = 0.708\ log_{10}(TP) + 0.774 \qquad (r^2 = 0.75,\ n = 18) \qquad \text{Eq. A-4}$$

$$log_{10}(FSB) = 5.692\ (M/z) + 28.7 \qquad (r^2 = 0.83,\ n = 20) \qquad \text{Eq. A-5}$$

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 A-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 Table A-6, produced a reasonable match.

Table A-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.

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 A-6.

$$log_{10}(FSB) = 1.07 + 1.14\ log_{10}(TP) \qquad (r^2 = 0.75,\ n = 22) \qquad \text{Eq. A-6}$$

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 Table A-7. At

intermediate TP concentrations, predictions of total fish biomass are similar between the two models.

Table A-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 risk screen, however, we need to assume a single value for fish productivity per unit area where TP concentration is an unknown.

A.4.3 Measured Total Fish Standing Biomass

The empirical models provided in Section A.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 surfaces such as rocks), 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) longer than 25 cm.

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) 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 (Table A-8).

Table A-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 (Table A-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 once per year in August. Most reservoirs were sampled at least once for 2 or more years between 1952 and 1975, with some sampled at least once for 10 to 20 years during that interval.

Leidy and Jenkins (1977) applied adjustment factors to correct for nonrecovery 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.

Table A-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.

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.

Table A-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 United States >202 ha (a)	61	41.3	3.2	133	30.9	NR
Appalachian Reservoirs, United States (b)	22	64.2	3.4	232	55.0	32
DeGray Lake, Arkansas, United States (c)	1	7.5	–	–	–	NR
Ranger and Mouse Lakes, Ontario (d)	2	3.3	2.7	3.8	–	NR
Lakes in United States (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).

A.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.

A.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, fat providing approximately twice as many calories as muscle, and 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 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 A.5.2) and measurements of fish biomass in different trophic groups (Section A.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 destructive.

A.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 appearing most similar to the TRIM.FaTE approach in compartmentalizing fish 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° and 55° latitude, and southern is defined as between 44° and 48° latitude. The model parameterization for "southern" locations would apply only to the more northern 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) 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 Table A-10 for the lake. The diets assigned to each species were not reported in the publications, so cannot be evaluated.

Table A-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, na = not applicable
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 with 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.

Table A-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 Table A-11 are in line with other estimates (see Table A-2 and Section A.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).

Table A-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 productivity (365 kg

annual production/kg average standing biomass) and turnover rates compared with other aquatic compartments. Zooplankton shows the next highest annual productivity rate (33.5 kg/kg), 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 kg/kg annually.

Table A-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/ Standing 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).

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 A.5.3).

A.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 A.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 destructive. Other approaches to inventorying 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 (Table A-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 Table A-12) by benthic or pelagic feeding habits. We pulled the data in Table A-12 from Appendix G in Leidy and Jenkins (1977).

Table A-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 Table A-12 that, on average, 18 percent of the fish biomass across the 61 reservoirs they examined was piscivorous (minimum 14 percent and maximum 24 percent, including both benthic and pelagic species; see bold values in Table A-13).

Table A-13. Proportion of Total Carrying Capacity, Proportion of 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	

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: Calculated from Table A-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 (pike >20 cm, pikeperch >40 cm) in a large, deep, stratifying reservoir in Germany (Table A-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, also listed in Table A-14. We also assigned a likely trophic level to each age/size-class based on our experience with estimating fish trophic levels, although we generally assigned a half-trophic level higher than the labels applied to simple aquatic food chains (U.S. EPA 2000). Our estimate is that the top trophic-level fish probably represent TL4.5 (WCC) and comprised 3.4 percent of the total fish biomass.

Table A-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	21.1	20.3	na	na
Water Column Omnivore (WCO/BO)	3.0	44.7	49.9	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) and biomass included in the WCC/BC group for TL3.5.

Source: Scharf (2008), Table 1.

The next trophic group (WCC/BC) comprised 17.6 percent of the total fish biomass in the Wupper Reservoir in Germany.

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.

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.

A.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 Table A-2, also is consistent with the data presented here.

A.6 DERIVATION OF LAKE SIZES FOR SUSTAINABLE WCC HARVEST

As stated in Section A.1, this appendix provides supporting information for Section 3.2.2.2 of the main report—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 A.6.1. Question 2—*How much fish can be harvested sustainably from lakes of different sizes?*—is answered in Section A.6.2.

A.6.1 Minimum Lake Size for Self-sustaining Population of WCC

As stated in Section A.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 A-7 and A-8. The standing biomass of WCC in a lake is calculated using Equation A-7. The assumption that the WCC fish compartment represents approximately 3.5 percent of the total fish standing biomass was documented in Section A.5.

$$WCC\_SB = Total\_SB \times Fraction\_WCC \qquad \text{Eq. A-7}$$

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 A-7 and the size of the lake (*Lake_Size*), the total number of WCC fish supported in the lake is calculated using Equation A-8:

$$Number\_WCC = (Lake\_Size \times WCC\_SB \times CF_1) / BW_{WCC} \quad \text{Eq. A-8}$$

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 A-7)

CF_1 = Unit conversion factor (4,047 m^2/acre)

BW_{WCC} = Body weight of adult WCC fish (2,000 g ww per individual; assumed)

Based on those two equations, we created a matrix that predicted *Number_WCC* in a lake as a function of both fish biomass per unit area and the overall lake size in Table A-15. The first 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 column in Table A-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 Table A-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 Table A-15 are the number of individual WCC fish predicted for each combination of total fish biomass and lake size.

In Table A-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.

A.6.2 Maximum Fish Ingestion Rate by Lake Size

The likely annual productivity of WCC fish (kg/year) in a lake (Table A-16) is estimated using Equation A-9.

$$Productivity\_WCC = (Lake\_Size \times WCC\_SB \times CF_1) / CF_2 \quad \text{Eq. A-9}$$

Table A-15. Number of WCC Adult Fish Supported by Lake Size (surface area in acres) and by Total Fish Biomass (TFB)

Fish Biomass (g ww/m^2)		Number of Adult Water-column Carnivores (WCC) (by lake surface area in acres)																						
TFB	**WCC**	**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.

Gray 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: Table A-16 and Table A-17 retain the same cell shading as Table A-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.

Table A-16. Total Standing Biomass of WCC Fish (kg) by Lake Size and Total Fish Biomass (TFB)

Fish Biomass (g ww/m^2)		Total Standing Biomass of Water-column Carnivores (WCC) (kg) (by lake surface area in acres)																					
TFB	**WCC**	**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 Table A-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.

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 A-7)

CF_1 = Unit conversion factor 1 (4,047 m^2/acre)

CF_2 = Unit conversion factor 2 (1,000 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 A-10. The equation assumes the angler consumes 50 percent WCC and 50 percent BC, represented by the factor of 2 in Equation A-10:

$$Max\_IR_{(BC+WCC)} = 2 \times (Productivity\_WCC \times FF \times HF \times CF_2) / CF_3 \quad \text{Eq. A-10}$$

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 A-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 (1,000 g/kg)

CF_3 = Unit conversion factor 3 (365 days/year)

Table A-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. Table A-17 is similar to Table 3-10 in the main report, 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.*

Table A-17. Estimated Maximum Fish-fillet-ingestion Rate (g/day) Associated with Sustainable Fishing of WCC by Lake Size and Total Standing Fish Biomass (TFB)

Fish Biomass (g ww/m^2)		Maximum Fish-fillet-ingestion Rate (g/day) for a Diet of 50% BC Plus 50% WCC Fish (by lake surface area in acres)																						
TFB	WCC	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 Table A-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. Note that the BC fish are more abundant; therefore, if the angler can consume 0.013 kg WCC fish/day, they 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 A-9 and A-10).

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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
BTAG	Biological Technical Assistance Group
bw	body weight
CAA	Clean Air Act
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
EC_{10}	effective concentration (10-percent response)
EC_{50}	effective concentration (50-percent response)
Eco-TEFs	ecological toxicity equivalency factors
ESL	ecological screening level
F	fluorine
FIR	food ingestion rate
FMR	free-living metabolic rate
GE	gross energy
GLNPO	Great Lakes National Program Office
GLWQI	Great Lakes Water Quality Initiative
GMAT	geometric mean
HAP	hazardous air pollutant
HCl	hydrogen chloride
HF	hydrogen fluoride
Hg	mercury
Hg^{++}	divalent mercury
HMW	high-molecular weight

IRIS	Integrated Risk Information System
ISQGs	interim sediment quality guidelines
LANL	Los Alamos National Laboratory
LC_{50}	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
NEL	no-effect level
NOAA	National Oceanic and Atmospheric Administration
NOEC	no-observed-effect concentration
NOEL	no-observed-effect levels
OEHHA	California Office of Environment and Health Hazard Assessment
OME	Ontario Ministry for the Environment
ORNL	Oak Ridge National Laboratory
OW	Office of Water
PAH(s)	polycyclic aromatic hydrocarbon(s)
PB-HAP	persistent bioaccumulative HAP
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
RTR	Risk Technology and Review
SAB	Science Advisory Board
SAV	secondary acute value
SCV	secondary chronic value
SESL	soil ecological screening level
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
TRV	toxicity reference value
UF	uncertainty factor
WEFH	Wildlife Exposure Factors Handbook
WQG	water quality guideline
ww	wet weight

B.1 INTRODUCTION

Pursuant to the Clean Air Act (CAA), 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 appendix provides materials supporting EPA's approach to the effects assessment, as described in Section 4.3 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 CAA. 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 appendix is organized in eight sections.

Section B.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 B.3. For POM and dioxins, we discuss derivation of toxicity equivalency factors (TEFs) for each group relative to their index chemicals, benzo[a]pyrene and 2,3,7,8-tetrachlorodibenzo-p-dioxin, respectively, for surface waters, sediment, and surface soils.

Section B.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 B.5. We discuss derivation of ecological toxicity equivalency factors (Eco-TEFs) for POM and dioxins relative to their index chemicals, benzo[a]pyrene and 2,3,7,8-tetrachlorodibenzo-p-dioxin, respectively, for TRVs for birds and mammals.

Section B.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 TRIM.FaTE estimates of chemical concentrations in smaller and larger fish.

Section B.7. Empirical data by which a bioaccumulation factor (BAF) was derived for arsenic in the water column and in benthic sediments are presented.

Section B.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 B.9. This appendix concludes with a list of the references cited.

B.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 4-2 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 4.3 of the main report. Section B.2.1 describes differences between "population-level" and "community-level" benchmarks in more detail than in Section 4 of the main report. Section B.2.2 provides supplemental information supporting the Section 4.3 derivation of ecological benchmarks for PB-HAPs. Section B.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 (SAB) review of the RTR assessment risk screens (U.S. EPA 2009a) and is not repeated here.

B.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 B.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 B.2.1.2). Three of EPA's twelve GEAEs (USEPA 2003a) were not used because established benchmarks are not available (Section B.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 B.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 B.2.1.5).

B.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 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 NAOELs 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.

B.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, 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.

B.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 4-2 of the main report. We evaluated, but did not use, the remaining three EPA GEAEs for the environmental risk screen (Table B-1):

- Animals exposed to airborne HAPs by inhalation,
- Microbial community in soils, and
- Amphibians and reptiles in their respective habitats.

Table B-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

B.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 toxicity reference values 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, EPA Region 4 values, National Oceanic and Atmospheric Administration [NOAA] 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., Office of Water, 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.

B.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-for-effects (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.

B.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 B.2.2.1), the sediment bed in lakes (Section B.2.2.2), and surface soils in terrestrial environments (Section B.2.2.3).

B.2.2.1 Water-Column Benchmarks

For organisms that live primarily in the water-column of aquatic ecosystems, EPA's National Ambient Water Quality Criteria (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 breadth of taxonomic groups required to establish NAWQC, EPA's Office of Water established a Tier II approach (not to be confused with the RTR ecological or human health Tier 2 assessment) that allows 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 is applied to the lowest acute and chronic toxicity value to estimate the Tier II SAVs and SCVs. EPA's Superfund program adopted the Tier II SAV methodology from the GLWQI, but on occasion varied its approach to calculating SCVs from SAVs when chronic aquatic toxicity data were 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 B-2.

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 4-4 of the main report).

Table B-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.

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 4-3 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 4-3 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 Office of Water (OW) decided to publish its NAWQC criteria for MeHg as concentrations in fish rather than as concentrations in water, because measured bioaccumulation factors 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 level) of 0.0028 µg/L and Tier II SAV (acute or PEL) of 0.099 µg/L (U.S. EPA 2015) (listed in Table 4-4

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 benzo[a]pyrene (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 level) 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).[1] 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 2,3,7,8-tetrachlorodibenzo-p-dioxin (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 4-3 of the main report).

B.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 B-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),

[1]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).

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 guidelines (ISQGs) 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).

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.

Table B-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

Sediment Screening Benchmark	mg/kg dw	Rationale for Not Using*
Washington State freshwater NEL	57	higher than other threshold levels
U.S. EPA Region 3 BTAG, freshwater	9.8	lower "threshold" available

Acronyms: AET = apparent effects threshold; ARCS = Assessment and Remediation of Contaminated Sediments (Program); BTAG = Biological Technical Assistance Group (Superfund); CLS = cleanup screening level; 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.

Values in Table B-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 level (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 4-3 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 4-3 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 4-3 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 4-3 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.

B.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 B-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.

Table B-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 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.

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[2] NOAELs for reproduction and growth to be 2.47 mg[As]/kg-day. One study

[2]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.

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 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 B-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 B-5). We calculated a NOAEL and LOAEL for piscivorous birds (in Section B.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 B.4.

Table B-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 B.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 4-3 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 B-6).

Table B-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

In a recent update of its ecological screening benchmarks, Region 4 cited the September 2012 release of the Los Alamos National Laboratory (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 B-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 uncertainty factor 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 B-7).

Table B-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 acceptable toxicant concentration; calculated in this table as the geometric mean of the SESL NOAEL and LOAEL. SESL = soil ecological screening levels

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 B-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 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 B-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 B-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 high-molecular-weight 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 ecological screening level (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.

B.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 4.3.2.2 of the main report). Section B.2.3.1 discusses three distinct approaches to setting limits for plant exposures to HF. Section B.2.3.2 lists existing regulatory benchmarks for HF in the United States and other countries. Section B.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).

B.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 B-8 presents the pros and cons of each method outlined by Hill (1969).

Table B-8. Overview of Three Approaches to HF Environmental Standards (Hill 1969)

Approach	Traditional Use/Benefit	Complicating Factors with HF
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.

B.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 B-9.

Table B-9. Governmental Air Criteria for Hydrogen Fluoride (HF) to Protect Plants

Reference	Specific Information[a]	Air Criteria for Hydrogen Fluoride (µg HF/m^3) 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	–	–

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
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.

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)

The Environment Canada criteria were based on regression analysis of exposure-concentration versus exposure-duration data from the studies shown in Section B.2.3.3 and from additional

unpublished studies.[3] 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. In addition, the value for "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 B-11, below), 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

[3]References cited by Environment Canada (1996) from conference proceedings abstracts or other nonpeer-reviewed/nonpublished sources are not included in this report.

Table B-9. The criteria are based on studies of agricultural crops, horticultural plants, and coniferous trees, as described in Section B.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 B-9. The Texas Commission on Environmental Quality (TCEQ) established effect screening levels for the protection of vegetation, cattle, and human health (TCEQ 2009, Table B-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 B-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 B-9. Section B.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.[4] Thus, the 90-day criteria for plants are lower than the reference exposure limit to protect human health from inhalation toxicity.

B.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).

[4]See http://oehha.ca.gov/air/chronic_rels/HyFluoCREL.html.

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 B-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).

Table B-10. Adverse Effects in Terrestrial Plants Following Short-term Exposures to HF

Species Tested	Study Protocol[a]	Results	LOEL (µg/m^3)	C × D (µg/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)

Species Tested	Study Protocol[a]	Results	LOEL (µg/m^3)	C × D (µg/m^3-d)[b]	Reference
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)
"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	McCune (1969b)
Gladiolus		Reduced growth or yield	6.0[d]	6.0	McCune (1969b)
Corn		Reduced growth or yield	10.5[d]	10.5	McCune (1969b)
Tomato		Increased foliar markings	12[d]	12	McCune (1969b)

[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 B-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 B-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 B-11) than for short-term exposures.

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 B-11).[5] Several studies of plants of agricultural importance are described in more detail below.

Table B-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)

[5]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.

Species Tested	Study Protocol	Results	LOEL (µg/m³)	Reference
Oat	0.01, 2.2, 4.3, or 9.1 µg F/m³, 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/m³, 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/m³ 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/m³ for 5 mo OR intermittent fumigation with 0.3 µg HF/m³ (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/m³ 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/m³ 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/m³ for 83 d	Fruit set reduced by 80%	0.54	Hill and Pack (1983)
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)

Species Tested	Study Protocol	Results	LOEL (µg/m³)	Reference
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).

Pack and Sulzbach (1976) fumigated nine species of agricultural crops with HF gas from seed through flowering to the time of harvest. Table B-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 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 B-1:

$$I = (A \times L)^{0.5} \quad \text{Eq. B-1}$$

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 B-10) and freesia, grapevine, and eucalyptus (Table B-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.

B.3 WILDLIFE TOXICITY REFERENCE VALUES

To assess risks to piscivorous wildlife, a toxicity reference value (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 B.2.2.3 and Section B.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 B.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 uncertainty factor (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 an uncertainty factor 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).

B.3.1 Derivation of TRVs for Piscivorous Wildlife in RTR Assessment

As described in Section 4.3.1.1 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 food ingestion rate 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 uncertainty factors, 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 B-2 below):

$$Dose_{wildlife} = Dose_{test\text{-}species} \times (BW_{test\text{-}species}/BW_{wildlife})^{1/4} \quad \text{Eq. B-2}$$

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.

B.3.2 Chemical-specific Wildlife TRVs for PB-HAPs

In the main report, Table 4-3 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.

B.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 an uncertainty factor 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 an uncertainty factor 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

B.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 food ingestion rates 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 4-4 in the main report). Given the similarity in size between mallard (1 kg) and American merganser (1.27 kg), no dose conversions were estimated.

B.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 B.3.2.4).

B.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 food ingestion rate 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 × food ingestion rate/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 food ingestion rate 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

B.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.

B.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

B.4 DERIVATION OF ECOLOGICAL TEFS FOR POM AND DIOXIN BENCHMARKS

Section B.4.1 covers the derivation of TEFs for POM relative to benzo[a]pyrene (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 B.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.

B.4.1 TEFs for POM for Surface Water, Sediments, and Soils

Table B-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 B-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

Polyaromatic Organic Matter	CAS RN	Surface Water TEF	Soil TEF	Sediment TEF	Mammalian TEF
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
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 B-12 are described in Sections B.4.1.1 through B.4.1.5. Chemical-specific derivations are described in Section B.4.1.6.

B.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 B.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 B-3 from an experimentally measured octanol-water partitioning coefficient(s) (K_{ow}) for the specified chemical (provided suitable empirical data are available):

$$SQB = f_{oc} \times K_{oc} \times WQB \qquad \textbf{Eq. B-3}$$

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.

B.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.

B.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.

B.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 $\log K_{ow}$ values over 6 are too large to readily penetrate membranes despite their lipophilicity. For example, the maximum $\log K_{ow}$ 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 $\log K_{ow}$ 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 B.4.1.6.

B.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).

B.4.1.6 TEFs for POM for Surface Waters, Soils, and Sediments

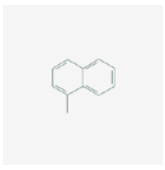
For POM chemicals for which we identified appropriate benchmarks to compare with the BaP benchmarks for surface water, sediment, or soil, we used the ratio of the benchmarks to calculate a TEF appropriate for the ecological assessment endpoint and environmental medium. Benchmarks were not available, however, for many individual POM chemicals. We therefore needed to use original toxicity tests to estimate TEFs.

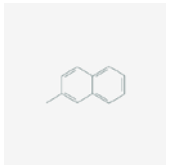
To calculate TEFs from individual toxicity tests requires a different approach than estimating TEFs from community benchmarks already calculated on the basis of many different species' tests and possibly some "uncertainty" factors based on available data. We compared individual ecotoxicity tests for 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 B-12, where

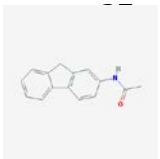
benchmarks were not already established. For each POM, we include its CAS Registry Number, its logK_{ow}, and a schematic of its chemical structure to help the reader follow our rationale.

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.

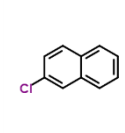
1-Methylnaphthalene (CAS # 90-12-0): LogK_{ow} = 3.87. In freshwater, the 4-day LC_{50} for fathead minnow (static test) is 9,000 µg/L (Mattson et al. 1976), compared with the BaP-exposed bluegill (*Lepomis macrochirus*) 4-day LC_{50} of 5 µg/L (Wu et al. 2016). The ratio of those values yields an acute 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.







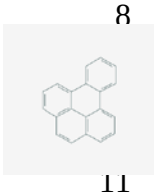
2-Acetylaminofluorene (CAS # 53-96-3): An aromatic amine with LogK_{ow} = 3.28. TEFs for surface water, soil, and sediment are from Region 5 ESL benchmarks relative to Region 5 ESLs for BaP for each medium, respectively (U.S. EPA 2003c).



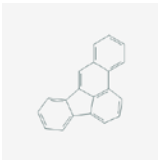
2-Chloronaphthalene (CAS # 91-48-7): $LogK_{ow}$ = 4.14. TEFs for surface water, soil, and sediment are from Region 5 ESLs relative to Region 5 ESLs for BaP for each medium, respectively (U.S. EPA 2003c).

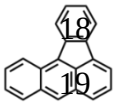
3-Methylcholanthrene (CAS # 56-49-5): $LogK_{ow}$ = 6.42. TEFs for surface water, soil, and sediment are from Region 5 ESLs relative to Region 5 ESLs for BaP for each medium, respectively (U.S. EPA 2003c).

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.

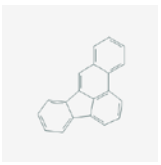


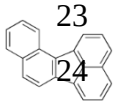
Benzo[a]fluoranthene (BaF, CAS # 203-33-8): $LogK_{ow}$ = 6.11, which is higher than the $logK_{ow}$ for benzo[b]fluoranthene (BbF, CAS # 205-99-2, $logK_{ow}$ = 5.78, figure to the right). BaF also has a higher $logK_{ow}$ than benzo[k]fluoranthene (BkF, $logK_{ow}$ 5.94) and a lower $logK_{ow}$ than 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).





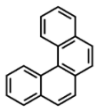
Benzo[j]fluoranthene (BjF, CAS # 205-82-3): $LogK_{ow}$ = 6.4, which is higher than for the other benzofluorenes. All benzo[x]fluorenes have the same molecular weight (252.3 g/mole). No toxicity data were identified for BjF for water, sediment, or soils. TEFs were set equal to those for BbF (figure right).



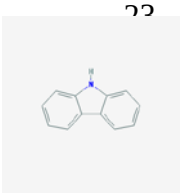


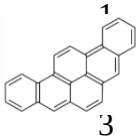
Benzo[g,h,i]fluoranthene (CAS # 203-12-3): $LogK_{ow}$ = 5.52 (molecular weight 226.3 g/mole). With $logK_{ow}$ close to 5.5, we used EPA EPI-Suite ECOSAR Version 4.11 (U.S. EPA 2012b) to estimate four freshwater TEFs: 48-hour LC_{50} for daphnids, 96-hour LC_{50} value for fish, chronic value for daphnids, 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.

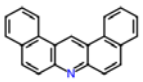


Carbazole (CAS # 86-74-8): $LogK_{ow}$ = 3.72. In freshwater, fathead minnow 4-day LC_{50} (flow-through design) = 930 µg/L (Brooke 1991) compared with BaP exposed bluegill (*Lepomis macrochirus*) 4-day LC_{50} of 5 µg/L (Wu et al. 2016) for a TEF of 0.0054. For soils, earthworm 28-day LC_{50} = 106/2 (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.



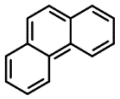


Dibenzo[a,i]pyrene (DaP, CAS #189-55-9): $LogK_{ow}$ = 7.28. Extremely low solubility. We assigned the same TEFs as indeno[1,2,3-c,d]pyrene, which has a K_{ow} value of 6.72. Aquatic toxicity benchmarks are higher than the limit of solubility.

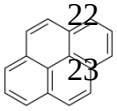


Dibenzo[a,j]acridine (CAS # 224-42-0): $LogK_{ow}$ = 5.63. With $logK_{ow}$ under 6.0, we used EPA EPI-Suite ECOSAR Version 4.11 to estimate four freshwater TEFs as explained for benzo[c]phenanthrene (above). The highest of the four corresponding TEFs (chronic fish TEF of 0.43) represents the surface 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.

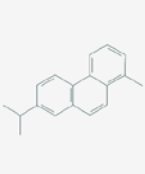
Perylene (CAS # 198-55-0): $LogK_{ow}$ = 5.82 (although the MSDS, http://datasheets.scbt.com/sc-206007.pdf, states 6.25 for $logK_{ow}$). Fish 4-day LC_{50} values range from 1.1 to 5.0 (MSDS). In a daphnid acute test, 0.6 µg/L kills 50% of individuals in 0.764 days (LT_{50} in renewal system). Thus, 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).

B.4.2 TEFs for Dioxins for Surface Water, Sediments, and Soils

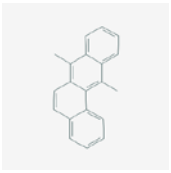


Table B-13 presents the TEFs for dioxins for surface water, soils, and sediments. Surface water TEFs are based on 1998 World Health Organization (WHO) TEFs for fish (from Van den Berg et al. (1998) as presented in Table 4 of *Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated 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.

Table B-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

Congener	CAS RN	Surface Water TEF	Soil TEF	Sediment TEF	Mammalian TEF[a]	Avian TEF
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).

B.5 TEFS FOR WILDLIFE TRVS

This section describes the derivation of toxicity equivalency factors (TEFs) for the wildlife TRVs for POM and for dioxins.

B.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).

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 (IRIS). 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 B-14 presents the data used to estimate TEFs for the RTR multipathway POM for mammalian wildlife. Insufficient data were available to calculate TEFs for POM for birds. The mammalian TEFs also are included in Table B-12 for comparison with other benchmark TEFs.

Table B-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

Compound (surrogate PAH)	CAS RN	NOAEL/LOAEL[a] (mg/kg-d)	Species/Effect (exposure regimen) [notes]	NOAEL-based TEF	Reference
7,12-Dimethylbenz[a] anthracene	57-97-6	ND	Set = BaP reproduction	1	none identified
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

Compound (surrogate PAH)	CAS RN	NOAEL/LOAEL[a] (mg/kg-d)	Species/Effect (exposure regimen) [notes]	NOAEL-based TEF	Reference
Dibenzo[a,j]acridine	224-42-0	ND	Less toxic than BaP	0.1	none identified
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.

B.5.2 TEFs for Wildlife for Dioxins

To estimate TEFs for dioxins mammals and birds (listed in Table B-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.

B.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 Section 4 of 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 food ingestion rate, 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 free-living metabolic rate 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 B.6.1) and American merganser (Section B.6.2).

B.6.1 Mink Exposure Factor Values and Assumed Diet

For mink (*Mustela vison or Neovison vison*), none of the measured food ingestion rates 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 B-15). The estimate of a free-living metabolic rate of 245 kcal/kg bw/day in Table B-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).

Although the TRV derived for mink assumed a 1-kg mink (Sample et al. 1996), which is representative of both eastern and western races of mink, a lower weight (0.8 kg) from the Midwest was used to estimate mink exposures to PB-HAPs in fish for the environmental risk screen. The lower body weight results in a slightly more conservative (i.e., higher) food ingestion rate as a proportion of body weight (i.e., 22.5 percent instead of 21.8 percent).

Table B-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)

Parameter	Value	Comments/References
Food assimilation efficiency (AE) for mammal consuming fish	0.91	U.S. EPA (1993c), Table 4-3
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 B-15). Based on the energy requirements (FMR) of mink and the ME per unit wet-weight prey, a food ingestion rate (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 B-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 B-16 shows the resulting FIR in three different units.

Table B-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 and assimilation efficiency 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.

B.6.2 Merganser Exposure Factor Values and Assumed Diet

For American merganser (*Mergus merganser americanus*), a few measured food ingestion rates 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,[6] we estimated a food ingestion rate 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 food ingestion rate equals 419 grams of fish, wet weight, per day per merganser (Table B-17).

Table B-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 B-18, are based on the reported lengths of fish caught in Michigan (Alexander 1977), with some consideration of studies from

[6]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.

other locations (e.g., White 1936, 1937; Huntington and Roberts 1959) and experimental choice studies (Latta and Sharkey 1966).

Table B-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
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).

B.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 B.7.1), bioaccumulation factors (Section B.7.2), and biota-sediment accumulation factors (Section B.7.3).

B.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 B-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 B-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)

Habitat	mg[As]/kg[fish dry weight]	mg[As]/kg[fish wet weight]	Species/location	Reference
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 trimethyl arsenic (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.

B.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 bioaccumulation factor 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., trophic levels 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 (OAQPS). 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 B-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]),[7] although most freshwater fish contained less than 200 µg[As]/kg[fish ww] across fish species, trophic levels, and sizes (Table B-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

[7]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.

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 B-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 B-20.

Table B-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		

Fish species, condition	Study Type	µg[As]/ L[water]	BAF or BCF (L/kg)	Location	Reference
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		
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 B-4) is close, with an r^2 of 0.82.

$$\textit{Field\_BAF}\ (\text{L/kg}) = 87.4 * \textit{Water\_Concentration}\ (\mu\text{g/L})^{-0.925} \qquad \textbf{Eq. B-4}$$

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 B-5; Williams et al. 2006). The relationship is close ($r^2 = 0.79$).

$$Lab\_BCF\ (L/kg) = 78.7 * Water\_Concentration\ (\mu g/L)^{-0.669} \quad \textbf{Eq. B-5}$$

The smaller exponent suggests that internal arsenic regulation might be impaired at higher water concentrations.

The trends shown in Table B-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 B-4 and Equation B-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).

B.7.3 BSAFs for Arsenic in Freshwater Benthic Invertebrates and Fish

As stated in Section 3.2 of the main report, 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 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.

B.8 ENVIRONMENTAL SCREENING THRESHOLD EMISSION RATES

As described in Section 4 of 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 B-21. The methods of changing thresholds for Tiers 2 and 3 also are described in Section 4 of the main report.

Table B-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

PB-HAP	Assessment Endpoint	Benchmark and Effect Level[a]	Tier 1 ESTER (TPY)
		Threshold Level – Invertebrate Community	1.27E+01
	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 methyl mercury (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
	Sediment Community	Threshold Level	2.08E+00

PB-HAP	Assessment Endpoint	Benchmark and Effect Level[a]	Tier 1 ESTER (TPY)
		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.

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