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Office of Chemical Safety and
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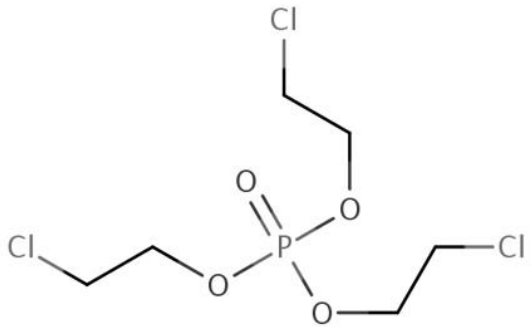
United States
Environmental Protection Agency

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP)

CASRN 115-96-8



September 2024

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As part of an intra-agency review, the TCEP Risk Evaluation was provided to multiple EPA Program Offices for review. Comments were submitted by Office of the Administrator/Office of Children's Health Protection, Office of Air and Radiation, Office of General Council, ORD, and Office of Water.

Docket
Supporting information can be found in the public docket, Docket IDs EPA-HQ-OPPT-2018-0476 and EPA-HQ-OPPT-2023-0265.

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This risk evaluation was reviewed and cleared for release by OPPT and OCSPP leadership.

EXECUTIVE SUMMARY

EPA has evaluated tris(2-chloroethyl) phosphate, or TCEP, under the Toxic Substances Control Act (TSCA). In this risk evaluation, **EPA has determined that TCEP presents an unreasonable risk of injury to human health and the environment under the conditions of use.**

In December 2019, EPA designated TCEP as a high-priority substance for TSCA evaluation and in August 2020 released the final scope of the risk evaluation. This risk evaluation assesses human health risk to workers, consumers, and the general population, as well as risk to the environment.

Although U.S. production of TCEP has decreased by about 99 percent since 2014, it is still used domestically as an additive flame retardant and plasticizer in polymers used in aerospace equipment and products and as an additive flame retardant in paint and coating manufacturing. In the past, TCEP was used in many products made in the United States, including fabrics and textiles, some types of foam, and construction materials—some of which may remain in use today. TCEP may still be found in goods that are imported into the United States.

Because TCEP is mixed into but not chemically bonded to materials, it can leach out of products and into the environment. TCEP that is released into the environment from manufacturing processes or leaching from products primarily ends up in water, sediment, soil, or dust. TCEP may leach out of materials dumped in landfills and reach groundwater or surface water—in particular from landfills that do not have an adequate liner system. It can also be released into the air from manufacturing, industrial processes, and open burning. If TCEP enters the atmosphere, it can be deposited in lakes and rivers through rain and snowfall. It can be transported long distances via air and water and has been detected in the Arctic. TCEP concentrations may be even higher indoors than outdoors because it can leach out of consumer products such as carpets or wooden furniture and attach to household dust. Although TCEP is persistent in the environment (*i.e.*, it does not easily degrade) and has been detected in organisms such as fish exposed to TCEP in surface water, it does not appear to bioaccumulate. This is because TCEP does not accumulate in people or animals at greater concentrations than exist in the environment.

Following the 2023 Draft Risk Evaluation, and in response to public and peer reviewer comments, EPA made the following key updates to the risk evaluation for TCEP:

1. Revised four existing conditions of use (COUs) to include "automotive articles and replacement parts containing TCEP" and added one new COU: "Industrial use – paints and coatings – paints and coatings," in response to the Auto Alliance comments.
2. Revised qualitative assessments of COUs not quantified in the draft risk evaluation based on similarity of exposure scenarios to COUs that have been quantified.
3. Updated the peer-reviewed literature search in February 2024 to fill data gaps for landfills (general population, consumer, and environmental hazard), environmental hazard, epidemiology, and inhalation (human health/animal toxicity).
4. Increased the range of the (Hazardous Waste) Delisting Risk Assessment Software (DRAS) analysis, bounding leachate concentration by solubility, and increasing loading rates to account for past disposal practices.
5. Revised concentration(s) of concern (COCs) for chronic aquatic hazards for both sediment and surface water compartments.
6. Changed the evidence integration/conclusion for developmental toxicity from likely to cause the effect to suggestive for the effect.

7. Summarized and evaluated epidemiology studies identified in the updated literature search and by peer reviewers for neurotoxicity as well as kidney, immune/hematological, thyroid, and lung/respiratory effects, body weight changes, developmental toxicity, and cancer.
8. Updated the human health risk characterization with revised analysis of the Consumer Exposure Model (CEM) for consumers to account for model updates and minor changes to parameter inputs.
9. Revised the analysis for Consumer paints and coatings COU, including consumer sensitivity analysis for minimum weight fraction values and revised dermal and oral ingestion estimates from soils to include soil concentrations from the Biosolids Screening Tool (BST) analysis.

Unreasonable Risk to Human Health

EPA concluded that kidney cancer, as well as neurological and reproductive systems and non-cancer effects in kidneys, are the key human health hazards from exposure to TCEP based on evaluation of human epidemiological studies and laboratory animal testing (see Sections 5.2.3.1 and 5.2.4). The Agency evaluated the risks of potential neurological effects from acute exposure, reproductive effects from intermediate and chronic exposures, and increased kidney cancer as relevant from exposure to TCEP at work, in the home, by breastfeeding, and by eating fish taken from TCEP-contaminated water. When determining unreasonable risk of TCEP to human health, EPA also accounted for potentially exposed and susceptible subpopulations—pregnant women, infants exposed through human milk, children and adolescents, people who experience high exposures or exposures from multiple routes (such as dermal and inhalation), people who live in fenceline communities near facilities that release TCEP, firefighters, and people and Tribes whose diets include large amounts of fish (see Section 5.3.3).

Workers with the greatest potential for exposure—both dermal and inhalation—to TCEP are those who spray TCEP-containing paints or coatings, or workers who are involved in processing a 2-part resin used in paints, coatings, and polyurethane resin castings for aerospace applications (see Section 5.3.2.1). Outside the workplace, adults, infants, and children may be most at risk if they breathe or ingest TCEP released from fabrics, textiles, foam, and wood products and that either attaches to dust or otherwise gets into indoor air (see Section 5.3.2.2). Infants and children may be at risk if they "mouth" products containing foam, textiles, or wood that contain TCEP (see Section 5.3.2.3). People who are subsistence fishers may be at risk if they eat TCEP-contaminated fish. Tribal people for whom fish is important dietarily and culturally have greater risk than the general population and subsistence fishers due to increased fish consumption (see Section 5.3.3).

EPA's assessment shows 10 of the 21 conditions of use to significantly contribute to the unreasonable risks presented by TCEP due to cancer and non-cancer health effects for (1) workers who handle or apply liquid formulations containing TCEP (due to dermal and inhalation exposures) as well as workers who fight structural fires; (2) people who breathe or ingest dust from TCEP released from consumer products; and (3) people who eat large amounts of fish contaminated with TCEP. For workers, there are certain activities where acute, intermediate, chronic, and lifetime exposures to TCEP, especially from contact with skin and inhalation of mists generated during the use of paints and coatings, contribute to unreasonable risk. Outside the work environment, TCEP presents unreasonable risk to adults, children, and infants with acute, intermediate/chronic, and lifetime exposure to TCEP—mainly from breathing or ingesting TCEP-containing dust or eating TCEP-contaminated fish. TCEP presents unreasonable risk to children and infants with acute and intermediate/chronic exposure from mouthing consumer products that contain TCEP. EPA also assessed whether breast-feeding infants of mothers from occupational, consumer, general population, subsistence fisher, and Tribal exposure scenarios were at higher risk than their mothers and determined that they are not.

Unreasonable Risk to the Environment

EPA assessed TCEP exposures to the aquatic environment when it leaches or is released into water through the manufacturing, processing, or use of TCEP or TCEP-containing materials. Aquatic hazard data were available for six fish species, four invertebrate species, five algal species, and from predictive models for sediment-dwelling organisms. **EPA's assessment shows that chronic exposure to TCEP to fish and to sediment-dwelling organisms under all six conditions of use that were quantitatively evaluated for risk to the environment significantly contribute to the unreasonable risk**. The Agency, however, has determined that acute exposure to TCEP does not present unreasonable risk to aquatic organisms (vertebrate and invertebrate species). Data on soil invertebrates and mammals indicate that acute and chronic exposure to TCEP does not present unreasonable risks to land-dwelling animals.

Considerations and Next Steps

The COUs evaluated for TCEP are listed in Table 1-1. The following 10 COUs significantly contribute to the unreasonable risk:

- Manufacturing (import);
- Processing – Incorporation into formulation, mixture, or reaction product – Paint and coating manufacturing;
- Processing – Incorporation into formulation, mixture, or reaction product – Polymers used in aerospace equipment and products;
- Processing – Incorporation into article – Aerospace equipment and products and automotive articles and replacement parts containing TCEP;
- Industrial use – Paints and coatings;
- Commercial use – Paints and coatings;
- Commercial use – Laboratory chemicals;
- Consumer use – Furnishing, cleaning, treatment/care products – Fabric and textile products;
- Consumer use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products; and
- Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites.

The following 11 COUs do *not* significantly contribute to the unreasonable risk:

- Processing – Recycling;
- Distribution in commerce;
- Industrial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP;
- Commercial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP;
- Commercial use – Furnishing, cleaning, treatment/care products – Fabric and textile products;
- Commercial use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products;
- Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation;
- Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites;
- Consumer use – Paints and coatings, including those found on automotive articles and replacement parts;

- Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation; and
- Disposal.

It is important to emphasize that the estimates of risk in the TCEP evaluation include assumptions and modeled predictions around which there are varying levels of uncertainty. That stated, the totality of information and weight of scientific evidence give EPA confidence that under the known, intended, and reasonably foreseen COUs that are subject to evaluation and regulation under TSCA, TCEP presents unreasonable risks to human health and the environment under the conditions of use.

Following issuance of this completed risk evaluation for TCEP, EPA will initiate risk management for TCEP by applying one or more of the requirements under TSCA section 6(a) to the extent necessary so that TCEP no longer presents an unreasonable risk. The risk management requirements will likely focus on the COUs significantly contributing to the unreasonable risk. However, under TSCA section 6(a), EPA is not limited to regulating the specific COUs found to significantly contribute to unreasonable risk and may select from among a suite of risk management options related to manufacture, processing, distribution in commerce, commercial use, and disposal to address the unreasonable risk.

1 INTRODUCTION

EPA has evaluated tris(2-chloroethyl) phosphate (TCEP) under the Toxic Substances Control Act (TSCA). TCEP is primarily used as an additive flame retardant and plasticizer in polymers used in aerospace equipment and products and as an additive flame retardant in paint and coating manufacturing. In the past, TCEP was processed in many products made in the United States, including fabrics and textiles, some types of foam, and construction materials—some of which may still be in use today. TCEP may also be imported in articles intended for consumer use. Section 1.1 provides production volume, life cycle diagram (LCD), conditions of use (COUs), and conceptual models used for TCEP; Section 1.2 includes an overview of the systematic review process; and Section 1.3 presents the organization of this risk evaluation. Figure 1-1 describes the major inputs, phases, and outputs/components of the TSCA risk evaluation process, from scoping to releasing the risk evaluation.

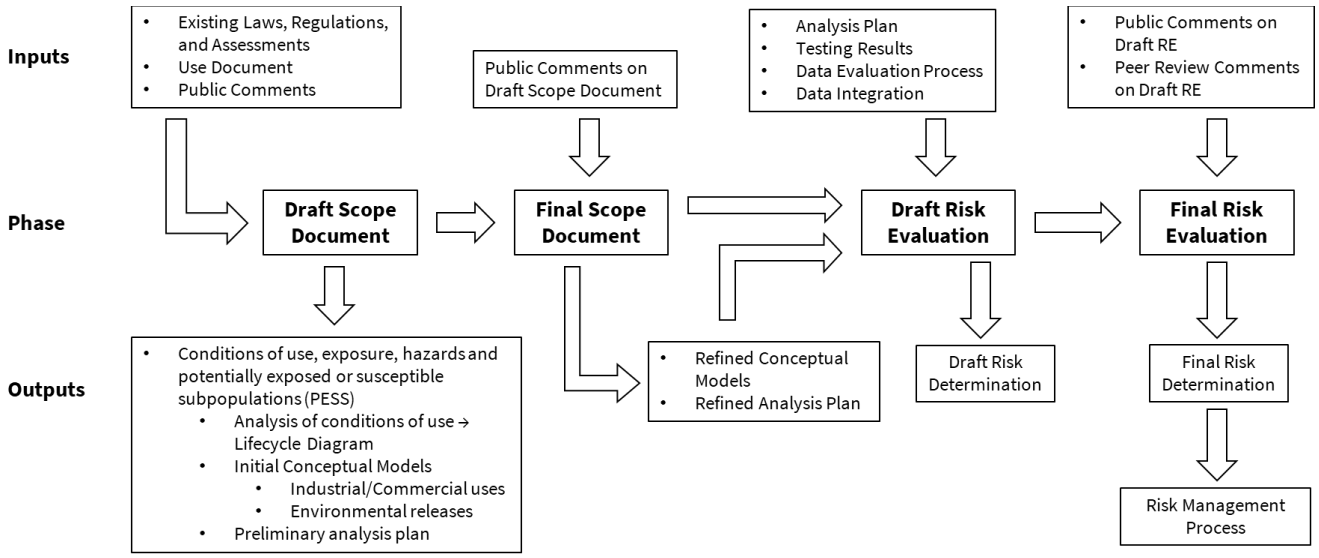


Figure 1-1. TSCA Existing Chemicals Risk Evaluation Process

1.1 Scope of the Risk Evaluation

EPA evaluated risk to human and environmental populations for TCEP. Specifically for human populations, the Agency evaluated risk to (1) workers and occupational non-users (ONUs) via inhalation and oral routes; (2) workers via dermal routes; (3) consumers via inhalation, dermal, and oral routes; and (4) the general population via oral, dermal, and inhalation routes. In this risk evaluation, the general population includes various subpopulations such as subsistence fishers, Tribal populations, and people who live in fenceline communities who live near facilities that emit TCEP. For environmental populations, EPA evaluated risk to (1) aquatic species via water and sediment, and (2) terrestrial species via air and soil leading to dietary exposure.

1.1.1 Life Cycle and Production Volume

The LCD shown below in Figure 1-2 depicts the COUs that are within the scope of the risk evaluation during various life cycle stages, including manufacturing, processing, use (industrial, commercial, consumer), distribution, and disposal. The LCD has been updated since it was included in the TCEP final scope document (U.S. EPA, 2020b) to correspond with minor updates to the COUs. The information in the LCD is grouped according to the Chemical Data Reporting (CDR) processing codes and use categories, including functional use codes for industrial uses and product categories for industrial, commercial, and consumer uses. The CDR Rule under TSCA requires U.S. manufacturers

(including importers) to provide EPA with information on the chemicals they manufacture or import into the United States. EPA collects CDR data approximately every 4 years with the latest collections occurring in 2006, 2012, 2016, and 2020, with reporting thresholds of 25,000 lb.

Descriptions of the industrial, commercial, and consumer use categories identified from the CDR are included in the LCD (Figure 1-2) (U.S. EPA, 2016d). The descriptions provide a brief overview of the use category; the Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S. EPA, 2024n) contains more detailed descriptions (*e.g.*, process descriptions, worker activities, process flow diagrams, equipment illustrations) for each manufacture, processing, use, and disposal category.

Because TCEP is also known to co-occur in formulation with other flame retardants, such as 2,2-bis(chloromethyl)-propane-1,3-diyltetrakis(2-chloroethyl) bisphosphate (V6), this risk evaluation evaluates TCEP when it co-occurs with other flame retardants in commercial and consumer products (*e.g.*, when it co-occurs with V6). However, it does not evaluate the other flame retardants.

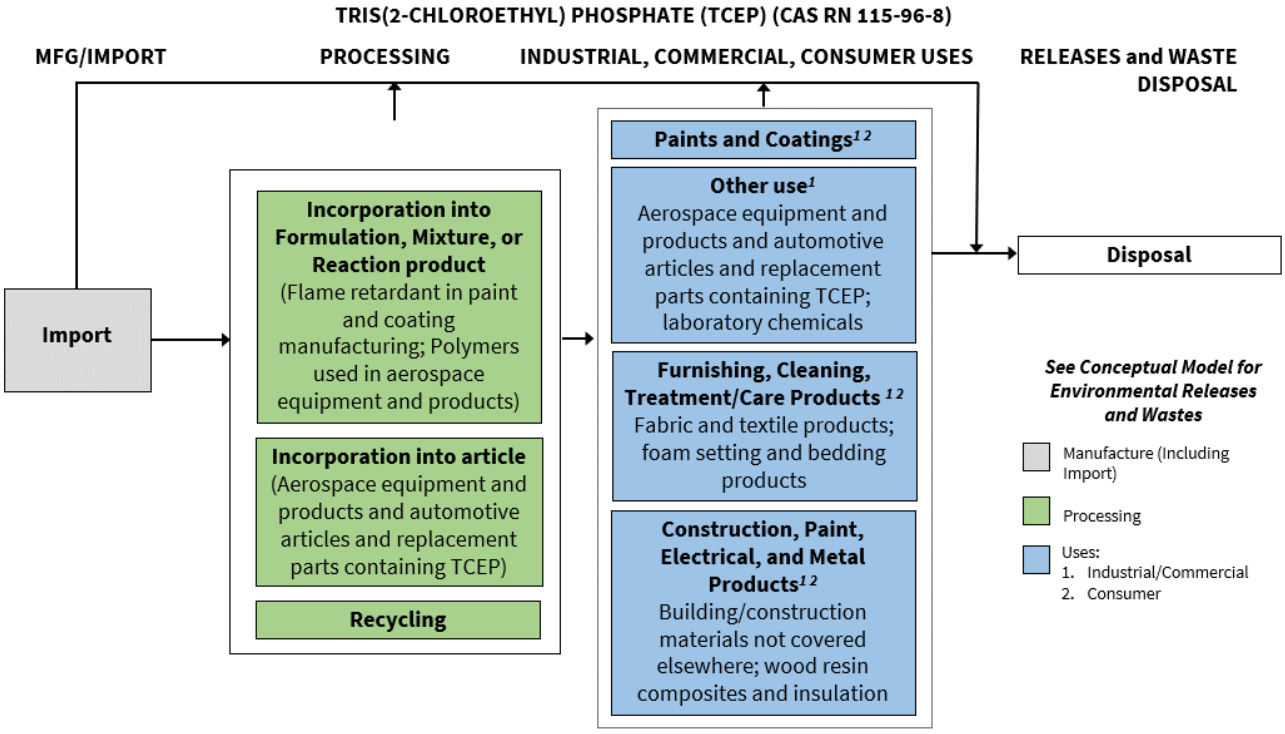


Figure 1-2. TCEP Life Cycle Diagram

[1] Due to lack of reasonably available data, including current CDR data, EPA cannot differentiate between import and processing sites.

[2] See Table 1-1 for additional details on TSCA conditions of use.

As evident in Figure 1-3, import, production volume, and uses of TCEP in the United States have curtailed in recent years. Although CDR data show production volumes for TCEP in chemical form in the tens of thousands of pounds from 2012 to 2015, the most recent updated 2020 CDR data showed that no company reported the manufacture (including import) of TCEP in the United States from 2016 to 2020. However, the reporting threshold for TCEP in CDR is 25,000 lb and some manufacturing could be

occurring below that threshold (U.S. EPA, 2020a).[1] The production volumes for TCEP reported to CDR for years 2012 to 2015 were all from one company, Aceto Corporation, a chemical manufacturer and supplier importing TCEP in chemical form. Aceto Corporation indicated to EPA that TCEP was imported and used as a flame retardant for unsaturated polyester resins and for aircraft furniture (U.S. EPA, 2020b). Note that prior to 2012, production volume in CDR was reported in ranges. From 1986 to 2002, the production volume reported to CDR (previously known as the Inventory Update Rule, or IUR) was between 1 and 10 million lb. In 2006, the production volume reported was between 500,000 and 1 million lb and in 2011 the production volume was withheld.

To supplement the CDR data, EPA also considered Datamyne import volume information that shows 593 lb of TCEP imported in 2020. Descartes Datamyne is a commercial searchable trade database that covers the import-export data and global commerce of more than 50 countries (across 5 continents) and includes cross-border commerce of the United States with over 230 trading partners (Descartes, 2020). The trade data are gathered from the U.S. Customs Automated Manifest System. Since 2014, total imports of TCEP in chemical form range in volume over the time from approximately 96,823 lb (in 2014) to 593 lb (in 2020) (Descartes, 2020). Note that for 2014, the Aceto Corporation data is included in the total production volume for CDR and Datamyne. For 2020, Sigma-Aldrich Corp reported the 593 lb.[2]

The 2016 CDR reporting data and Datamyne import volume data for TCEP in chemical form are provided in Figure 1-3. TCEP imported in articles is not captured in these data. Note, EPA only recently added TCEP to the Toxics Release Inventory (TRI) with the first year of reporting from facilities due July 1, 2024. As of September 2024, there are no TRI entries for TCEP.

[1] Note that because CDR generally does not include information on impurities or manufacturing solely in small quantities for research and development, and because small manufacturers are exempt from 2020 CDR reporting, some manufacturing could be occuring at small manufacturers. However, EPA does not consider domestic manufacturing of TCEP to be reasonably foreseeable. Lastly, TCEP imported in articles would not be captured in CDR.

[2] Due to the nature of Datamyne data, some shipments containing TCEP may be excluded due to being categorized under other names that were not included in the search terms. There also may be errors in the data that prevent shipment records containing the chemical from being located. Datamyne does not include articles/products containing the chemical unless the chemical name is included in the description; however, based on descriptions provided on the bills of lading, Figure 1-3 provides an estimate of the volume of TCEP imported as the chemical (not in an identified product) from 2012 to 2020.

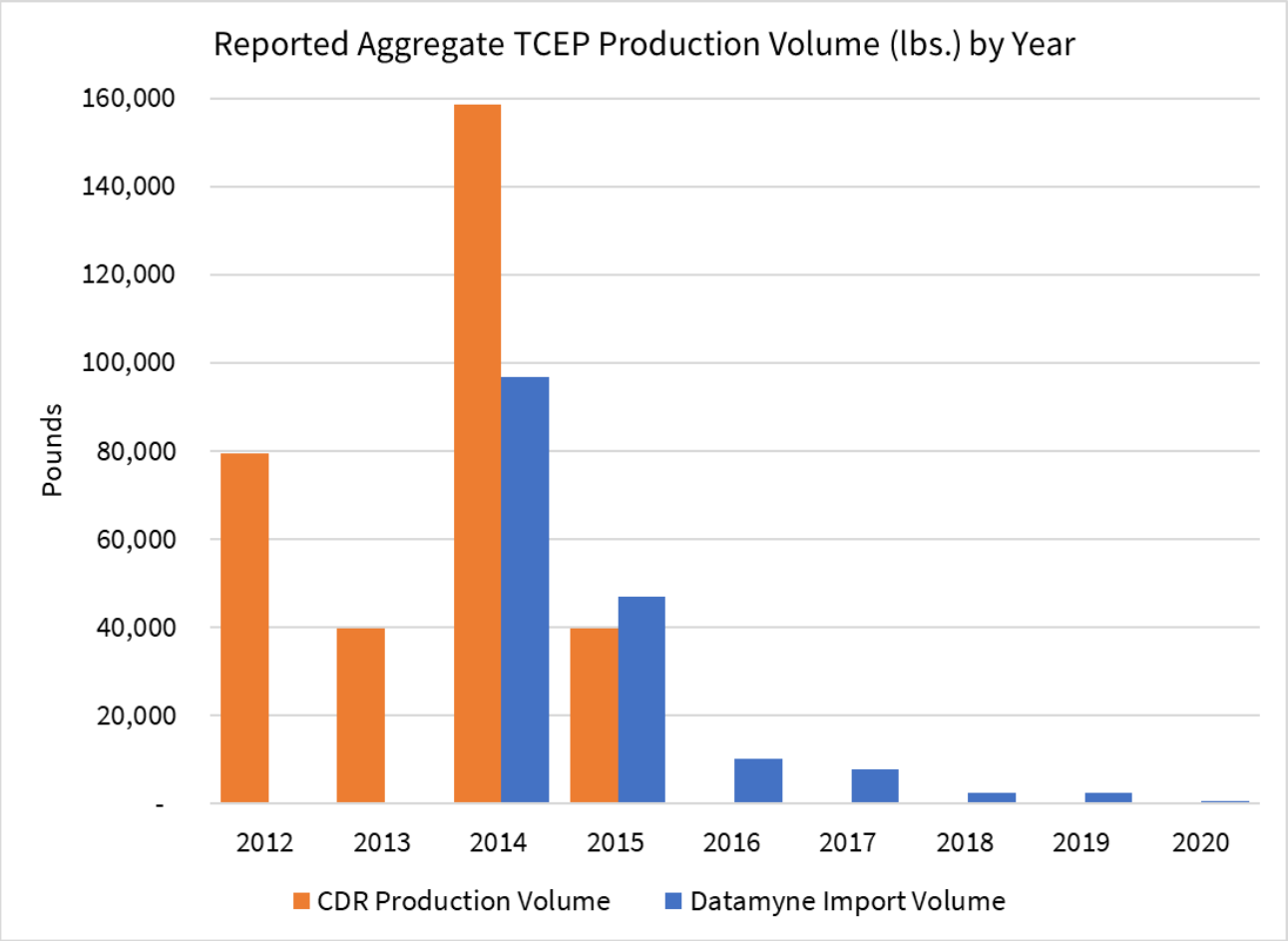


Figure 1-3. Reported Aggregate TCEP Production Volume (lb) 2012–2020
Note: CDR data for the 2016 reporting period is available via ChemView. Because of an ongoing CBI substantiation process required by TSCA, the CDR data available in this risk evaluation is more specific than currently provided in ChemView (U.S. EPA, 2019a). For 2014, Aceto Corporation's production volume is included in both the CDR data and the Datamyne data.

Given the uncertainties in the current production volume for TCEP, EPA used two production volumes in its analyses for this risk evaluation: 2,500 and 25,000 lb. The 2,500 lb production volume is used as a more realistic estimate reflecting current production volumes, while 25,000 lb is used as an upper bound estimate based on the 2020 CDR reporting threshold. There are several reasons why EPA considers 2,500 lb to be a more realistic production volume. First, the decreasing aggregate TCEP production volumes according to CDR and Datamyne, as shown in Figure 1-3, suggest that the production volume is now somewhere below the 2020 CDR reporting threshold of 25,000 lb, with Datamyne showing 593 lb of TCEP imported in 2020 and generally the most recent Datamyne information (2017–2020) in the low thousands of pounds or lower. Additionally, EPA received public comments (EPA-HQ-OPPT-2018-0476-0041) on the final scope document (U.S. EPA, 2020b) confirming industry's transition away from the domestic use of TCEP.

Communication with industry further supported the declining use of TCEP as many companies have since discontinued or reformulated products that contained TCEP, even though TCEP is still in use for several commercial and consumer COUs (EPA-HQ-OPPT-2018-0476-0056). However, there is no federal ban on the manufacture, process, or use of TCEP that would prevent production volumes from increasing again (see Appendix B for the regulatory history of TCEP). Therefore, EPA used these two

production volumes to characterize what is possible and what is realistic given reasonably available information. Given EPA's research, the 25,000 lb upper bound production volume is believed to be an overestimate of current production volumes in the United States. For these reasons, the 2,500 lb production volume is used throughout this risk evaluation as EPA has more confidence that it is reflective of current production volumes. Estimates using the upper bound of 25,000 lb are presented in appendices and supplemental files.

1.1.2 COUs Included in the Risk Evaluation

The *Final Scope of the Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) CASRN 115-96-8* (U.S. EPA, 2020b) identified and described the life cycle stages, categories and subcategories that comprise COUs that EPA planned to consider in the risk evaluation. All COUs for TCEP included in this risk evaluation are reflected in the LCD (Figure 1-2) and conceptual models (Section 1.1.2.1). Table 1-1 below presents all COUs for TCEP.

In this risk evaluation, EPA made edits to the COUs listed in the final scope document. These edits reflect EPA's improved understanding of the COUs based on further outreach and public comments received, which have been added to the reference(s) column of Table 1-1. Changes include removing "flame retardant" as the exclusive functional use in the processing conditions of use; editing industrial and commercial use in "aircraft interiors and products" to "aerospace equipment and products"; and improved the description of the COU to avoid using the "products not covered elsewhere" description from the CDR reporting codes. EPA did not receive public comments on additional commercial uses that fall into the "Other use" category aside from laboratory chemicals, the Agency removed "*e.g.,*" from the COU, "Commercial use – other use – *e.g.,* laboratory chemicals."

All COUs assessed in this Risk Evaluation are considered on-going uses. However, there are several COUs for which part of the life cycle has ceased, such as manufacturing (including import) and processing. However, other parts of the life cycle including recycling, commercial or consumer use, and disposal are on-going. These COUs are identified in Table 1-1 and include four COUs for commercial use and five COUs for consumer use.

EPA reviewed comments submitted for docket EPA-HQ-OPPT-2023-2012-0001 Significant New Use Rule for Certain Non-ongoing Uses; Flame Retardants, for certain non-ongoing uses of TCEP and made corresponding edits to Table 1-1, including the addition of a new COU, Industrial use – Paints and coatings, and "automotive articles and replacement parts containing TCEP" to some processing, industrial, and commercial uses. Appendix D contains a description of the COUs.

Table 1-1. Conditions of Uses in the Risk Evaluation for TCEP

Life Cycle Stage[a]	Category[b]	Subcategory[c]	Reference(s)
Manufacturing	Import	Import	U.S. EPA (2016d)
Processing	Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	U.S. EPA (2019a); Duratec (2018); PPG (2010); PPG (2016); U.S. EPA (2017c) Flame Control Coatings_meeting memo
	Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	EPA-HQ-OPPT-2018-0476-0015; EPA-HQ-OPPT-2018-0476-0012; BJB Enterprises (2017); EPA-HQ-OPPT-2018-0476-0045; Summary of email exchanges
	Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	EPA-HQ-OPPT-2018-0476-0006; EPA-HQ-OPPT-2018-0476-0045; Boeing meeting memo; EPA-HQ-OPPT-2023-0265-0043; EPA-HQ-OPPT-2023-0012-0010
	Recycling	Recycling	U.S. EPA (2019a)
Distribution in Commerce	Distribution in commerce	Distribution in commerce	
Industrial Use	Other use	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	EPA-HQ-OPPT-2018-0476-0006; Boeing meeting memo; EPA-HQ-OPPT-2023-0265-0043; EPA-HQ-OPPT-2023-0012-0010
	Paints and coatings	Paints and coatings	EPA-HQ-OPPT-2023-0265-0043; EPA-HQ-OPPT-2023-0012-0010
Commercial Use	Other use	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	EPA-HQ-OPPT-2018-0476-0006; EPA-HQ-OPPT-2023-0265-0043; EPA-HQ-OPPT-2023-0012-0010
	Paints and coatings	Paints and coatings	U.S. EPA (2019a); Alliance for Automotive Innovation
	Laboratory chemicals	Laboratory chemical	TCI America (2018)
	Furnishing, cleaning, treatment/care products	Fabric and textile products[d]	EPA-HQ-OPPT-2018-0476-0015
	Furnishing, cleaning, treatment/care products	Foam seating and bedding products[d]	Stapleton et al. (2011); Stapleton & Hammel meeting memo
	Construction, paint, electrical, and metal products	Building/construction materials – insulation[d]	EPA-HQ-OPPT-2018-0476-0015; EPA-HQ-OPPT-2018-0476-0041; EC (2009), citing IARC (1990)

Life Cycle Stage[a]	Category[b]	Subcategory[c]	Reference(s)
Commercial Use	Construction, paint, electrical, and metal products	Building/construction materials – wood and engineered wood products – wood resin composites[d]	EC (2009), citing IARC (1990), OECD (2006) and IPCS (1998)
Consumer Use	Paints and coatings	Paints and coatings[d], including those found on automotive articles and replacement parts	U.S. EPA (2019a); Alliance for Automotive Innovation; EPA-HQ-OPPT-2023-0265-0043; EPA-HQ-OPPT-2023-0012-0010
	Furnishing, cleaning, treatment/care products	Fabric and textile products[d]	EPA-HQ-OPPT-2018-0476-0015
	Furnishing, cleaning, treatment/care products	Foam seating and bedding products[d]	Stapleton et al. (2011); Stapleton & Hammel meeting memo
	Construction, paint, electrical, and metal products	Building/construction materials – insulation[d]	EPA-HQ-OPPT-2018-0476-0015; EPA-HQ-OPPT-2018-0476-0041; EC (2009), citing IARC (1990)
	Construction, paint, electrical, and metal products	Building/construction materials –wood and engineered wood products – wood resin composites[d]	EC (2009), citing IARC (1990), OECD (2006), and IPCS (1998)
Disposal	Disposal	Disposal[e]	

[a] Life Cycle Stage Use Definitions (40 CFR 711.3)
– "Industrial Use" means use at a site at which one or more chemicals or mixtures are manufactured (including imported) or processed.
– "Commercial Use" means the use of a chemical or a mixture containing a chemical (including as part of an article) in a commercial enterprise providing saleable goods or services.
– "Consumer Use" means the use of a chemical or a mixture containing a chemical (including as part of an article, such as furniture or clothing) when sold to or made available to consumers for their use.
– Although EPA has identified both industrial and commercial uses here for purposes of distinguishing scenarios in this document, the Agency interprets the authority over "any manner or method of commercial use" under TSCA section 6(a)(5) to reach both.
[b] These categories of COU appear in the LCD, reflect CDR codes, and broadly represent COUs of TCEP in industrial and/or commercial settings and for consumer uses.
[c] These subcategories reflect more specific COUs of TCEP.
[d] Domestic manufacturing and processing for these COUs has ceased.
[e] This COU includes associated disposal of those COUs for which domestic manufacturing and/or processing have ceased.

1.1.2.1 Conceptual Models

The conceptual model in Figure 1-4 presents the exposure pathways, exposure routes, and hazards to human populations from industrial and commercial activities and uses of TCEP. Figure 1-5 presents the conceptual model for consumer activities and uses, Figure 1-6 presents general population exposure pathways and hazards for environmental releases and wastes, and Figure 1-7. presents the conceptual model for ecological exposures and hazards from environmental releases and wastes.

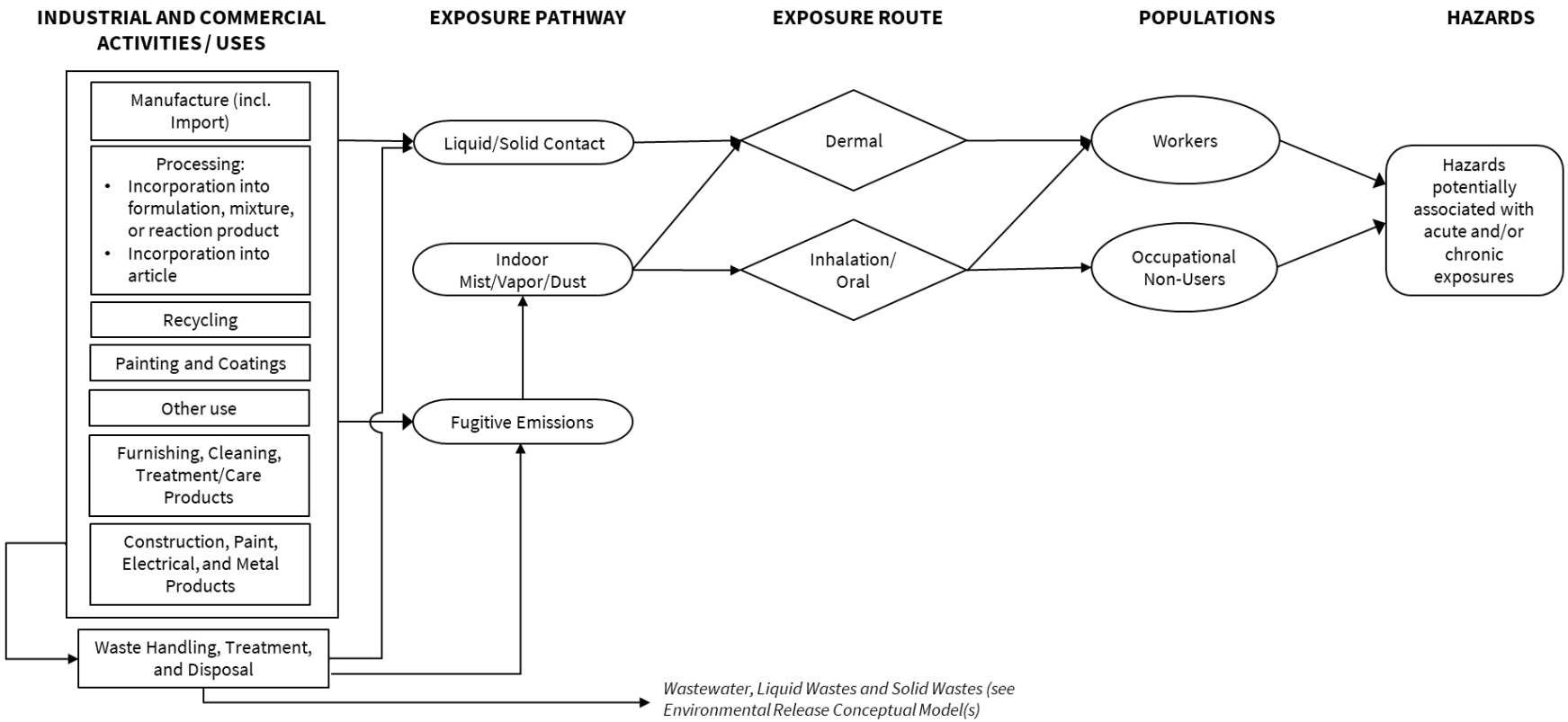


Figure 1-4. TCEP Conceptual Model for Industrial and Commercial Activities and Uses: Potential Exposure and Hazards
The conceptual model presents the exposure pathways, exposure routes, and hazards to human populations from commercial activities and uses of TCEP.

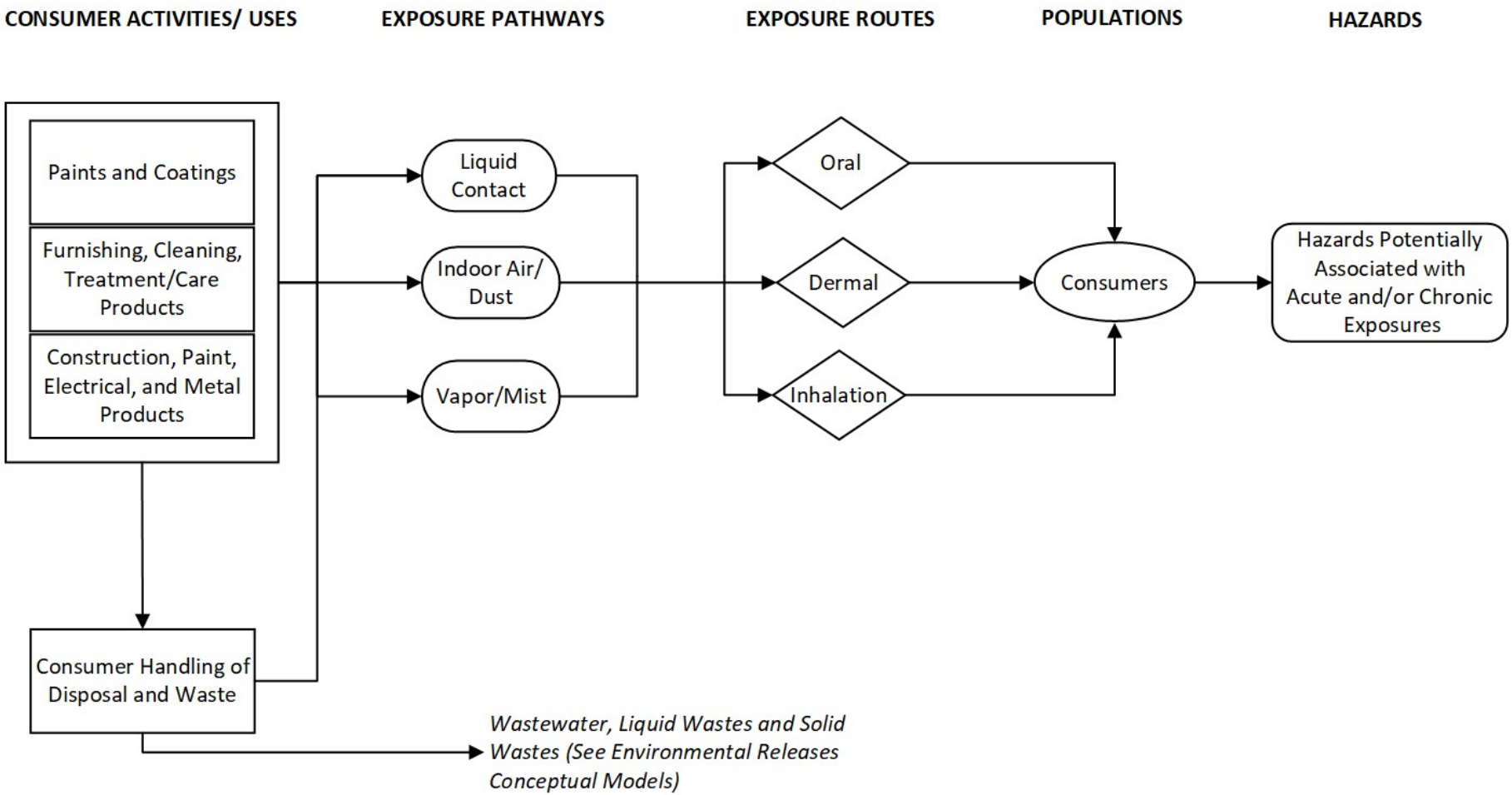


Figure 1-5. TCEP Conceptual Model for Consumer Activities and Uses: Potential Exposures and Hazards
The conceptual model presents the exposure pathways, exposure routes, and hazards to human populations from consumer activities and uses of TCEP.

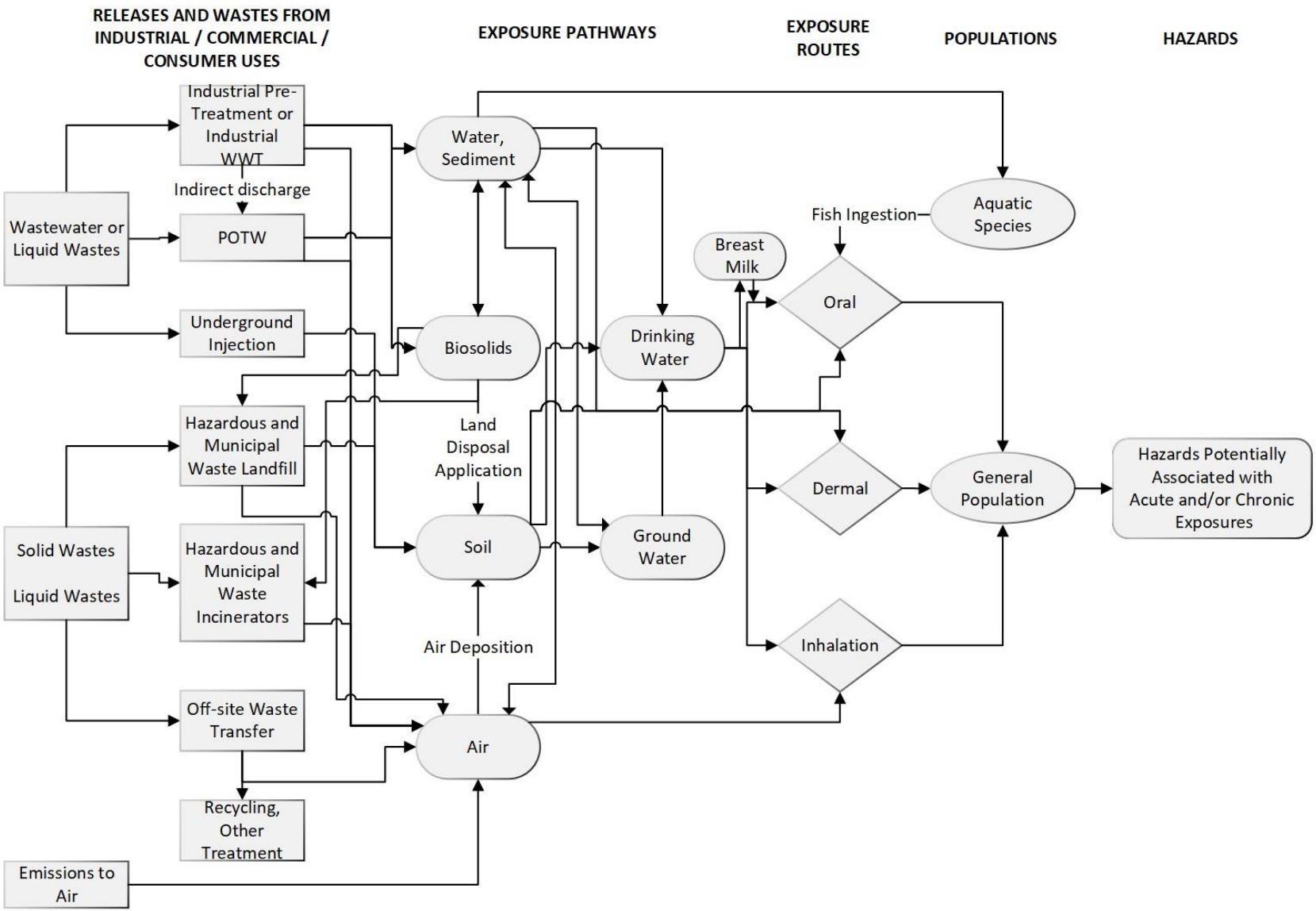


Figure 1-6. TCEP Conceptual Model for Environmental Releases and Wastes: General Population Hazards
The conceptual model presents the exposure pathways, exposure routes, and hazards to human populations from releases and wastes from industrial, commercial, and/or consumer uses of TCEP.

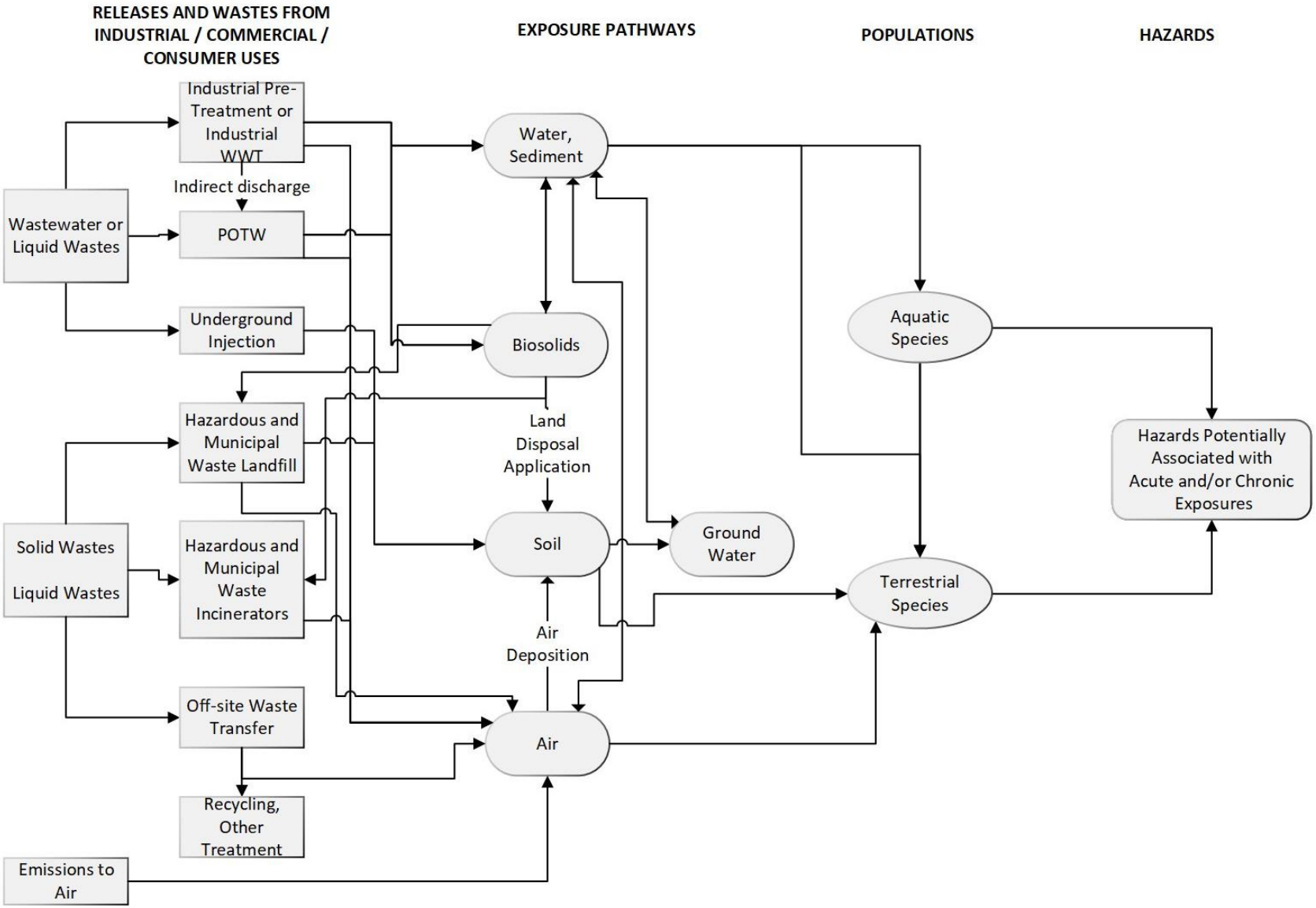


Figure 1-7. TCEP Conceptual Model for Environmental Releases and Wastes: Ecological Exposures and Hazards
The conceptual model presents the exposure pathways, exposure routes, and hazards to environmental populations from releases and wastes from industrial, commercial, and/or consumer uses of TCEP.

1.1.3 Populations Assessed

Based on the conceptual models presented in Section 1.1.2.1, Figure 1-8 presents the human and ecological populations assessed in this risk evaluation. Specifically for humans, EPA evaluated risk to workers and ONUs via inhalation routes and risk to workers via dermal routes; risk to consumers via inhalation, dermal, and oral routes; risk to the general population via oral, dermal, and inhalation routes. For environmental populations, EPA evaluated risk to aquatic species via water and sediment, and risk to terrestrial species via air, soil, and water leading to dietary exposure. Human health risks were evaluated for acute, intermediate, chronic, and lifetime exposure scenarios as appropriate, and environmental risks were evaluated for acute and chronic exposure scenarios, as applicable based on reasonably available exposure and hazard data as well as the relevant populations for each. All consumers of products containing TCEP were considered users of those products, and bystanders were not assessed separately because all the consumer COUs assessed were article scenarios. For the purposes of article exposures, consumers and bystanders are considered the same.

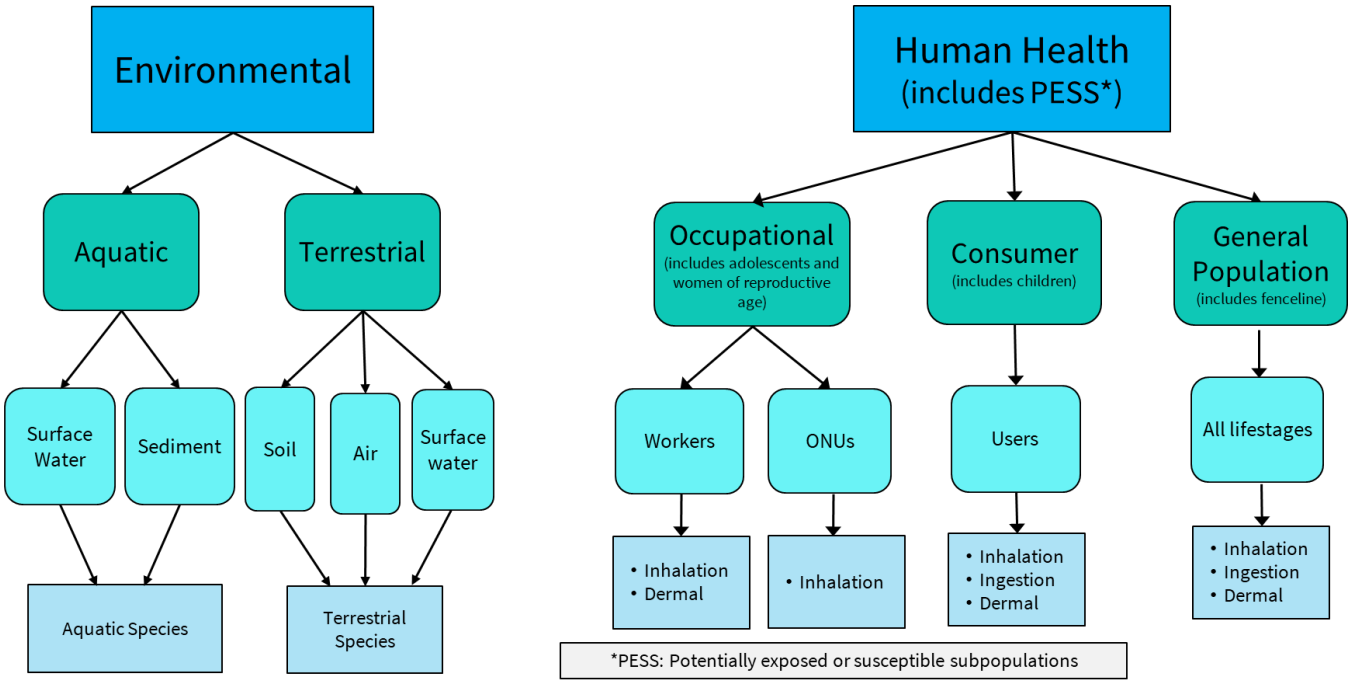


Figure 1-8. Populations Assessed in this Risk Evaluation

1.1.3.1 Potentially Exposed or Susceptible Subpopulations

TSCA section 6(b)(4)(A) requires that risk evaluations "determine whether a chemical substance presents an unreasonable risk of injury to health or the environment, without consideration of costs or other non-risk factors, including an unreasonable risk to a potentially exposed or susceptible subpopulation identified as relevant to the risk evaluation by the Administrator, under the conditions of use." TSCA section 3(12) states that "the term '*potentially exposed or susceptible subpopulation*' means a group of individuals within the general population identified by the Administrator who, due to either greater susceptibility or greater exposure, may be at greater risk than the general population of adverse health effects from exposure to a chemical substance or mixture, such as infants, children, pregnant women, workers, or the elderly."

This risk evaluation considers potentially exposed or susceptible subpopulations (PESS) throughout the human health risk assessment (see Section 5). Considerations related to PESS can influence the selection

of relevant exposure pathways, the sensitivity of derived hazard values, the inclusion of particular human populations, and the discussion of uncertainties throughout the assessment. Evaluation of the qualitative and quantitative evidence for PESS begins as part of the systematic review process, where any available relevant published studies and other data are identified. If adequate and complete, this evidence informs the derivation of exposure estimates and human health hazard endpoints/values that are protective of PESS.

EPA has identified a list of specific PESS factors that may contribute to a group having increased exposure or biological susceptibility, such as lifestage, occupational and certain consumer exposures, nutrition, and lifestyle activities. For TCEP, the Agency identified how the risk evaluation addressed these factors as well as any remaining uncertainties. For the TCEP risk evaluation, EPA accounted for the following PESS groups: infants exposed through human milk from exposed individuals, children and male adolescents who use consumer articles or among the exposed general population, subsistence fishers, Tribal populations, pregnant women, workers and consumers who experience aggregated or sentinel exposures, people living in fenceline communities near facilities that emit TCEP, and firefighters. See Section 5.3.3 and Appendix D for details related to this analysis.

1.2 Systematic Review

The U.S. EPA's Office of Pollution Prevention and Toxics (EPA/OPPT) applies systematic review principles and approaches in the development of risk evaluations. TSCA section 26(h) requires EPA to use scientific information, technical procedures, measures, methods, protocols, methodologies, and models consistent with the best available science and base decisions under section 6 on the weight of scientific evidence. Systematic review supports the risk evaluation through data searching, screening, evaluation, extraction, and evidence integration and is used to develop the exposure and hazard assessments based on reasonably available information. EPA defines "reasonably available information" to mean information that EPA possesses or can reasonably generate, obtain, and synthesize for use in risk evaluations, considering the deadlines for completing the evaluation (40 CFR 702.33).

In response to comments received by the National Academies of Sciences, Engineering, and Medicine (NASEM), TSCA Scientific Advisory Committee on Chemicals (SACC) and public, EPA developed the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances* (U.S. EPA, 2021a) (also referred to as the "2021 Draft Systematic Review Protocol") to describe systematic review approaches implemented in TSCA risk evaluations. In response to recommendations for chemical specific systematic review protocols, the *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Protocol* (U.S. EPA, 2024p) (also referred to as the "TCEP Systematic Review Protocol") describes clarifications and updates to approaches outlined in the 2021 Draft Systematic Review Protocol that reflect NASEM, SACC, and public comments as well as chemical-specific risk evaluation needs. For example, EPA has updated the data quality evaluation process and will not implement quantitative methodologies to determine both metric and overall data or information source data quality determinations. Screening decision terminology (*e.g.*, "met screening criteria" as opposed to "include") was also updated for greater consistency and transparency and to more appropriately describe when information within a given data source met discipline-specific title and abstract or full-text screening criteria. Additional updates and clarifications relevant for TCEP data sources are described in greater detail in the TCEP Systematic Review Protocol (U.S. EPA, 2024p).

The systematic review process is briefly described in Figure 1-9 below. Additional details regarding these steps are available in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Literature inventory trees and evidence maps for each discipline (*e.g.*, human health hazard) displaying results of

the literature search and screening, as well as sections summarizing data evaluation, extraction, and evidence integration are included in the TCEP Systematic Review Protocol (U.S. EPA, 2024p).

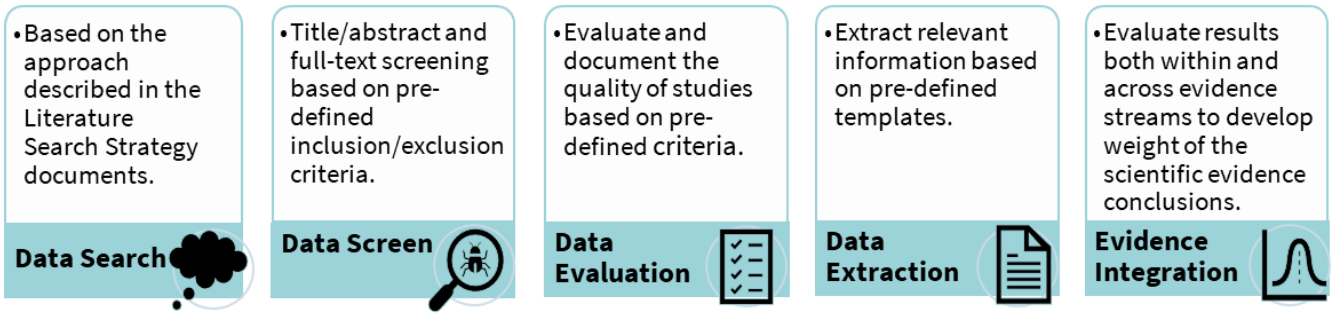


Figure 1-9. Diagram of the Systematic Review Process

EPA used reasonably available information, defined in Title 40 Code of Federal Regulations (40 CFR) Section 702.33, in a fit-for-purpose approach, to develop a risk evaluation that relies on the best available science and is based on the weight of scientific evidence in accordance with TSCA sections 6 and 26. EPA reviewed reasonably available information and evaluated the quality of the methods and reporting of results of the individual studies using the evaluation strategies described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and the TCEP Systematic Review Protocol (U.S. EPA, 2024p).

EPA also identified key assessments conducted by other EPA programs and other U.S. and international organizations. Depending on the source, these assessments may include information on COUs (or the equivalent), hazards, exposures, and potentially exposed or susceptible subpopulations. Some of the most pertinent assessments that were consulted for TCEP include the following:

- U.S. EPA's 2009 *Provisional Peer-Reviewed Toxicity Values (PPRTV) for Tris(2-chloroethyl)phosphate (TCEP) (CASRN 115-96-8)*;
- 2009 *European Union Risk Assessment Report: CAS: 115-96-8: Tris (2-chloroethyl) phosphate, TCEP*;
- Environment Canada and Health Canada's 2009 *Screening Assessment for the Challenge Ethanol, 2-chloro-, phosphate (3:1) (Tris(2-chlrorethyl) phosphate [TCEP])*;
- Australia's 2016 *Ethanol, 2-chloro-, phosphate (3:1): Human health tier II assessment*;
- Australia's 2017 *Ethanol, 2-chloro-, phosphate (3:1): Human health tier III assessment*;
- ATSDR's 2012 *Toxicological Profile for Phosphate Ester Flame Retardants*;
- NTP's 1991 Technical Report on *Toxicology and Carcinogenesis Studies of Tris(2-chloroethyl) Phosphate (CASRN 115-96-8) in F344/N Rats and B6C3F1 Mice (Gavage Studies)*; and
- IARC's 1999 *Monographs on the Evaluation of Carcinogenic Risks to Humans Volume 71*.

An update to the peer literature search to capture information published since 2019 was performed in February 2024. The search criteria, resources, and search strings used in the literature search update for TCEP can be found in Section 3.1 of the TCEP Systematic Review Protocol (U.S. EPA, 2024p). After the search update was complete, additional filtering steps were performed to produce subsets of literature relevant to targeted information areas to fill known data gaps, namely: landfills (general population, consumer, and environmental hazard; n = 11); environmental hazard (n = 13); epidemiology (n = 73); and inhalation (human health/animal toxicity; n = 50). Subsequent discipline-specific title, abstract and full-text screening criteria (*e.g.*, PECO/PESO/RESO) were leveraged for further

prioritization of references for use in the risk evaluation, as described in Section 4 of the TCEP Systematic Review Protocol (U.S. EPA, 2024p).

1.3 Organization of the Risk Evaluation

This risk evaluation for TCEP includes five additional sections, a list of REFERENCES, and several APPENDICES. Section 2 summarizes basic physical-chemical characteristics as well as the fate and transport of TCEP. Section 3 includes an overview of releases and concentrations of TCEP in the environment. Section 4 provides a discussion and analysis of the environmental risk assessment, including the environmental exposure, hazard, and risk characterization based on the COUs for TCEP. Section 5 presents the human health risk assessment, including the exposure, hazard, and risk characterization based on the COUs. Section 5 also includes a discussion of PESS based on both greater exposure and/or susceptibility, as well as a description of aggregate and sentinel exposures. Sections 4 and 5 both discuss any assumptions and uncertainties and how they impact the risk evaluation. Finally, Section 6 presents EPA's risk evaluation of whether the chemical presents an unreasonable risk to human health or the environment under the assessed COUs.

Appendix A provides a list of abbreviations and acronyms as well a glossary of select terms used throughout this risk evaluation. Appendix B provides a brief summary of the federal, state, and international regulatory history of TCEP. Appendix C lists all separate supplemental files associated with this risk evaluation, which can be accessed through hyperlinks included in the references. All subsequent appendices include more detailed analysis and discussion than are provided in the main body of this risk evaluation for TCEP.

2 CHEMISTRY AND FATE AND TRANSPORT

Physical and chemical properties determine the behavior and characteristics of a chemical that inform its condition of use, environmental fate and transport, potential toxicity, exposure pathways, routes, and hazards. Environmental fate and transport include environmental partitioning, accumulation, degradation, and transformation processes. Environmental transport is the movement of the chemical within and between environmental media, such as air, water, soil, and sediment. Transformation or degradation occur through reaction of the chemical in the environment. Thus, understanding the environmental fate of TCEP informs the determination of the specific exposure pathways, and potential human and environmental populations that EPA considered in this risk evaluation.

2.1 Physical and Chemical Properties

EPA gathered and evaluated physical and chemical property data and information according to the process described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). During the evaluation of TCEP, EPA considered both measured and estimated physical and chemical property data/information summarized in Table 2-1, as applicable. More details are given in Appendix F.1. Information on the full, extracted dataset is available in the supplemental file *Risk Evaluation for Tris (2-chloroethyl) Phosphate (TCEP) – Systematic Review of Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties* (U.S. EPA, 2024v).

TCEP is a clear, transparent liquid with a slight odor (DOE, 2016; U.S. EPA, 2015b; ECB, 2009; Lewis and Hawley, 2007; Weil, 2001) and low viscosity (IARC, 1990). TCEP is a chlorinated phosphate ester that is used as a flame-retardant additive and plasticizer. It melts around −55 °C and begins to decompose at 320 °C (DOE, 2016; U.S. EPA, 2015b; Toscano and Coleman, 2012; ECB, 2009; IARC, 1990). TCEP is soluble in water with a water solubility of 7,820 mg/L at 20 °C (ECB, 2009) and hydrophilic with a logarithmic octanol:water partition coefficient (log K_{OW}) value of 1.78 (ECB, 2009). TCEP is reported to have low volatility with a vapor pressure of 0.0613 mm Hg at 25 °C (Dobry and Keller, 1957) and a boiling point of 330 °C (U.S. EPA, 2019b; DOE, 2016; U.S. EPA, 2015a; Haynes, 2014; Toscano and Coleman, 2012). However, it will become more volatile when the temperature increases (0.5 mm Hg at 145 °C) (Toscano and Coleman, 2012; NTP, 1992). Because of its high boiling point, low volatility, and a Henry's Law constant of 2.945×10^{-6} atm·m^3/mol at 25 °C (U.S. EPA, 2017a), TCEP is categorized as a semi-volatile organic compound (SVOC) (ECHA, 2018; TERA, 2015).

Table 2-1. Physical and Chemical Properties of TCEP

Property	Selected Value[a]	Reference(s)	Overall Quality Determination[b]
Molecular formula	$C_6H_{12}Cl_3O_4P$	NLM (2019)	High
Molecular weight	285.49 g/mol	Haynes (2014)	High
Physical form	Clear, transparent liquid with slight odor	Weil (2001); Lewis and Hawley (2007); ECB (2009); U.S. EPA (2015b); DOE (2016)	High
Melting point	−55 °C	Toscano and Coleman (2012), as cited in ATSDR (2012), NLM (2015), U.S. EPA (2015a), U.S. EPA (2015b), and DOE (2016)	High

Property	Selected Value[a]	Reference(s)	Overall Quality Determination[b]
Boiling point	330 °C	Toscano and Coleman (2012); Haynes (2014); ATSDR (2012); NLM (2015); U.S. EPA (2015a); DOE (2016); U.S. EPA (2019b)	High
Density	1.39 g/cm^3 at 25 °C	Toscano and Coleman (2012); Haynes (2014); DOE (2016)	High
Vapor pressure	0.0613 mm Hg at 25 °C	Dobry and Keller (1957) as cited in Verbruggen et al. (2005), ATSDR (2012), NLM (2015), U.S. EPA (2015a), and U.S. EPA (2019b)	High
Vapor density	9.8 (air = 1)	ILO (2019)	High
Water solubility	7,820 mg/L at 20 °C	ECB (2009) as cited in Verbruggen et al. (2005), EC (2009), U.S. EPA (2015b), U.S. EPA (2015a), and NLM (2015)	High
Logarithmic octanol:water partition coefficient (log K_{OW})	1.78	ECB (2009) as cited in Verbruggen et al. (2005), EC (2009), U.S. EPA (2015b), U.S. EPA (2015a), and NLM (2015)	High
Logarithmic octanol:air partition coefficient (log K_{OA})	7.86–7.93	Okeme et al. (2020)	High
Henry's Law constant	2.945E−06 atm·m^3/mol at 25 °C (calculated)	U.S. EPA (2017a)	High
Flash point	225 °C (closed cup)	ECB (2000) as cited in U.S. EPA (2015a)	High
Autoflammability	480 °C	ECB (2009); ILO (2019)	Medium
Viscosity	45 cP at 20 °C	IARC (1990)	High
Refractive index	1.4721 at 20 °C	Haynes (2014) as cited in NLM (2015)	High

[a] Measured unless otherwise noted
[b] "Overall Quality Determination" apply to all references listed in this table

2.2 Environmental Fate and Transport

Environmental Fate and Transport (Section 2.2):
Key Points

EPA evaluated the reasonably available information to characterize the environmental fate and transport of TCEP, the key points are summarized below:

- TCEP exists in both gaseous and particle phases under environmentally relevant conditions and partitions to organic carbon in the air. TCEP is not expected to undergo significant direct photolysis, but TCEP in the gaseous phase will rapidly degrade in the atmosphere ($t_{1/2}$ = 5.8 hours).
- TCEP is not expected to undergo abiotic degradation processes such as photolysis and hydrolysis in aquatic environments under environmentally relevant conditions. However, TCEP's rate of hydrolysis is highly dependent on pH and temperature conditions.
- TCEP does not biodegrade in water under aerobic conditions but will volatilize from surface water despite its low Henry's Law constant (2.945×10^{-6} atm·m^3/mol at 25 °C).
- TCEP can be transported to sediment from overlying surface water through advection and dispersion of dissolved TCEP and deposition of suspended solids containing TCEP. Based on its log K_{OC} values (3.23–3.46), TCEP is expected to partition to organic matter in suspended and benthic sediments. However, TCEP may partition between surface water and sediments to varying degrees because of its high water solubility (7,820 mg/L at 20 °C).
- TCEP accumulation in soil is unlikely because of its log K_{OC} values (2.08–2.52). Due to its high water solubility and its low Henry's Law constant, TCEP in moist soil will both migrate to groundwater and volatilize.
- TCEP will be minimally removed via conventional drinking water and wastewater treatment and will be retained in wastewater effluents with a low fraction being adsorbed into sludge.
- TCEP has been detected in surface water and groundwater samples; point sources include wastewater effluents and landfill leachates.
- TCEP has been detected in surface water, air, and snow in remote locations with no known source of releases but is known to undergo long-range transport through atmospheric, plastic debris, and other natural processes.
- TCEP does not bioaccumulate in aquatic fish but may in benthic fish. When TCEP concentrations are transferred to higher trophic levels in the food web, trophic dilution occurs.
- Overall, TCEP appears to be a persistent mobile organic compound (PMOC). PMOCs can dissolve in water or bind to particles, resulting in longer environmental half-lives and greater potential for long-range transport—especially in the air, water, and sediment compartments.

2.2.1 Fate and Transport Approach and Methodology

Reasonably available environmental fate data—including biotic and abiotic biodegradation rates, removal during wastewater treatment, volatilization from lakes and rivers, and logarithmic organic carbon:water partition coefficient (log K_{OC})—are the parameters used in the current risk evaluation. In assessing the environmental fate and transport of TCEP, EPA considered the full range of results from data sources that were rated high-quality. Information on the full extracted dataset is available in the

supplemental file *Risk Evaluation for Tris (2-chloroethyl) Phosphate (TCEP) – Systematic Review of Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport* (U.S. EPA, 2024t). Other fate estimates were based on modeling results from Estimation Programs Interface (EPI) Suite™ (U.S. EPA, 2017a), a predictive tool for physical and chemical properties and environmental fate estimation.[3] Information regarding the model inputs is available in Appendix F.

Table 2-2 provides selected environmental fate data that EPA considered while assessing the fate of TCEP and were updated after publication of *Final Scope of the Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) CASRN 115-96-8* (U.S. EPA, 2020b) with additional information identified through the systematic review process.

Table 2-2. Environmental Fate Properties of TCEP

Property or Endpoint	Value[a]	Reference	Overall Quality Determination
Indirect photodegradation	$t_{1/2}$ = 5.8 hours (based on ·OH rate constant of 2.2E−11 cm^3/mole-sec at 25 °C and 12-hour day with 1.5E06 ·OH/cm^3; estimated) [b]	U.S. EPA (2017a)	High
Direct photodegradation	Not expected to be susceptible to direct photolysis by sunlight because the chemical structure of TCEP does not contain chromophores that absorb at wavelengths >290 nm	HSDB (2015)	High
Hydrolysis half-life	$t_{1/2}$ = 2 years at pH 8 and 25 °C (estimated)	Saint-Hilaire et al. (2011)	High
	$t_{1/2}$ = 0.083 days at pH 13; no significant degradation observed over 35 days at pH 7, 9, and 11	Su et al. (2016)	
Aerobic biodegradation	Water: 13% and 4%/28 days (OECD 301B) at 10 and 20 mg/L test substance concentration in activated domestic sludge, adaption not specified	Life Sciences Research Ltd (1990b)	High
	Soil: DT_{50} = 17.7 days; 78%/40 days based on test substance concentration of 50 µg/kg	Hurtado et al. (2017)	
Anaerobic biodegradation	No degradation observed	Pang et al. (2018)	High
Bioconcentration factor (BCF) (L/kg, unless noted)	Whole body BCF = 0.31 ± 0.06, 0.16 ± 0.03, and 0.34 ± 0.04 at test substance concentrations of 0.04, 0.2, and 1.0 mg/L, respectively in the muscle of juvenile Atlantic salmon (*Salmo salar*)	Arukwe et al. (2018)	High
	BCF = 1.0 ± 0.1 (muscle), 4.3 ± 0.2 (liver), 2.6 ± 0.2 (brain), 1.6 ± 0.1 (gill), and 1.6 ± 0.1 (kidney) at test substance concentration of 9.1 µg/L for juvenile common carp (*Cyprinus carpio*) (OECD 305)	Tang et al. (2019)	
	BCF = 0.8 ± 0.1 (muscle), 2.4 ± 0.1 (liver), 2.2 ± 0.1 (brain), 1.9 ± 0.2 (gill) at test substance concentration of 893 µg/L, respectively for zebrafish (*Danio rerio*) (OECD 305)	Wang et al. (2017a)	

[3] See EPI Suite™ for additional information and supporting documents about this freely available, online suite of programs, which was reviewed by the EPA Science Advisory Board (SAB, 2007).

Property or Endpoint	Value[a]	Reference	Overall Quality Determination
Bioaccumulation factor (BAF) (L/kg, unless noted)	Mean BAF = 794 (muscle), 1,995 (liver), 1,995 (kidney), and 1,995 (gill)	Bekele et al. (2021)	High
	Mean BAF = 30.7 (muscle) and 70.7 (liver) for crucian carp (*Carassius auratus*)	Choo et al. (2018)	
	Mean BAF = 2,198 at test substance concentration of 0.464 ng/L for walleye (*Sander vitreus*)	Guo et al. (2017b)	
	Mean BAF = 1,248 for snakehead (*Ophiocephalus argus*), 191 for catfish (*Clarias batrachus*), 109–202 for mud carp (*Cirrhinus molitorella*), 207 for crucian carp (*Carassius auratus*), and 463 for Oriental River prawn (*Macrobrachium nipponense*)	Liu et al. (2019a)	
	Mean BAF = 6,310 for benthic invertebrates (soft tissue); 2,690 for pelagic fish (organ); 4,270 for benthic fish (organ and whole body)	Wang et al. (2019b)	
Logarithmic organic carbon:water partition coefficient (log K_{OC}) (soil)	2.08–2.52	Cristale et al. (2017)	High
Logarithmic organic carbon:water partition coefficient (log K_{OC}) (sediment)	3.23 ± 0.23	Wang et al. (2018a)	High
	3.32 (mean; range 2.5–4.06)	Zhang et al. (2021)	
	3.46 ± 0.48	Zhang et al. (2018b)	
Removal in wastewater treatment	Approximately −5% removal after primary treatment; −19.1% overall removal	Kim et al. (2017)	High
Trophic magnification factor (TMF)	Benthic food web: 2.6 (tentative due to small sample size, n = 15)	Brandsma et al. (2015)	High
	No significant relationship with pelagic food web and total food web		
	Antarctic food chain: 5.2	Fu et al. (2020)	
	No significant relationship with trophic level	Zhao et al. (2018)	
Biota-sediment accumulation factor (BSAF)	Mean BSAF (L/kg): 1.09 (muscle) and 2.49 (liver) for Crucian carp (*Carassius auratus*)	Choo et al. (2018)	High
	Mean BSAF: 0.015–0.171	Liu et al. (2019a)	
	Mean BSAF: 2.19E−03 for benthic invertebrates and 1.48E−03 for benthic fishes	Wang et al. (2019b)	

[a] Measured unless otherwise noted
[b] Information estimated using EPI Suite™ (U.S. EPA, 2017a)

2.2.2 Summary of Fate and Transport Assessment

Numerous studies have described TCEP as a "ubiquitous" contaminant because it is commonly found in various environmental compartments such as indoor air and dust, outdoor air, surface water, drinking

water, groundwater, soil, sediment, biota, and even precipitation all over the world (Awonaike et al., 2021; Ma et al., 2021; Propp et al., 2021; Choo and Oh, 2020; Li et al., 2019b; Tan et al., 2019; Arukwe et al., 2018; Kim and Kannan, 2018; Cao et al., 2017; Hurtado et al., 2017; Wang et al., 2017a; Bradman et al., 2014; Padhye et al., 2014; Cristale et al., 2013; Bradman et al., 2012; Regnery and Püttmann, 2010b; Benotti et al., 2009; Fries and Puttmann, 2003, 2001). This is because TCEP is primarily used as an additive plasticizer and flame retardant. When used as an additive, TCEP is added to manufactured materials via physical mixing rather than chemical bonding and as a result, TCEP can easily leach or diffuse into its surrounding environment (Qi et al., 2019; Liu et al., 2014; Wei et al., 2014; ATSDR, 2012; van der Veen and de Boer, 2012; EC, 2009; ECB, 2009; NICNAS, 2001). TCEP's physical and chemical properties suggests that its main mode of distribution in the environment is through water and soil, depending on where it is being released (Figure 2-1; see also Appendix F.2.1.2) (U.S. EPA, 2017a; TERA, 2015; Regnery and Püttmann, 2010b; Zhang et al., 2009).

Multiple studies have identified urban sources as sources of TCEP in the environment through fugitive emissions to air (Abdollahi et al., 2017; Luo et al., 2015; Möller et al., 2011). The exact sources of TCEP emissions from urban environment are unknown, however they are likely the articles that were treated with or containing TCEP (Abdollahi et al., 2017; Luo et al., 2015; Wei et al., 2014; Möller et al., 2011; Aston et al., 1996). Compared to outdoor air, TCEP concentrations are significantly higher in indoor air, because TCEP has the potential to volatilize from treated products and diffuse into air, as well as partition onto dust, due to its use as an additive (Qi et al., 2019; TERA, 2015; Liu et al., 2014; ATSDR, 2012; EC, 2009; NICNAS, 2001). Atmospheric deposition has been identified as an important source of TCEP to surface water and soil, especially in urban areas. Several studies showed that higher TCEP concentrations in precipitation were generally seen in densely populated areas with high traffic volume (Kim and Kannan, 2018; Regnery and Püttmann, 2010b; Regnery and Puettmann, 2009; Marklund et al., 2005b). In addition, storm water and urban runoff can contribute to additional emissions to surface water.

TCEP can be transported to sediment from overlying surface water by advection and dispersion of dissolved TCEP and by deposition of suspended solids containing TCEP. However, TCEP may partition between surface water and sediments to varying degrees because of its log K_{OC} values (Zhang et al., 2021; Wang et al., 2018a; Zhang et al., 2018b) and water solubility (Lee et al., 2018; Ma et al., 2017; Brandsma et al., 2015; Cao et al., 2012), which could contribute to its mobility in the environment. Higher concentrations of TCEP in sediment are expected to be found at potential source locations (*e.g.*, near urban and industrialized areas) (Chokwe and Okonkwo, 2019; Tan et al., 2019; Lee et al., 2018; Wang et al., 2018a; Cao et al., 2017; Maruya et al., 2016; Cristale et al., 2013). Precipitation events, such as rain and snow may enhance soil concentrations of TCEP, but accumulation in soil is expected to be unlikely. TCEP may either volatilize from moist soil or migrate through the soil zone (Mihajlović et al., 2011). Due to its water solubility (7,820 mg/L) and soil log K_{OC} values (2.08–2.52), dissolved TCEP was observed to be mobile and migrated to groundwater by common soil transport processes such as advection and diffusion (Propp et al., 2021; Buszka et al., 2009; Barnes et al., 2004). TCEP in the soil was seen to be vertically transported to deeper soil horizons, causing TCEP concentration in the surface soil to be lower (He et al., 2017; Bacaloni et al., 2008).

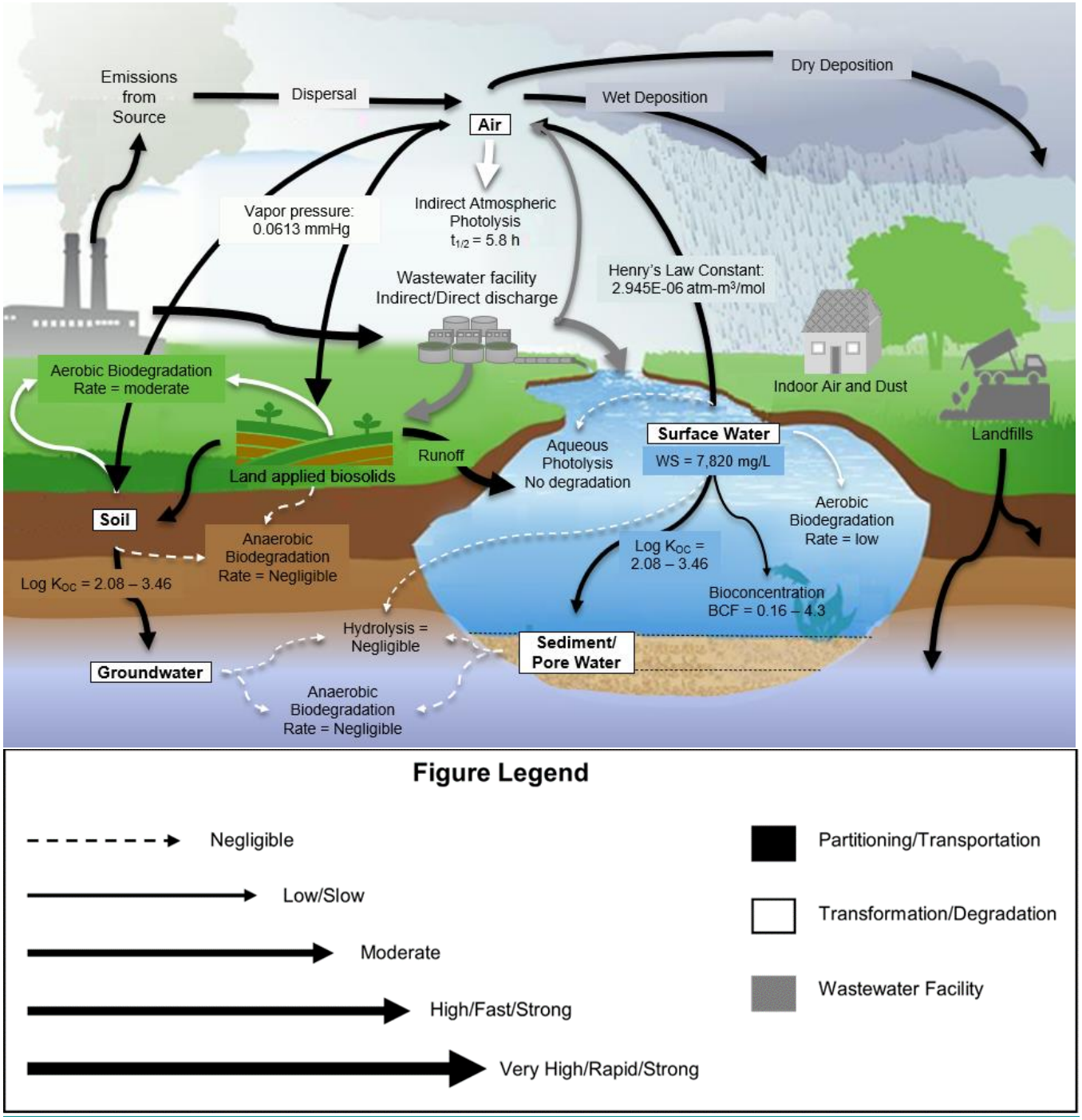


Figure 2-1. Transport, Partitioning, and Degradation of TCEP in the Environment[a]

[a] The diagram depicts the distribution (grey arrows), transport and partitioning (black arrows), and the transformation and degradation (white arrows) of TCEP in the environment. The width of the arrow is a qualitative indication of the likelihood that the indicated partitioning will occur or the rate at which the indicated degradation will occur (*i.e.*, wider arrows indicate more likely partitioning or more rapid degradation).

Most flame retardants that have "High" or "Very High" persistence designations, such as TCEP, are persistent because they are expected to be stable by design to maintain their flame-retardant properties throughout its lifetime in products (U.S. EPA, 2015a). Based on multiple monitoring studies, TCEP appears to be a persistent mobile organic compound (PMOC). PMOCs can dissolve in water or bind to particles, resulting in longer environmental half-lives and greater potential for long-range transport (Blum et al., 2019; Rodgers et al., 2018; Reemtsma et al., 2016). TCEP was detected in both lake and

marine waters of the Arctic, where TCEP was quantified in water and air far from human settlements (>500 km). Atmospheric deposition and watershed runoff may be the primary sources of TCEP in these remote waters where TCEP is unlikely to be rapidly transformed by hydrolysis, photolysis, or biodegradation (Na et al., 2020; McDonough et al., 2018; Li et al., 2017b). These findings indicate that TCEP has the potential to undergo long-range transport in air and water. TCEP's long-range transport potential (LRTP) was seen to be significantly underestimated when using its physical and chemical properties in quantitative structure-activity relationship (QSAR) models because the behavior of TCEP in the environment often does not align with its physical and chemical properties. A detailed summary of physical and chemical properties and a fate and transport assessment of TCEP is available in Appendix F.

2.2.3 Weight of Scientific Evidence Conclusions for Fate and Transport

2.2.3.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the Fate and Transport Assessment

Given the consistent results from numerous high-quality studies, there is a robust confidence that TCEP:

- is not expected to undergo significant direct photolysis (see Appendix F.2.2);
- will partition to organic carbon in the air (see Appendix F.2.2);
- will exist in both the gas and particle phases (see Appendix F.2.2);
- showed no significant degradation after undergoing hydrolysis but hydrolysis rate was seen to increase with increasing pH (see Appendix F.2.3.1);
- does not undergo biodegradation in water under aerobic conditions (see Appendix F.2.3.1);
- will volatilize from surface water (see Appendix F.2.3.1) and moist soil (see Appendix F.2.4.1);
- produces hazardous byproducts when undergoing thermal degradation (see Appendix F.2.5.1);
- will not be removed after undergoing wastewater treatment and will be retained in effluents with low fraction being adsorbed onto sludge (see Appendix F.2.5.2);
- is minimally removed after undergoing conventional drinking water treatment (see Appendix F.2.5.3); and
- has the ability to undergo long-range transport (see Appendices F.2.2 and F.2.3.1).

As a result of limited studies identified, there is a moderate confidence that TCEP:

- does not undergo biodegradation in water and sediment under anaerobic conditions (see Appendixes F.2.3.1 and F.2.3.2);
- will partition to organic carbon in sediment (see Appendix F.2.3.2) and soil (see Appendix F.2.4.1);
- will enter surface water and groundwater from landfills (see Appendix F.2.4.3);
- will not bioaccumulate in fish residing in the water column (see Appendix F.2.6);
- may bioaccumulate in benthic fish (see Appendix F.2.6); and
- does not bioaccumulate when TCEP concentrations are transferred to higher trophic levels in the food web (see Appendix F.2.6).

A detailed discussion of strengths, limitations, assumptions, and key sources of uncertainty for the fate and transport assessment of TCEP is available in Appendix F.

3 RELEASES AND CONCENTRATIONS IN THE ENVIRONMENT

EPA estimated environmental releases of TCEP. Section 3.1 describes the approach and methodology for estimating releases. Estimates of environmental releases are presented in Section 3.2. Section 3.3 presents the approach, methodology, and estimates of environmental concentrations that result from environmental releases of TCEP.

3.1 Approach and Methodology

3.1.1 Industrial and Commercial

EPA categorized the COUs listed in Table 1-1 into occupational exposure scenarios (OESs) (see Table 3-1). EPA developed the OESs to group processes or applications with similar sources of release and occupational exposures that occur at industrial and commercial workplaces within the scope of the risk evaluation. For each OES, occupational exposure and environmental release results are provided and expected to be representative of the entire population of workers and sites involved for the given OES in the United States. Note that EPA may define only a single OES for multiple COUs, while in other cases multiple OESs may be developed for a single COU. For example, the paint and coating manufacturing COU has two associated OESs—a 1-part coatings scenario and a 2-part reactive coatings scenario. EPA makes this determination by considering variability in release and use conditions and whether the variability can be captured as a distribution of exposure or instead requires discrete scenarios. Specifically, the 1-part coatings tend to be water-based formulations and could potentially have a greater release to water whereas the 2-part reactive coatings could have greater release to incineration or landfill. Further information on specific OESs is provided in Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S. EPA, 2024n).

All COUs assessed in this Risk Evaluation are considered on-going uses. However, there are several COUs for which part of the life cycle has ceased, such as manufacturing (including import) and processing. However, other parts of the life cycle including recycling, commercial or consumer use, and disposal are on-going. These COUs are identified in Table 3-1 and include four COUs for commercial use and five COUs for consumer use.

Table 3-1. Crosswalk of COUs to OESs Assessed

COU			OES
Life Cycle Stage[a]	**Category[b]**	**Subcategory[c]**	
Manufacture	Import	Import	Repackaging
Processing	Incorporated into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings
			Incorporation into paints and coatings – 2-part reactive coatings
	Incorporated into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resins
	Incorporated into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article
	Recycling	Recycling	Recycling e-waste
Distribution	Distribution	Distribution in commerce	Distribution in commerce[f]
Industrial Use	Other use	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Installation of article
	Paints and coatings	Paints and coatings	Use of paints and coatings – Spray application OES
Commercial Use	Other use	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Use and/or maintenance of aerospace equipment and products and automotive articles and replacement parts containing TCEP
	Paints and coatings	Paints and coatings	Use of paints and coatings – Spray application OES
	Other use	Laboratory chemicals	Lab chemical – Use of laboratory chemicals
	Furnishing, cleaning, treatment/care products	Fabric and textile products[d]	End of service life disposal[d] (releases and exposures not quantified)
		Foam Seating and Bedding Products[d]	End of service life disposal[d] (releases and exposures not quantified)
	Construction, paint, electrical, and metal products	Building/construction materials – Insulation[d]	End of service life disposal[d] (releases and exposures not quantified)
		Building/construction materials – Wood and engineered wood products – Wood resin composites[d]	End of service life disposal[d] (releases and exposures not quantified)

COU			OES
Life Cycle Stage[a]	**Category[b]**	**Subcategory[c]**	
Disposal	Disposal	Disposal[e]	Waste disposal (landfill or incineration, covered in each COU/OES as opposed to a separate COU)

[a] Life Cycle Stage Use Definitions (40 CFR 711.3)

– "Industrial Use" means use at a site at which one or more chemicals or mixtures are manufactured (including imported) or processed.

– "Commercial Use" means the use of a chemical or a mixture containing a chemical (including as part of an article) in a commercial enterprise providing saleable goods or services.

– "Consumer Use" means the use of a chemical or a mixture containing a chemical (including as part of an article, such as furniture or clothing) when sold to or made available to consumers for their use.

– Although EPA has identified both industrial and commercial uses here for purposes of distinguishing scenarios in this document, the Agency interprets the authority over "any manner or method of commercial use" under TSCA section 6(a)(5) to reach both.

[b] These categories of COUs appear in the LCD, reflect CDR codes, and broadly represent COUs of TCEP in industrial and/or commercial settings and for consumer uses.

[c] These subcategories reflect more specific COUs of TCEP.

[d] This COU includes associated disposal of those COUs for which domestic manufacturing and/or processing have ceased.

[e] Section 3.2 provides details on these OESs.

[f] Distribution in Commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP or sealed packages of TCEP containing products.

The 2016 CDR data (U.S. EPA, 2019a) included a single reporting site, Aceto Corporation in Port Washington, New York, importing TCEP, with no downstream industry sectors identified. TCEP was not reported in the 2020 CDR (U.S. EPA, 2020a). EPA did identify other data on current import volumes and possible import sites from Datamyne, as presented in Figure 1-3, which showed some TCEP imports below the CDR threshold of 25,000 lb/site-yr. Nevertheless, processors of TCEP may be purchasing the chemical from importers (see Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S. EPA, 2024n) for details). Therefore, EPA assumed TCEP may still be imported at volumes below the CDR reporting threshold and EPA assessed the following two potential scenarios: (1) one site importing 25,000 lb, and (2) one site importing 2,500 lb. EPA modeled environmental releases and occupational exposures for these hypothetical scenarios. For each OES, where monitoring data were not available, daily releases were estimated per media of release based on EPA Standard Models, Generic Scenarios (GSs), and/or Emission Scenario Documents (ESDs) to generate annual releases and for the estimation of associated release days. TCEP is not listed on the National Emissions Inventory (NEI) and was only recently added to TRI, with the first year of reporting from facilities due July 1, 2024. As of September 2024, there are no TRI entries for TCEP. EPA describes its approach and methodology for estimating daily releases and for detailed facility level results in Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S. EPA, 2024n).

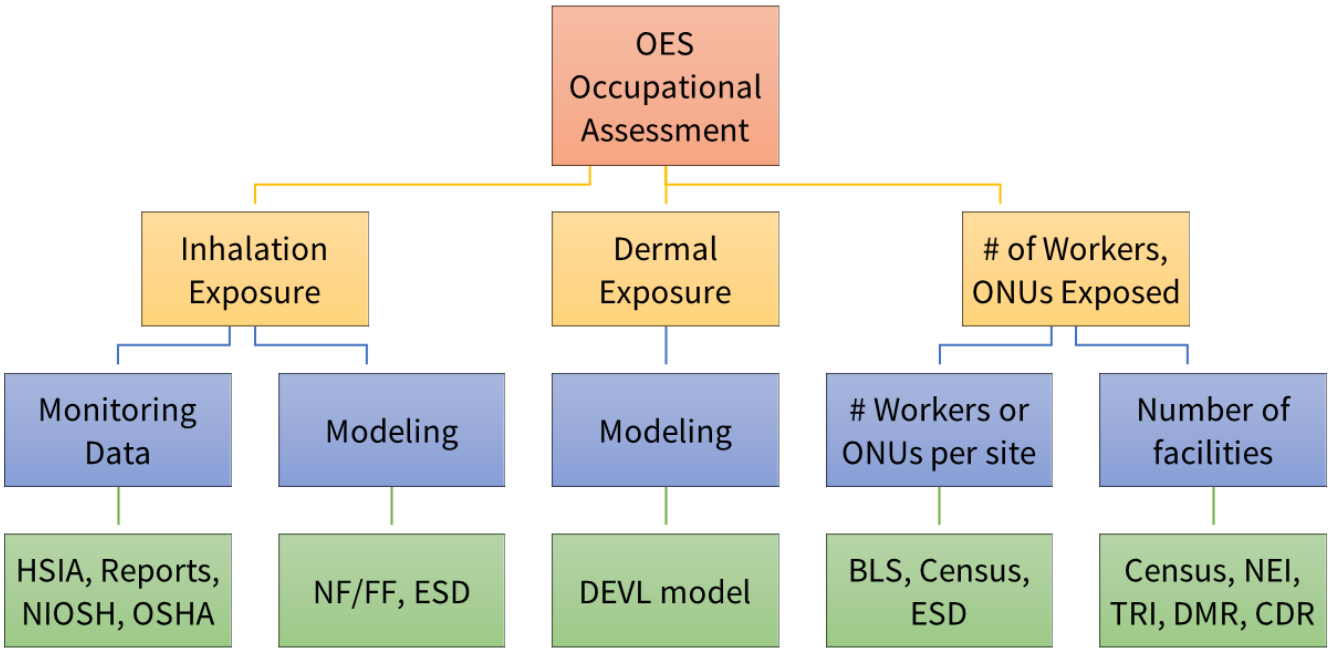


Figure 3-1. An Overview of How EPA Estimated Daily Releases for Each OES
BLS = Bureau of Labor Statistics; DEVL = Dermal Exposure to Volatile Liquids model; DMR = Discharge Monitoring Report; ELG = Effluent Limitation Guidelines; HSIA = Halogenated Solvents Industry Alliance; NF/FF = Near-Field/Far Field; NIOSH = National Institute for Occupational Safety and Health; OSHA = Occupational Safety and Health Administration

The releases of TCEP were estimated for each media applicable to the OES. For TCEP, releases could occur to water, air, or disposal to land. TCEP released could be in the form of liquid (neat or in formulation), vapor, and/or solid waste.

3.2 Environmental Releases

Environmental Releases (Section 3.2):
Key Points

EPA evaluated the reasonably available information for releases of TCEP to the environment. The key points of the environmental releases are summarized below:

- EPA assessed environmental releases of TCEP from industrial and commercial sources as well as consumer products.
  - For industrial and commercial sources, EPA used data from literature, relevant ESDs, or GSs to estimate environmental releases to air, surface water, and waste disposal from a generic facility for each OES. Some OESs could not be quantified due to insufficient data. Of the OESs that could be quantified, the highest release estimates were from:
    - Incorporation into paints and coatings – 1-part coatings
    - Incorporation into paints and coatings – 2-part reactive coatings
    - Formulation of TCEP-containing reactive resins (for use in 2-part systems)
    - Use of paints and coatings – spray application OES.
  - For consumer products, EPA did not have enough information to assess environmental releases quantitatively. However, the Agency acknowledges that there may be TCEP releases to the environment via the demolition and disposal of consumer articles, as well as to wastewater via domestic laundry. These releases were assessed qualitatively. EPA included anecdotal information from peer-reviewed literature on releases from consumer articles in Section 5.1.2.2.5.

3.2.1 Industrial and Commercial

EPA combined its estimates for each activity that is reasonably expected to occur during each OES. These activities were based on using data from literature, relevant ESDs or GSs. Once these activities were identified, existing EPA models and parameters (*e.g.*, the EPA/OPPT Mass Transfer Coefficient model, EPA/OPPT Penetration model, ChemSTEER User Guide, etc.) were used in a Monte Carlo simulation to create a distribution of releases. From this distribution EPA provides a high-end (95th percentile) and central tendency (50th percentile) release values as well as a range of potential release days. The releases presented are assumed to be representative of what would be reasonably expected to occur at an individual generic site. In some cases, where it was not reasonable to assume a single generic site due to throughput constrictions presented in the relevant source (*e.g.*, it is not reasonable to assume that a single paint application site or laboratory would use the entire PV of 25,000 lb), a range of potential number of sites is presented in Table 5-2. A summary of these ranges of releases across OESs is presented in Table 3-2. See Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S. EPA, 2024n) for more details on deriving the overall confidence score for each OES. For some OESs, EPA was not able to estimate or did not anticipate there to be releases; for example:

- EPA was not able to quantify disposal of articles that historically contained TCEP with reasonably available information. This was assessed qualitatively.
- Installation of articles is not expected to lead to significant releases because the articles are expected to already be in final form (*e.g.*, electronic potting) and not expected to undergo further processing (*i.e.*, shaping, sanding, cutting, etc.).

- EPA was not able to quantify releases of TCEP that could occur during the recycling of electronic waste (e-waste). Sources used for this provided monitoring data from breathing zone measurements from various locations within a facility that recycles e-waste that contained very small amounts of TCEP dust. The source of TCEP was not identified and the source further stated that TCEP is rarely used in electronics. EPA expects releases that could occur during this activity to be minimal and only potentially occur at a small subset of facilities.
- EPA lacks production volume data to assess TCEP exposure from distribution into commerce due to the declining production and manufacturing in recent years. Although manufacturing, processing, and distribution into commerce of TCEP is declining (see Section 1.1.1, Table 3-1); distribution into commerce that has occurred, is ongoing, or is likely to occur during a COU subject to evaluation; and exposure to human or ecological populations has occurred or is likely to occur; will be included in the risk evaluation as an exposure associated with a COU.

3.2.1.1 Summary of Daily Environmental Release Estimates

Table 3-2 and Table 3-3 provide estimated releases that could occur during each OES, the expected media of release if releases are expected to occur during that OES, and possible number of sites where releases could occur. The estimated daily releases are based on a 2,500 lb production volume. For most cases, the number of sites is based on a single generic site; however, in some cases, such as use of paints and coatings and laboratory chemicals, a distribution of the number of sites was created. The distributions for number of sites were created for these OESs to provide variability in the potential number of sites and is further explained in the Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S. EPA, 2024n).

Table 3-2. Summary of EPA's Daily Release Estimates for Each OES and EPA's Overall Confidence in these Estimates for 2,500 lb Production Volume

COU	OES	Estimated Daily Release Range across Sites (kg/site-day)		Type of Discharge,[a] Air Emission,[b] or Transfer for Disposal[c]	Estimated Release Frequency Range across Sites (days)[d]		Number of Facilities[e]	Overall Confidence
		Central Tendency	High-End		Central Tendency	High-End		
Manufacture (Import)	Repackaging	6.35	9.89	Surface water	4	4	1 generic site	Medium
		3.18E−04	6.03E−04	Fugitive or stack air	4	4		
		N/A	N/A	Waste disposal (landfill or incineration)	N/A	N/A		
Processing	Incorporation into paints and coatings – 1-part coatings	1.02E01	3.52E01	Surface water	6	2	1 generic site	High
		1.56E−03	9.60E−03	Fugitive or stack air	6	4		
		1.53	9.27	Waste disposal (landfill or incineration)	7	2		
Processing	Incorporation into paints and coatings – 2-part reactive coatings	2.71E01	3.19E01	Surface water	1	1	1 generic site	High
		3.65E−03	7.90E−03	Fugitive air	1	1		
		3.75E−03	1.99E−02	Stack air	1	1		
		3.40E01	3.40E01	Waste disposal (landfill or incineration)	1	1		
Processing	Formulation of TCEP-containing reactive resins (for use in 2-part systems)	2.52E01	3.15E01	Surface water	1	1	1 generic site	High
		3.25E−03	8.83E−03	Fugitive air	1	1		
		2.73E−03	2.07E−02	Stack air	1	1		
		3.40E01	3.40E01	Waste disposal (landfill or incineration)	1	1		
Processing	Processing into 2-part resin article	N/A	N/A	Surface water	N/A	N/A	1 generic site	High
		3.30E−04	9.90E−04	Fugitive or stack air	55	113		
		3.98E−01	2.50	Waste disposal (landfill or incineration)	92	17		
Processing	Recycling e-waste	EPA did not have sufficient data to estimate these releases						
Distribution	Distribution in commerce	Distribution in Commerce [h]						
Industrial Use	Installation of articles	Releases expected to be negligible						

COU	OES	Estimated Daily Release Range across Sites (kg/site-day)		Type of Discharge,[a] Air Emission,[b] or Transfer for Disposal[c]	Estimated Release Frequency Range across Sites (days)[d]		Number of Facilities[e]	Overall Confidence
		Central Tendency	High-End		Central Tendency	High-End		
Industrial and commercial use	Industrial and Commercial Use of paints and coatings – Spray application[g]	2.37	2.32E01	Surface water	1	2	95th Percentile: 2,031 50th Percentile: 281	Medium
		1.25E01	1.14E02	Fugitive air	1	2		
		N/A	N/A	Waste disposal (landfill or incineration)	N/A	N/A		
Commercial Use	Use and/or maintenance of aerospace equipment and products and automotive articles and replacement parts containing TCEP	Releases expected to be negligible						
	Lab chemical – Use of laboratory chemicals	3.96E−01[f]	8.83E−01[f]	Surface water	220	214	13 (1st percentile) – 6 (5th percentile)	High
		6.47E−05[f]	7.99E−05[f]	Fugitive or stack air	220	214		
		N/A	N/A	Waste disposal (landfill or incineration)	N/A	N/A		
	Furnishing, cleaning, treatment/care products • Fabric and textile products • Foam seating and bedding products	Manufacturing and Processing of these COU's has ceased, EPA does not have sufficient data to estimate the releases that may occur during disposal of already existing products						
	Construction, paint, electrical, and metal products • Building/construction materials – Insulation • Building/construction materials – Wood and engineered wood products – Wood resin composites							

COU	OES	Estimated Daily Release Range across Sites (kg/site-day)		Type of Discharge,[a] Air Emission,[b] or Transfer for Disposal[c]	Estimated Release Frequency Range across Sites (days)[d]		Number of Facilities[e]	Overall Confidence
		Central Tendency	High-End		Central Tendency	High-End		
Disposal	Disposal	Waste Disposal (Landfill or Incineration, covered in each COU/OES as opposed to a separate COU)						

[a] Direct discharge to surface water; indirect discharge to non-POTW; indirect discharge to POTW
[b] Emissions via fugitive air; stack air; or treatment via incineration
[c] Transfer to surface impoundment, land application, or landfills
[d] Where available, EPA used peer-reviewed literature (*e.g.,* GSs or ESDs) to provide a basis to estimate the number of release days of TCEP within a COU.
[e] Where available, EPA used peer reviewed literature (*e.g.,* emission scenario documents) data to provide a basis to estimate the number of sites using TCEP within a condition of use.
[f] "High-end" is the 5th percentile and "Central Tendency" is the 1st percentile. See Section 3.10 of Engineering Supplemental file for rationale of using the 1st and 5th percentiles.
[g] Multiple throughput and site scenarios are presented in Table 5-1 of the Engineering Supplemental file.
[h] Distribution in Commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP or sealed packages of TCEP containing products.

Table 3-3. Summary of EPA's Release Estimates for Each COU/OES and EPA's Overall Confidence in these Estimates

Life Cycle Stage	Category	Subcategory	OES	Surface Water	Air		Waste Disposal		Overall Confidence	Sources
					Fugitive Air	Stack Air	Landfill	Incineration		
Manufacture (Import)	Import	Import	Repackaging	☑	☑	☑	☒	☒	Medium	Peer-reviewed literature[e] (GS/ESD)
Processing	Incorporated into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	☑	☑	☑	☑	☒	High	Peer-reviewed literature[e] (GS/ESD)
			Incorporation into paints and coatings – 2-part coatings	☑	☑	☑	☒	☑	High	Peer-reviewed literature[e] (GS/ESD)
	Incorporated into formulation, mixture, or reaction product	Polymers used in aerospace equipment and	Formulation of TCEP-containing reactive resins (for use in 2-part systems)	☑	☑	☑	☒	☑	High	Peer-reviewed literature[f] (GS/ESD)
	Incorporated into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	☒	☑	☑	☑	☒	High	Peer-reviewed literature[e] (GS/ESD)
	Recycling	Recycling	Recycling e-waste	☐	☐	☐	☐	☐	Medium	NIOSH HHE's used for exposure estimates; insufficient data to estimate releases

Life Cycle Stage	Category	Subcategory	OES	Surface Water	Air		Waste Disposal		Overall Confidence	Sources
					Fugitive Air	Stack Air	Landfill	Incineration		
Distribution	Distribution	Distribution in commerce	Distribution in Commerce	Distribution (Table 3-1)						
Industrial Use	Other use	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Installation of article	☒	☒	☒	☒	☒	Medium	Releases not expected to occur during handling of aerospace articles
	Paints and coatings	Paints and coatings	Use of paints and coatings – Spray application OES 1,000 kg daily throughput	☑	☑	☒	☒	☒	Medium	Peer-reviewed literature[e]

Life Cycle Stage	Category	Subcategory	OES	Surface Water	Air		Waste Disposal		Overall Confidence	Sources
					Fugitive Air	Stack Air	Landfill	Incineration		
Commercial Use	Other use	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Use and/or maintenance of aerospace equipment and products	☒	☒	☒	☒	☒	Medium	Releases not expected to occur during handling of aerospace articles
	Paints and coatings	Paints and coatings	Use of paints and coatings – Spray application OES 1,000 kg daily throughput	☑	☑	☒	☒	☒	Medium	Peer-reviewed literature[e]
	Other use	Laboratory chemicals	Lab chemical – Use of laboratory chemicals	☑	☑	☑	☒	☒		Peer-reviewed literature[e]
	Furnishing, cleaning, treatment/care products	Fabric and textile products		☐	☐	☐	☐	☐	Medium	Peer-reviewed literature[e]
		Foam seating and bedding products		☐	☐	☐	☐	☐	Medium	Peer-reviewed literature[e]
	Construction, paint, electrical, and metal products	Building/ construction materials – Insulation		☐	☐	☐	☐	☐	Medium	Peer-reviewed literature[e]
		Building/ construction materials – Wood and engineered wood products – Wood resin composites		☐	☐	☐	☐	☐	Medium	Peer-reviewed literature[e]
Disposal			Disposal	Evaluated as part of each OES as opposed to a standalone OES						

☑= Estimated releases ☒= Estimated releases but not anticipated ☐= Releases not quantified, assessed qualitatively

3.2.2 Consumer Releases

Environmental releases to the environment may occur from consumer articles containing TCEP via the end-of-life disposal and demolition of consumer articles in the built environment, as well as from the associated down-the-drain release of TCEP from domestic laundry that removes TCEP containing dust from clothing to wastewater. It is difficult for EPA to quantify these ends-of-life and down-the-drain laundry exposures due to limited reasonably available information on source attribution of the consumer COUs. In previous assessments, EPA has considered down-the-drain analysis for consumer products scenarios where there is reasonably foreseen exposure scenario where it can be assumed the consumer product (*e.g.*, drain cleaner, lubricant, oils) will be discarded directly down-the-drain. Although EPA acknowledges that there may be TCEP releases to the environment via the demolition and disposal of consumer articles and the release of TCEP to wastewater via domestic laundry, the Agency did not quantitatively assess these scenarios due to lack of reasonably available information. EPA instead assessed them qualitatively. Anecdotal information in the peer-reviewed literature helps qualitatively describe how TCEP may be potentially released to the environment from consumer articles (see Section 5.1.2.2.5).

3.2.3 Weight of Scientific Evidence Conclusions for Environmental Releases from Industrial, Commercial, and Consumer Sources

For each OES, EPA considered the assessment approach, the quality of the data and models, and uncertainties in assessment results to determine a level of confidence as presented in Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S. EPA, 2024n). EPA determined that the various GSs and ESDs had overall quality determinations of high or medium, depending on the GS/ESD. The GSs and ESDs are documents developed by EPA or OECD that are intended to provide an overview of an industry and identify potential release and exposure points for that industry; they cover processes and are not specific to any chemical. This lack of chemical specificity creates an uncertainty in the overall release estimate—the assessed parameter values may not always be representative of applications specific to TCEP use in each OES. Another uncertainty is lack of consideration for release controls. The GS/ESDs assume that all activities occur without any release controls and in an open-system environment where vapor and particulates freely escape. Actual releases may be less than estimated if facilities utilize pollution control methods. Although TCEP monitoring data would be preferred to modeled estimates from generic scenarios, monitoring data were not available for almost all the OESs included in the risk evaluation. EPA strengthened modeled estimates by using Monte Carlo modeling to allow for variation in environmental release calculation input parameters according to the GS/ESD and other literature sources. The Agency was unable to quantitatively assess releases to the environment from consumer products containing TCEP. See section 5.3.2.3.2 for additional details.

3.2.3.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the Environmental Release Assessment

Use of Reporting Year-Release Trends Analysis

The 2016 CDR only had one reporter of TCEP while the 2020 CDR had no reporters; it is assumed that TCEP has been largely phased out of products it was historically used in such as flexible and rigid foam products. EPA expects that any remaining current users of TCEP do not surpass the CDR reporting threshold of 25,000 lb per site-year (*i.e.*, <25,000 lb/year is used at any given site).

EPA searched the Discharge Monitoring Report (DMR) database for TCEP monitoring data from 2010 to 2021. Monitoring data were available for locations in California; however, TCEP was not detected in any of the effluents of the POTWs that were monitored (U.S. EPA, 2022a). DMR data are submitted by

NPDES permit holders to states or directly to EPA according to the monitoring requirements of the facility's permit. States are required to load only major discharger data into DMR and may or may not load minor discharger data. The definition of major vs. minor discharger is set by each state and could be based on discharge volume or facility size. Due to these limitations, some sites that discharge may not be included in the DMR dataset. It is uncertain the extent to which sites not captured in these databases release TCEP into the environment or whether the releases are to water, air, or landfill. TCEP was officially added to TRI at the end of 2022. However, companies will not have to report on their possible management and/or use of TCEP until July 2024. As of September 2024, there are no TRI entries for TCEP.

EPA also searched other databases including the Water Quality Portal (WQP), where monitoring trends indicate a downward trend of TCEP concentrations in surface water (see Figure 3-9).

Use of Generic Scenario and Emission Scenario Documents for Number of Facilities

In some cases, the number of facilities for a given OES was estimated using GSs and ESDs, which are peer-reviewed. These documents typically attempt to find and map applicable North American Industry Classification System (NAICS) codes to an OES. This is done by identifying keywords relevant to that OES and entering them into the search tool on the U.S. Census Bureau's website. The results are reviewed for relevancy and the most applicable NAICS codes are selected. It is possible that the NAICS codes selected may not fully represent all potential types of sites for a given OES.

Uncertainties Associated with Number of Release Days Estimate

EPA did not have site specific data for the number of release days for most OESs. Typically, in these cases, the Agency assumed that an activity occurs once per day (*e.g.*, a facility may process a single batch per day). In the event that this assumption leads to a number of operating days that exceeds 365 days, it may be assumed that a site will be processing more than one batch per day. Given the relatively small production volume of TCEP being assessed this situation was not encountered. However, it is possible that this could lead to either an under or over estimation of the number of release days. In certain circumstances, EPA chose 250 days a year as the upper bound of possible number of operating days because that is considered the maximum number of days a worker would be exposed, but for most OESs the number of release days was well under this value.

Concentrations in the Environment (Section 3.3):
Key Points

EPA evaluated the reasonably available information on concentrations of TCEP in the environment. The key points on environmental concentrations are summarized below:

- EPA assessed environmental concentrations of TCEP in air, water, and land (soil, biosolids and groundwater).
  - For the air pathway, measured data from a variety of locations within and outside of the United States provided TCEP concentrations near facilities and locations that would represent general population exposure, as well as in remote locations. EPA also modeled ambient air concentrations and deposition from facilities releasing TCEP to air. The Agency expects dry and wet air deposition of TCEP from air to land and surface waters may be an important source of TCEP to the ambient environment.
  - For the water pathway, EPA found measured data on TCEP in surface water, precipitation, groundwater, wastewater, and the sediment compartment. The Agency also modeled TCEP concentrations in surface water and sediment, including air deposition contributions to each, near facilities releasing TCEP. EPA expects surface water and sediment to be the main environmental exposure pathways for aquatic organisms.
  - For the land pathway, EPA found measured concentrations of TCEP in soil, biosolids, and groundwater. The Agency modeled soil concentrations from air deposition and biosolids as well as groundwater concentrations from landfill leachate. EPA does not expect TCEP concentrations to accumulate in soil; rather, TCEP in soil is expected to migrate to groundwater.

3.3 Concentrations in the Environment

The environmental exposure characterization focuses on aquatic and terrestrial releases of TCEP from hypothetical facilities that use, manufacture, or process TCEP under industrial and/or commercial COUs subject to TSCA regulations. To characterize environmental exposure, EPA assessed point estimate exposures derived from both measured and predicted concentrations of TCEP in ambient air, surface water, and landfills in the United States.

A literature search was also conducted to identify sources of TCEP monitoring and reported modeled data. The tornado plots in the subsequent sections are a summary of the monitoring for the various environmental media. The plots provide the range of media concentrations in monitoring various studies. The plots are split by U.S. and non-U.S. data, fraction (*e.g.*, vapor, gas, particle; see Figure 3-2 and the studies are ordered from top to bottom from newer to older data. The plots are colored to indicate general population, remote, near facility, and unknown population information.

For more information on TCEP monitoring data, please see the following documents:

- *Environmental Monitoring Concentrations Reported by Media Type* (U.S. EPA, 2024i);
- *Environmental Monitoring and Biomonitoring Concentrations Summary Table* (U.S. EPA, 2024h);
- *Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure.* (U.S. EPA, 2024x); and
- *Data Extraction Information for General Population, Consumer, and Environmental Exposure* (U.S. EPA, 2024r).

3.3.1 Ambient Air Pathway

EPA conducted systematic review to obtain concentrations of TCEP in ambient air. Section 3.3.1.1 displays the aggregated results of reported monitoring concentrations for ambient air found from the systematic review. Section 3.3.1.2 reports EPA modeled ambient air concentrations and deposition fluxes.

Ambient air concentrations of TCEP were measured in seven studies in the United States (Figure 3-2). Bradman et al. (2014) recorded a maximum concentration of 1.60 ng/m^3 at 14 early childhood education facilities in California between May 2010 and May 2011. Peverly et al. (2015) sampled TCEP in ambient air at 13 locations across Chicago, Illinois. They demonstrated that TCEP ambient air concentrations (maximum of 0.570 ng/m^3) were slightly higher nearer to downtown Chicago than suburban Chicago.

Moran et al. (2023) detected atmospheric concentrations of TCEP at Troutman Lake, AK of up to 2.8 ng/m^3. The air concentrations were in the vapor phase. TCEP was also found to be in deposition to Troutman Lake at magnitudes of 290 to 1,300 ng/m^2/day (Moran et al., 2023). Although TCEP potential for LRTP has been described in the literature, other comparable Arctic sites suggest local point sources may contribute to the vapor phase TCEP concentrations around Troutman Lake. Furthermore, passive samplers deployed near the Native Village of Savoonga, 63 km from Troutman Lake, reported TCEP below the detection limit of (0.01 ng/m3), one to two orders of magnitude lower than the concentrations reported by (Moran et al., 2023). Troutman lake lies directly south of the Village of Gambell on the northwest corner of Sivuqaq. The island is home to the Sivuqaq Yupik people who practice a traditional subsistence lifestyle. During the Cold War, a military instillation was developed on Sivuqaq due to its proximity to Russia. Upon closure of the installation, debris and chemical waste were buried by the military in disposal sites around the village and the adjacent Troutman Lake. Moran et al. (2023) suggests that the occurrence of these organophosphate flame retardants is unlikely to be associated with formerly used defense sites, because the defense site precedes the use of these chemicals. Rather, Moran et al. (2023) suggests that local waste disposal practices and nearby open-air landfills may represent important sources of TCEP concentrations to Troutman Lake. Although the Gambell West landfill was 900 m and 1,400 m from the northwest and northeast sampling locations, respectively, the concentration reported in this study (2.8 ng/m^3) was similar to those reported in Kerric et al. (2021), which described mean combined vapor and gas concentrations of 2.4 ng/m^3 around a municipal landfill near Montreal, Canada. The Gambell West Landfill is a class III community active trench and fill landfill that uses burn units for treatment. Deposition of TCEP was higher on the north end of the lake than the south end of the lake. The northwest site had the most deposition with a magnitude of 1,300 ng/m^2/day.

3.3.1.1 Measured Concentrations in Ambient Air

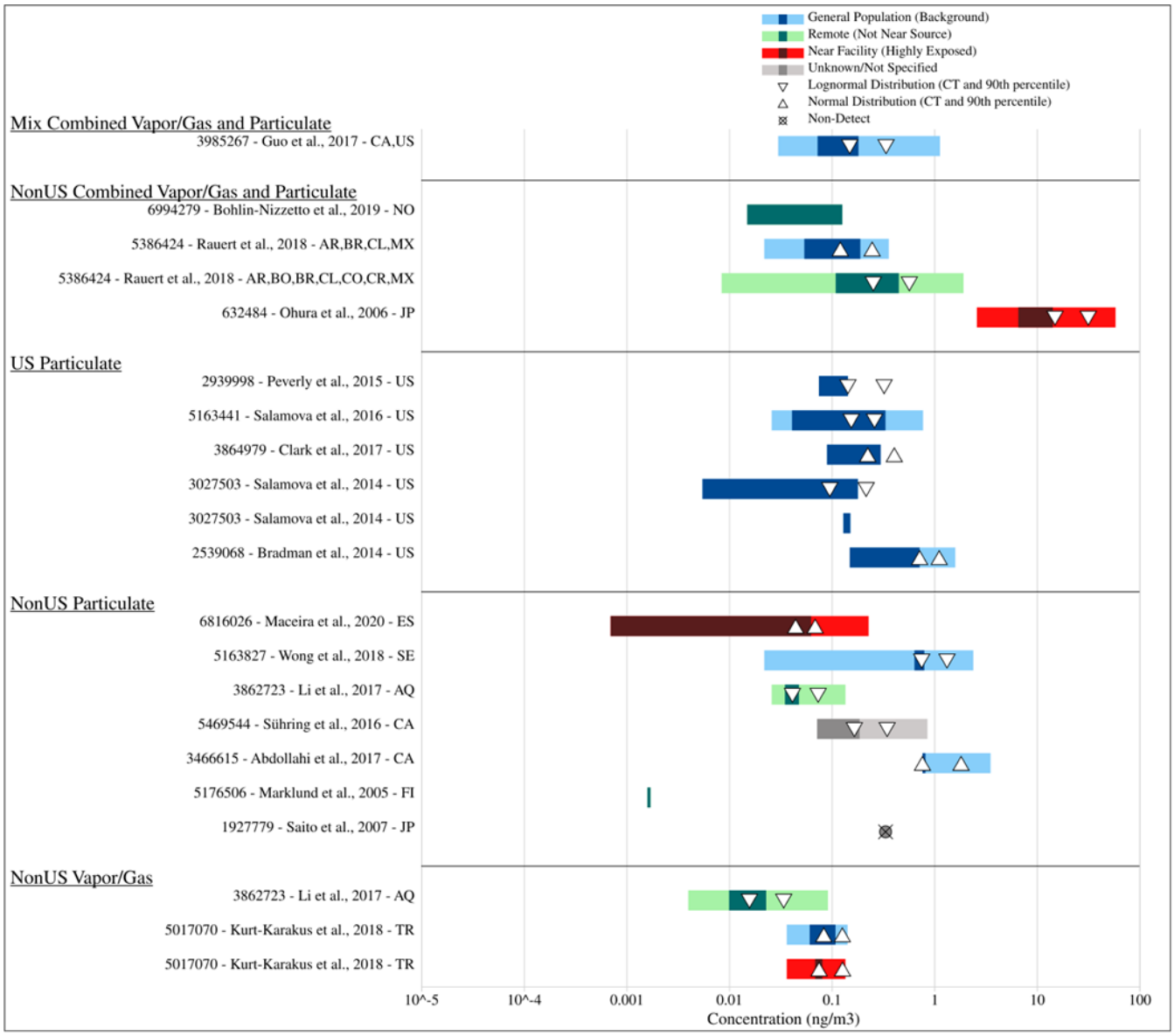


Figure 3-2. Concentrations of TCEP (ng/m^3) in Ambient Air from 2000 to 2019

3.3.1.2 EPA Modeled Concentrations in Ambient Air and Air Deposition (IIOAC/AERMOD)

EPA used the Integrated Indoor-Outdoor Air Calculator (IIOAC), and the American Meteorological Society (AMS)/EPA Regulatory Model (AERMOD) to estimate ambient air concentrations and air deposition of TCEP from facility releases. IIOAC uses pre-run results from a suite of AERMOD dispersion scenarios at a variety of meteorological and land-use settings, as well as release emissions, to estimate particle deposition at different distances from sources that release chemical substances to the air. AERMOD, a higher tier model, was utilized to incorporate refined parameters for gaseous as well as particle deposition. AERMOD is a steady-state plume model that incorporates air dispersion based on planetary boundary layer turbulence structure and scaling concepts, including treatment of both surface and elevated sources, and both simple and complex terrain.

Industrial and commercial release estimates are presented in Section 3.2. Table 3-3 provides the following COUs/OESs that have ambient air releases (stack or fugitive). These facility releases were utilized to model ambient air concentrations and deposition via AERMOD and IIOAC.

The full set of inputs and results of IIOAC and AERMOD are presented in Appendix I.3. For the initial IIOAC runs, EPA modeled each of the fugitive air and stack air release scenarios for the seven relevant OESs. In addition, due to initial uncertainties in the particle size, EPA ran IIOAC for both fine and coarse particle settings for TCEP. In IIOAC, all calculated air concentrations of fine and coarse particles are capped by an upper limit equal to the National Ambient Air Quality Standards (NAAQS) for particulate matter (PM). These limits are 35 and 150 μg/m^3 for fine and coarse particles (*i.e.*, the NAAQS for PM2.5 and PM10), respectively. These limits were met for all the OESs with stack emissions. In addition, this limit was reached for the fine particle size, fugitive emissions run for the commercial use of paints and coatings (see Appendix I.3).

A further limitation of IIOAC is that it does not model gaseous deposition. Due to the inability to model gaseous deposition, and due to the initial screening results meeting the NAAQS caps, EPA decided to run a higher tier model (AERMOD) for the ambient air pathway.

AERMOD is a steady-state Gaussian plume dispersion model that incorporates air dispersion based on planetary boundary layer turbulence structure and scaling concepts, including treatment of both surface and elevated sources and both simple and complex terrain. AERMOD can incorporate a variety of emission source characteristics, chemical deposition properties, complex terrain, and site-specific hourly meteorology to estimate air concentrations and deposition amounts at user-specified population distances and at a variety of averaging times. Readers can learn more about AERMOD, equations within the model, detailed input and output parameters, and supporting documentation by reviewing the *AERMOD Users' Guide* (U.S. EPA, 2018).

Additional parameters were required to run the higher tier model, AERMOD. EPA reviewed available literature and referenced the fenceline methodology, *Draft Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities Version 1.0* (U.S. EPA, 2022b), to select input parameters for deposition, partitioning factors between the gaseous and particulate phases, particle sizes, meteorological data, urban/rural designations, and physical source specifications. A full description of the input parameters selected for AERMOD and details regarding post-processing of the results are provided in Appendix I.3.3.

AERMOD was run under two land categories: suburban forested and bodies of water. A limited set of AERMOD tests suggested suburban-forest was a reasonable and appropriately health-protective default land-cover selection when land-cover analysis is not possible. Bodies of water typically led to the highest deposition values. Ambient air concentrations for both land categories for each OES are presented in Appendix I.3.3. Table 3-4 is an excerpt of the modeled annual air release data for the Use of paints and coatings – spray application OES, 2,500 lb production volume, 95th percentile release estimate, suburban forest land category scenario. The ambient air modeled concentrations and deposition values are presented for two meteorology conditions (Sioux Falls, South Dakota, for central tendency meteorology [MetCT]; and Lake Charles, Louisiana, for higher-end meteorology [MetHIGH]), 10 distances, and 3 percentiles (10th, 50th and 95th percentiles). These results indicate a maximum ambient air concentration of 2.55 ng/m^3 at 10 m from the facility and maximum deposition of 17.5 g/m^2 at 30 m from the facility for the Use of paints and coatings – spray application OES, 2,500 lb production volume, 95th percentile release estimate, suburban forest land category scenario.

Table 3-4. Excerpt of Ambient Air Modeled Concentrations and Deposition for the Use of Paints and Coatings – Spray Application OES, 2,500 lb Production Volume, 95th Percentile Release Estimate, Suburban Forest Land Category Scenario

Meteorology[a]	Distance (m)	Concentration (ng/m$^{3)}$ by Percentile			Deposition (g/m^2) by Percentile		
		10th	50th	95th	10th	50th	95th
MetCT	10	4.98E−01	9.27E−01	1.11	3.29	7.00	8.14
MetCT	30	1.11E−01	2.84E−01	4.16E−01	2.80	5.90	7.67
MetCT	30–60	5.80E−02	1.34E−01	2.86E−01	1.22	2.67	5.78
MetCT	60	3.40E−02	9.42E−02	1.58E−01	8.46E−01	1.87	2.58
MetCT	100	1.15E−02	3.36E−02	6.45E−02	2.82E−01	6.68E−01	9.63E−01
MetCT	100–1,000	1.09E−04	5.21E−04	4.90E−03	2.21E−03	9.07E−03	8.13E−02
MetCT	1,000	5.92E−05	1.82E−04	7.95E−04	1.39E−03	3.43E−03	9.51E−03
MetCT	2,500	7.91E−06	2.39E−05	1.49E−04	1.86E−04	4.53E−04	1.78E−03
MetCT	5,000	2.29E−06	8.21E−06	4.83E−05	5.36E−05	1.71E−04	6.49E−04
MetCT	10,000	7.68E−07	2.56E−06	1.76E−05	1.85E−05	5.44E−05	2.68E−04
MetHIGH	10	5.90E−01	1.03	2.55	5.88	1.04	3.29
MetHIGH	30	1.12E−01	2.71E−01	7.05E−01	2.74	6.69	17.5
MetHIGH	30–60	4.87E−02	1.27E−01	4.32E−01	1.29	3.17	11
MetHIGH	60	2.88E−02	8.69E−02	2.23E−01	7.09E−01	2.06	5.33
MetHIGH	100	8.77E−03	3.08E−02	8.21E−02	2.13E−01	7.06E−01	1.93
MetHIGH	100–1,000	6.85E−05	4.23E−04	4.60E−03	1.61E−03	9.60E−03	1.06E−01
MetHIGH	1,000	3.25E−05	1.62E−04	6.08E−04	7.75E−04	3.68E−03	1.47E−02
MetHIGH	2,500	4.54E−06	2.52E−05	9.06E−05	1.06E−04	5.21E−04	2.19E−03
MetHIGH	5,000	1.30E−06	9.54E−06	2.87E−05	3.03E−05	1.97E−04	6.75E−04
MetHIGH	10,000	2.74E−07	4.19E−06	1.32E−05	7.09E−06	8.75E−05	2.99E−04

[a] MetCT refers to meteorological conditions from Sioux Falls, South Dakota, and MetHIGH refers to meteorological conditions from Lake Charles, Louisiana. Because the scenarios are not at real locations, they were modeled twice using two different meteorological stations. These central tendency and high-end estimates were determined during the development of EPA's IIOAC.

3.3.1.2.1 Partitioning between Gaseous Phase and Particulate Phase

Dry and wet air deposition of TCEP to land and surface waters may be an important source of TCEP to the ambient environment. Air deposition may be the result of particle deposition and/or gaseous deposition.

There is conflicting information about the particle size of TCEP and whether TCEP is present in the gas or particle phase. A study of offices in China suggests that the mass median aerodynamic diameters (MMAD) of TCEP is coarse, between 4 and 5 µm, and that the contribution of TCEP is due to indoor rather than outdoor air (Yang et al., 2014). Another Chinese study suggests that only 22 percent of TCEP is found among particle size fractions of dust samples less than 43 µm (He et al., 2018c). A third

Chinese study indicates that the MMAD of TCEP is fine, between 1 and 2 µm (Cao et al., 2019). Schreder et al. (2016) indicates that TCEP is not detected in respirable particulate fractions (<4 µm). A team of Canadian scientists sought to make sense of these discrepancies by examining the gas-particle partitioning of organophosphate esters. Okeme (2018) evaluated gas-particle partitioning in indoor and outdoor air by using a group of single-parameter and poly-parameter models. Their predictions suggest that TCEP should be in the gas phase, contrary to measurements. Okeme (2018) suggests that the unexpectedly high particle fractions reported in many studies is due to sampling artifact. Okeme (2018) argues that many of the studies with high particle fractions do not account for safe sampling volumes, and that gas-phase sorption could be contributing substantially to the mass of TCEP captured on the filters. EPA adopted the recommendation of Okeme (2018) that many studies with the exception of Wolschke et al. (2016) and a few others, have likely mischaracterized the gas-particle partitioning of TCEP in air due to sampling artifact. Therefore, EPA selected a proportion of emissions in gaseous phase of 82 percent and the proportion in particle phase of 18 percent based on Wolschke et al. (2016).

3.3.2 Water Pathway

EPA conducted systematic review to obtain concentrations of TCEP in surface water, precipitation, and sediment. Sections 3.3.2.1, 3.3.2.3, 3.3.2.7, and 3.3.2.8 display the aggregated results of reported monitoring and reported modeled concentrations for surface water, precipitation, and sediment found as a result of systematic review. Section 3.3.2.4 provides surface water concentrations as a result of surface water databases. Sections 3.3.2.5, 3.3.2.6, 3.3.2.9, and 3.3.2.10 report EPA modeled surface water and sediment concentrations.

3.3.2.1 Geospatial Analyses of Environmental Releases

No location information is available for facilities that produce, manufacture, or use TCEP. The surface water data from the WQP shows TCEP concentration distributed across the United States. Figure 3-3 indicates the detected water concentrations from the WQP from 1995 to 2022. Many additional sample sites recorded non-detects, which are not shown in this figure.

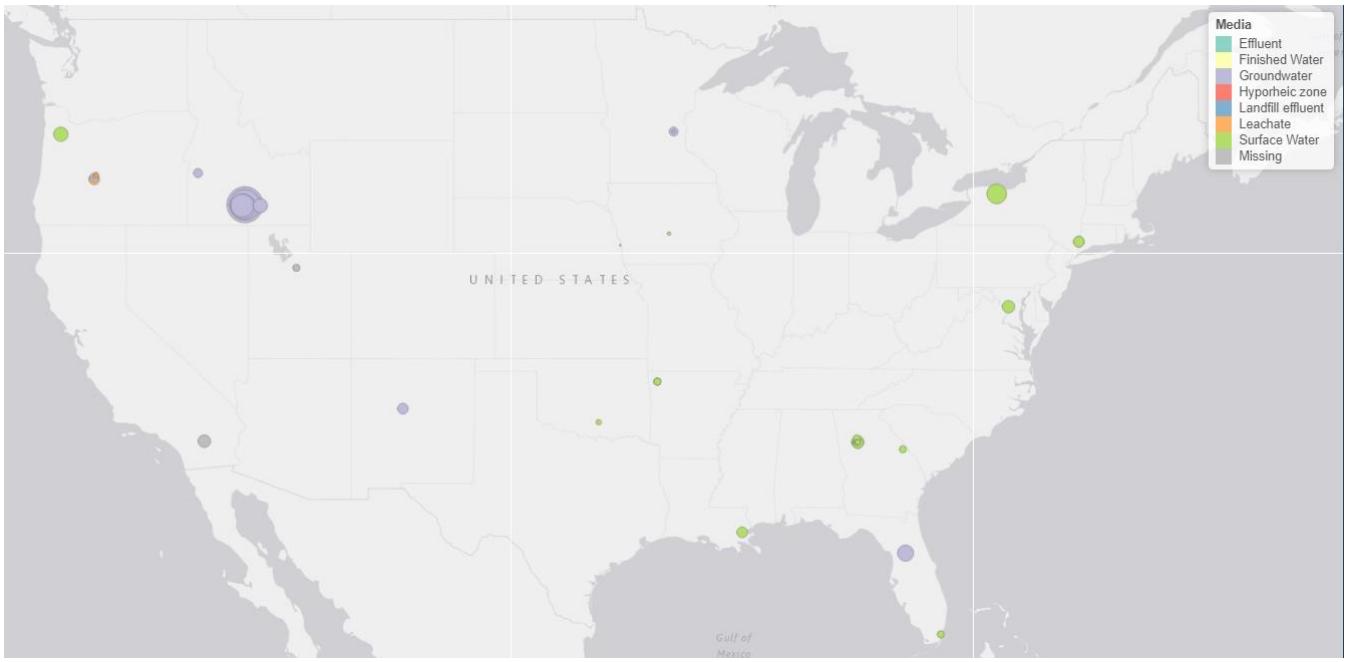


Figure 3-3. Map of Nationwide Measured TCEP Water Concentrations Retrieved from the WQP from 1995 to 2022

Source: EPA Accessible Link to Interactive Figure.

Size of the dots indicate magnitude of concentration; see Appendix I.2.1 for more details.

3.3.2.1.1 Geospatial Analysis for Tribal Exposures

Although EPA did not identify facilities that release TCEP on or near Tribal lands, TCEP has been detected in surface water and/or groundwater on or near Tribal lands. Groundwater samples collected in 2000 downgradient of the Norman Landfill had TCEP concentrations between 0.22 to 0.74 µg/L. Figure 3-4 indicates that the Norman Landfill was also located within a few miles from the Chickasaw Tribal Lands in Oklahoma. The landfill closed in 1985, was covered with a clay cap, and vegetated (Barnes et al., 2004).

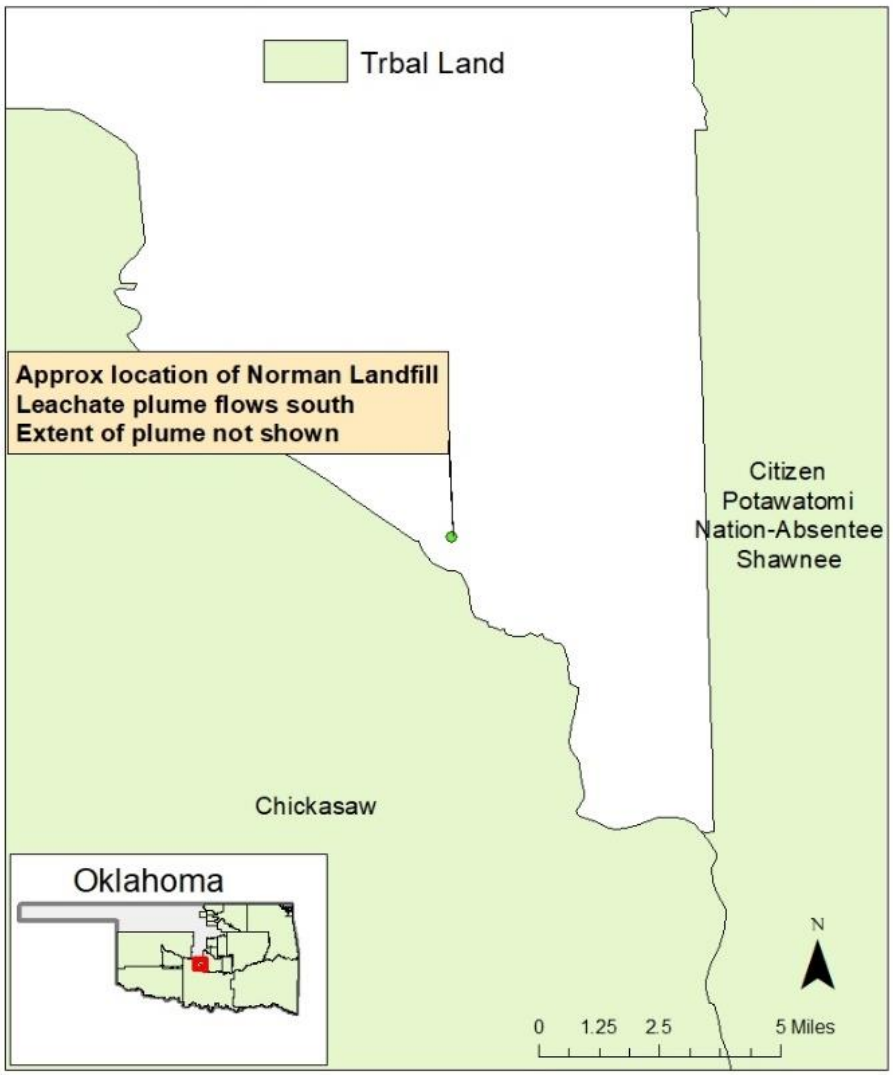


Figure 3-4. Map Indicating Norman Landfill in Proximity to Tribal Lands

In 2018, concentrations in groundwater of up to 2.4 µg/L were detected at the Twenty-Nine Palms Band of Missions Indians in Coachella, California (Figure 3-5). These concentration data were provided by EPA's STORage and RETrieval (STORET) Data Warehouse rather than collected as part of landfill monitoring efforts like the example above. This site was monitored again in 2019 (0.24 µg/L) and twice in 2021 (0.79 to 0.84 µg/L) (STORET via (NWIS et al., 2022)).

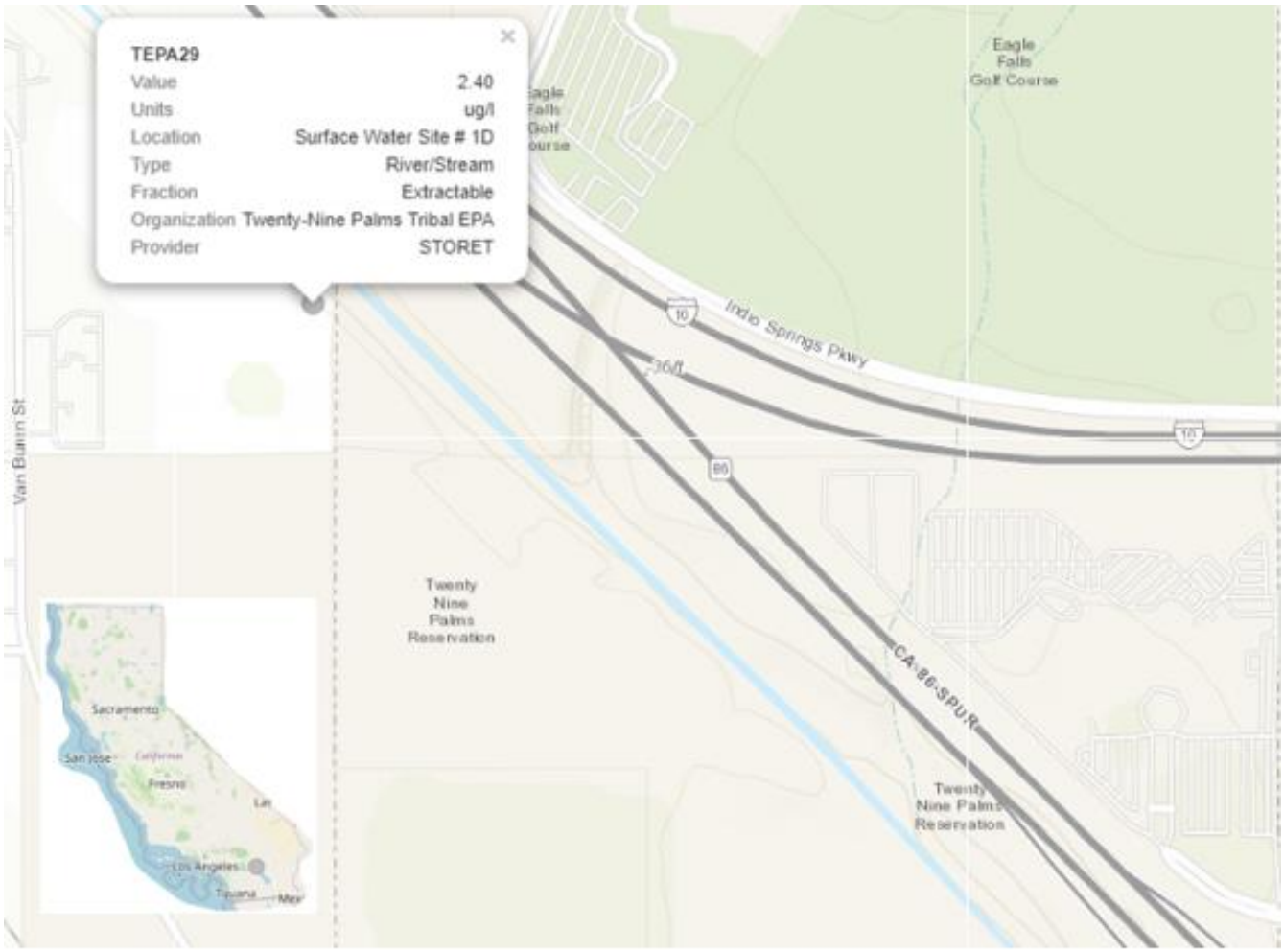


Figure 3-5. Groundwater Concentration of TCEP Reported near Twenty-Nine Palms Reservation near Coachella, California
Source: EPA Accessible Link to Interactive Figure.
See Appendix I.2.1 for more details.

3.3.2.2 Measured Concentrations in Surface Water

A summary of surface water monitoring studies is provided in Figure 3-6. Six U.S. studies were identified (five in the "US Not Specified" section and one in the "Mix Not Specified"). Sengupta et al. (2014) reported TCEP concentrations at 581 ng/L in October 2011 and 785 ng/L in July 2011 in the Los Angeles and San Gabriel Rivers during low flow conditions. TCEP concentrations in the Santa Clara River, California, were recorded up to 810 ng/L during low flow events in 2013 (Maruya et al., 2016).

A Korean study found midstream concentrations of TCEP 9 times higher than upstream values (234 vs. 15.0 ng/L) (Choo et al., 2018). This study suggested that a potential cause of the elevated TCEP concentrations was due to an industrial complex involving fiber manufacture being located near the midstream site.

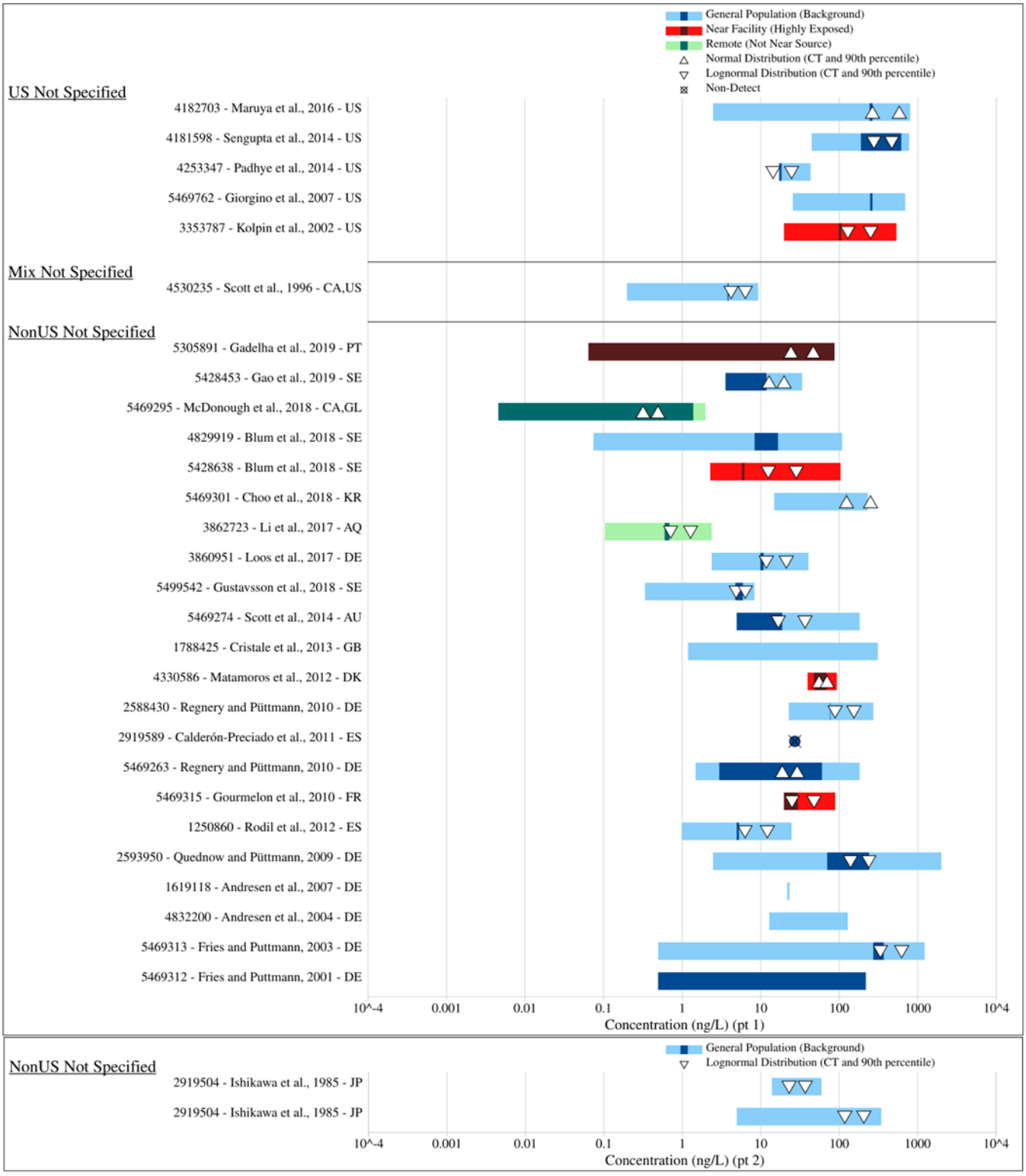


Figure 3-6. Concentrations of TCEP (ng/L) in Surface Water from 1980 to 2017

3.3.2.3 Measured Concentrations in Precipitation

Scott et al. (1996) recorded concentrations of TCEP in precipitation samples from 14.4 to 52.3 ng/L in Ontario, Canada, collected in 1994 (Figure 3-7).

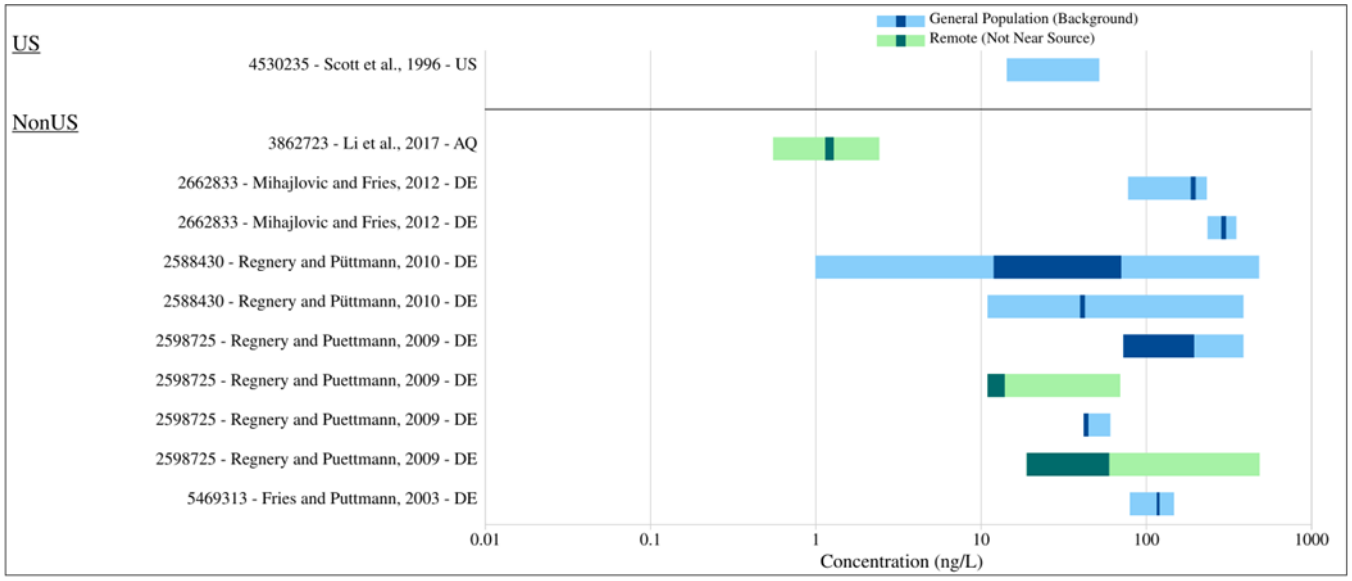


Figure 3-7. Concentrations of TCEP (ng/L) in Precipitation from 1994 to 2014

3.3.2.4 Measured Concentrations in Surface Water Databases

Measured surface water concentrations were obtained from EPA's Water Quality Exchange (WQX) using the WQP Tool, which is the nation's largest source of water quality monitoring data and includes results from EPA's STORage and RETrieval (STORET) Data Warehouse, the U.S. Geological Survey (USGS) National Water Information System (NWIS), and other federal, state, and Tribal sources.

The complete record of national monitoring of surface water reported by the WQP were reviewed to summarize the prevalence of TCEP in raw surface water (NWIS et al., 2022). Data retrieved in January 2023 included sampling dates from 2001 to 2022 and resulted in 9,892 available sample results (Figure 3-8.). Full details of the retrieval and processing of ambient surface water monitoring data from the WQP are presented in Appendix I.2. Figure 3-8. shows the range of TCEP concentrations detected in surface water samples the lowest detected sample concentrations within the data set are 0.02 µg/L. Most of the sample records available (95%) had no level of TCEP detected above the reported detection limit for the analysis (referred to as "non-detects"). The highest detection limit was 0.5 µg/L. The 466 detected values ranged from 0.47 to 7.66 µg/L, with a median of 0.23 µg/L.

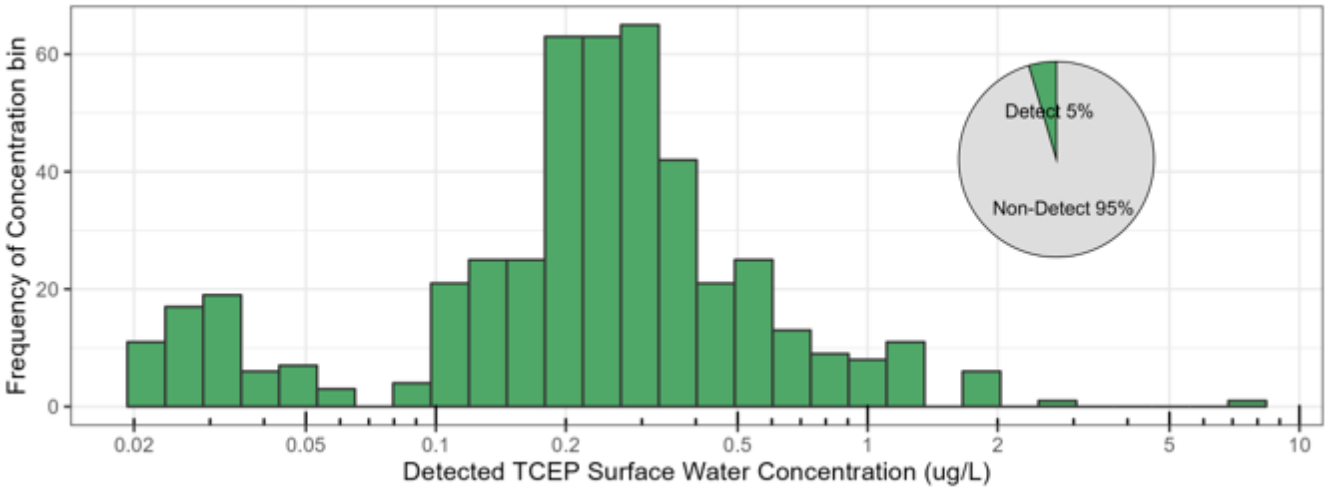


Figure 3-8. Frequency of Nationwide Measured TCEP Surface Water Concentrations Retrieved from the WQP from 2003 to 2022

The highest concentrations of TCEP detected in surface water in the United States is 7.66 µg/L, detected in August 2013 in Rochester, New York (NWIS via [WQP]) with a detection limit of 0.16 µg/L. This monitoring location is on the Genesee river at Ford Street bridge within 1,500 feet downstream of an abandoned Vacuum Oil plant on the west bank of the Rochester's Plymouth-Exchange neighborhood. The Vacuum Oil plant is a brownfield site that is being managed by the New York State Department of Environmental Conservation (DEC). EPA lacks data to confirm whether Vacuum Oil is the source of TCEP. Concentrations of up to 2.55 µg/L have been detected in Oregon as recent as October 2020 (STORET via [WQP]). Figure 3-9 demonstrates that surface water concentrations of TCEP have been decreasing over the past two decades.

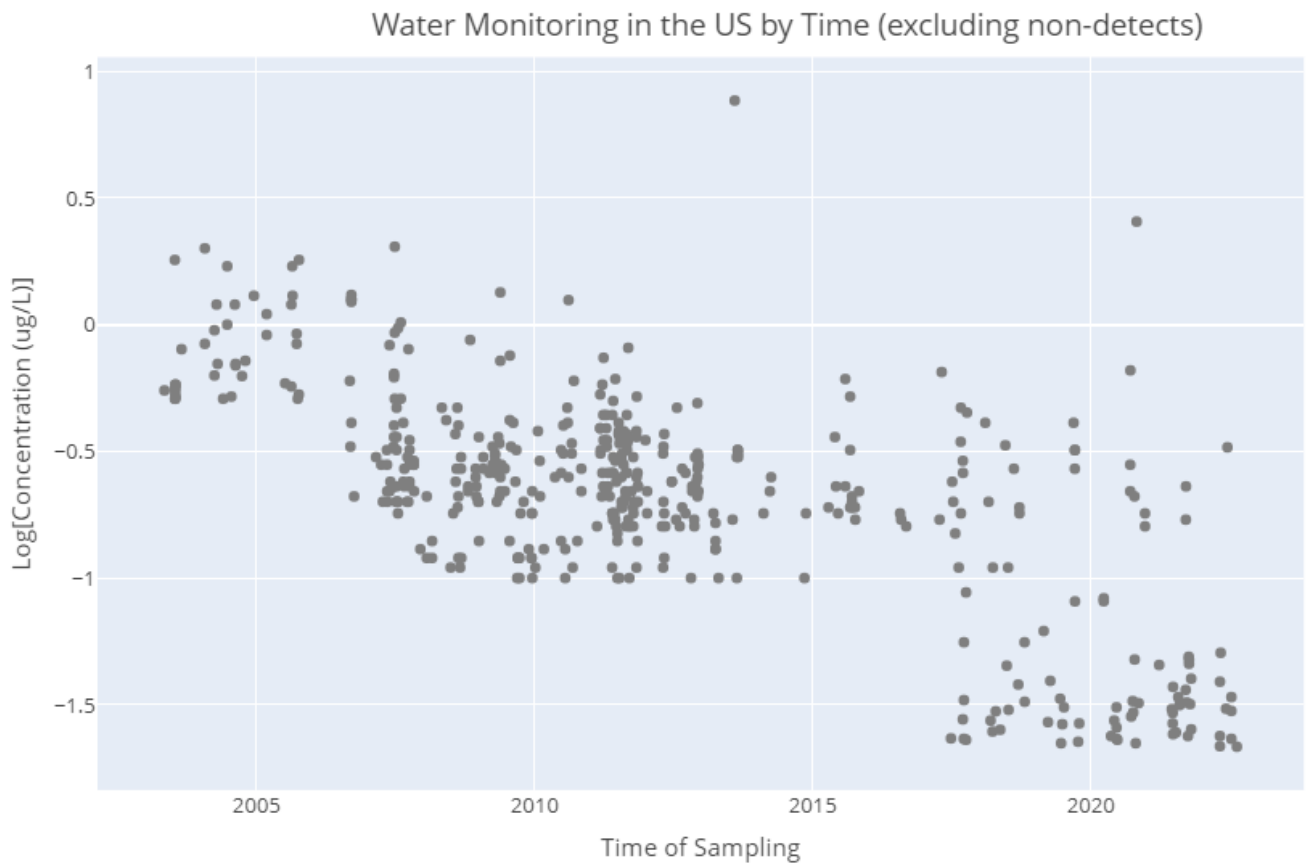


Figure 3-9. Time Series of Nationwide Measured TCEP Surface Water Concentrations Retrieved from the WQP from 2003 to 2022
See Appendix I.2.1 for more details.

3.3.2.5 EPA Modeled Surface Water Concentrations (E-FAST 2014, VVWM-PSC)

A tiered modeling approach was implemented for estimating surface water concentrations of TCEP. EPA's Exposure and Fate Assessment Screening Tool, version 2014 (E-FAST 2014) (U.S. EPA, 2007b), a simple dilution-based model, was first used to estimate total chemical surface water concentrations in streams. As E-FAST 2014 does not consider chemical partitioning into various media due to physical and chemical properties (K_{OW}, K_{OC}), it tends to overestimate total surface water concentrations and underestimate the chemical concentration that is sorbed to soil. Because TCEP's physical and chemical properties lends it to potentially partitioning into various media (see Section 2.2.2), E-FAST 2014-derived exposures that were greater than the most conservative environmental- or human health-relevant point of departure (POD) were triaged for further modeling using the VVWM-PSC model which incorporates partitioning and degradation. The VVWM-PSC model was also used to estimate settled sediment in the benthic region of streams.

Predicted surface water concentrations were modeled for facility releases as detailed in Section 3.2. The aquatic modeling was conducted with E-FAST 2014 using hypothetical annual release/loading amounts (kg/yr) and estimates of the number of days per year that the annual load is released. As appropriate, two scenarios were modeled per release: release of the annual load over an estimated maximum number of operating days per year. Additionally, the Probabilistic Dilution Model (PDM), a module of E-FAST 2014, was run to predict the number of days a stream concentration will exceed the designated COC value.

Table 3-5 release estimates are presented based on a 2,500 lb per site-year, high-end estimate release scenarios, the only deviation from this is the Use of paints and coatings and the Lab chemical OESs. These deviations are due to single site throughput constraints within the models used, in these cases, the PV of 2,500 lb/year was used to create a distribution of the possible number of sites. The 2,500 lb was not divided by COU, rather the full 2,500 lb was considered for each COU. Because CDR reporting is done on a per site-year basis, EPA estimated a 2,500 lb per site-year. Section 3.2 provides a summary of the release estimates for each COU/OES. For the maximum days of release scenarios, surface water concentrations under 7Q10 (*i.e.,* the lowest 7-day average flow that occurs [on average] once every 10 years) flow conditions for E-FAST 2014 ranged from 5.70×10^{2} to 1.11×10^{4} for the various exposure scenarios. Results for VVWM-PSC are overall slightly lower for all scenarios because VVWM-PSC accounts for additional sink effects that are not accounted for in E-FAST 2014. For more information on E-FAST 2014 and VVWM-PSC, including information on input parameters, see Appendix I.2.

Table 3-5. Summary of Modeled Surface Water Concentrations for the 2,500 lb, High-End Release Estimates

Life Cycle Stage	Category	Subcategory	OES	Inputs			E-FAST 2014	VVWM-PSC
				Days of Release	Estimated 7Q10 Flow (m^3/day)	Daily Pollutant Load (kg/day)	Daily Concentration – 7Q10 (µg/L)	Daily Concentration – 7Q10 (µg/L)
Manufacture	Import	Import	Repackaging	4	4,130	9.88	2,392	2,390
Processing	Incorporated into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	2	3,380	35.18	10,407	10,200
			Incorporation into paints and coatings – 2-part coatings	1	3,380	31.89	9,436	8,280
		Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resins	1	2,850	31.54	11,066	9,190
Commercial Use	Paints and coatings	Paints and coatings	Use of paints and coatings – Spray application	2	4,130	23.26	5,631	5,590
	Other use	Laboratory chemicals	Lab chemical – Use of laboratory chemicals	182	4,130	0.40	96	96
7Q10 = the lowest 7-day average flow that occurs (on average) once every 10 years								

3.3.2.6 EPA Modeled Surface Water Concentrations via Air Deposition (AERMOD)

A study in the lower great lakes suggested that TCEP undergoes net gas phase deposition to lakes at a flux of −3,980 ng/m^2 per day (Ma et al., 2021). Other studies in the open ocean have suggested that the air-water gas exchanges were dominated by volatilization from seawater to air for TCEP 146 ± 239 ng/m^2 per day (Li et al., 2017b). Moran et al. (2023) recently reported TCEP deposition to Troutman Lake, Alaska, at magnitudes of 290 to 1300 ng/m^2/day. The likely source of TCEP was suggested to be from nearby open-air landfills and local disposal sites (Moran et al., 2023).

EPA used IIOAC and AERMOD to estimate air deposition from facility releases and to calculate a resulting pond water concentration near a hypothetical facility. Pond water concentrations from air deposition were estimated for the COUs with air releases. Air deposition modeling was conducted using IIOAC and AERMOD. Due to limitations of IIOAC in incorporating gaseous and particulate deposition, deposition results from the AERMOD were utilized in calculating pond water concentrations. A description of the ambient air modeling and the deposition results are provided in Section 3.3.1.2. Using the modeled deposition rates, the TCEP concentration in pond water was calculated with the following equations:

Equation 3-1.

$$AnnDep = TotDep \times Ar \times CF$$

Where:

$AnnDep$	=	Total annual deposition to water body catchment (µg)
$TotDep$	=	Annual deposition flux to water body catchment (g/m^2)
Ar	=	Area of water body catchment (m^2)
CF	=	Conversion of grams to micrograms

Equation 3-2.

$$PondWaterConc = \frac{AnnDep}{Ar \times Pond\ Depth}$$

Where:

$PondWaterConc$	=	Annual-average concentration in water body (µg/L)
$AnnDep$	=	Total annual deposition to water body (µg)
Ar	=	Area of water body (m^2); default = 10,000 m^2 from EPA OPP standard farm pond scenario
Pond Depth	=	Depth of pond; default = 2 m from EPA OPP standard farm pond scenario
CF	=	Conversion of cubic meters to liters

Appendix I.3.3 presents the range of calculated pond water concentrations for the different emission scenarios. The highest estimated 95th percentile pond water concentration, across all exposure scenarios, for the 2,500 lb production volume, high-end estimate was for commercial use of paints and coatings scenario:

- 1.07×10^3 µg/L or 1,070 µg/L at 100 m from the source; and
- 8.10 µg/L at 1,000 m from the source.

3.3.2.7 Measured Concentrations in Wastewater

Laundry wastewater may be the primary source of TCEP to wastewater treatment plant influent and subsequently to the aquatic environment. This theory suggests that the TCEP in the indoor environment is transferred to indoor dust that is subsequently transferred to clothing. The dust is removed from the clothing during laundry and this wastewater reaches the wastewater treatment plants. Not all wastewater treatment plants are fully effective in removing TCEP, and the subsequent effluent may result in higher concentrations in the aquatic environment (Schreder and La Guardia, 2014). Wastewater monitoring data from multiple locations in Emeryville, California corroborates this theory, as the highest levels of TCEP were shown to come from industrial laundry services at levels of 3.72 µg/L in wastewater (Jackson and Sutton, 2008). A study in Albany, New York, between 2013 and 2015 indicated mean influent concentrations of 1,430 ng/L and effluent concentrations of 1,100 ng/L of TCEP (Kim et al., 2017). The monitoring data suggests that U.S. values of TCEP in wastewater appear to be higher than concentrations in other high-income countries as shown in Figure 3-10.

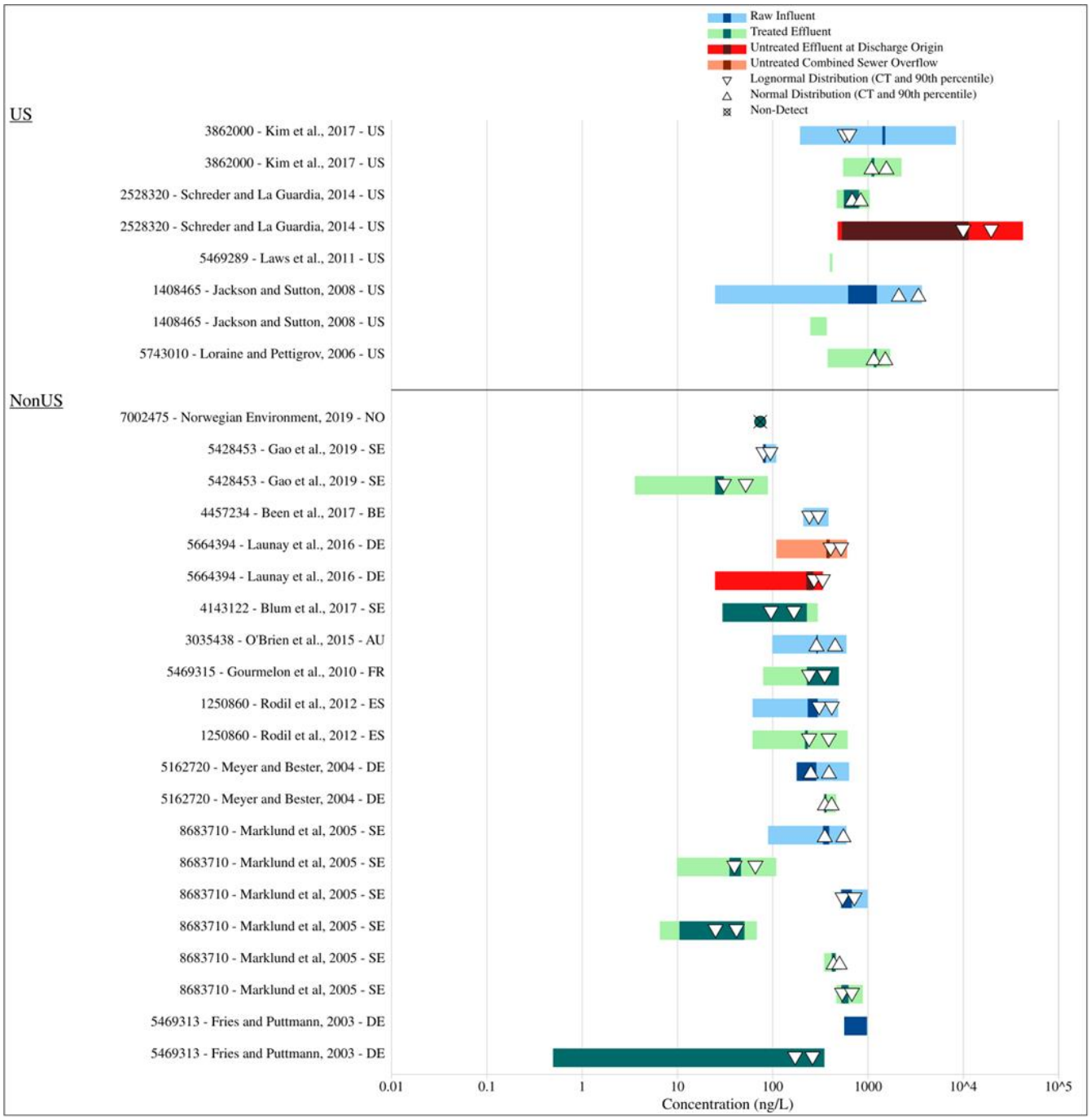


Figure 3-10. Concentrations of TCEP (ng/L) in Wastewater from 2001 to 2018

3.3.2.8 Measured Concentrations in Sediment

Limited information was available on measured concentrations of TCEP in sediment in the United States. Maruya et al. (2016) detected TCEP in coastal embayments at up to 6.98 ng/g dry weight in Marina Del Ray, Los Angeles, California, in 2013. The mean sediment TCEP concentration was 2.2 ng/g with a 90th percentile value of 4.0 ng/g Maruya et al. (2016). Concentrations of TCEP were reported at a maximum of 41 ng/g in sediment samples of the Elbe River at the mouths of five tributaries after a flooding event in Europe in August 2002 (Stachel et al., 2005).

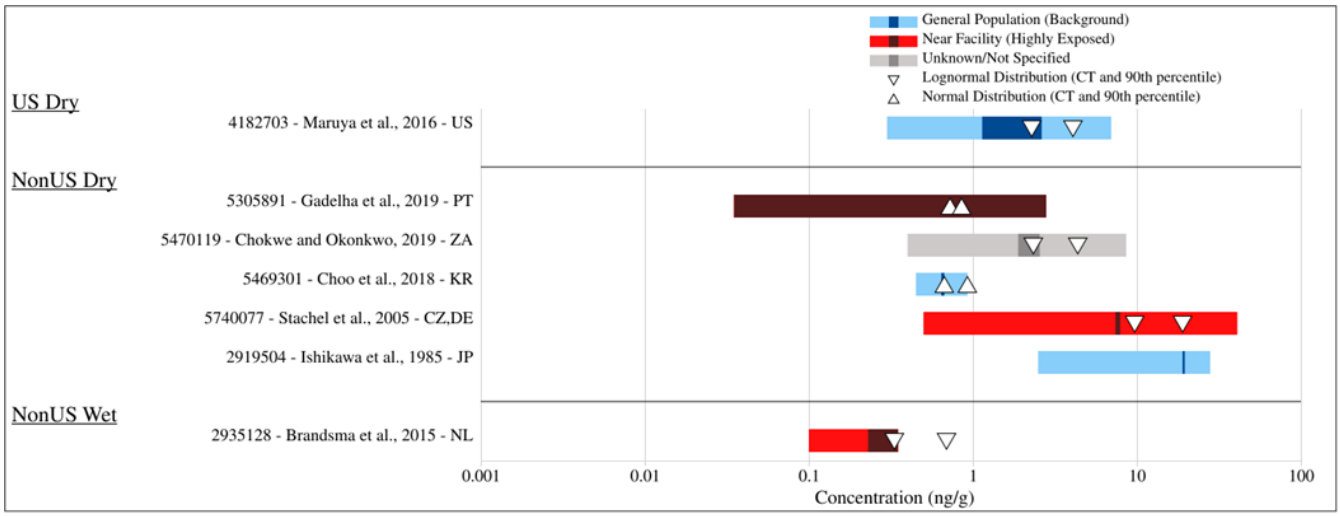


Figure 3-11. Concentrations of TCEP (ng/g) in Sediment from 1980 to 2017

Kawagoshi et al. (1999) reported TCEP concentrations up to 7,395 ng/g from waste disposal sites to the surrounding sea in Osaka, Japan. Although this study was published in 1999, this study explains that disposal sites may be important sources of TCEP in sediment concentrations.

3.3.2.9 EPA Modeled Sediment Concentrations (VVWM-PSC)

A summary of the benthic pore water and sediment concentrations modeled using VVWM-PSC are summarized by COU/OES in Table 3-6. Modeled estimates are presented for the 2,500 lb production volume, high-end estimate release scenarios. Section 3.2.2 provides a summary of the release estimates for each COU/OES. For the maximum day of release scenarios, sediment concentrations ranged from 8.94×10^{2} to 5.04×10^{3} µg/kg for the 2,500 lb production volume, high-end estimate release scenarios.

Table 3-6. Summary of Modeled Benthic Pore Water and Sediment Concentrations for the 2,500 lb Production Volume, High Estimate Releases

Life Cycle Stage	Category	Subcategory	OES	Inputs			VVWM-PSC	
				Days of Release	Estimated 7Q10 Flow (m^3/day)	Daily Pollutant Load (kg/day)	Benthic Pore Water Concentration (µg/L)	Sediment Concentration (ng/g)
Manufacture	Import	Import	Repackaging	4	4,130	9.88	153	894
Processing	Incorporated into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	2	3,380	35.18	334	1,960
			Incorporation into paints and coatings – 2-part coatings	1	3,380	31.89	152	893
		Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resins	1	2,850	31.54	182	1,070
Commercial Use	Paints and coatings	Paints and coatings	Use of paints and coatings – Spray application OES	2	4,130	23.26	177	1,040
	Other use	Laboratory chemicals	Lab chemical – Use of laboratory chemicals	182	4,130	0.40	90	380

For more information on the VVWM-PSC methodology, including inputs used, please see Appendix I.2.4.

3.3.2.10 EPA Modeled Sediment Concentrations via Air Deposition (AERMOD)

EPA used AERMOD to estimate air deposition from facility releases and calculate a resulting sediment concentration near a hypothetical facility. Sediment concentrations from air deposition were estimated for the condition of use scenarios with air releases. Air deposition modeling was conducted using IIOAC and AERMOD. Due to limitations of IIOAC in incorporating gaseous and particulate deposition, deposition results from the AERMOD were utilized in calculating sediment concentrations. A description of the modeling and the deposition results is provided above in Section 3.3.1.2. Additional details on IIOAC and AERMOD are presented in Appendix I.3.3. Using the modeled deposition rates, the TCEP concentration in sediment was calculated with the following equations:

Equation 3-3.

$$AnnDep = TotDep \times Ar \times CF$$

Where:

$AnnDep$	=	Total annual deposition to water body catchment (µg)
$TotDep$	=	Annual deposition flux to water body catchment (g/m^2)
Ar	=	Area of water body catchment (m^2)
CF	=	Conversion of grams to micrograms

Equation 3-4.

$$Sediment\ Concentration\left(\frac{\mu g}{kg}\right) = \frac{AnnDep}{Ar \times Mix\ \times\ Dens}$$

Where:

$Sediment\ Conc$	=	Annual-average concentration in water body (µg/kg)
$AnnDep$	=	Total annual deposition to water body (µg)
Ar	=	Area of water body (m^2); default = 10,000 m^2 from EPA OPP standard farm pond scenario
Pond Depth	=	Depth of pond; default = 2 m from EPA OPP standard farm pond Scenario
Mix	=	Mixing depth (m); default = 0.1 m
Dens	=	Density of sediment; default = 1,300 kg/m^3 from the European Commission Technical Guidance Document (ECB, 2003).

Appendix I.3.3I.3.3 presents the range of calculated sediment concentrations for the different emission scenarios. Equation 3-4 is conservative as it does not include a water solubility parameter. The highest estimated 95th percentile sediment concentration amongst all exposure scenarios was for the 2,500 lb production volume, high-end estimate release commercial use of paints and coatings scenario:

- 1.64×10^4 µg/kg or 16,400 µg/kg at "fenceline" population (100 m from the source); and
- 1.25×10^2 µg/kg or 125 µg/kg at "community" population (1,000 m from the source).

3.3.3 Land Pathway

EPA conducted systematic review to obtain concentrations of TCEP in soil, biosolids, and groundwater. Sections 3.3.3.1, 3.3.3.3, and 3.3.3.5 display the aggregated results of reported monitoring and reported modeled concentrations for soil, sediment, and groundwater found as a result of systematic review.

Section 3.3.3.7 provides groundwater concentrations from water databases. Sections 3.3.3.2, 3.3.3.4, and 3.3.3.8 report EPA modeled and estimated soil and groundwater concentrations.

3.3.3.1 Measured Concentrations in Soil

There are no reported soil concentrations of TCEP in the United States. A research team in Germany observed concentrations of TCEP from 5.07 to 23.48 ng/g dry weight. Snow melt appears to be a contributor to amplified soil concentrations. The highest soil concentrations were observed 1 day after snow melt at 23.48 ng/g, whereas soil concentrations at the same location before snowfall were below 8 ng/g. The meltwater generated at the snow surface percolated downwards due to gravity picking up chemicals present at the snow grain edge (Mihajlovic and Fries, 2012). These authors suggested that the source of the TCEP may be due to its use in cars (Mihajlović et al., 2011). TCEP levels ranged from 1.03 to 2.30 ng/g dry weight in Bursa, Turkey, a city known for its textile and automotive parts manufacturing (Kurt-Karakus et al., 2018).

3.3.3.2 EPA Modeled Soil Concentrations via Air Deposition (AERMOD)

EPA used AERMOD to estimate air deposition from facility releases and calculate a resulting soil concentration near a hypothetical facility.

Soil concentrations from air deposition were also estimated for the COUs with air releases (see Table 3-3 for a crosswalk of COU/OES with air releases). The air deposition modeling was conducted using IIOAC and then AERMOD. A description of the modeling and the deposition results is provided above in Section 3.3.1.2. Using the modeled deposition rates, the TCEP concentration in soil was calculated with the following equations:

Equation 3-5.

$$AnnDep = TotDep \times Ar \times CF$$

Where:

$AnnDep$	=	Total annual deposition to soil (µg)
$TotDep$	=	Annual deposition flux to soil (g/m^2)
Ar	=	Area of soil (m^2)
CF	=	Conversion of grams to micrograms

Equation 3-6.

$$SoilConc = \frac{AnnDep}{Ar \times Mix \times Dens}$$

Where:

$SoilConc$	=	Annual-average concentration in soil (µg/kg)
$AnnDep$	=	Total annual deposition to soil (µg)
Mix	=	Mixing depth (m); default = 0.1 m from the European Commission Technical Guidance Document (TGD) (ECB, 2003)
Ar	=	Area of soil (m^2)
$Dens$	=	Density of soil; default = 1,700 kg/m^3 from TGD (ECB, 2003)

The above equations assume instantaneous mixing with no degradation or other means of chemical reduction in soil over time and that TCEP loading in soil is only from direct air-to-surface deposition (*i.e.*, no runoff).

Appendix I.3.3 presents the range of calculated soil concentrations corresponding to the emission

scenarios considered. From the table, the highest estimated 95th percentile soil concentration amongst all exposure scenarios was for the commercial use of paints and coatings scenario:

- 1.14×10^{4} µg/kg at "fenceline" population (100 m from the source); and
- 8.65×10^{1} µg/kg at "community" population (1,000 m from the source)

3.3.3.3 Measured Concentrations in Biosolids

Wastewater and liquid waste treatment can result in effluent discharge to water and land application of biosolids. A study of a wastewater treatment plant in New York reported means of combined sludge concentrations (40.1 ng/g dry weight), ash (47.7 ng/g dry weight), and sludge cake (78.9 ng/g dry weight) (Kim et al., 2017). Wang et al. (2019c) reported mean and median TCEP concentrations of 10.6 ng/g dry weight and 2.46 ng/g dry weight respectively from a nationwide survey of sewage sludge sampled in 2006-2007. TCEP was detected up to 317 ng/g dry weight in this study which collected sewage sludge from wastewater treatment plants across 35 states in the United States (Wang et al., 2019c). Due to its persistence, and recalcitrance to anaerobic and aerobic degradation (Table 2-2. Environmental Fate Properties of TCEPTable 2-2), it is likely that dissolved TCEP will eventually reach surface water and groundwater via runoff after the land application of biosolids. TCEP has been found at concentrations of 4 ng/g in Canada in biosolids (Woudneh et al., 2015).

3.3.3.4 EPA Calculated Soil Concentrations via Biosolids

Section 2.2.3.1 indicates that TCEP will not be removed after undergoing wastewater treatment and will be retained in effluents with a low fraction being adsorbed onto sludge.

To assess soil concentrations resulting from biosolid applications, EPA relied upon modeling work conducted in Canada (EC/HC, 2011) that used Equation 60 from TGD (ECB, 2003), as follows:

Equation 3-7.

$$PEC_{soil} = \frac{C_{sludge} \times AR_{sludge}}{D_{soil} \times BD_{soil}}$$

Where:

PEC_{soil}	=	Predicted environmental concentration (PEC) for soil (mg/kg)
C_{sludge}	=	Concentration in sludge (mg/kg)
AR_{sludg}	=	Application rate to sludge amended soils (kg/m2/yr); default = 0.5 from Table A-11 of TGD
D_{soil}	=	Depth of soil tillage (m); default = 0.2 m in agricultural soil and 0.1 m in pastureland from Table A-11 of TGD
BD_{soil}	=	Bulk density of soil (kg/m3); default = 1,700 kg/m3 from Section 2.3.4 of TGD

The concentration in sludge was assumed as 0.079 mg/kg dry weight based on Kim et al. (2017). Using these assumptions, the estimated soil concentrations after the first year of application were 0.116 µg/kg in tilled agricultural soil and 0.232 µg/kg in pastureland.

A limitation of Equation 3-7 is that it assumes no losses from transformation, degradation, volatilization, erosion, or leaching to lower soil layers. Section 3.3.3.8 describes the potential leaching of TCEP from landfills. Additionally, it is assumed there is no input of TCEP from atmospheric deposition and there are no background TCEP accumulations in the soil. EPA has also assumed that there is only one application of biosolids per year.

3.3.3.5 EPA Modeled Soil Concentrations via BST

EPA modeled soil concentrations from EPA Office of Water's (OW) Biosolids Screening Tool (BST). The BST is a multimedia, multi-pathway, multi-receptor deterministic screening-level model that can estimate potential exposures with land application of biosolids. The BST was peer reviewed by the EPA Science Advisory Board in 2023 (EPA-SAB-24-001). More information is available in the BST Supporting Documents. The BST captures a 40-year application (one application every year) of biosolids to pastureland. Equation 3-7 calculates only one application.

Using BST, EPA estimates soil concentrations of 0.1412 mg/kg for the 2,500 lb, high-end estimate for the Incorporation into paints and coatings – 1-part coatings OES, and 0.5293 mg/kg for the 25,000 lb, central tendency estimate for the Incorporation into paints and coatings – 2-part reactive coatings OES.

Due to several uncertainties, EPA relied on an additional model and a few assumptions to streamline this analysis. A key input required in the BST is the input of dry biosolids concentration µg/g in dry weight. EPA used the RIVM SimpleTreat 4.1 model to estimate dry weight biosolids concentrations from the environmental release estimates (see Section 3.2.1.1). For this screening analysis, only COUs with the highest release estimates were modeled. A default annual biosolids land application rate of 1 kg/m^2/year and a TCEP biosolids concentration of 22.3 mg/kg, estimated using the SimpleTreat 4.0 wastewater treatment plant model, were used as input to the BST.

Table 3-7. BST Modeled Soil Concentrations for Incorporation into Paints and Coatings

OES	PV (lb)	Release Estimate (kg/day)	Release Estimate Divided by 365 (kg/day)	SimpleTreat 4.1 Combined Sludge (mg/kg)	BST Soil Concentration (mg/kg)
Incorporation into paints and coatings – 1-part coatings	2,500	35.2	0.20	5.95	0.1412
Incorporation into paints and coatings – 2-part reactive coatings	25,000	65.9	0.75	22.3	0.5293

There are uncertainties with how long biosolids are gathered before they are eventually land applied. It would be unreasonable to assume that the total TCEP released from a facility per year (whether it be a few days of release or many days of release) would be applied to biosolids all at once. Thus, the release estimate was divided by 365 to be more reflective of the land application scenario (1 application every year).

The resulting BST soil concentrations are one order of magnitude above the soil concentrations from modeled air deposition at 1,000 m (0.0865 mg/kg) and one order of magnitude above the soil concentrations observed in a Germany after snow melt (0.235 mg/kg) (Mihajlovic and Fries, 2012). The BST soil concentrations are three orders of magnitude higher than the Canadian analysis described in Equation 3-7 (0.0002 mg/kg). For a series of inputs and assumptions used in the BST analysis please see the *Supplemental File: Biosolids Screening Tool Modeling Results* (U.S. EPA, 2024d).

3.3.3.6 Measured Concentrations in Groundwater

TCEP was detected in a groundwater plume downgradient (0.22–0.74 µg/L) of the Norman Landfill, Oklahoma. The Norman Landfill is a municipal unlined landfill (subtitle D) established in 1920 and closed in 1985 (Barnes et al., 2004). One domestic well in Elkhart, Indiana, reported TCEP

concentrations of 0.65 to 0.74 µg/L between 2000 and 2002. This domestic well was near Himco Dump, a historical waste site, used for disposal until 1976 (Buszka et al., 2009). A study from Fort Devens, Massachusetts, reported concentrations of 0.28 to 0.81 µg/L at monitoring wells down-gradient of a land application facility (Hutchins et al., 1984). These studies suggest that there is potential for TCEP to migrate to groundwater and domestic wells from nearby non-hazardous waste landfills (*e.g.*, Norman Landfill) or historical waste sites (*e.g.*, Himco Dump, Indiana, Fort Devens, Massachusetts).

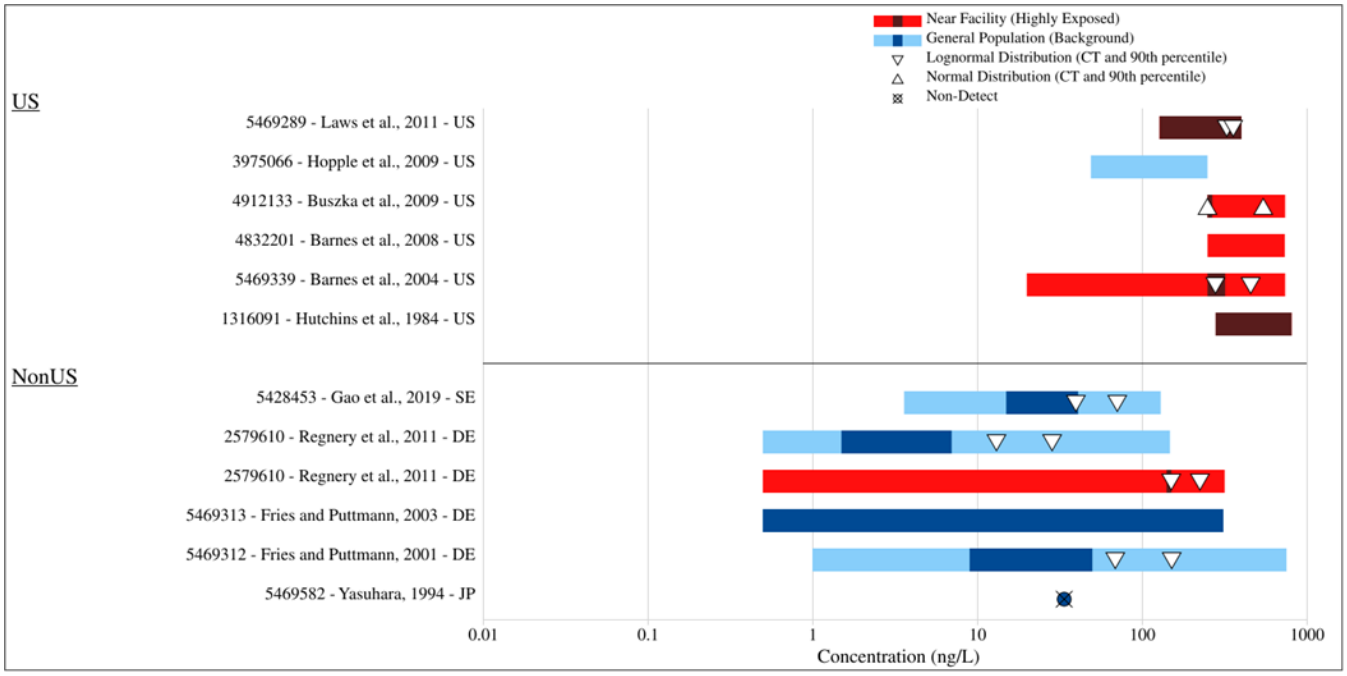


Figure 3-12. Concentrations of TCEP (ng/L) in the Not Specified Fraction of Groundwater from 1978 to 2017

A case study in Brazil, quantified TCEP up to 7.96 ng/L in well water samples downstream of waste area (Cristale et al., 2019). The waste area included bulky waste containing discarded upholstered furniture, and mattresses. This study demonstrates that consumer discarded waste may be a source of TCEP in landfills and subsequent migration to groundwater.

3.3.3.7 Measured Concentrations in Groundwater Databases

Data were retrieved from the WQP to characterize observed concentrations of TCEP in groundwater. These monitored values may or may not represent locations used as a source for drinking water and are analyzed to characterize the observed ranges of TCEP concentrations in groundwater—irrespective of the reasons for sample collection. Data retrieved in January 2023 included sampling dates from 1995 to 2021 and resulted in 51 detected results. Figure 3-13 shows most (98%, n = 3,325) of the sample records available had no TCEP detected above the reported detection limit for the analysis (referred to as "non-detects"). The 51 detects had a median value of 0.21 µg/L. Full details of the retrieval and processing groundwater monitoring data from the WQP are presented in Appendix I.2.

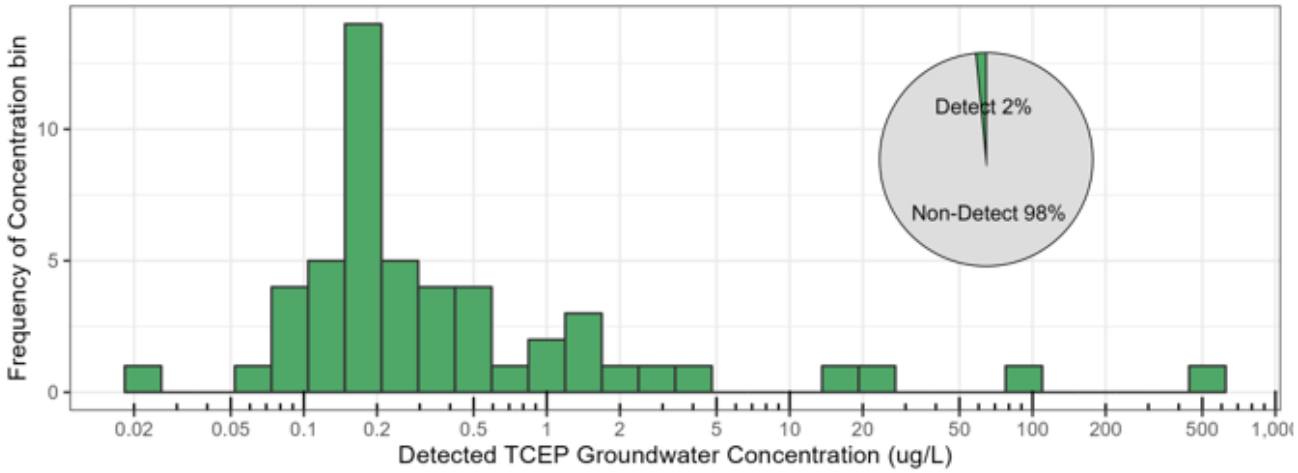


Figure 3-13. Frequency of Nationwide Measured TCEP Groundwater Concentrations Retrieved from the WQP from 1995 to 2021

The highest concentrations of TCEP detected in groundwater in the United States is 610 µg/L, detected in April 2002 in Idaho. Other samples at similar locations in April 2004 were an order of magnitude lower (2.8 to 94 µg/L) (NWIS et al., 2022). These estimates are from groundwater wells along the Gooding Milner Canal in the Magic Valley. Also in 2002, TCEP was detected in groundwater in Belleview, Florida, at a concentration of 3.5 µg/L. A more recent value (May 2017) detected TCEP in groundwater at a concentration of 2.4 µg/L in New Mexico. The New Mexico monitoring location is a well in the Four Hills Village in Albuquerque, New Mexico, which is about 1 to 2 miles from the Kirtland Air Force Base (AFB) Landfill. Generally, based on the WQP data, concentrations of TCEP in groundwater have been decreasing over the last two decades (Figure 3-14).

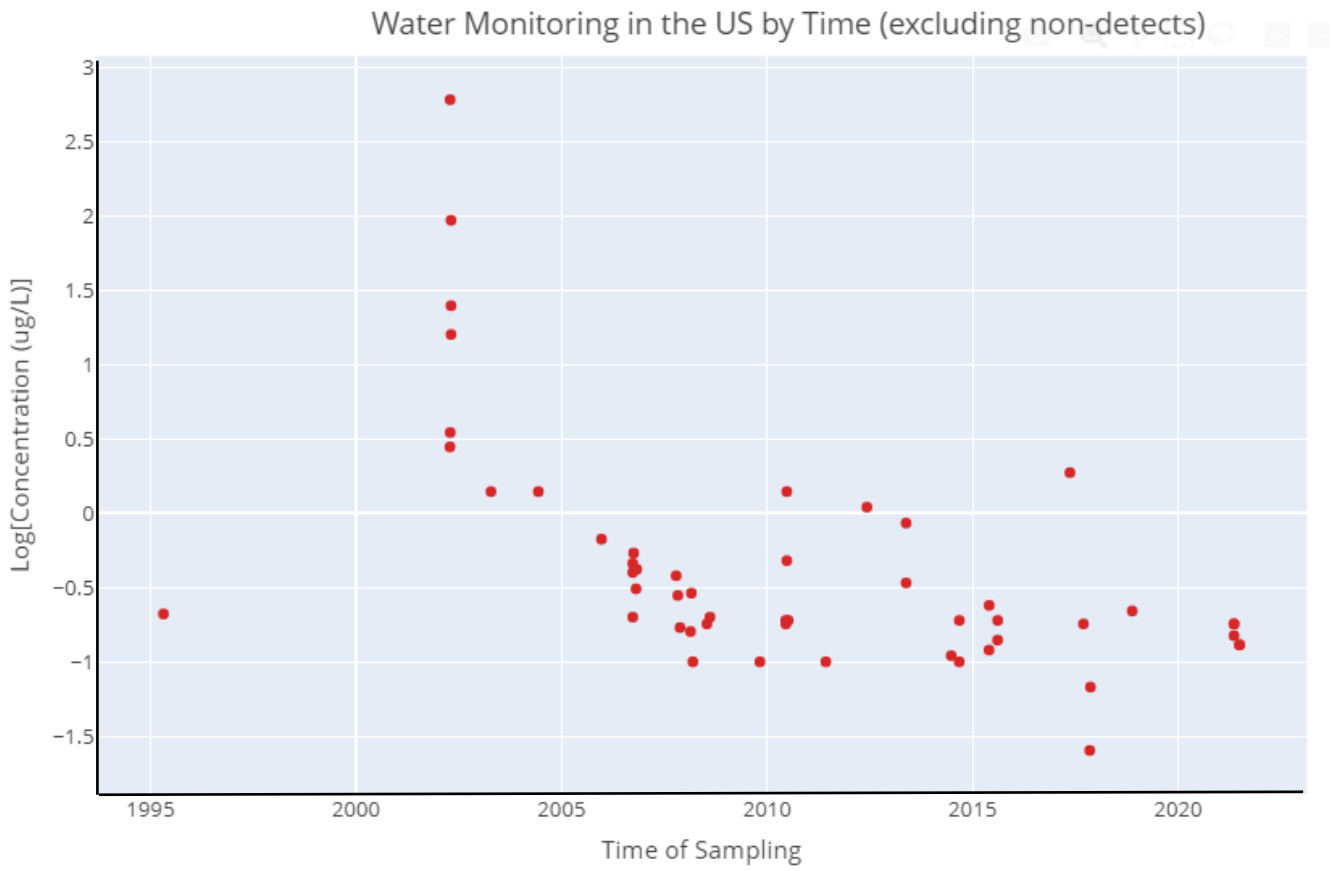


Figure 3-14. Time Series of Nationwide Measured TCEP Groundwater Concentrations Retrieved from the WQP from 1995 to 2021
See Appendix I.2.1 for more details.

3.3.3.8 EPA Modeled Groundwater Concentrations via Leaching (DRAS)

Landfills may have various levels of engineering controls to prevent groundwater contamination. These can include industrial liners, leachate capturing systems, and routine integration of waste. However, groundwater contamination from disposal of consumer, commercial, and industrial waste streams continue to be a prominent issue for many landfills throughout the United States (Li et al., 2015; Li et al., 2013). These contaminations may be attributed to perforations in the liners, failure of the leachate capturing system, or improper management of the landfills. Groundwater contamination with TCEP may occur when the chemical substance is released to landfills, underground injection wells, or surface impoundments. Due to its physical and chemical properties (*e.g.*, water solubility, Henry's Law constant) and fate characteristics (*e.g.*, biodegradability, half-life in groundwater), TCEP is anticipated to persist in groundwater for substantially longer than in other media.

Several sources of TCEP may contribute to groundwater concentrations including industrial facility releases and disposal of consumer products in landfills. With many manufacturing and processing uses phased out, EPA expects environmental releases of TCEP from industrial facilities to be declining. In fact, EPA has seen concentrations in surface water and groundwater generally declining over time. However, environmental releases from landfills may remain (or increase). EPA considered the potential for groundwater contamination following disposal of waste containing TCEP to landfills.

This assessment was completed using the (Hazardous Waste) Delisting Risk Assessment Software (DRAS). DRAS was specifically designed to address the Criteria for Listing Hazardous Waste identified

in 40 CFR 261.11(a)(3), a requirement for evaluating proposed hazardous waste delisting. In this assessment, DRAS is being utilized to determine potential groundwater concentrations of TCEP after TCEP-containing consumer products have been disposed of into a non-hazardous waste landfill. To understand possible exposure scenarios from these ongoing practices, EPA modeled groundwater concentrations of TCEP leaching from landfills where TCEP or consumer products containing TCEP have been disposed. The greatest potential for release of disposed TCEP to groundwater is from landfills that do not have an adequate liner system.

Potential groundwater concentrations resulting from disposal of TCEP to landfills vary across landfill loading rates and concentrations of TCEP in leachate. Estimated exposures presented here are therefore based on varying landfill conditions. Production volumes of 2,500 lb (1,134 kg) and 2,500,000 lb. (1,134,000 kg) are used as potential loading rates. To account for the uncertainties in estimating current loading rates, EPA varied loading rates over four orders of magnitude. While current production volumes are anticipated to be lower (2,500 lb), usage of TCEP was higher in the past and the leaching occurring from landfills may be a result of past disposal practices. Furthermore, these loading rates conservatively assume that a combination of raw TCEP and TCEP in commercial and consumer goods all goes to a single landfill each year.

The highest leachate concentrations observed in the literature was 177 µg/L (Masoner et al., 2014a). Masoner et al. (2014a) analyzed leachate concentrations from various landfills across the United States in 2011 and 2012. In 2011, the reported range of TCEP in leachate concentrations in these landfills ranged from 8.0×10^{-1} to 1.8×10^{2} µg/L, with a median of 1.0×10^{1} µg/L and a detection frequency of 35 percent. In 2012, the maximum leachate concentration was 9.1×10^{-1} µg/L with a detection frequency of 27 percent (Masoner et al., 2016). A value of 3.5 µg/L was observed from wastewater sampled from an aerospace/aircraft modification facility in Washington State in 2021 (WSDE, 2022).

To account for the uncertainties in these estimates a range of leachate concentrations were selected for the DRAS model from 1.0×10^{-4} to 1.0×10^{3} mg/L. The top of the range was bounded by TCEP's solubility. DRAS calculates a weight adjusted dilution attenuation factor (DAF) based on loading rates. The leachate concentration is divided by the DAF to calculate the groundwater concentrations. The resulting groundwater concentrations are potential concentrations that people living within one mile of a landfill might be exposed if the release were not identified and remediated. For more information on the DRAS model please see Appendix I.6.

Table 3-8. Potential Groundwater Concentrations (µg/L) of TCEP Found in Wells within 1 Mile of a Disposal Facility Determined Using the DRAS Model

Leachate Concentration (mg/L)	Loading Rate (kg)			
	1.13E−03	**1.13E−04**	**1.13E−05**	**1.13E−06**
1.00E–04	1.08E–06	1.01E–05	9.90E–05	9.43E–04
1.00E–03	1.08E–05	1.01E–04	9.90E–04	9.43E–03
1.00E–02	1.08E–04	1.01E–03	9.90E–03	9.43E–02
1.00E–01	1.08E–03	1.01E–02	9.90E–02	9.43E–01
1.00	1.08E–02	1.01E–01	9.90E–01	9.43
1.00E01	1.08E–01	1.01	9.90	9.43E01
1.00E02	1.08	1.01E01	9.90E01	9.43E02

Leachate Concentration (mg/L)	Loading Rate (kg)			
	1.13E−03	1.13E−04	1.13E−05	1.13E−06
1.00E03	1.08E01	1.01E02	9.90E02	9.43E03
Note: Concentrations are organized by potential loading rates (kg) and potential leachate concentrations (mg/L). Groundwater concentrations are in µg/L.				

EPA believes 2,500 lb (1.13×10^{3} kg) is the most suitable production volume for current uses of TCEP. The highest leachate concentrations observed in the literature was 177 µg/L (Masoner et al., 2014a). Using these two values (1.13×10^{3} kg and 0.177 mg/L), the expected groundwater concentrations are 1.08×10^{-3} to 1.08×10^{-2} µg/L. Disposals of TCEP containing materials were likely a few orders of magnitude higher than current levels when the TCEP production volume was higher and TCEP was more commonly used. If taking a production volume of 250,000 lb (1.13×10^{5} kg), and 0.177 mg/L leachate concentration the expected groundwater concentrations are 9.90×10^{-2} to 9.90×10^{-1} µg/L.

These estimates are within the range of the groundwater concentrations reported in the monitoring literature: 0.28 to 0.81 µg/L in a study from Fort Devens, MA monitoring wells down-gradient of a land application facility (Hutchins et al., 1984), 0.65 to 0.74 µg/L between 2000 and 2002 at a domestic well in Elkhart, Indiana near Himco Dump (Buszka et al., 2009). In addition, these values are below groundwater concentrations of 2.4 µg/L detected in May 2017 in New Mexico reported in the WQP. The New Mexico monitoring location is a well in the Four Hills Village in Albuquerque, New Mexico, which is about 1 to 2 miles from the Kirtland AFB Landfill.

3.4 Concentrations of TCEP in the Indoor Environment

TCEP – Concentrations in the Indoor Environment (Section 3.4): Key Points

EPA evaluated the reasonably available information for concentrations of TCEP in the indoor environment. The key points are summarized below:

- The indoor environment exposure characterization focused on consumer uses, disposals, and background exposures of TCEP.
  - Indoor air monitoring data show TCEP in particulate or vapor/gas form with concentrations primarily between 1×10^{-2} and 1×10^{4} ng/m^3.
  - Indoor dust is an important exposure pathway for TCEP. EPA found monitoring data showing a range of TCEP concentrations in indoor dust in residential spaces, public spaces, and vehicles, with concentrations as high as 167,532 ng/g in homes.

The indoor environment exposure characterization focuses on consumer uses, disposals, and background exposures of TCEP. In addition to the contribution from consumer uses, indoor environment TCEP concentrations were estimated from ambient contributions for air.

Note that indoor air and dust concentrations from consumer uses are presented in Section 5.1.2.

For more information on TCEP indoor monitoring and reported indoor modeling data, please see:

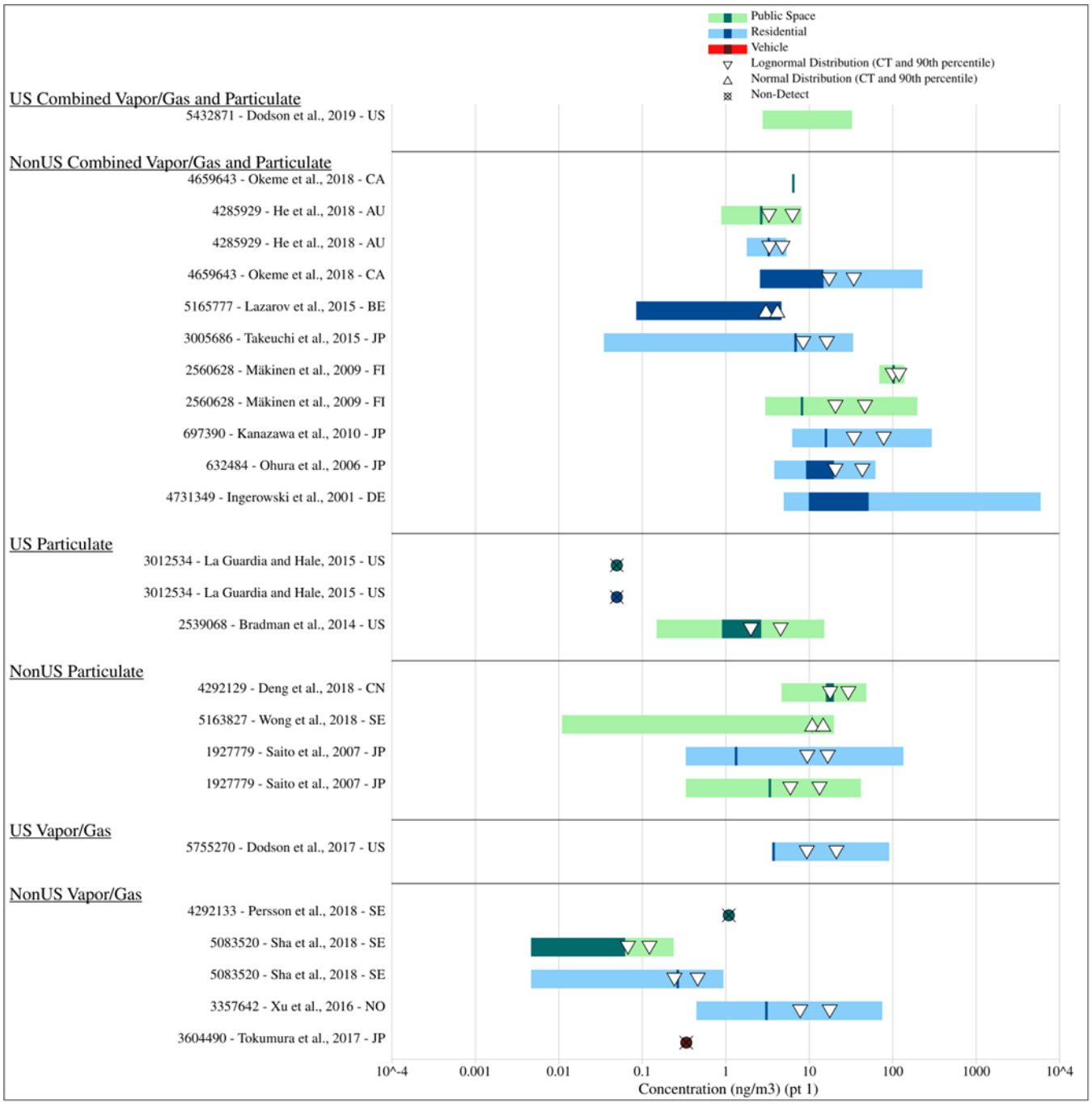
- *Environmental Monitoring Concentrations Reported by Media Type* (U.S. EPA, 2024i);

- *Environmental Monitoring and Biomonitoring Concentrations Summary Table* (U.S. EPA, 2024h);
- *Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure.* (U.S. EPA, 2024x); and
- *Data Extraction Information for General Population, Consumer, and Environmental Exposure* (U.S. EPA, 2024r).

3.4.1 Indoor Air Pathway

3.4.1.1 Measured Concentrations in Indoor Air

The indoor air monitoring data indicates indoor air concentrations primarily between 1×10^{-2} and 1×10^{4} ng/m^3 ranges. One study indicated particulate concentrations of TCEP of up to 1.1×10^{7} ng/m^3 max in PM2.5 (Wallner et al., 2012). This study may have had issues with sampling artifacts due to the use of glass filters as described by Okeme (2018) (see Section 3.3.1.2 for more details). There was only one study on vapor/gas in the United States. Dodson et al. (2017) reported a 95th percentile concentration of 37 ng/m^3 TCEP in vapor/gas.


Public Space
Residential
Vehicle
Lognormal Distribution (CT and 90th percentile)
Normal Distribution (CT and 90th percentile)
Non-Detect
US Combined Vapor/Gas and Particulate
5432871 - Dodson et al., 2019 - US
NonUS Combined Vapor/Gas and Particulate
4659643 - Okeme et al., 2018 - CA
4285929 - He et al., 2018 - AU
4285929 - He et al., 2018 - AU
4659643 - Okeme et al., 2018 - CA
5165777 - Lazarov et al., 2015 - BE
3005686 - Takeuchi et al., 2015 - JP
2560628 - Mäkinen et al., 2009 - FI
2560628 - Mäkinen et al., 2009 - FI
697390 - Kanazawa et al., 2010 - JP
632484 - Ohura et al., 2006 - JP
4731349 - Ingerowski et al., 2001 - DE
US Particulate
3012534 - La Guardia and Hale, 2015 - US
3012534 - La Guardia and Hale, 2015 - US
2539068 - Bradman et al., 2014 - US
NonUS Particulate
4292129 - Deng et al., 2018 - CN
5163827 - Wong et al., 2018 - SE
1927779 - Saito et al., 2007 - JP
1927779 - Saito et al., 2007 - JP
US Vapor/Gas
5755270 - Dodson et al., 2017 - US
NonUS Vapor/Gas
4292133 - Persson et al., 2018 - SE
5083520 - Sha et al., 2018 - SE
5083520 - Sha et al., 2018 - SE
3357642 - Xu et al., 2016 - NO
3604490 - Tokumura et al., 2017 - JP
10^-4
0.001
0.01
0.1
1
10
100
1000
10^4
Concentration (ng/m3) (pt 1)

(continued)

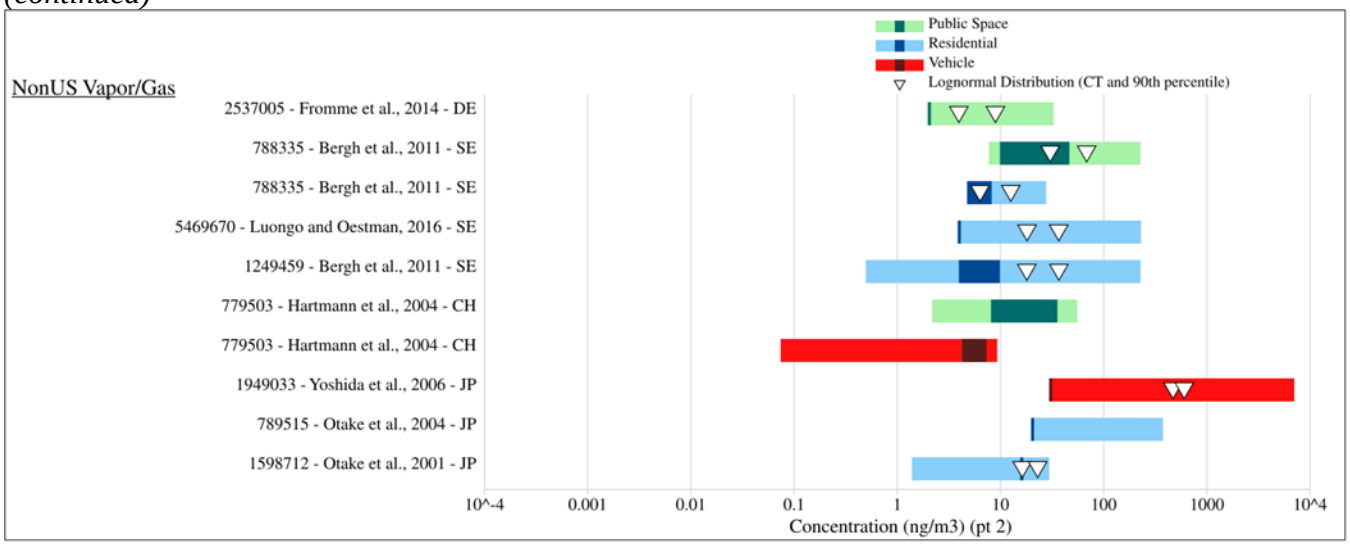


Figure 3-15. Concentrations of TCEP (ng/m^3) in Indoor Air from 2000 to 2016

3.4.1.2 Measured Concentrations in Personal Air

Two studies measured TCEP in personal air in the U.S. Personal air refers to the area within the breathing zone. Schreder et al. (2016) conducted a study on white-collar workers in urban, suburban, and rural areas of Washington State. Participants were instructed to wear an Institute of Occupational Medicine (IOM) sampler affixed to a shirt collar within the breathing zone continually during a 24-hour day during normal activities, including at home and at work, traveling to and from home and work, shopping, and socializing, and to wear or hang the sampler at breathing zone level during sleep. Schreder et al. (2016) reported mean and maximum inhalable (>4 µm) TCEP concentrations of 19.1 ng/m^3 and 77.8 ng/m^3 respectively, detected in 8/9 participants. La Guardia and Hale (2015) conducted a study measuring flame retardants among the personal air of four gymnastics coaches at their workplace and their homes. TCEP was not detected in the personal air of these coaches. Okeme et al. (2018) reported a median personal air concentration of three Canadian office workers of 34 ng/m^3. Polydimethylsiloxane (silicone rubber) brooches were used for the sampling methodology, and the three participants wore the samplers for 7 days.

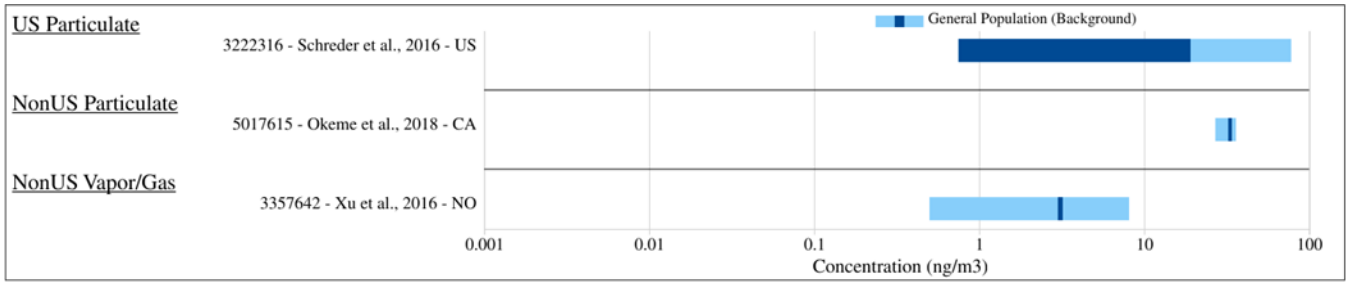


Figure 3-16. Concentrations of TCEP (ng/m^3) in Personal Inhalation in General Population (Background) Locations from 2013 to 2016

3.4.1.3 EPA Modeled Indoor Concentrations as a Ratio of Ambient Air

IIOAC calculates a mean and high-end indoor air concentration based on the outdoor/ambient air concentration and the mean and high-end indoor-outdoor ratios. In IIOAC, indoor-outdoor ratios of 0.65 and 1 are used for the mean and high-end ratios, respectively. The indoor-outdoor ratio of 0.65 is used to calculate indoor air concentrations corresponding to the mean outdoor air concentration for each

potentially exposed population. The indoor-outdoor ratio of 1 is used to calculate the indoor air concentration corresponding to the 95th percentile of outdoor air concentration of each potentially exposed population.

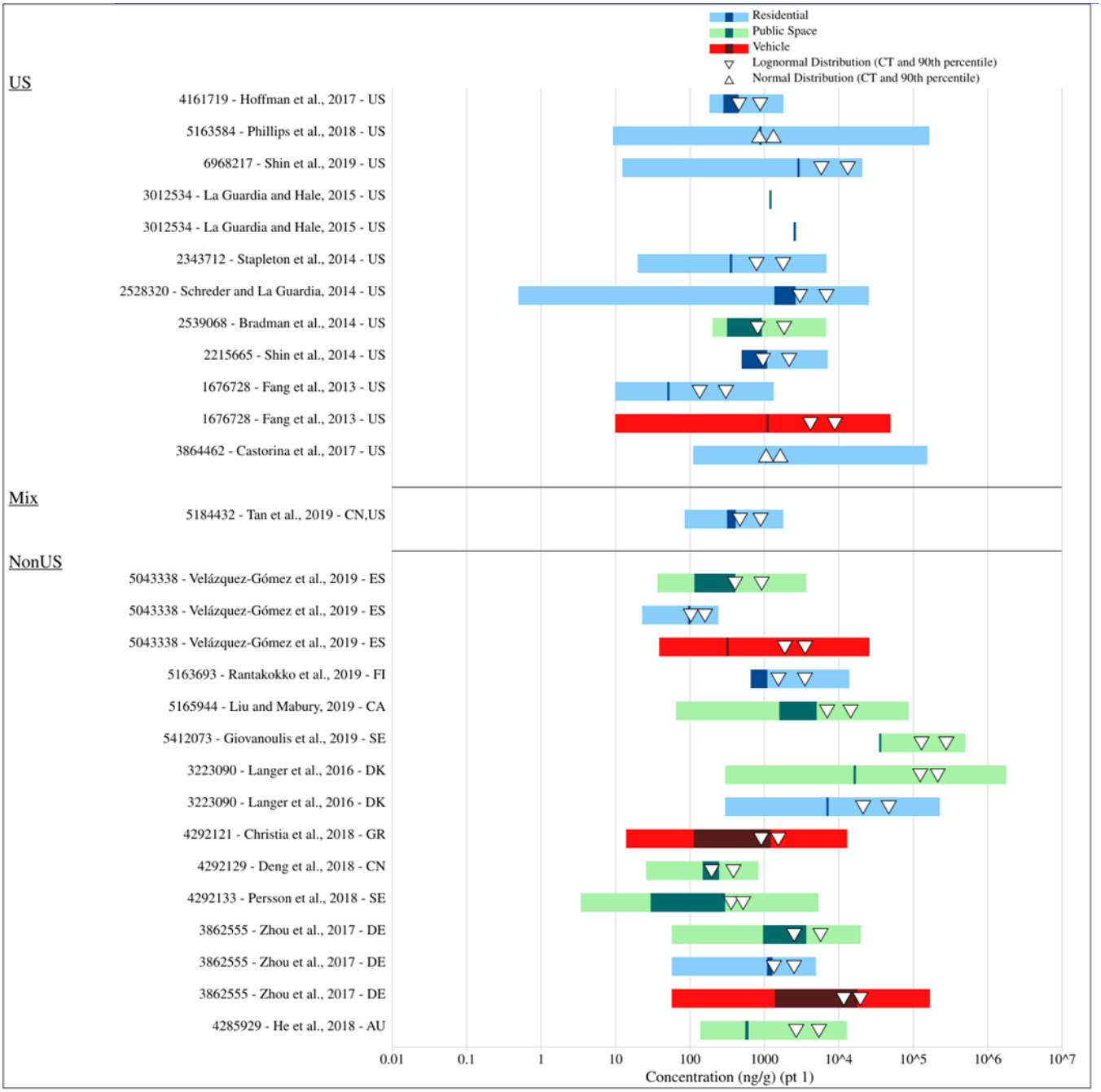
IIOAC was used as a tier 1 screening model before estimating ambient exposures via AERMOD. Results of IIOAC are presented in Appendix I.3.

3.4.1.4 Reported Modeled Concentrations in Indoor Air

Shin et al. (2014) reported TCEP emission rates in a whole house of 48.417 mg/day. Emission rate refers to the amount of chemical emitted per unit time. Shin et al. (2014) utilized fugacity-based indoor mass balance models to estimate whole-house emission rates of various SVOCS including TCEP.

3.4.2 Indoor Dust Pathway

3.4.2.1 Measured Concentrations in Indoor Dust



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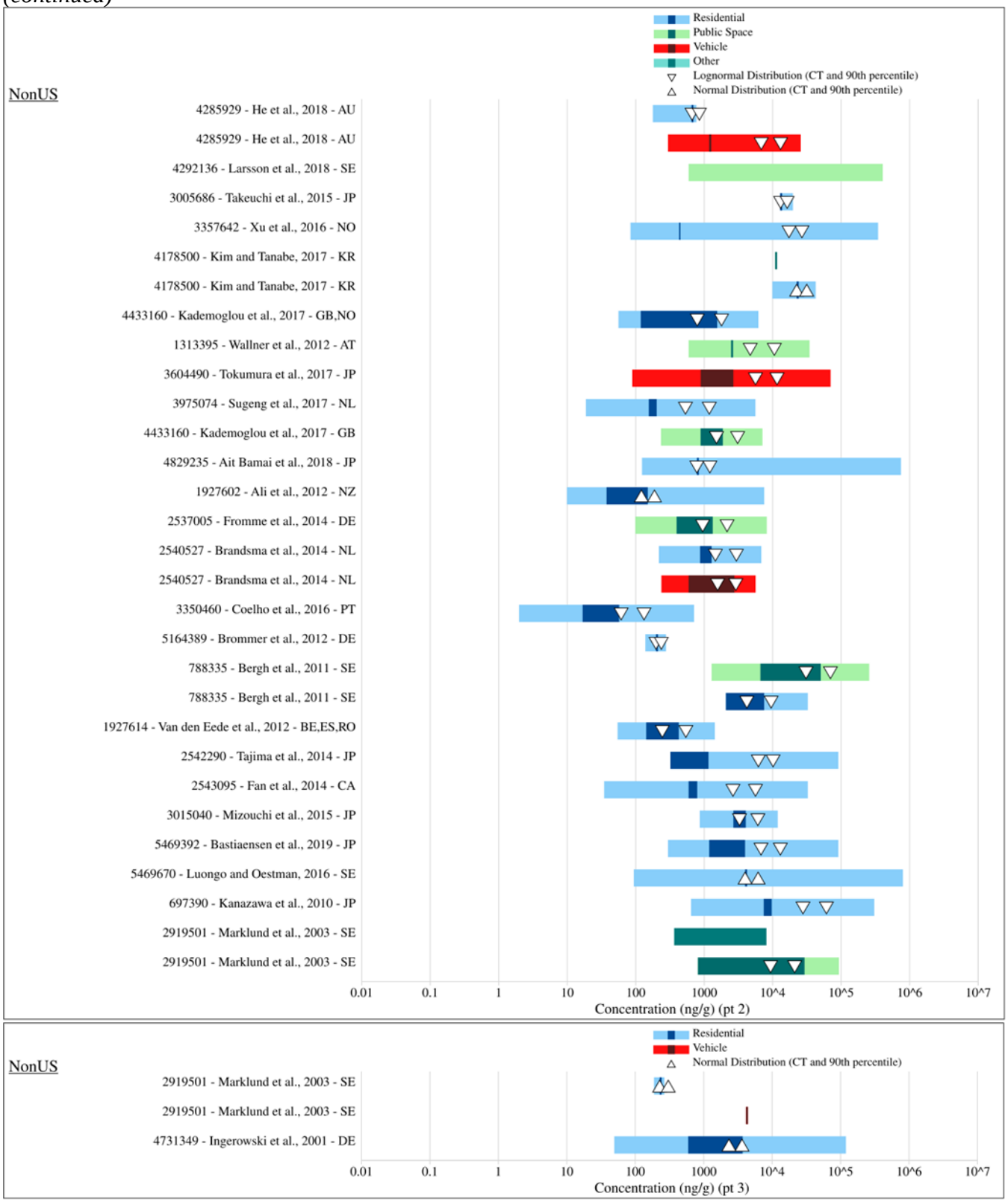


Figure 3-17. Concentrations of TCEP (ng/g) in Indoor Dust from 2000 to 2019

Concentrations of TCEP in dust were significantly higher in facilities with napping equipment (*e.g.*, foam beds and mats) made from foam (Bradman et al., 2014). Correlations between organophosphate

esters in dust and consumer products containing foams, furniture, and electronics strongly implicate household items as sources of these chemicals (Abafe and Martincigh, 2019). In the United States, concentrations of TCEP in dust are reported at 50.2 ng/g in houses and up to 1,080 ng/g in cars (Fang et al., 2013). Phillips et al. (2018) reported maximum concentrations of TCEP of 167,532 ng/g and a geometric mean of 864.1 ng/g in North Carolina homes from September 2014 to April 2016 as part of the Toddler's Exposure to SVOCs in the Indoor Environment (TESIE) study. A study of the Center for the Health Assessment of Mothers and Children of Salinas (CHAMACOS) cohort in California reported similar concentrations of TCEP as the TESIE cohort. It found that TCEP levels in dust are significantly associated with the presence of extremely worn carpets (Castorina et al., 2017).

3.4.2.2 Reported Modeled Concentrations in Indoor Dust

Castorina et al. (2017) reported modeled oral doses of 0.064 µg/kg-day for pregnant women via residential indoor dust in Salinas Valley, California. Schreder et al. (2016) reported 50th percentile modeled intakes for children (82.8 ng/day) and adults (41.4 ng/day). Ingerowski et al. (2001), a low-quality study, reported a range of dust intakes of from 0.2 to 2 µg/day.

Rantakokko et al. (2019) modeled inhalation, dermal, and oral intakes of TCEP in children from indoor dust. 50th percentile intakes were highest for dust ingestion (2.9 ng/kg-day) vs. dermal absorption (1.3 ng/kg/day) and inhalation (0.023 ng/kg-day). This suggests that for children's exposure to dust, oral routes may be the most important avenue of exposure. Kademoglou et al. (2017) modeled adult and toddler daily dust intakes from European homes and offices. They reported mean toddler dust intakes of 14.195 ng/kg/day for the high intake rate and 3.549 ng/kg/day in houses located in the United Kingdom. Adult intakes were higher in houses (0.624 ng/kg bw with high intake rate) vs. offices (0.0214 ng/kg bw with high intake for 8 hours spent in offices). The highest observed modeled dust intakes (1.38 µg/kg-day) were reported for children at a kindergarten in Hong Kong (Deng et al., 2018b).

4 ENVIRONMENTAL RISK ASSESSMENT

EPA assessed environmental risks of TCEP exposure to aquatic and terrestrial species. Section 4.1 describes the environmental exposures through surface water, sediment, soil, air, and diet via trophic transfer. Environmental hazards for aquatic and terrestrial species are described in Section 4.2, while environmental risk is described in Section 4.3. Updates since the draft risk evaluation within the Environmental Risk Assessment section include: (1) Integrated updated peer-reviewed literature search in February 2024 to fill data gaps for environmental hazard; (2) Revised concentration(s) of concern (COCs) for chronic aquatic hazards for both sediment and surface water compartments; (3) Calculated risk estimates from soils to include soil concentrations from the BST analysis.

4.1 Environmental Exposures

Environmental Exposures (Section 4.1):
Key Points

EPA evaluated the reasonably available information for environmental exposures of TCEP to aquatic and terrestrial species. The key points of the environmental exposure assessment are summarized below:

- EPA expects the main environmental exposure pathways for TCEP to be surface water, sediment, and soil. The ambient air exposure pathway was also assessed for its contribution via deposition to these media.
- TCEP exposure to aquatic species through surface water and sediment were modeled to estimate concentrations near industrial and commercial uses. These results were compared to measured concentrations of TCEP from databases (*i.e.,* WQP) or published literature from a variety of locations.
  - Modeled data estimate surface water concentrations in the low thousands of ppb (Table 4-11) and pore water concentrations low hundreds of ppb (Table 4-12) near industrial and commercial uses.
  - Monitoring data show TCEP surface water concentrations in the United States generally decreasing over the last two decades.
  - While EPA does not expect TCEP to bioaccumulate in higher trophic levels in the food web, biomonitoring from the published literature show TCEP in the tissue of several aquatic species including fish in the Great Lakes and harbor seals in San Francisco Bay.
  - EPA also estimated fish tissue concentrations by COU using the modeled water releases from industrial and commercial uses.
- TCEP exposure to terrestrial species through soil, air, and surface water was also assessed using modeling and monitoring data.
  - TCEP exposure to terrestrial organisms occurs primarily through diet via the soil pathway, with deposition from air to soil being a source. Exposure through diet was assessed through a trophic transfer analysis, which estimated the transfer of TCEP from soil through the terrestrial food web using representative species.
  - TCEP exposure to terrestrial organisms from surface water ingestion is typically ephemeral. Therefore, the trophic transfer analysis for terrestrial organisms assumed TCEP exposure concentrations for wildlife water intake are equal to TCEP soil concentrations for each corresponding exposure scenario.
  - Direct exposure of TCEP to terrestrial receptors via air was not assessed quantitatively because dietary exposure was determined to be the driver of exposure to wildlife. The contribution of TCEP exposure from inhalation relative to the ingestion exposure route is not expected to drive risk because of dilution associated environmental conditions.

4.1.1 Approach and Methodology

Soil and surface water are the major environmental compartments for TCEP (see Section 2.2.2). The environmental exposure assessment focuses on TCEP concentrations in surface water, sediment, and soil

as these are the media used to determine risks to aquatic and terrestrial organisms (see Section 4.3). Ambient air was also assessed for its contribution via deposition to these media.

Monitoring information for aquatic and terrestrial species are presented in Sections 4.1.2 and 4.1.3 below. Reported monitoring information on environmental media (*e.g.,* surface water, sediment, air) are presented in Section 3.3. When available, measured TCEP concentrations from databases (*i.e.,* WQP) or published literature were used as comparative exposure concentrations for risk quotient (RQ) calculations and are presented in Section 4.3.

EPA utilized various models to assess the environmental concentrations resulting from the industrial and commercial release estimates (see Section 3.3). These models are E-FAST 2014, VVWM-PSC, IIOAC, and AERMOD. Additional information on these models is available in Section 3.3. TCEP surface water concentrations (ppb) were modeled by E-FAST 2014 and VVWM-PSC. TCEP pore water and benthic concentrations were modeled using VVWM-PSC as described in Section 3.3.2.9. TCEP concentrations in soil, surface water, and sediment via air deposition at the community level (1,000 m from the source) were modeled as described in Sections 3.3.3.2, 3.3.2.6, and 3.3.2.10, respectively. Reported and modeled surface water and sediment concentrations were used to assess TCEP exposures to aquatic species.

Measured and modeled soil concentrations were utilized to assess risk to terrestrial species via trophic transfer (see Section 4.1.4). Specifically, trophic transfer of TCEP and potential risk to terrestrial animals was based on modeled soil data from AERMOD and concentrations reported within Mihajlovic and Fries (2012). Potential risk to aquatic dependent wildlife utilized surface water concentrations modeled via VVWM-PSC for each COU in combination TCEP fish concentrations calculated using the whole body BCF reported within (Arukwe et al., 2018). Exposure factors for terrestrial organisms used within the trophic transfer analyses are presented in Section 4.1.4. Application of exposure factors and hazard values for organisms at different trophic levels is detailed within Section 4.3 and utilized equations as described in the *U.S. EPA Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a).

For more information on TCEP monitoring data in aquatic and terrestrial species, please see the following supplemental documents:

- *Environmental Monitoring Concentrations Reported by Media Type* (U.S. EPA, 2024i);
- *Environmental Monitoring and Biomonitoring Concentrations Summary Table* (U.S. EPA, 2024h);
- *Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure.* (U.S. EPA, 2024x); and
- *Data Extraction Information for General Population, Consumer, and Environmental Exposure* (U.S. EPA, 2024r).

4.1.2 Exposures to Aquatic Species

4.1.2.1 Measured Concentrations in Aquatic Species

A graphical survey of TCEP concentrations in fishes within reasonably available published literature (seven studies) is presented in Figure 4-1. Guo et al. (2017b) measured concentrations of TCEP in fish samples in the Great Lakes Basin using the Great Lakes Fish Monitoring and Surveillance Program (GLFMSP) sampling protocol. TCEP was found in more than 50 percent of the fish samples at a geometric mean of 13.3 ng/g lipid, including lake trout (*Salvelinus namaycush*) or walleye (*Sander*

vitreus). The lipid-based concentrations of TCEP in Lake Erie fish were significantly higher than those of the other four Great Lakes. These concentrations are in line with lipid-based concentrations from Sundkvist et al. (2010), who measured TCEP in mussels (*Mytilus edulis*), herring (*Clupeidae*), eelpout (*Zoarces viviparus*), salmon (*Salmo salar*), and perch *(Perca fluviatilis*) in Swedish lakes and coastal areas.

TCEP has been recorded in the blubber of harbor seal (*Phoca vitulina*) within the San Francisco Bay at a median concentration of 3.4 ng/g (Sutton et al., 2019). Sutton et al. (2019) indicated that blubber might not be a good indicator of exposure to hydrophilic phosphate-based flame retardants due to degradation and metabolism. Two European studies present lipid concentrations of TCEP in aquatic mammals at similar levels to the lipid concentrations in fish shown above (Sala et al., 2019; Hallanger et al., 2015).

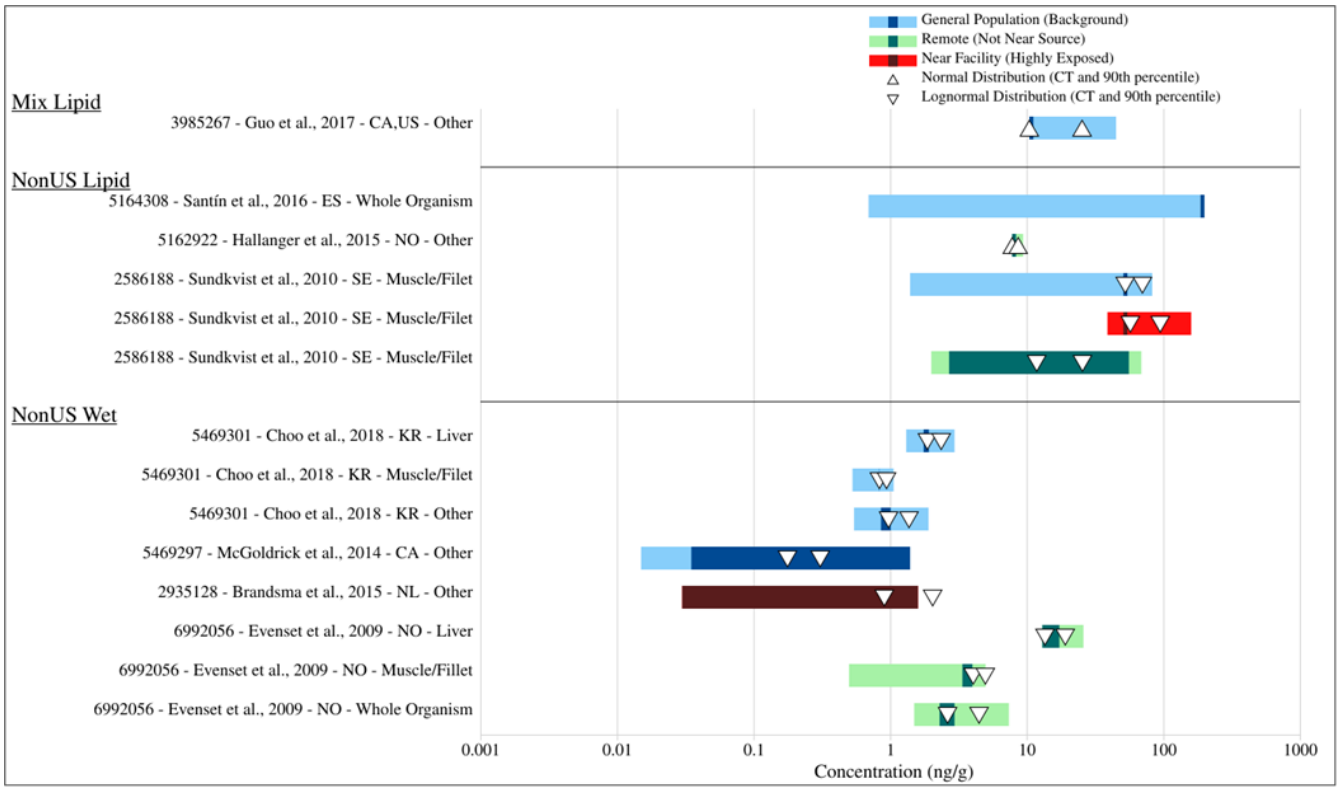


Figure 4-1. Measured Concentrations of TCEP (ng/g) in Aquatic Species – Fish from 2003 to 2016

4.1.2.2 Calculated Concentrations in Aquatic Species

In addition to considering monitoring data from published literature, EPA modeled concentrations in fish for each industrial and commercial release scenario (Table 4-1). Concentrations of TCEP in fish were calculated by multiplying the VVWM-PSC modeled surface water concentrations for each industrial and commercial releases scenario by the BCF of 0.34 L/kg (Arukwe et al., 2018) (Table 2-2). These conservative whole fish TCEP concentrations were utilized within the screening-level assessment for trophic transfer as described in Section 4.1.4.

Table 4-1. TCEP Fish Concentrations Calculated from VVWM-PSC Modeled Industrial and Commercial TCEP Releases

Scenario Name	Production Volume (lb/year)	Release Distribution[a]	Surface Water Concentration (µg/L)	Fish Concentration (ng/g)
Import and repackaging	2,500	High-End	2,350	799
Incorporation into paints and coatings – 1-part coatings	2,500	High-End	10,000	3,400
Incorporation into paints and coatings – 2-part reactive coatings	2,500	High-End	8,150	2,771
Use of paints and coatings – spray application	2,500	High-End	5,500	1,870
Formulation of TCEP containing reactive resin	2,500	High-End	9,040	3,073
Laboratory chemicals	2,500	High-End	96	32

[a] Production volume of 2,500 lb TCEP/year uses high-end estimates (95th percentile for all COUs except the laboratory chemicals COU that uses the 1st percentile).

These calculated whole fish results are one to three orders of magnitude higher than the reported fish concentrations in Guo et al. (2017b), who reported a geometric mean of 35.6 ng/g lipid in Lake Erie. Guo et al. (2017b) also reported a geometric mean concentration of TCEP in Great Lakes water of 4.64×10^{-4} µg/L via Venier et al. (2014), while Arukwe et al. (2018) used a water concentration of 7.75×10^{2} µg/L to derive the BCF within laboratory-controlled experiments. The current TCEP surface water concentrations modeled via VVWM-PSC are one to two orders of magnitude greater that values reported in Arukwe et al. (2018); however, it is important to consider that modeled concentration are intended to represent COU-based source release concentrations.

4.1.2.3 Modeled Concentrations in the Aquatic Environment

E-FAST 2014 was used to estimate total TCEP surface water concentration within lotic (*i.e.*, flowing) systems and represents TCEP concentration within the water column. The days of exceedance modeled in E-FAST 2014 are not necessarily consecutive and could occur throughout a year at different times. Days of exceedance is calculated as the probability of exceedance multiplied by the total modeled days of release. While both E-FAST 2014 and VVWM-PSC consider dilution and variability in flow, the VVWM-PSC model can estimate a time-varying surface water concentration, partitioning to suspended and settled sediment, and degradation within compartments of the water column. VVWM-PSC considers model inputs of physical and chemical properties of TCEP (*i.e.*, K_{OW}, K_{OC}, water column half-life, photolysis half-life, hydrolysis half-life, and benthic half-life), allowing EPA to model predicted pore water and sediment concentrations.

The VVVM-PSC model utilized relatively low stream orders (*i.e.*, depth of 2 m) as a conservative approach for modeling stream reach. Results within PSC are reported as the maximum concentration value of the investigated chemical over the specified averaging periods (*e.g.*, 1-day, 3-day, etc.) as well as a time-series graph of surface water and benthic pore water concentrations (U.S. EPA, 2019e). TCEP surface water concentrations (ppb) were modeled by E-FAST 2014 and VVWM-PSC and are presented in Table 4-11 for each COU at a production volume of 2,500 lb per year. TCEP pore water concentration modeled by VVWM-PSC are presented within Table 4-11 and Table 4-12, respectively.

EPA used IIOAC and AERMOD to estimate air deposition from facility releases and calculate a resulting pond water concentration near a hypothetical facility. Pond water concentrations from air deposition were estimated for the COUs with air releases (Table 4-9). AERMOD results indicate air deposition to water are not drivers of risk and have significantly reduced TCEP concentrations when compared to TCEP when modeled within the water column, pore water, and sediment modeling via E-FAST 2014 and VVWM-PSC. For example, the highest estimated 95th percentile pond water concentration from annual deposition from air to water, across all exposure scenarios, was 8.1 µg/L for the Commercial use of paints and coatings scenario at an annual production volume of 2,500 lb. This highest modeled concentration (8.1 µg/L) within a pond at 1,000 m from a point source was approximately 12 times lower than the lowest surface water concentration modeled using VVWM-PSC (96 µg/L as a maximum 1-day average concentration for the laboratory chemicals scenario at an annual production volume of 2,500 lb). Although the IIOAC and AERMOD were applied to a generic farm pond setting to calculate concentrations of TCEP in pond surface water and pond sediment, these models do not account for media exchange of the chemical of interest as VVWM-PSC does.

4.1.3 Exposures to Terrestrial Species

4.1.3.1 Measured Concentrations in Terrestrial Species

Two studies (see Figure 4-2) have reported concentrations of TCEP and a TCEP metabolite bis(2-chloroethyl) phosphate (BCEP) in bird eggs (Guo et al., 2018; Stubbings et al., 2018). From these two studies the mean concentration of TCEP in birds by wet weight is 5.3 ng/g with a 90th percentile value of 9.7 ng/g. BCEP was among the most abundant metabolites (0.38 to 26 ng/g ww) in bald eagle (*Haliaeetus leucocephalus*) eggs. These values are results of the Michigan Bald Eagle Biosentinel Program archive that sampled bald eagles in the Great Lakes Region from 2000 to 2012.

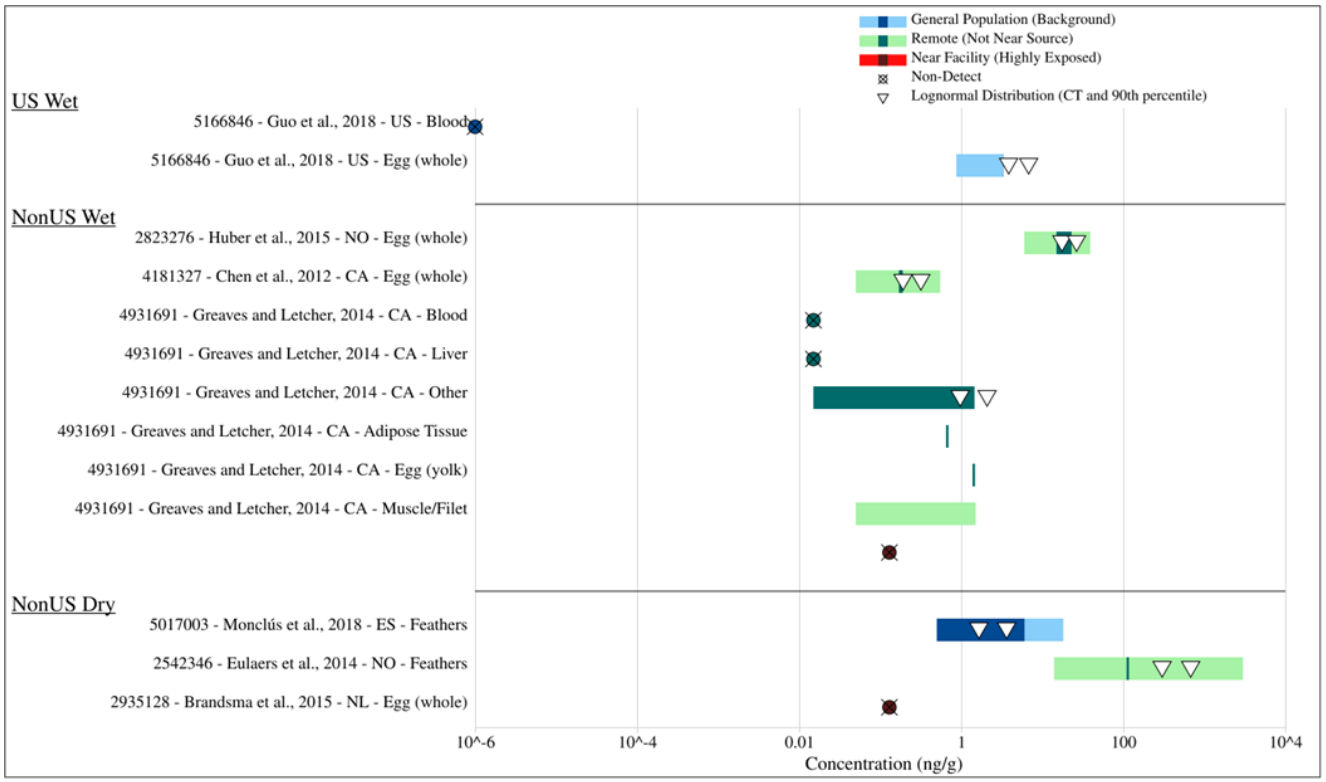


Figure 4-2. Measured Concentrations of TCEP (ng/g) in Terrestrial Species – Bird from 2000 to 2016

Aston et al. (1996) reported TCEP in pine needles (*Pinus ponderosa)* at six out of nine collection sites in the Sierra Nevada Foothills in the mid-1990s with a geometric mean TCEP concentration of 142 ng/g and a range of 10 ng/g to 1,950 ng/g (Figure 4-3). Although the source of the TCEP is unknown, the authors suspected that concentrations may have been due to aerial transport and deposition from nearby point sources such as incinerators. Samples reported within Aston et al. (1996) were collected in 1993 and 1994 with concentrations from this study representing a period with significantly higher concentrations of TCEP in production and use (see Section 1.1.1).

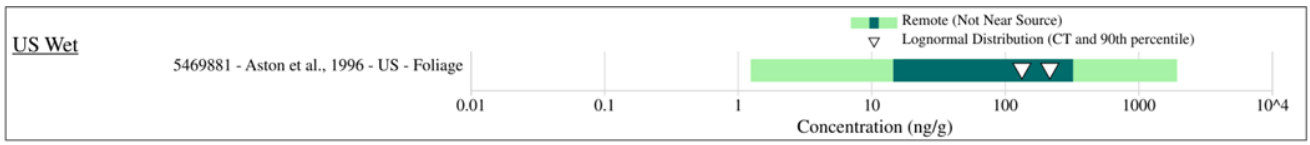


Figure 4-3. Measured Concentrations of TCEP (ng/g) in the Wet Fraction of Terrestrial Species – Plant in Remote (Not Near Source) Locations from 1993 to 1994

4.1.3.2 Modeled Concentration in the Terrestrial Environment

The contribution of exposure risk from inhalation relative to the ingestion exposure route is not expected to drive risk because of dilution associated environmental conditions (U.S. EPA, 2003a, b). In addition, TCEP is not persistent in air due to its short half-life in the atmosphere ($t_{1/2}$ = 5.8 hours) and because particle-bound TCEP is primarily removed from the atmosphere by wet or dry deposition (U.S. EPA, 2017a). Air deposition to soil modeling is described in Section 3.3.3.2. EPA determined the primary exposure pathway for terrestrial organisms is through soil via dietary uptake via trophic transfer. As described in Section 3.3.3.2, IIOAC and subsequently AERMOD were used to assess the estimated release of TCEP via air deposition from specific exposure scenarios to soil. Estimated concentrations of TCEP that could be in soil via air deposition at the community level (1,000 m from the source) exposure scenarios have been calculated and are presented in Appendix H.2.

4.1.4 Trophic Transfer Exposure

Trophic transfer is the process by which chemical contaminants can be taken up by organisms through dietary and media exposures and transferred from one trophic level to another. EPA has assessed the available studies collected in accordance with the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) relating to the biomonitoring of TCEP.

TCEP is released to the environment by various exposure pathways (see Figure 2-1). The exposure pathway for terrestrial organisms is through soil; deposition of TCEP from air to soil is the primary exposure pathway. A secondary source of TCEP contamination in soil is from the application of biosolids. However, the concentration of TCEP in soil from biosolids is two orders of magnitude less than the TCEP soil concentration from air deposition (see Section 3.3). Therefore, biosolid application is not expected to drive risk within the terrestrial environment. The exposure pathway for water includes runoff from soil (*e.g.*, after a rain event), deposition from air, and direct releases from water treatment plants. Sediment TCEP concentrations determined by VVMW-PSC modeling range from 2.6- to 108.8-fold greater than surface water concentration across all COUs (see Section 3.3.2.9). Indicating that sediment acts as a sink for TCEP and a source of elevated exposure to TCEP through the dietary exposure pathway for higher trophic levels in the water column that feed on benthic organisms. Trophic magnification is not expected in the water column or terrestrial environments but may occur where TCEP concentrations are high (*i.e.*, in the benthic zone) (Table 2-2).

Representative avian and mammal species are chosen to connect the TCEP transport exposure pathway via terrestrial trophic transfer from earthworm (*Eisenia fetida*) uptake of TCEP from contaminated soil

through invertivore avian (American woodcock [*Scolopax minor*]) and mammal (short-tailed shrew [*Blarina brevicauda*]) species, to the American kestrel (*Falco sparverius*) that feeds on invertebrates, avian, and small terrestrial vertebrates.

American woodcocks primarily feed on invertebrates with a preference for earthworms. When earthworms are not available, other soil invertebrates and a small proportion of vegetation may be consumed. Depending on the location and season, earthworms may comprise 58 to 99 percent of American woodcock diet (U.S. EPA, 1993b). Short-tailed shrews primarily feed on invertebrates with earthworms comprising approximately 31 percent (stomach volume) to 42 percent (frequency of occurrence) of their diet. American kestrels have a varied diet that includes invertebrates and vertebrates (mammal, avian, and reptile). The proportion of prey type will vary by habitat and prey availability. For trophic transfer analysis, the American kestrel diet comprised equal proportions of the three representative prey species (*i.e.*, one-third earthworm, one-third American woodcock, and one-third short-tailed shrew), which approximates the dietary composition of the American kestrel winter diet reported in Meyer and Balgooyen (1987). The calculations for assessing TCEP exposure from soil uptake by earthworms and the transfer of TCEP through diet to higher trophic levels are presented in Section 4.3.1.10. Because surface water sources for wildlife water ingestion are typically ephemeral, the trophic transfer analysis for terrestrial organisms assumed TCEP exposure concentration for wildlife water intake are equal to soil concentrations for each corresponding exposure scenario (U.S. EPA, 2003a, b).

The representative semi-aquatic terrestrial species is the American mink (*Mustela vison*), whose diet is highly variable depending on their habitat. In a riparian habitat, American mink derive 74 to 92 percent of their diet from aquatic organisms, which includes fish, crustaceans, birds, mammals, and vegetation (Alexander, 1977). Similar to soil concentrations used for terrestrial organisms, the highest modeled surface water TCEP concentrations with a production volume of 25,000 lb/year was used as a surrogate for the TCEP concentration found in the American mink's diet in the form of both water intake and a diet of fish. For trophic transfer, fish concentrations shown in Table 4-1 are used in conjunction with trophic transfer calculations in Section 4.3.1.1.

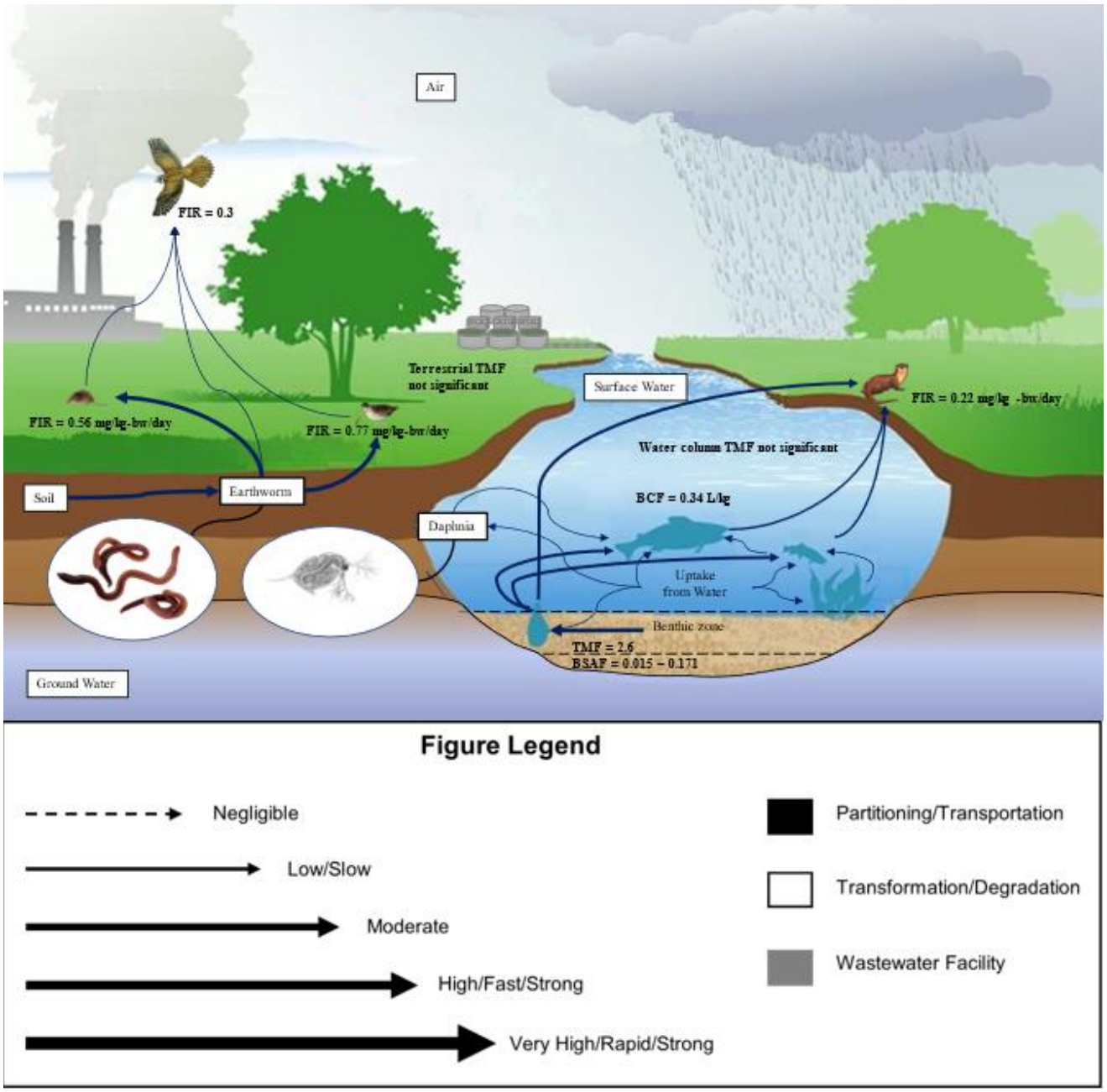


Figure 4-4. Trophic Transfer of TCEP in Aquatic and Terrestrial Ecosystems[a]

[a] The diagram demonstrates uptake from media to biota and trophic transfer through the food web (blue arrows). The width of the arrows shows relative chemical transport between biota or media. Within the aquatic environment, the benthic zone is bounded by dashed black lines from the bottom of the water column to sediment surface and subsurface layers. The depth that the benthic environment extends into subsurface sediment is site specific. The conceptual model illustrates BCFs, BSAFs, and TMFs for aquatic organisms as shown in Appendix F.2.6. Food intake rates (FIRs) are shown for terrestrial vertebrates.

4.1.5 Weight of Scientific Evidence Conclusions for Environmental Exposures

4.1.5.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the Environmental Exposure Assessment

Concentrations of TCEP in environmental and biological media are expected to vary. Release from industrial facilities, indoor sources, and long-range transport may all contribute to concentrations of

TCEP in the environment. Determining the source apportionment of TCEP from each is complex. Proximity to facilities and other sources is likely to lead to elevated concentrations compared to locations that are more remote. No manufacturing or processing facility locations were identified for releases to TCEP. The inability to locate releases in proximity to facility locations contributes to a layer of uncertainty when selecting model input parameters that are typically informed by location (*e.g.,* meteorological data, land cover parameters for air modeling, flow data for water modeling).

Limited monitoring data are available for aquatic and terrestrial species in the United States. In addition, monitoring data collected in previous years when production volume and associated releases of TCEP into the environment are expected to have been higher than they are currently and expected to be in the future. When considering older monitoring data and monitoring data from international sources, there are uncertainties associated with using these data because it is unknown whether those sampling sites are representative of current sites within the United States. Recent and future estimated levels of TCEP in the area may be lower than past levels due to reported reductions in releases over time. The predicted concentrations may be lower than concentrations that consider more years of releases or releases associated with higher production volumes.

In modeling environmental concentrations of TCEP, EPA acknowledges the conservative nature of the E-FAST 2014 model and the additional refinement provided by the VVWM-PSC model. Water dilution models can be used to determine the concentration of a chemical in the surface water after a source releases the chemical into a water body. Because the E-FAST 2014 model default values encompass either a combination of upper percentile and mean exposure parametric values, or all upper percentile parametric values, the resulting model predictions represent high-end exposures estimates. A simple dilution model such as E-FAST 2014 provides exposure estimates that are derived from a simple mass balance approach and does not account for partitioning between compartments within a surface water body or degradation over time in different media, parameters which are relevant to TCEP. For these reasons, EPA utilized a two-tier approach by complementing the E-FAST 2014 modeling with more refined estimates from the PSC model to describe further environmental exposures.

When modeling using E-FAST 2014, EPA assumed that primary treatment removal at POTWs occurred with 0 percent removal efficiency. EPA recognizes that this is a conservative assumption that results in no removal of TCEP prior to release to surface water. Section 2.2.1 and Appendix F.2.5.2 discusses the recalcitrance of TCEP to wastewater treatment systems. This assumption reflects both the uncertainty of the type of wastewater treatment that may be in use at a direct discharging facility and the TCEP removal efficiency in that treatment.

EPA used a combination of chemical-specific parameters and generic default parameters when estimating surface water, sediment, soil, and fish-tissue concentrations. For estimated soil concentrations from air deposition, specifically, EPA recognizes that different default parameters for gaseous vs. particle partitioning, may result in concentrations of a higher magnitude. However, EPA used central tendency, high production volume, and high-end, central tendency production volume values to characterize the variability within and across scenarios. To estimate soil concentrations, EPA also used central tendency and high-end meteorological inputs.

Comparison of model outputs with monitored values offers one way to ground truth the combination of model inputs and outputs used. EPA compared monitoring and modeled surface water, sediment, soil, and fish-tissue concentration estimates. Estimates of fish-tissue concentrations are further discussed in Section 5.1.3.4.2. In summary, EPA compared monitored and modeled fish tissue concentrations and found modeled fish concentrations were two to three orders of magnitude higher than those reported for

whole fish within published literature (see Section 4.1.2.2). The conservative approach for calculated fish tissue concentrations presented in Section 4.1.2.2 was utilized for trophic transfer analysis to semi-aquatic mammals (see Section 4.3.1.10). In comparison to measured values reported within published literature, these calculated values should be viewed as organisms with direct proximity to source of TCEP release as calculated using VVWM-PSC.

EPA conducted modeling of TCEP concentrations in surface water, pore water, and sediment based on the assumption that releases entered lotic (flowing) aquatic systems. Although EPA did not consider the potential impact of persistence and longer-term sinks in lake and estuary environments, localized deposition of TCEP within 1,000 m from hypothetical release sites from air to soil, water, and sediment were modeled for each applicable COU via IIOAC and AERMOD.

4.2 Environmental Hazards

Environmental Hazards (Section 4.2):
Key Points

EPA evaluated the reasonably available information for environmental hazard endpoints associated with TCEP exposure. The key points of the environmental hazard assessment are summarized below:

- Aquatic species hazard:
  - Aquatic hazard data were available for TCEP for six species of fish, four invertebrate species, and five algae species.
  - To estimate hazards (mortality) from acute exposures, EPA supplemented the empirical data with hazard predictions from an EPA predictive tool, Web-based Interspecies Correlation Estimation. These data were used with the empirical fish and daphnid data to create a species sensitivity distribution (SSD) and calculate a TCEP concentration of concern (COC) for acute exposures of aquatic species (16,700 ppb) representing the lower 95th percentile of an HC05 (Table 4-6).
  - EPA applied Web-based Interspecies Correlation Estimation (Web-ICE) to empirical hazard data on algae to create an SSD and calculated a COC for aquatic plants of 66,000 ppb (Table 4-6).
  - EPA also calculated a COC for chronic exposures (survival of yellow catfish) to aquatic species (2.8 ppb) using empirical fish data (Table 4-6).
- Terrestrial species hazard:
  - Terrestrial hazard data for TCEP were available for soil invertebrates, mammals, and avian species.
  - Based on empirical toxicity data for nematodes and earthworms, the chronic hazard threshold for terrestrial invertebrate is 612 mg/kg soil (Table 4-7).
  - Empirical toxicity data for mice and rats were used to estimate a chronic toxicity reference value (TRV) for terrestrial mammals of 44 mg/kg-bw/day (Table 4-7).

4.2.1 Approach and Methodology

During the scoping process, EPA reviewed potential environmental hazards associated with TCEP exposure and identified 14 sources of environmental hazard data shown in Figure 2-10 of *Final Scope of the Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) CASRN 115-96-8* (U.S. EPA, 2020b).

EPA completed the review of environmental hazard data/information sources during risk evaluation using the data quality evaluation metrics and the data quality criteria described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Studies were assigned an overall quality determination of high, medium, low, or uninformative.

EPA assigned an overall quality determination of high or medium to 23 acceptable aquatic toxicity and 17 acceptable terrestrial toxicity studies. For the aquatic studies on vertebrates and invertebrates, seven species had appropriate endpoint concentrations (LC50) for assessing the effects from acute exposures of TCEP. Five empirical hazard values were available for aquatic plants, represented by green algae and diatom species. The modeling approach, Web-ICE, Version 3.3, can both predict toxicity values for environmental species that are absent from a dataset and provide a more robust dataset to estimate

toxicity thresholds. EPA used Web-ICE to supplement empirical acute hazard data for aquatic organisms. Details outlining the method are included in Appendix G. For terrestrial species, all mammal studies were from mice and rats used as human health model organisms. These studies were used to calculate a toxicity reference value (TRV) for mammals, which is expressed as doses in units of mg/kg-bw/day. Although the TRV for TCEP is derived from laboratory mice and rat studies, because body weight is normalized, the TRV can be used with ecologically relevant wildlife species to evaluate chronic dietary exposure to TCEP. Representative wildlife species chronic hazard thresholds are evaluated in the trophic transfer assessments using the TRV.

4.2.2 Aquatic Species Hazard

Toxicity to Aquatic Organisms

EPA assigned an overall quality determination of high or medium to 23 acceptable aquatic toxicity studies. These studies contained relevant aquatic toxicity data for: brine shrimp (*Artemia* sp.), diatoms (*Phaeodactylum tricornutum*) and (*Skeletonema costatum*), green alga (*Dunaliella salina*), (*Platymonas subcordiformis*), and (*Raphidocelis subcapitata*), Japanese seabass (*Lateolabrax maculatus*), Japanese medaka (*Oryzias latipes*), Manila clam (*Ruditapes philippinarum*), mrigal carp (*Cirrhinus mrigala*), mysid shrimp (*Neomysis awatschensis*), rainbow trout (*Oncorhynchus mykiss*), water flea *(Daphnia magna)*, yellow catfish (*Pelteobagrus fulvidraco*), and zebrafish (*Danio rerio*). EPA identified 12 aquatic toxicity studies, displayed in Table 4-2, Table 4-3, and Table 4-4 as the most relevant for quantitative assessment. The remaining studies were represented by results at a sub-organ or mechanistic level, considered to be separated from direct population level effects or did not demonstrate effect(s) at the test concentrations employed within their study concentrations gradients. The Web-ICE application was used to predict LC50 toxicity values for 46 aquatic species (22 fish, 1 amphibian, 9 aquatic invertebrates, and 14 benthic invertebrates) using empirical acute toxicity data from rainbow trout, zebrafish, and *Daphnia magna* (Raimondo and Barron, 2010). Invertebrate and vertebrate species with empirical hazard values (n = 7) and predicted species (n = 46) toxicity data were subsequently used to calculate the distribution of species sensitivity to acute TCEP exposure (Appendix G). The Web-ICE application was used to predict EC50 values for three algal species using empirical hazard values from one freshwater green algae species (*Raphidocelis subcapitata*) and two marine diatoms (*Phaeodactylum tricornutum* and *Skeletonema costatum*). The five empirical and three predicted species hazard values were subsequently used to calculate the distribution of species sensitivity for aquatic plants exposed to TCEP (Appendix G).

Aquatic Vertebrates

Relevant acute toxicity studies for fish that included LC50 data were assigned an overall quality determination of high for three 96-hour static condition fish toxicity studies (Zhang et al., 2024; Alzualde et al., 2018; Life Sciences Research Ltd, 1990a) evaluating the median LC50 from exposure to TCEP. The acute 96-hour LC50 values for fish were 249 mg/L for rainbow trout (Life Sciences Research Ltd, 1990a), 279 mg/L for zebrafish embryo (Alzualde et al., 2018), and 46 mg/L for juvenile Japanese seabass (Zhang et al., 2024). The LC50 study for rainbow trout did not meet the assumptions of the Probit test. Therefore, a non-linear interpolation was used to approximate the LC50 value. The zebrafish embryo study by Alzualde et al. (2018) used a non-linear regression test (sigmoidal dose-response curve) to calculate the LC50. Zhang et al. (2024) used log probability curves estimate 96-hourr LC50 values and 95 percent confidence intervals (CIs) for Japanese seabass.

The ChV is the geometric mean of the lowest-observed-effect-concentration (LOEC) and no-observed-effect-concentration (NOEC). The overall quality determination for relevant studies with acute exposure duration ChV values were high for two 96-hour studies for rainbow trout and zebrafish (Alzualde et al., 2018; Life Sciences Research Ltd, 1990a). The 96-hour rainbow trout had a ChV of 70.7 mg/L for

mortality (Life Sciences Research Ltd, 1990a) while the 96-hour zebrafish embryo had a ChV of 139.7 mg/L for development and growth (Alzualde et al., 2018).

Six studies on fishes were available and represented TCEP exposure with subchronic and chronic durations ranging from 14 to 120 days. All six studies were assigned an overall quality determination of high for the apical assessment endpoints of regulatory interest (*i.e.*, impaired growth, survival, or reproduction). The shortest exposure duration was a study on Japanese Medaka encompassing 14-day TCEP exposures across approximately 9 days of embryo development followed by approximately 5 days of larval development (Sun et al., 2016). Daily TCEP renewal of 90 percent of the treatment and solvent control waters was conducted every 24 hours throughout the exposure period with no analytical verification of TCEP concentrations. The duration of this experimental exposure covering all of embryogenesis and 5 days of larval development representing sensitive lifestages for fishes. Reporting nominal concentrations, the 14-day exposure to TCEP resulted in a ChV of 0.559 mg/L for development and growth with significant differences in length compared to control groups (Sun et al., 2016). Mean hatchability, reported as a percent, was not significantly different among TCEP treatments and decreased with increased TCEP treatment concentrations of 1.25 mg/L and 6.25 mg/L resulting in hatchability of 92.6 ± 7.4 and 90.6 ± 5.8, respectively, compared to the control percent hatchability of 96.3 ± 3.7. Similarly, increasing TCEP treatment concentrations of 1.25 mg/L and 6.25 mg/L resulting in non-significant increases in percent gross abnormality rates of 5.8 ± 2.4 and 5.9 ± 2.4, respectively, compared to the control gross abnormality rate of 3.2 ± 1.0. Authors reported TCEP-exposed larvae were observed to have increasing trends in swimming speeds for both dark and light phases, however, these quantified movements were not significantly different from control.

Juvenile mrigal carp were exposed to TCEP for 21 days at nominal concentrations of 0.04, 0.2, and 1 mg/L with daily TCEP renewal of 75 percent of the treatment and control waters (Sutha et al., 2020). Structural abnormalities based on histology were observed within the lowest TCEP treatment concentration (0.04 mg/L) for gill, liver, and kidney tissues with the severity of abnormalities increasing with increasing TCEP treatment concentrations. Authors described the greatest incidences of abnormalities occurring within gill and liver tissues. For example, gill tissue abnormalities included but were not limited to: epithelial lifting, hyperplasia, and degeneration of cells in primary lamellae; while abnormalities observed within liver tissue included but were not limited to: necrosis, pyknotic nuclei, and increased sinusoids vessels. Sutha et al. (2020) also observed significant differences in plasma thyroid hormones and antioxidant enzyme activities at the lowest treatment concentration (0.04 mg/L). Although authors reported non-quantified behavioral changes associated with TCEP exposure compared to control treatments the study did not record any changes in growth or survival from a 21-day TCEP exposure.

Two studies were conducted with 30-day TCEP exposures to juvenile yellow catfish with daily 66 percent water replacement and analytically verified TCEP concentrations (Hu et al., 2022; Zhao et al., 2021). Water samples were collected twice a week for analytical verification of TCEP using Liquid chromatography-mass spectrometry. Hu et al. (2022) reported nominal TCEP treatment concentrations of 0, 0.001, 0.01, and 0.1 mg/L with analytically verified concentrations of 0, 0.00087 ± 0.00012, 0.00924 ± 0.00131, and 0.08915 ± 0.00463 mg/L, respectively. Hazard values are represented by analytically verified TCEP concentrations as reported by the authors and rounded to represent 0.0009, 0.009, and 0.09 mg/L. Survival was significantly decreased at a TCEP treatment concentration of 0.09 mg/L compared to the solvent (0.01% DMSO) control with a NOEC of 0.009 mg/L resulting in a ChV of 0.028 mg/L. Final weight and length were significantly reduced compared to control groups at a TCEP concentration of 0.009 mg/L. Final length and weight at the lowest TCEP treatment concentration of 0.0009 mg/L were not significantly different from the control treatment. Specific growth rate was

significantly higher at 0.0009 mg/L and was significantly reduced compared to control at both the 0.009 mg/L and 0.09 mg/L treatment concentrations. Gill histology indicated shortened secondary gill lamellae and epithelial hyperplasia at the lowest nominal TCEP concentration of 0.0009 mg/L, however, authors did not perform severity scoring or report the proportions of individuals affected Zhao et al. (2021). Severity scoring and the precent of individuals affected was documented with histological analysis of the liver conducted by Hu et al. (2022). Liver tissues displayed no histological changes for the control and 0.0009 mg/L TCEP treatment group, while the 0.009 mg/L treatment group displayed cytoplasmic vacuolization (10–50% affected), cellular peripheral nucleus (<10% affected), and karyolysis (<10% affected). Authors reported observing increase incidences of karyolysis (10–50% affected), cellular degeneration (10–50% affected), and focal necrosis (<10% affected) within the liver at the 0.09 mg/L TCEP treatment.

Two studies were conducted on zebrafish with 120-day TCEP exposures beginning at the embryo stage and measuring outcomes within sexually mature adults (Hu et al., 2023) and outcomes from resulting embryos and larvae (Wang et al., 2022). The solvent DMSO was used at a final concentration of 0.0001 percent within both solvent control and TCEP treatment groups. TCEP and solvent control treatments had daily 50 percent water replacement and water samples were collected before and after renewal at day 119 of the exposure period for analytical verification using liquid chromatography-mass spectrometry. Authors reported nominal TCEP treatment concentrations of 0, 0.0008, 0.004, 0.020, 0.100 mg/L and analytically verified concentrations after renewal of 0, 0.00072 ± 0.00014, 0.00372 ± 0.00031, 0.01868 ± 0.00155, and 0.09377 ± 0.00629 mg/L, respectively. Hazard values are represented by analytically verified TCEP concentrations as reported by the authors and rounded to represent 0.0007, 0.004, 0.019, and 0.09 mg/L. The concentration of TCEP within newly fertilized embryos resulting from parental exposures was reported for each concentration within Wang et al. (2022). Body mass and length for adults exposed to TCEP for 120 days were significantly reduced compared to controls at TCEP concentrations of 0.004 mg/L and greater (Hu et al., 2023). Adult hepatosomatic index, percent weight of the liver relative to body weight, was significantly reduced in all treatment concentrations compared to control treatment (Hu et al., 2023). The effects of adult 120-day TCEP exposure on newly fertilized embryos and larvae reared in TCEP free water were reported within Wang et al. (2022), with apical endpoints recorded at 120 hours post fertilization (hpf) related to survival, hatch rate, heart rate, body length, and incidence of malformations. Heart rate was significantly increased compared to control groups at an adult 120-day TCEP exposure concentration of 0.09 mg/L. Survival at 120 hpf was significantly different from control for adult 120-day TCEP exposure concentrations at and above 0.004 mg/L. Categories of malformations observed within 120 hpf larvae included pericardial edema, yolk sac edema, tail deformation, and spinal curvature. The incidence of malformations at 120 hpf in larvae were significantly greater than control for all adult 120-day TCEP exposure concentrations.

Table 4-2. Aquatic Vertebrate Environmental Hazard Studies for TCEP

Duration	Test Organism (Species)	Endpoint(s)	Hazard Values (mg/L) [a]	Geometric Mean[b] (mg/L)	Effect	Citation (Data Evaluation Rating)
Acute	Fish: rainbow trout (*Oncorhynchus mykiss*)	96-hour LC50 96-hour NOEC/LOEC	249 50/100	70.7	Mortality	(Life Sciences Research Ltd, 1990a) (High)
	Fish: Japanese Seabass (*Lateolabrax maculatus*)	96-hour LC50	46	–	Mortality	(Zhang et al., 2024) (High)
	Fish: zebrafish embryo (*Danio rerio*)	96-hour LC50	279	–	Mortality	(Alzualde et al., 2018) (High)
		96-hour EC50 96-hour NOEC/LOEC	118 114/171	139.7	Developmental/ Growth	
Subchronic /Chronic	Fish: Japanese medaka (*Oryzias latipes)*	14-day NOEC/LOEC	0.25/1.25	0.559	Developmental/ Growth	(Sun et al., 2016) (High)
	Fish: mrigal carp (*Cirrhinus mrigala*)	21-day LOEC	0.04	–	Renal/Kidney	(Sutha et al., 2020) (High)
					Hepatic/Liver	
					Respiratory	
	Fish: yellow catfish (*Pelteobagrus fulvidraco*)	30-day NOEC/LOEC	0.009/0.09	0.028	Mortality	(Hu et al., 2022) (High)[c d]
			0.0009/0.009	0.0028	Developmental/ Growth	
			0.0009/0.009	0.0028	Hepatic/Liver	
	Fish: zebrafish (*Danio rerio*)	120-day NOEC/LOEC 120-day LOEC	0.0007/0.004	0.0016	Developmental/ Growth	(Hu et al., 2023) (High)[d]
			0.0007	–	Hepatic/Liver	
	Fish: zebrafish embryo/ larvae (*Danio rerio*)	120-day F0 exposure; F1 NOEC/LOEC	0.0007/0.004[e]	0.0016 [e]	Mortality	(Wang et al., 2022) (High)[d]
		LOEC	0.09[e]	–	Cardiovascular	
		LOEC	0.0007[d]	–	Reproductive/ Teratogenic	

[a] Hazard value represented by the corresponding "Endpoint" in the adjacent column (*e.g.*, LC50, NOEC, LOEC).
[b] Geometric mean of definitive values only.
[c] Hu et al. (2022) has the same data and data quality ranks for mortality and growth/development outcomes reported within Zhao et al. (2021).
[d] Hazard values are represented by analytically verified concentrations as reported by the authors.
[e] TCEP concentrations represent adult 120-day exposures with progeny (embryo/larvae) reared for 120 hours in water containing no TCEP. Maternal transfer of TCEP was confirmed from analytical chemistry performed on eggs from each treatment and control groups.

Amphibians
No amphibian studies were reasonably available to assess potential hazards from TCEP exposure. However, modeled data from Web-ICE predicted a bullfrog (*Lithobates catesbeianus*) 96-hour LC50 with a geometric mean of 264 mg/L from two surrogate species. Therefore, amphibians are accounted for within the Web-ICE and species sensitivity distribution (SSD) results.

Aquatic Invertebrates
Three studies were available on aquatic invertebrates and represented with acute exposures to different species and chronic exposures to *Daphnia magna* (Table 4-3). Two related studies conducted on *Daphnia magna* represent both acute and chronic exposures of TCEP. The 48-hour TCEP exposure resulted in an EC50 for immobilization of 171 mg/L with a LOEC for immobilization at 48-hours of 117 mg/L (Toray Research Center, 1997a). A chronic study with 21-day exposure of TCEP to *Daphnia magna* resulted in a NOEC and LOEC for the cumulative number of offspring produced per parent of 10 mg/L and 17 mg/L, respectively (Toray Research Center, 1997c). Working with marine species, Zhang et al. (2024) used log probability curves to estimate 48-hr LC50 values for brine shrimp and 96-hr LC50 values for mysid shrimp and Manila clams.

Table 4-3. Aquatic Invertebrate Environmental Hazard Studies for TCEP

Duration	Test Organism (Species)	Endpoint	Hazard Values (mg/L) [a]	Geometric Mean[b] (mg/L)	Effect	Citation (Data Evaluation Rating)
Acute	Aquatic Invertebrate: brine shrimp (*Artemia sp.*)	48-hour LC50	97	–	Mortality	(Zhang et al., 2024) (High)
	Aquatic Invertebrate: mysid shrimp (*Neomysis awatschensis*)	96-hour LC50	40	–	Mortality	
	Mollusk: Manila clam (*Ruditapes philippinarum*)	96-hour LC50	312	–	Mortality	
	Aquatic Invertebrate: *Daphnia magna*	48-hour EC50	171	–	Immobilization	(Toray Research Center, 1997a) (High)
		48-hour NOEC/LOEC	76/117	94.2	Immobilization	
Chronic	Aquatic Invertebrate: *Daphnia magna*	21-day LC50	83.1	–	Mortality	(Toray Research Center, 1997c) (High)
		21-day EC50	29.6	–	Reproduction	
		21-day NOEC/LOEC	10/17	13	Reproduction	

[a] Hazard value represented by the corresponding "Endpoint" in the adjacent column (*e.g.*, LC50, NOEC, LOEC)
[b] Geometric mean of definitive values only.

Aquatic Plants
Two studies were available and represent TCEP exposures to two species of marine diatoms, two species of marine green algae, and one species of freshwater algae (Table 4-4). One study represented a 72-hour exposure of TCEP to the freshwater green algae, *Raphidocelis subcapitata* (Toray Research Center, 1997b). Exposure of TCEP to green algae for 72 hours resulted in a NOEC and LOEC for the growth inhibition of 65 mg/L and 116 mg/L, respectively (Toray Research Center, 1997b). Zhang et al.

(2024) conducted TCEP exposures on marine microalgae and used log probability curves to estimate 96-hr EC50 values for growth on two diatom species, *Phaeodactylum tricornutum* and *Skeletonema costatum*, and two marine green algae species, *Dunaliella salina* and *Platymonas subcordiformis*.

Table 4-4 Aquatic Plant Environmental Hazard Studies for TCEP

Test Organism (Species)	Endpoint	Hazard Values (mg/L) [a]	Geometric Mean[b] (mg/L)	Effect	Citation (Data Evaluation Rating)
Diatom: *Phaeodactylum tricornutum*	96-hour EC50	76	–	Growth	(Zhang et al., 2024) (High)
Diatom: *Skeletonema costatum*	96-hour EC50	353	–	Growth	
Green Algae: *Dunaliella salina*	96-hour EC50	265	–	Growth	
Green Algae: *Platymonas subcordiformis*	96-hour EC50	395	–	Growth	
Green Algae: *Raphidocelis subcapitata*[c]	72-hour EC50	212	–	Growth	(Toray Research Center, 1997b) (High)
	72-hour NOEC/LOEC	65/116	87	Growth	

[a] Hazard value represented by the corresponding "Endpoint" in the adjacent column (*e.g.*, LC50, NOEC, LOEC)
[b] Geometric mean of definitive values only.
[c] Test species formerly known as *Selenastrum capricornutum* and *Pseudokirchneriella subcapitata*.

4.2.3 Terrestrial Species Hazard

EPA assigned an overall quality determination of high or medium to 17 acceptable terrestrial toxicity studies. These studies contained relevant terrestrial toxicity data for two Norway rat (*Rattus norvegicus*) strains (F334 and Sprague-Dawley), two mouse (*Mus musculus*) strains (CD-1 IGS and B6C3F1), one earthworm (*Eisenia fetida*), and one nematode (round worm; *Caenorhabditis elegans*). EPA identified a total of seven terrestrial toxicity studies, displayed in Table 4-5, as the most relevant for quantitative assessment.

Terrestrial Vertebrates

Five relevant chronic toxicity studies for terrestrial vertebrates that included no-observed-effect level (NOEL) and/or lowest-observed-effect level (LOEL) data were assigned an overall quality determination of high or medium with reproduction, mortality, and/or neurotoxicity (*e.g.*, lesions to hippocampus) endpoints for rodents (n = 4) and thyroid effects for the single avian toxicity study. One study with a medium overall quality determination was for the reproduction endpoints reported within Matthews et al. (1990). Mortality endpoints within the same study received an overall quality determination of high.

Similarities among mammalian studies with ecologically relevant, population-level effects were observed. Of the three studies that included mice, two studies resulted in LOEL values. Reproductive effects (NOEL = 175 mg/kg, LOEL = 700 mg/kg) due to reduced sperm count was shown in Matthews et al. (1990). An initial dose gradient for a single dose reproduction study found that the lowest test dose with mortality effects in mice was LOEL = 1,000 mg/kg (Hazleton Laboratories, 1983). Additionally, ataxia and tremors were noted shortly after dosing of the mice, which may be related to neurotoxicity.

Male rats were more sensitive (NOEL = 88 mg/kg, LOEL = 175 mg/kg) to TCEP exposure through the oral route for mortality endpoints than females (NOEL = 175 mg/kg, LOEL = 350 mg/kg) (Matthews et al., 1990). The 2-year studies for neurotoxicity (degenerative lesions of cerebrum and brain stem) and mortality endpoints showed a NOEL of 44 mg/kg and a LOEL of 88 mg/kg (NTP, 1991b). A 60-day Sprague-Dawley rat study also resulted in neurotoxicity with lesions in the hippocampus (Yang et al., 2018a). These studies indicate that neurotoxicity of the brain may be a mode of action (MOA) for TCEP exposures in rodents.

For avian species, one high-quality study was available for the American kestrel (Fernie et al., 2015). The study reported statistically significant increases in the plasma free thyroid hormones triiodothyronine (T3) and thyroxine (T4) (LOEL = 0.0025 mg/kg-bw/day) with no effects on body weight or food consumption from 21-day TCEP exposure through the diet.

Soil Invertebrates

Relevant chronic toxicity studies for soil invertebrates included two studies that were assigned an overall quality determination of high. The earthworm had a NOEL of 0.1 mg/kg soil and a LOEL of 1.0 mg/kg soil at 3, 7, and 14 days of exposure to TCEP that showed a significant dose response relationship with degradation of the digestive tract and exfoliation of the typhlosole (Yang et al., 2018b). The nematode study results show a NOEL of 500 mg/kg soil and a LOEL of 750 mg/kg soil at 3 days exposure to TCEP for reduced growth and shortened lifespan, and an LC50 of 1,381 mg/kg soil at 6 days exposure to TCEP (Xu et al., 2017).

Terrestrial Plants

No terrestrial plants studies were available to assess potential hazards from TCEP exposure.

Table 4-5. Terrestrial Organisms Environmental Hazard Studies Used for TCEP

Duration	Test Organism	Endpoint	Hazard Values (mg/kg)[a] [b]	Geometric Mean[c] (mg/kg)	Effect	Citation (Data Evaluation Rating)
Mammals						
Chronic	F344/N rats (*Rattus norvegicus*)	2-year NOEL/LOEL	44/88	62.2	Neurotoxicity/ mortality	(NTP, 1991b) (High)
		16-week NOEL/LOEL	Female:175/350 Male: 88/175	247.5 124.1	Mortality	(Matthews et al., 1990) (High)
	B6C3F1 mice (*Mus musculus*)	16-week NOEL/ LOEL	175/700	495.0	Reproduction	(Matthews et al., 1990) (Medium)
	Sprague-Dawley rat (*Rattus norvegicus*)	60-day NOEL/LOEL	50/100	70.7	Neurotoxicity	(Yang et al., 2018a) (High)
Acute	CD-1 IGS outbred mice (*Mus musculus*)	8-day LOEL	1,000	NA	Mortality	(Hazleton Laboratories, 1983) (High)
Avian						
Chronic	American kestrel (*Falco sparverius*)	14-day LOEL	0.0025	NA	Thyroid	(Fernie et al., 2015) (High)
Soil invertebrates						
Chronic	Earth worm (*Eisenia fetida*)	3, 7, 14-day, NOEC/LOEC	0.1/1.0	0.3	Gastrointestinal	(Yang et al., 2018b) (High)
Acute	Nematode (*Caenorhabditis elegans*)	3-day NOEC/LOEC 6-day LC50	500/750 1,381	612.4 NA	Growth/mortality	(Xu et al., 2017) (High)

[a] Hazard values for mammals and avian are in mg/kg-bw/day.
[b] Hazard value represented by the corresponding "Endpoint" in the adjacent column (*e.g.*, LC50, NOEC, LOEC)
[c] Geometric means of definitive values only (*i.e.*, >48 mg/kg was not used in the calculation).

4.2.4 Environmental Hazard Thresholds

EPA calculates hazard thresholds to identify potential concerns to aquatic and terrestrial species. For aquatic species, the hazard threshold is called a concentration of concern (COC), and for terrestrial species, the hazard threshold is called a hazard value or toxicity reference value (TRV). These terms (COC, TRV, and hazard value) describe how the hazard thresholds are derived and can encompass multiple taxa or ecologically relevant groups of taxa as the environmental risk characterization serves populations of organisms within a wide diversity of environments. See Appendix G for more details about how EPA weighed the scientific evidence. Hazard thresholds are then used to calculate RQs in the risk characterization step of the environmental risk evaluation. After weighing the scientific evidence, EPA selects the appropriate toxicity value from the integrated data to use as a hazard threshold for each assessment type.

For aquatic species, EPA estimates hazard by calculating a COCs for a hazard threshold. COCs can be calculated using a deterministic method by dividing a hazard value by an assessment factor (AF) according to EPA methods (U.S. EPA, 2016e, 2014b, 2012b).

Equation 4-1.

$$COC = \frac{toxicity\ value}{AF}$$

The AF approach is one-size-fits-all and does not take data availability or species sensitivity into account (U.S. EPA, 2013, 2991006). COCs can also be calculated using probabilistic methods. For example, an SSD can be used to calculate a hazardous concentration for 5 percent of species (HC05). The HC05 estimates the concentration of TCEP that is expected to be protective for 95 percent of species. This HC05 can then be used to derive a COC, and the lower bound of the 95 percent CI of the HC05 can be used to account for uncertainty instead of dividing by an AF. The application of ICE models eliminates the need for AFs by extrapolating toxicity to a diversity of species representing a wide range of aquatic taxa with surrogate species sensitivity (Awkerman et al., 2014). Aquatic hazard values within Section 4.2.2 are presented in mg/L, while the subsequent section will demonstrate the calculation of acute and chronic COC in µg/L or ppb to conform with modeled and monitored environmental media concentrations presenting within Section 4.3.

4.2.4.1 Aquatic Species COCs Using Empirical and SSD Data

For the acute COC, EPA used the 96-hour LC50 toxicity data from rainbow trout, zebrafish, and *Daphnia magna* studies as surrogate species to predict LC50 toxicity values for 46 additional aquatic organisms (22 fish, 1 amphibian, 9 aquatic invertebrates, 14 benthic invertebrate species) using the Web-ICE application (Raimondo and Barron, 2010). Species with empirical hazard values (n = 7) and predicted species (n = 46) toxicity data were subsequently used to calculate the distribution of species sensitivity to TCEP exposure through the SSD toolbox as shown in Appendix G.2.1 (Etterson, 2020). The calculated HC05 was 31.6 mg/L with a 95 percent CI of 16.7 mg/L to 57.0 mg/L (Figure_Apx G-4). The lower 95 percent CI of the HC05 was then multiplied by 1,000 to convert mg/L to µg/L (or ppb) resulting in 16,700 µg/L. The chronic COC was derived from the ChV of the 30-day LOEC/NOEC of 0.028 mg/L for yellow catfish with the application of an AF of 10. The ChV for yellow catfish represents TCEP treatment mortality compared to control treatments observed within juveniles exposed to TCEP for 30 days (Hu et al., 2022). The ChV for growth and observations of liver histopathological alterations identified from the same study from 30 day TCEP exposure is 0.0028 mg/L (Hu et al., 2022). For aquatic plants, Web-ICE and SSD were applied to five empirical and three estimated EC50 values as detailed in Appendix G.2.1. Two model fit distributions (normal and logistic) demonstrated the best fit

and both resulted in a lower 95 percent CI of the EC50 of 66 mg/L. This value was then multiplied by 1,000 to convert mg/L to µg/L (or ppb) resulting in 66,000 µg/L.

The acute COC derived from the lower 95th percent confidence interval of the HC05 for TCEP is 16,700 µg/L or ppb.

For the chronic COC, the ChV of the 30-day LOEC/NOEC of 0.028 mg/L for yellow catfish, based on mortality. Therefore, the chronic COC = 0.028 mg/L/(AF of 10) = 0.0028 mg/L × 1,000 = 2.8 µg/L or ppb.

For the aquatic plants COC derived from the lower 95th percent confidence interval of the EC05 for TCEP is 66,000 µg/L or ppb.

4.2.4.2 Terrestrial Species Hazard Values

For terrestrial species, EPA estimates hazard by using a hazard value for soil invertebrates, a deterministic approach, for calculating a TRV for mammals. The TRV is expressed as doses in units of mg/kg-bw/day. Although the TRV for TCEP is derived from laboratory mice and rat studies, body weight is normalized, therefore, the TRV can be used with ecologically relevant wildlife species to evaluate chronic dietary exposure to TCEP. Representative wildlife species chronic hazard thresholds were evaluated in the trophic transfer assessments using the TRV. The following criteria were used to select the data to calculate the TRV with NOEL and/or LOEL data (U.S. EPA, 2007a).

Step 1: At least three results and two species tested for reproduction, growth, or mortality general end points.

- The minimum dataset required to derive either a mammalian or avian TRV consists of three results (NOEL or LOEL values) for reproduction, growth, or mortality for at least two mammalian or avian species. If these minimum results are not available, then a TRV is not derived.

Step 2: Are there three or more NOELs in reproduction or growth effect groups?

- Calculation of a geometric mean requires at least three NOEL results from either the reproduction or growth effect groups.
- Because there was a single reproduction effect result and no growth effect results, then proceed to Step 3.

Step 3: If there is at least one NOEL result for the reproduction or growth effect groups:

- Then the TRV is equal to the lowest reported no-observed-adverse-effect level (NOAEL) for any effect group (reproduction, growth, or mortality), except in cases where, the NOEL is higher than the lowest bounded LOEL.
- Then the TRV is equal to the highest bounded NOEL below the lowest bounded LOEL.

For TCEP, the NOEL for reproduction is 350 mg/kg-bw/day, and the lowest mortality LOEL is 88 mg/kg-bw/day with a NOEL of 44 mg/kg-bw/day. For more details see Appendix G.2.2.

Toxicity Reference Value (TRV) for Terrestrial Toxicity

The chronic TRV for mammals is 44 mg/kg-bw/day.

For soil invertebrates, EPA estimates hazard by calculating the ChV for a hazard threshold. The ChV is the geometric mean of the NOEC and LOEC values. Although the most sensitive adverse outcome from

TCEP exposure is for earthworm gastrointestinal damage, the ecologically relevant effects for soil invertebrates are for reproduction, population, and growth. The nematode NOEC (500 mg/kg soil) and LOEC (750 mg/kg soil) for reduced growth and shortened lifespan are used to calculate the ChV. The ChV for soil invertebrates is 612.4 mg/kg soil.

4.2.5 Summary of Environmental Hazard Assessment

Overall, EPA has moderate confidence in the evidence that TCEP presents hazard potential to aquatic species (Table 4-8). For acute aquatic exposures to TCEP within vertebrates, the 96-hour LC50 toxicity values are 46, 249.0, and 279.1 mg/L for Japanese seabass, rainbow trout and zebrafish, respectively, from three high-quality studies (Zhang et al., 2024; Alzualde et al., 2018; Life Sciences Research Ltd, 1990a). Additional acute TCEP exposure within invertebrates resulted in empirical data for four species: brine shrimp (*Artemia* sp.), Mysid shrimp (*Neomysis awatschensis*), Manila clam (*Ruditapes philippinarum*), and *Daphnia magna* (Zhang et al., 2024; Toray Research Center, 1997a). Empirical acute hazard data from TCEP exposure is available from three fish and four invertebrate species and when used with the Web-ICE application resulted in predicted species (n = 46) to calculate the distribution of species sensitivity to acute TCEP exposure (Appendix G).

Subchronic exposures from one study and chronic exposure effects from five studies within aquatic vertebrates represent a variety of exposure durations, life history stages, and fish species (Table 4-2). Subchronic and chronic duration exposures of TCEP to aquatic vertebrates produced effects including: mortality (Hu et al., 2022; Wang et al., 2022); growth/development (Hu et al., 2023; Hu et al., 2022; Wang et al., 2022; Sun et al., 2016); organ level effects within liver (Hu et al., 2023; Hu et al., 2022; Sutha et al., 2020), gills (Sutha et al., 2020), kidney (Sutha et al., 2020), and heart rate (Wang et al., 2022). Chronic TCEP exposure in *Daphnia magna* produced effects on both mortality with a 21-day LC50 of 83.1 mg/L and reproduction with a 21-day ChV of 13 mg/L (Toray Research Center, 1997c). EPA identified two studies with overall quality determinations of high for green algae, resulting in five hazard values for diatoms and green algae (Zhang et al., 2024; Toray Research Center, 1997b). Although no amphibian studies were available to assess potential hazards from TCEP exposure, modeled data from Web-ICE provided acute hazard values from two surrogate species for bullfrog resulting in a geometric mean LC50 of 264 mg/L.

EPA calculated COCs for aquatic organisms, which are summarized in Table 4-6. These COCs were utilized to determine risk to aquatic organisms from modeled and published concentrations of TCEP in surface water and benthic pore water. EPA calculated an acute COC from the lower 95 percent CI of the HC05 of 16,7500 ppb for aquatic organisms based on the LC50 toxicity values from 3 test species and 19 fish, 1 amphibian, 7 aquatic invertebrates, and 10 benthic invertebrate species using Web-ICE (Raimondo and Barron, 2010). Empirical and predicted species acute hazard data consisted of 25 fish, 1 amphibian, 12 aquatic invertebrates, and 15 benthic invertebrates used to calculate the distribution of species sensitivity to TCEP exposure through the SSD toolbox (Etterson, 2020). The calculated HC05 was 31,600 µg/L. The acute COC equals the lower 95 percent CI of the HC05, or 16,700 µg/L. For the chronic COC, the ChV of the 30-day LOEC/NOEC of 0.028 mg/L for yellow catfish based on mortality (Hu et al., 2022), was used with the application of an AF of 10, resulting in 2.8 ppb. The ChV for growth and observations of liver histopathological alterations identified from the same study from 30 day TCEP exposure is 0.0028 mg/L (Hu et al., 2022). The landscape of hazard values for aquatic plants was largely represented within marine species with the resulting Web-ICE and SSD analysis utilizing empirical and predicted species hazard values from two freshwater species and six saltwater species. Two of the best fitting models both resulted in lower 95 percent CI value of the HC05, resulting in 66,000 µg/L.

Overall, EPA has robust confidence in the evidence that TCEP presents hazard to terrestrial mammals via dietary exposure, and moderate confidence that TCEP poses hazard to soil invertebrates (Table 4-7). For chronic terrestrial mammalian exposures to TCEP, the NOEL, and/or LOEL toxicity data ranged from a rat NOEL of 50 mg/kg-bw/day to a mouse LOEL of 1,000 mg/kg-bw/day for reproduction, mortality, and/or neurotoxicity endpoints. These were assigned an overall quality determination of high for all five studies with the exception of one medium overall quality determination for a reproduction endpoint (Yang et al., 2018a; Matthews et al., 1993; NTP, 1991b; Matthews et al., 1990; Hazleton Laboratories, 1983). EPA calculated chronic toxicity to mammals from TCEP exposure using a TRV. The TRV is equal to the highest NOAEL below the lowest LOAEL for mortality. The chronic TRV for mammals is 44 mg/kg-bw/day (Table 4-7). The TRV is used as the chronic hazard threshold for representative species during the trophic transfer assessments.

For soil invertebrate exposure to TCEP, a NOEC of 500 mg/kg soil and a LOEC of 750 mg/kg soil at 3 days exposure to TCEP was expressed for reduced growth and shortened lifespan of nematodes. The ChV is 612 mg/kg soil for growth and reduced lifespan (Xu et al., 2017) (Table 4-7).

Hazard threshold values for earthworms and American kestrels (Table 4-7) are represented by toxicity endpoints, including degradation of the digestive track in earthworms and increases in plasma thyroid hormones in kestrels. Although the most sensitive adverse outcome within soil invertebrates from TCEP exposure is for earthworm, the ecologically relevant effects for soil invertebrates are for reduced growth and shortened lifespan with a ChV of 612 mg/kg soil, from which an RQ value can be calculated. Similarly, while the hazard value for the American kestrel within this analysis is based on elevated plasma free thyroid concentrations at 7 days, the study did not detect any effects on free thyroid concentrations, kestrel growth (*i.e.*, body weight), nor food consumption at the conclusion of the 21-day dietary exposure study with TCEP (Fernie et al., 2015). Because the apical assessment endpoint of growth was not affected, it is difficult to assess the ecological relevancy of the change.

Table 4-6. Environmental Hazard Thresholds for Aquatic Environmental Toxicity

Environmental Aquatic Toxicity	Hazard Value (µg/L)	Assessment Factor (AF)	COC (µg/L)
Acute aquatic exposure: Lower 95% CI of HC05 from SSD	16,700	N/A[a]	16,700
Chronic aquatic exposure: based on fish ChV	28	10	2.8
Aquatic Plants COC: Lower 95% CI of HC05 from SSD	66,000	N/A[a]	66,000
[a] Used lower 95% CI of the HC05 or EC05 to account for uncertainties rather than an AF.			

Table 4-7. Environmental Hazard Thresholds for Terrestrial Environmental Toxicity

Environmental Terrestrial Toxicity	Hazard Value or TRV
Mammal	44 mg/kg-bw/day
American Kestrel (*Falco sparverius*)	0.0025 mg/kg-bw/day
Nematode (*Caenorhabditis elegans*)	612 mg/kg soil
Earthworm (*Eisenia fetida*)	0.3 mg/kg soil

4.2.6 Weight of Scientific Evidence Conclusions for Environmental Hazards

EPA uses several considerations when weighing and weighting the scientific evidence to determine confidence in the environmental hazard data. These considerations include the quality of the database, consistency, strength and precision, biological gradient/dose response, and relevance (see Appendix G.2.3.1) and are consistent with the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Table 4-8 summarizes how these considerations were determined for each environmental hazard threshold. Overall, EPA considers the evidence for chronic mammalian hazard thresholds robust, the evidence for aquatic vertebrate and invertebrate and terrestrial invertebrates hazard thresholds moderate, and the evidence for chronic avian hazard thresholds slight. A more detailed explanation of the weight of scientific evidence, uncertainties, and overall confidence levels is presented in Appendix G.2.3.1.

4.2.6.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the Environmental Hazard Assessment

Quality of the Database; and Strength (Effect Magnitude) and Precision

All the studies used to calculate COCs (aquatic fish and algae), TRVs (terrestrial mammals), and hazard thresholds (terrestrial invertebrates) received a high overall quality determination from the systematic review data quality evaluation. Effect magnitude was not reported for mammal studies. Effect magnitude was reported for aquatic fish, invertebrates, algae, and nematode studies using LC50s.

For aquatic organisms, six fish species, four invertebrate species, and five algal species were represented in the empirical data from systematic review; seven invertebrate and vertebrate species had data appropriate for the SSD model and all five algal species were appropriate for an SSD on aquatic plants. EPA was able to supplement the dataset for 46 aquatic species for TCEP with predictions from Web-ICE, which included predictions for 22 fish, 1 amphibian, 9 aquatic invertebrates, 14 benthic invertebrates. The use of three surrogate species available as inputs for the Web-ICE application reduces the confidence in the Web-ICE and subsequent SSD output. However, the use of the probabilistic approach within this risk evaluation increases confidence compared to a deterministic approach using the two studies on fishes and one study on *Daphnia magna* with acute hazard study endpoints. The use of the lower 95 percent CI instead of a fixed AF of 5 also increases confidence as it is a more data-driven way to account for uncertainty.

A total of seven studies represent subchronic and chronic exposure of TCEP to aquatic organisms and were assigned moderate confidence in the overall quality of the database. The one chronic study on invertebrates is a 21-day TCEP exposure to *Daphnia magna* reported effects on reproduction and survival (Toray Research Center, 1997c). The remaining six studies are from TCEP exposures to fishes (Japanese medaka, mrigal carp, yellow catfish, and zebrafish) for varying durations (14, 21, 30, and 120 days) and among different lifestages such as embryo/larvae (Wang et al., 2022; Sun et al., 2016), juvenile (Hu et al., 2022; Zhao et al., 2021; Sutha et al., 2020), and adult (Hu et al., 2023). Several of the chronic duration TCEP exposure studies recorded outcomes for apical assessment endpoints (*i.e.*, survival, growth, reproduction) and sub-organ level assessment endpoints (*i.e.*, hormone concentrations, gene expression, oxidative stress). A study with a 30-day TCEP exposure with a ChV as an endpoint of mortality was used to calculate the chronic COC (Hu et al., 2022). The 30-day exposure was conducted with analytical verification of TCEP concentrations throughout the duration of the exposure and was accompanied by recording outcomes associated with effects on survival and growth in addition to and mechanistic impacts on endocrine function and gene expression (Hu et al., 2022). There were no reasonably available empirical toxicity data available for benthic organisms. Using the acute and chronic COCs creates an additional uncertainty associated with extrapolating water column organism sensitivity from TCEP exposure.

The database for the aquatic plant assessment consisted of two studies on three species of green algae and two diatom species both with an overall quality determination of high (Zhang et al., 2024; Toray Research Center, 1997b). Slight confidence was assigned to the overall quality of the database due to the relatively limited number of studies and species represented. The previously conducted ECHA risk assessment on TCEP summarizes hazards for two species of algae (*Scenedesmus subspicatus* and *Selenastrum capricornutum*) represented with five studies and reported a wide range of hazard values for 48 to 96 hour exposures within algae (ECB, 2009). Moderate confidence was assigned to the strength as precision consideration for the algal assessment with the use of analytically verified TCEP concentrations and thorough protocols employed within the hazard study (Toray Research Center, 1997b).

For terrestrial mammal species, no wildlife studies were available from systematic review; however, four high-quality level studies with two species, mice and rats, represented were used from human health animal model studies. A TRV derived from the mammal studies was used to calculate the hazard threshold in mg/kg-bw.

For avian species, a single, high-quality level study was available for the American kestrel. The avian study detected transient differences in thyroid hormone level with no apparent effects on body weight or food consumption. Although the test did not detect any effects on apical assessment endpoints of regulatory interest (*i.e.*, impaired growth, survival, or reproduction) and the ecological relevancy of change in thyroid hormone level is uncertain, the study is still useful for the trophic transfer assessment. For example, if the results of the trophic transfer show that exposure from TCEP is lower than (*i.e.*, is protective for) the hazard threshold for effect on thyroid hormones, then a qualitative assertion can be made that the exposure levels from TCEP do not indicate risk.

For soil invertebrates, two high-quality level soil invertebrate studies were available. The earthworm study did not have an ecologically relevant endpoint effect, although the earthworm is still useful for assessing trophic transfer hazards both because of its direct ingestion of soil and because the earthworm is expected to be part of the diet of other trophic levels (short-tailed shrew, woodcock, and American kestrel).

Consistency: For aquatic fish species, the behavior effect of hypoactivity under dark phase stimulation and development/growth effects was similar in Japanese medaka and zebrafish. Behavioral differences between treatment and control groups were observed within mrigala carp from chronic 21-day exposures to TCEP characterized by behaviors such as fast swimming and inability to feed (Sutha et al., 2020). Activity under light and dark phases, as well as development/growth effects, were not tested with rainbow trout. Mortality effects for NOEC/LOEC and LC50s were similar for zebrafish and rainbow trout while the LC50 for the Japanese seabass was lower indicating more sensitivity to TCEP. Many chronic duration studies conducted aquatic vertebrates recorded histology, oxidative stress, gene expression, endocrine function, and apical outcomes including but not limited to growth and survival. Moderate confidence was assigned to the consistency of both acute and chronic exposure TCEP effects on aquatic species because the outcomes were consistently observed within aquatic species.

For terrestrial mammal species, human health animal model studies (rats) are in agreement with respect to neurotoxicity effects resulting from lesions to the brain. Confidence is robust on the MOA for rats on exposure to TCEP via diet due to neurotoxic effects with lesions to the brain. Three studies included mice; however, only a single study resulted in a LOEL for mortality. The maximum dose in all the studies that included both rats and mice were all below the single study for mice where the lowest test concentration resulted in the LOEL.

The single aquatic plant, avian, earthworm, and nematode studies were insufficient to characterize consistency in their respective outcomes.

Biological Gradient/Dose-Response

A dose response was reported for all studies used for calculating hazard thresholds as well as the earthworm study used in trophic transfer. However, because the American kestrel study only had one dose concentration, no dose-response was reported.

Biological Relevance: Behavior and developmental/growth effects were in agreement between species tested, zebrafish, yellow carp, mrigala carp and Japanese medaka (Hu et al., 2023; Hu et al., 2022; Sutha et al., 2020; Alzualde et al., 2018; Sun et al., 2016). Mortality effects from acute TCEP exposure were also in agreement between species tested (*Daphnia magna*, brine shrimp, Manila clam, Japanese seabass, mysid shrimp, zebrafish, and rainbow trout). All rat studies across multiple strains exhibited brain lesions from TCEP exposure that was associated with the mortality endpoint. Data were insufficient to observe correspondence of adverse outcomes across species within taxa group for avian of terrestrial invertebrates.

Physical/Chemical Relevance: Empirical data were on the effects of the chemical of interest, which increases confidence. TCEP was identified, including source, for all organisms. Purity was either not reported or not analytically verified for rainbow trout, earthworm, one of the mouse/rat studies (Matthews et al., 1990), and the American kestrel study (Fernie et al., 2015).

Environmental Relevance: Additional uncertainty is associated with laboratory to field variation in exposures to TCEP are likely to have some effect on hazard threshold; that is, gavage vs. natural forage diet for mammals (rats and mice) and invertebrate substrate (*i.e.*, nematodes maintained on nematode growth medium and earth worms on artificial soil). The five species of green algae and diatoms representing aquatic plant hazard were represented by four saltwater species (Zhang et al., 2024) and one freshwater species (Toray Research Center, 1997b). Test conditions for aquatic species correspond well with natural environmental conditions.

Table 4-8. TCEP Evidence Table Summarizing the Overall Confidence Derived from Hazard Thresholds

Types of Evidence	Quality of the Database	Consistency	Strength and Precision	Biological Gradient/ Dose-Response	Relevance[a]	Hazard Confidence[b]
Aquatic						
Acute aquatic assessment	++	++	++	+++	+++	Moderate
Chronic aquatic assessment	++	++	++	+++	+++	Moderate
Algal assessment	+	Not applicable	++	++	+	Slight
Terrestrial						
Chronic avian assessment	+	Not applicable	+	+	++	Slight
Chronic mammalian assessment	++	+++	+++	+++	+++	Robust
Terrestrial invertebrates	++	Not applicable	++	++	+++	Moderate

[a] Relevance includes biological, physical/chemical, and environmental relevance.
[b] Hazard Confidence reflects the overall confidence in the conclusions about the presence or absence of hazard thresholds and the weight of support and uncertainties around all the available data and does not necessarily represent a summation of the individual evidence properties.
+++ Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the hazard estimate.
++ Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize hazard estimates.
+ Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.

4.3 Environmental Risk Characterization

Environmental Risk Characterization (Section 4.3):
Key Points

EPA evaluated the reasonably available information to support environmental risk characterization. The key points of the environmental risk characterization are summarized below:

- For aquatic species, the 5 of 20 COUs resulted in chronic RQs greater than 1 with over 30 days of exceedance within surface water (Table 4-21).
- For aquatic species, chronic RQs are above 1 and have corresponding days of exceedance greater than 30 days within the sediment compartment (benthic pore water) for 5 of 20 COUs (Table 4-21). Because of TCEP's affinity to bind to sediment and persistence in the aquatic compartment, there could be a lasting effect on benthic biota and potential community-level impacts from chronic TCEP exposure. EPA has moderate confidence in the RQ inputs for the acute and chronic aquatic assessment.
- Monitoring data show RQs from mean TCEP surface water concentrations within the WQP database or published literature below 1 (Table 4-13). However, differences in magnitude between modeled and measured concentrations may be due to measured concentrations not being geographically or temporally close to releases of TCEP from a facility.
- For terrestrial species, EPA did not identify RQs greater than or equal to 1.
    - RQs for soil invertebrates or terrestrial mammals were less than 1 using either modeled soil concentrations or concentrations taken from the very limited monitoring data set available (from an urban area of Germany) (Table 4-22). EPA has moderate confidence in the RQ inputs for the terrestrial invertebrate assessment.
    - RQs were below 1 for all representative species and corresponding trophic level using TCEP soil concentrations from available published literature. RQs were below 1 for semi-aquatic terrestrial receptors via trophic transfer from fish and using the highest modeled TCEP surface water concentrations (Table 4-22). EPA has moderate confidence in the RQ inputs for the screening level trophic transfer assessment.

EPA considered fate, exposure, and environmental hazard to characterize the environmental risk of TCEP. For environmental receptors, EPA estimated (1) risks to aquatic species via water and sediment, and (2) to terrestrial species via exposure to soil by air deposition and through diet via trophic transfer. Risk estimates to aquatic-dependent terrestrial species included exposures to TCEP through water and diet. As described in Section 2.2.2, TCEP is described as a "ubiquitous" contaminant because it is commonly found in various environmental compartments such as surface water, soil, sediment, and biota. TCEP's physical and chemical properties suggests that its main mode of distribution in the environment is water and soil, depending on the media of release (Figure 2-1; see Appendix F.2.1.2). TCEP has the potential to undergo long-range transport in air and water (LTRP) that could be significantly underestimated when using its physical and chemical properties in QSAR models. TCEP's behavior in the environment often does not align with its physical and chemical properties. It can be transported to sediment from overlying surface water by advection and dispersion of dissolved TCEP and by deposition of suspended solids containing TCEP. However, TCEP can partition between surface water and sediments due to its log K_{OC} values (3.23 to 3.46) (Zhang et al., 2021; Wang et al., 2018a; Zhang et al., 2018b) and water solubility (7,820 mg/L) (U.S. EPA, 2015b; EC, 2009; ECB, 2009), which could contribute to its mobility in the environment. For example, TCEP in the soil was seen to be

vertically transported to deeper soil horizons, causing TCEP concentrations in the surface soil to be lower (He et al., 2017; Bacaloni et al., 2008). TCEP does not undergo hydrolysis under environmentally relevant conditions and is considered persistent in water (see Appendix F.2.3.1), sediment (see Appendix F.2.3.2), and soil (see Appendix F.2.4.1).

Direct exposure of TCEP to terrestrial receptors via air was not assessed quantitatively because dietary exposure was determined to be the driver of exposure to wildlife (Section 4.1.3). The contribution of exposure risk from inhalation relative to the ingestion exposure route is not expected to drive risk because of dilution-associated environmental conditions (U.S. EPA, 2003a, b). The gaseous phase of TCEP is expected to have a short half-life in the atmosphere ($t_{1/2}$ = 5.8 hours) with a log K_{OA} value of 7.86 to 7.93 (Okeme et al., 2020), suggesting this compound would adsorb to organic carbon present in airborne particles (Okeme et al., 2020; Ji et al., 2019; U.S. EPA, 2017a; Wang et al., 2017b). The resulting particle-bound TCEP would be expected to be removed from the atmosphere through wet or dry deposition. Annual air deposition to water and soil was modeled using AERMOD for applicable COUs (see Table 4-9), and these modeled values are included as components within the current environmental risk characterization.

EPA quantitatively assessed TCEP concentrations in surface water, pore water, sediment, and soil for aquatic and terrestrial receptors via modeled concentrations (EFAST 2014, VVWM-PSC, AERMOD) representing COU-based releases of TCEP. As reported in Section 3.3.2.5, EPA estimated surface water concentrations from COU based releases of TCEP and reported from 2,350 ppb (or µg/L) to 10,000 ppb with a production volume of 2,500 lb/year using high-end 95th percentile estimates. Considered to be a minor component, annual air deposition of TCEP to water was modeled using AERMOD indicating deposition to a lentic (*i.e.*, relatively static) system at 1,000 m from the source at 0.49 ppb with a production volume of 2,500 lb/year using high-end 95th percentile estimates, which is four orders of magnitude less than the lowest surface water concentration modeled using the model, VVWM-PSC. median TCEP surface water concentrations in ambient water were 0.23 ppb and ranged from 0.47 ppb to 7.66 ppb for 466 detected values in the WQP from 2003 to 2022. TCEP water concentrations in published literature were reported in Section 3.3.2 and represent ambient TCEP concentrations from surface waters and are not associated with direct environmental releases of TCEP. Maximum TCEP concentrations in surface waters were collected near urban environments recorded at 0.581, 0.785, and 0.810 ppb during low-flow conditions in the Los Angeles, San Gabriel, and Santa Clara Rivers in California, respectively (Maruya et al., 2016; Sengupta et al., 2014).

As reported in Section 3.3.2.9, modeled benthic pore water TCEP concentrations ranged from 90 to 334 ppb for the production volume of 2,500 lb/year. Modeled sediment concentrations ranged from 893 ppb (or µg/kg) to 1,960 ppb for the production volume of 2,500 lb/year. Air deposition to sediment, as reported in Section 3.3.2.10, indicated the highest annual deposition at 1,000 m was 125 ppb, which is almost 7 times lower than the lowest sediment TCEP value modeled with VVWM-PSC (Incorporation into paints and coatings – solvent borne at 893 ppb) and about 40 times lower than the highest PSC value for laboratory chemicals (5,040 ppb). As reported in Section 3.3.3.2, calculated TCEP soil concentrations resulting from modeled air deposition 1,000 m from the source with a production volume of 2,500 lb/year ranged from 1.49×10^{-6} to 0.0039 mg/kg and 1.92×10^{-6} to 0.0055 mg/kg for central tendency and high-end meteorology conditions.

Section 4.2 details available environmental hazard data and indicates that TCEP presents hazard to aquatic and terrestrial organisms. For acute exposures, TCEP is a hazard to aquatic animals at 16,700 ppb based on the lower 95 percent CI of the HC05 resulting from an SSD utilizing EPA's Web-ICE (Raimondo et al., 2023) and SSD toolbox applications (Etterson, 2020). For chronic exposures, TCEP is

a hazard to aquatic organisms with a ChV of 2.8 ppb for fish. For terrestrial exposures, TCEP is a hazard to mammals at 44 mg/kg-bw/day and a hazard to soil invertebrates with a ChV of 612 mg/kg. In addition, TCEP presented sub-organ level hazard values for birds at doses of 0.0025 mg/kg-bw/day and for soil invertebrates at 0.3 mg/kg soil and will serve to supplement terrestrial receptors via a conservative approach to estimate risk from trophic transfer.

EPA assigned an overall quality determination of high or medium to 23 acceptable aquatic toxicity studies and 17 acceptable terrestrial toxicity studies (see *Risk Evaluation for Tris(2-chloroethyl) Phosphate – Systematic Review Supplemental File: Data Quality Evaluation of Environmental Hazard Studies* (U.S. EPA, 2024w)). The *Risk Evaluation for Tris(2-chloroethyl) Phosphate – Systematic Review Supplemental File: Data Quality Evaluation of Environmental Hazard Studies* (U.S. EPA, 2024w) presents details of the data evaluations for each study, including evaluations of each metric and overall study quality level. As detailed in Section 4.2.6, EPA considers the evidence for terrestrial chronic mammalian robust, the evidence for aquatic hazard thresholds and terrestrial invertebrates moderate, and the evidence for the algae and terrestrial chronic avian slight.

4.3.1 Risk Characterization Approach

EPA characterized the environmental risk of TCEP using RQs (U.S. EPA, 1998b; Barnthouse et al., 1982), which are defined as follows:

Equation 4-2.

$$RQ = \frac{Environmental\ Exposure\ Concentration}{Hazard\ Threshold}$$

Environmental exposure concentrations for each compartment (*i.e.*, surface water, pore water, sediment, and soil) were based on measured (*i.e.*, monitored data and/or reasonably available literature) and/or modeled (*i.e.*, E-FAST 2014, VVMW-PSC, AERMOD) concentrations of TCEP from Section 3.3. EPA calculates hazard thresholds to identify potential concerns to aquatic and terrestrial species. These terms describe how the values are derived and can encompass multiple taxa or ecologically relevant groups of taxa as the environmental risk characterization serves populations of organisms within a wide diversity of environments. For hazard thresholds, EPA used the COCs calculated for aquatic organisms, and the hazard values or TRVs calculated for terrestrial organisms as detailed within Section 4.2.

RQs equal to 1 indicate that environmental exposures are the same as the hazard threshold. If the RQ is above 1, the exposure is greater than the hazard threshold. If the RQ is below 1, the exposure is less than the hazard threshold. RQs derived from modeled data for TCEP are shown in Table 4-11 and Table 4-12 for aquatic organisms, and Table 4-15 for terrestrial organisms. For aquatic species, acute risk is indicated when the RQ is greater than or equal to 1 for acute exposures, or chronic risk is indicated with a RQ greater than or equal to 1 with days of exceedance at or above 30 days for chronic exposures. The chronic COC was derived from a 30-day exposure; therefore, the days of exceedance to demonstrate risk reflects the exposure period for that hazard value. For terrestrial species, RQ values are calculated from the hazard value for soil invertebrates (nematode) and TRV for mammals as detailed in Section 4.2.4, and risk is indicated when the RQ greater than or equal to 1.

EPA used modeled (*e.g.*, E-FAST 2014, VVWM/PSC, AERMOD) and measured (*e.g.*, monitoring information from peer-reviewed literature or relevant databases) data to characterize environmental concentrations for TCEP and to calculate the RQ. Table 4-9 represents the COUs with relevant environmental releases represented in the current risk characterization on aquatic and terrestrial receptors. Exposure data are especially helpful to characterize exposures from facilities and/or COUs. In

the absence of facility-specific releases for TCEP, estimated releases were generated for a generic facility for each COU with production volume scenarios set at 2,500 lb/year (Table 4-9). Exposure data and corresponding RQ values produced with a production volume of 25,000 lb/year are presented within Appendix H. Surface water monitoring data on TCEP from available databases such as the WQP and published literature were used as additional approaches to characterize risk to aquatic receptors. The purpose of using monitored data and published literature, when available, was to determine if concentrations in the ambient environment exceeded the identified hazard benchmarks for aquatic and terrestrial receptors while also providing support for or concurrence with modeled concentrations.

As described in Section 3.3.3.2, IIOAC and subsequently AERMOD were used to assess the estimated release of TCEP via air deposition from specific exposure scenarios to soil (Table 4-9). Estimated concentrations of TCEP that could be in soil via air deposition at the community level (1,000 m from the source) exposure scenarios have been calculated.

Table 4-9. Risk Characterization to Corresponding Aquatic and Terrestrial Receptors Assessed for the Following COUs

COU (Life Cycle Stage/Category/Subcategory)	OES	RQ Values Calculated for Aquatic Receptors[a]	RQ Values Calculated for Terrestrial Receptors[b]
Manufacture/Import/Import	Repackaging	Yes	Yes
Processing/Incorporation into formulation, mixture, or reaction product/Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	Yes	Yes
Processing/Incorporation into formulation, mixture, or reaction product/Paint and coating manufacturing	Incorporation into paints and coatings – 2-part reactive coatings	Yes	Yes
Processing/Incorporation into formulation, mixture, or reaction product/Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resins	Yes	Yes
Processing/Incorporation into article/Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	N/A[d]	Yes
Processing/Recycling/Recycling	Recycling e-waste	EPA did not have sufficient data to estimate these releases[c]	
Distribution in Commerce/Distribution in commerce	Distribution in commerce	Distribution in commerce[d]	
Industrial use/Other use/Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Installing article (containing 2-part resin) for aerospace applications (electronic potting)	Releases expected to be negligible [c]	
Commercial use/ Other use/Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Installing article (containing 2-part resin) for aerospace applications	Releases expected to be negligible [c]	
Commercial use/Paints and coatings/Paints and coatings	Use of paints and coatings – Spray application	Yes	Yes

COU (Life Cycle Stage/Category/Subcategory)	OES	RQ Values Calculated for Aquatic Receptors[a]	RQ Values Calculated for Terrestrial Receptors[b]
Commercial use/Laboratory chemicals/Laboratory chemicals	Lab chemical – Use of laboratory chemicals	Yes	Yes
Commercial use/Furnishing, cleaning, treatment care products/Fabric and textile products		End of service life disposal (Releases and exposures not quantified) [c]	
Commercial use/Furnishing, cleaning, treatment care products/Foam seating and bedding products		End of service life disposal (Releases and exposures not quantified) [c]	
Commercial use/construction, paint, electrical, and metal products/Building/construction materials – Insulation		End of service life disposal (Releases and exposures not quantified) [c]	
Commercial use/Construction, paint, electrical, and metal products/Building/construction materials – Wood and engineered wood products – Wood resin composites		End of service life disposal (Releases and exposures not quantified) [c]	
Consumer use/Paints and coatings/Paints and coatings, including those found on automotive articles and replacement parts		No quantified environmental releases from consumer uses[e]	
Consumer use/Furnishing, cleaning, treatment care products/Fabric and textile products		No quantified environmental releases from consumer uses[e]	
Consumer use/Furnishing, cleaning, treatment care products/Foam seating and bedding products		No quantified environmental releases from consumer uses[e]	
Consumer use/Construction, paint, electrical, and metal products/Building/construction materials – Insulation		No quantified environmental releases from consumer uses[e]	
Consumer use/Construction, paint, electrical, and metal products/Building/construction materials – Wood and engineered wood products – Wood resin composites		No quantified environmental releases from consumer uses[e]	
Disposal/Disposal/Disposal		Waste disposal (Landfill or Incineration, covered in each COU/OES as opposed to a separate COU) [c]	

[a] RQ values calculated for aquatic receptors based on TCEP releases from wastewater, WQP database, and published literature
[b] RQ values calculated for terrestrial receptors based on TCEP releases as fugitive air and stack air deposition to soil, trophic transfer, and published literature
[c] Section 3.2 provides details on these OESs
[d] Distribution in Commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP or sealed packages of TCEP containing products (Section 3.1.1).
[e] Section 5.1.2.2.5 details the lack of information to characterize exposures for disposal of consumer wastes

EPA used IIOAC and AERMOD to estimate annual air deposition from hypothetical facility releases and calculate resulting surface concentrations to a pond. Air deposition to surface water for the 2,500 lb/year production volume scenario resulted in the highest annual deposition at 1,000 m of 0.49 µg/L which is approximately 200 times lower than the lowest surface water TCEP value modeled with VVWM-PSC (laboratory chemicals at 96 µg/kg). RQs for each relevant COU listed in Table 4-9 were

calculated for annual air deposition to surface water at 1,000 m and are presented within Appendix H for both production volumes and meteorological conditions. All RQ values for the production volume scenario of 2,500 lb/year were less than 1, with the highest RQ at 0.17 for TCEP use in paints and coatings at job sites. The higher production volume scenario of 25,000 lb/year also resulted in RQs that were less than 1 for all COUs expect the paints and coatings at job sites COU. The low production volume scenario modeling used high-end estimates for at 95th percentile of the mean. RQs for the mean (50th percentile) air to water deposition with the AERMOD for both meteorological models were below 1. It is not anticipated that air deposition to water will significantly contribute as TCEP concentrations within the water column, pore water, and sediment.

Frequency and duration of exposure can affect the potential for adverse effects in aquatic receptors. Within the aquatic environment, a two-tiered modeling approach was employed to predict surface water, pore water, and sediment TCEP concentrations. If E-FAST 2014 predicted 7Q10 surface water concentrations were greater than the chronic or acute COCs, the VVWM-PSC model was then used to confirm whether the predicted surface water concentration days of exceedance as determined by the acute COC and chronic COC. For TCEP, all six applicable OESs (Table 4-9) modeled in E-FAST 2014 produced chronic RQ values greater or equal to 1, prompting the use of VVWM-PSC for greater ecological resolution on TCEP concentrations and days of exceedance within the water column and benthic compartments. VVWM-PSC considers model inputs of physical and chemical properties of TCEP (*i.e.*, K_{OW}, K_{OC}, water column half-life, photolysis half-life, hydrolysis half-life, and benthic half-life) allowing EPA to model predicted benthic pore water and sediment concentrations.

Environmental RQ values by exposure scenario with TCEP surface water concentrations (ppb) were modeled by E-FAST 2014 and VVWM-PSC and are presented in Table 4-11. The max day average concentrations produced by VVWM-PSC represent the maximum concentration (ppb) over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate. Environmental RQ values by exposure scenario for aquatic organisms with TCEP pore water concentration modeled by VVWM-PSC are presented within Table 4-12. Scenarios and production volume allow for the calculation of RQs and days of exceedance that for risk estimation to aquatic organisms (scenarios with an acute RQ greater than or equal to 1, or a chronic RQ greater than or equal to 1 and 30 days or more of exceedance for the chronic COC).

EPA considers the biological relevance of species that COCs or hazard values are based on when integrating these values with the location of the surface water, pore water, and sediment concentration data to produce RQs. Life-history and habitat of aquatic organisms influence the likelihood of exposure above the hazard threshold in an aquatic environment. EPA has identified COC values associated with aquatic hazard values and include acute COC and chronic COC. The acute COC for aquatic species is the lower 95 percent CI of the HC05 of an SSD, a modeled probability distribution of toxicity values from multiple taxa inhabiting the water column. The chronic COC is represented by a mortality endpoint from 30-day exposures to TCEP within the water column. Calculated RQ values for pore water are represented with acute and chronic COCs. The confidence in these RQ inputs were described as moderate confidence determinations for the acute COC and chronic COC.

4.3.1.1 Risk Characterization Approach for Trophic Transfer

Trophic transfer is the process by which chemical contaminants can be taken up by organisms through dietary and media exposures and transfer from one trophic level to another. Chemicals can be transferred from contaminated media and diet to biological tissue and accumulate throughout an organisms' lifespan (bioaccumulation) if they are not readily excreted or metabolized. Through dietary consumption of prey, a chemical can subsequently be transferred from one trophic level to another. If biomagnification occurs,

higher trophic level predators will contain greater body burdens of a contaminant compared to lower trophic level organisms.

EPA conducted screening-level approaches for aquatic and terrestrial risk estimation based on exposure via trophic transfer using conservative assumptions for factors such as: area use factor, TCEP absorption from diet, soil, and water. Section F.2.5 details persistence as this compound is expected to persist within aquatic and terrestrial environments. Under laboratory conditions, mean whole body BCF for juvenile Atlantic Salmon (*Salmo salar*) is reported as 0.34 L/kg wet weight for an experimental exposure concentration of 1.0 mg/L (Arukwe et al., 2018). TCEP is not considered bioaccumulative; however, geometric mean concentrations within biota in Lake Erie have been reported at concentrations of 35.6 ng/g lipid as reported by Guo et al. (2017b) in Section 4.1.2. Section 4.1 reports measured concentrations of TCEP within biota with seven studies indicating TCEP concentrations within whole fish and lipid (see Section 4.1.2.1), one study within a marine mammal (see Section 4.1.2.1), and two studies with terrestrial organisms (see Section 4.1.3.1). A screening-level analysis was conducted for trophic transfer and formulation of RQ values from aquatic and terrestrial hazard values. If RQ values were greater than or equal to 1, risk estimation based on potential trophic transfer of TCEP is indicated from this screening-level approach and further refined analysis is warranted. If an RQ value is less than 1, risk based on potential trophic transfer of TCEP is not indicated from screening-level approach and no further assessment is necessary. The screening-level approach employs a combination of conservative assumptions (*i.e.*, conditions for several exposure factors included within Equation 4-3 below) and utilization of the maximum values obtained from modeled and/or monitoring data from relevant environmental compartments.

A secondary source of TCEP contamination in soil is from the application of biosolids. For this screening analysis, the COU with the highest release estimates were modeled with methods described within Section 3.3.3.5. Using BST, EPA estimated soil concentrations of 0.1412 mg/kg for the 2,500 lb/year production volume high-end estimate for the Incorporation into paints and coatings – 1-part coatings OES, and 0.5293 mg/kg for the 25,000 lb/year production volume, central tendency estimate for the Incorporation into paints and coatings – 2-part reactive coatings OES.

Following the basic equations as reported in Chapter 4 of the *U.S. EPA Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a), wildlife receptors may be exposed to contaminants in soil by two main pathways: incidental ingestion of soil while feeding, and ingestion of food items that have become contaminated due to uptake from soil. The general equation used to estimate the risk from exposure via these two pathways is provided below:

Equation 4-3.

$$RQ_j = \frac{\left(\left[Soil_j * P_s * \mathrm{FIR} * \mathrm{AF}_{sj}\right] + \left[\sum_{i=1}^{N} B_{ij} * P_i * [\mathrm{FIR} + \mathrm{WIR}] * \mathrm{AF}_{ij}\right]\right) * \mathrm{AUF}}{HT_j}$$

Where:

RQ_j	=	Risk quotient for contaminant (j) (unitless)
$Soil_j$	=	Concentration of contaminant (j) in soil (mg/kg dry weight)
N	=	Number of different biota type (i) in diet
B_{ij}	=	Concentration of contaminant (j) in biota type (i) (mg/kg dry weight)
P_i	=	Proportion of biota type (i) in diet
FIR	=	Food intake rate (kg of food [dry weight] per kg body weight per day)
WIR	=	Water intake rate (kg of water per kg body weight per day)

AF_{ij} = Absorbed fraction of contaminant (j) from biota type (i) (for screening purposes set equal to 1)
AF_{sj} = Absorbed fraction of contaminant (j) from soil (s) (for screening purposes set equal to 1)
HT_j = Hazard Threshold (mg/kg-BW[wet weight]/day)
P_s = Proportion of total food intake that is soil (kg soil/kg food)
AUF = Area use factor (for screening purposes set equal to 1)

Table 4-10. Terms and Values Used to Assess Potential Trophic Transfer of TCEP for Terrestrial Risk Characterization

Term	Earthworm (*Eisenia fetida*)	Short-Tailed Shrew (*Blarina brevicauda*)	American Woodcock (*Scolopax minor*)	American Kestrel (*Falco sparverius*)	American Mink (*Mustela vison*)
$Soil_j$ [a]	0.0055 mg/kg[b] TCEP	0.0055 mg/kg[b] TCEP	0.0055 mg/kg[b] TCEP	0.0055 mg/kg[b] TCEP	10.3 mg/L[c] TCEP
N	1	1	1	3	1
B_{ij}	0.0055 mg/kg[b] TCEP (soil)	0.0055 mg/kg TCEP (worm)	0.0055 mg/kg TCEP (worm)	0.0055 mg/kg TCEP (worm) 0.0046 mg/kg TCEP (short-tailed shrew) 0.0057 mg/kg TCEP (woodcock)	3.71 mg/kg[d] TCEP (Fish)
P_i	1	1	1	0.33	1
FIR	1	0.55[e]	0.77[e]	0.30[d]	0.22[e]
WIR	1	0.223[e]	0.1[e]	Dietary hydration	0.104[e]
AF_{ij}	1	1	1	1	1
AF_{sj}	1	1	1	1	1
HT_j	0.3 mg/kg-soil/day	44 mg/kg-bw/day	N/A[f]	0.0025 mg TCEP/kg-bw/day	44 mg TCEP/kg-bw/day
P_s	1	0.03[g]	0.164[g]	0.057[g]	1
AUF	1	1	1	1	1

[a] TCEP concentration in surface water for Mink
[b] Highest soil concentration of TCEP obtained using AERMOD modeling (2,500 lb/year)
[c] Highest surface water concentration of TCEP obtained using VVWM-PSC modeling (2,500 lb/year)
[d] Highest fish concentration (mg/kg) calculated from surface water concentration TCEP (VVWM-PSC) and whole body BCF of 0.34 (Arukwe et al., 2018)
[e] Exposure factors (FIR and WIR) sourced from EPA's *Wildlife Exposure Factors Handbook* (U.S. EPA, 1993b)
[f] No TCEP hazard threshold value for this representative species is available
[g] Soil ingestion as proportion of diet represented at the 90th percentile sourced from EPA's *Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a)

Terrestrial hazard data are available for soil invertebrate and mammals using hazard values detailed in Section 4.2.4. Representative avian and mammal species are chosen to connect the TCEP transport exposure pathway via trophic transfer from earthworm uptake of TCEP from contaminated soil through invertivore avian (American woodcock) and mammal (short-tailed shrew) species, to the American kestrel that feeds on invertebrates as well as avian and small terrestrial vertebrates.

At the screening-level, the conservative assumption is that the invertebrate diet for the American woodcock and short-tailed shrew comprises 100 percent earthworms from contaminated soil. Similarly, the dietary assumptions for the American kestrel are 100 percent of the invertebrate, avian, and mammal diet are from the earthworm, American woodcock, and short-tailed shrew, respectively. Additionally, the screening-level analysis uses the highest modeled or monitored soil contaminate level to determine if a more detailed assessment is required. Because surface water sources for wildlife water ingestion are typically ephemeral, the trophic transfer analysis for terrestrial organism assumed TCEP exposure concentration for wildlife water intake are equal to soil concentrations for each corresponding exposure scenario.

Exposure factors for food intake rate (FIR) and water intake rate (WIR) were sourced from the EPA's *Wildlife Exposure Factors Handbook* (U.S. EPA, 1993b). The proportion of total food intake that is soil (P_s) is represented at the 90th percentile for representative taxa (short-tailed shrew, woodcock, and hawk) and was sourced from calculations and modeling in EPA's *Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a). Additional assumptions for this analysis have been considered to represent conservative screening values (U.S. EPA, 2005a). Within this model, incidental oral soil exposure is added to the dietary exposure resulting in total oral exposure greater than 100 percent. In addition, EPA assumes that 100 percent of the contaminant is absorbed from both the soil (AF_{sj}) and biota representing prey (AF_{ij}). The proportional representation of time an animal spends occupying an exposed environment is known the area use factor (AUF) and has been set at 1 for all biota within this equation (Table 4-10).

The following hazard values were used for trophic transfer of TCEP from media (soil) through trophic levels: earthworm ChV of 0.3 mg/kg soil, mammal TRV dose of 44 mg/kg-bw/day, and American kestrel LOEL at doses of 0.0025 mg/kg-bw/day. Short-tailed shew and American mink hazard threshold values were calculated from the mammal TRV (44 mg/kg-bw/day). It is important to reiterate that hazard values within this screening-level trophic transfer analysis for earthworm and American kestrel are represented by endpoints of gastrointestinal damage and increased plasma thyroid hormones, respectively. Although the most sensitive adverse outcome within soil invertebrates from TCEP exposure is for earthworm, the ecologically relevant effects for soil invertebrates are for reduced growth and shortened lifespan with a ChV of 612 soil mg/kg from which an RQ value can also be calculated. The inclusion of earthworms and kestrels from this screening-level analysis represent an additional conservative approach for estimating risk to terrestrial organisms via trophic transfer.

For semi-aquatic terrestrial species, the TRV was used with the American mink for the screening-level assessment (Table 4-10). Similar to the above soil concentrations used as term $Soil_i$ in Equation 4-1, the highest surface water concentration modeled via VVWM-PSC was used as a surrogate for the TCEP concentration found in the American mink's diet, which is highly variable depending on habitat. In a riparian habitat, mink derive 74 to 92 percent of their diet from aquatic organisms, which includes fish, crustaceans, birds, mammals, and vegetation (Alexander, 1977). The American mink was used as the representative species for semi-aquatic mammals. As a conservative assumption, 100 percent of the American mink's diet is predicted to come from fish. Fish concentration (mg/kg) was calculated using surface water concentrations of TCEP from VVWM-PSC assuming a BCF of 0.34 as reported for whole body values from 1 mg/L TCEP exposures under laboratory conditions (Arukwe et al., 2018).

4.3.2 Risk Characterization for Aquatic Receptors

The VVWM-PSC model identified substantial deposition of TCEP to the sediment and resulting pore water (Table 4-12) with a production volume of 2,500 lb/year. A major concern centered around the RQs within pore water is the lasting effects on benthic biota and potential community-level impacts

from chronic TCEP exposure within this aquatic compartment. Acute risk estimates for both surface and pore water were not at or above one for any COU, however, chronic risk estimates were greater than one for 5 out of 20 COUs with quantified releases to water for both surface water (Table 4-11) and pore water (Table 4-12) using high-end estimates for the 2,500 lb/year production volume scenario. Furthermore, RQ values are also greater than one for both surface water (Table_Apx H-2) and pore water (Table_Apx H-3) the same 5 out of 20 COUs when using the central tendency estimates for the 2,500 lb/year production volume scenario.

The physical and chemical properties of TCEP and its persistence translate to removal from the water column by particulate and sediment organic matter and persistence within sediment (see Section 2.2.2). TCEP may partition between water and sediment due to its physical and chemical properties and, as a result, exposure of TCEP and the duration of that exposure to organisms dwelling within the sediment could be elevated. Many benthic invertebrates are detritivores, meaning they feed on dead plant and animal material or contribute to the liberation of additional nutrient resources by further breaking down these materials. Detritivorous benthic invertebrates often serve as an important food source for many juvenile fishery and non-game resident species. In several cases, days of exceedance were greater in pore water (Table 4-12) than the surface water (Table 4-11), further indicating that TCEP would be a more persistent hazard to benthic dwelling organisms with increased durations of exposure.

Listed below are the 5 out of 20 COUs (Life Cycle Stage/Category/Subcategory with their respective OES) evaluated, RQs for chronic duration exposures were greater than or equal to one with more than 30 days of exceedance within surface water and pore water.

Manufacture/Import/Import/Import and Repackaging
Surface Water: Surface water acute RQ values for import and packaging TCEP was less than 1 via both E-FAST 2014 and VVWM-PSC modeling. VVWM-PSC demonstrated a chronic RQ greater than 1 at 112.5 with 34 days of exceedance.

Pore Water: The pore water acute RQ for importing and repackaging TCEP was less than one the acute COC. The chronic RQ for importing and repackaging TCEP was greater than one for the chronic COC at 44.3. The corresponding days of exceedance for the chronic COC was 252 days.

Processing/Incorporation into Formulation, Mixture, or Reaction Product/Paints and Coating Manufacturing/Incorporation into Paints and Coatings – 1-Part Coatings
Surface Water: Surface water acute RQ values for TCEP incorporation into paints and coatings – 1-part coatings were less than 1 via both E-FAST 2014 and VVWM-PSC modeling. VVWM-PSC demonstrated a chronic RQ greater than 1 at 244.3 with 80 days of exceedance.

Pore Water: The pore water acute RQ for TCEP incorporation into paints and coatings – 1-part coatings was less than one for the acute COC. The chronic RQ for importing and repackaging TCEP was greater than one for the chronic COC at 96.8. The corresponding days of exceedance for the chronic COC was 298 days.

Processing/Incorporation into Formulation, Mixture, or Reaction Product/Paints and Coating Manufacturing/Incorporation into Paints and Coatings – 2-Part Coatings
Surface Water: Surface water acute RQ values for TCEP incorporation into paints and coatings – resins/solvent-borne were less than 1 via both E-FAST 2014 and VVWM-PSC modeling. VVWM-PSC demonstrated a chronic RQ greater than 1 at 110 with 33 days of exceedance.

Pore Water: The pore water acute RQ for TCEP incorporation into paints and coatings – resins/solvent-borne was less than one for the acute COC. The chronic RQ for importing and repackaging TCEP was greater than one for the chronic COC at 43.9. The corresponding days of exceedance for the chronic COC was 251 days.

Commercial use/Paints and coatings/Paints and coatings/Use in Paints and Coatings – Spray application

Surface Water: Surface water acute RQ values for TCEP use in paints and coatings at job sites were less than 1 via both E-FAST 2014 and VVWM-PSC modeling. VVWM-PSC demonstrated a chronic RQ greater than 1 at 132.5 with 33 days of exceedance.

Pore Water: The pore water acute RQ for TCEP use in paints and coatings at job sites was less than one for the acute COC. The chronic RQs for paints and coatings at job sites was greater than one for the chronic COC at 52.5. The corresponding days of exceedance for the chronic COC was 260 days.

Processing/Incorporated into Formulation, Mixture, or Reaction Product/Polymers Used in Aerospace Equipment and Products/Formulation of TCEP into 2-Part Reactive Resins

Surface Water: Surface water acute RQ values for formulation of TCEP into 2-part reactive resins were less than 1 via both E-FAST 2014 and VVWM-PSC modeling. VVWM-PSC demonstrated a chronic RQ greater than 1 at 129.3 with 52 days of exceedance.

Pore Water: The pore water acute RQ for formulation of TCEP into 2-part reactive resins was less than one for the acute COC. The chronic RQ for 2-part reactive resins was greater than one for the chronic COC at 51.4. The corresponding days of exceedance for the chronic COC was 262 days.

Commercial Use/Laboratory Chemicals/Laboratory Chemicals/Laboratory Chemicals

Surface Water: Within the water column, acute RQ values for laboratory chemicals were less than 1 via both E-FAST 2014 and VVMM-PSC modeling. VVMW-PSC modeling demonstrated a chronic RQ greater than 1 of 34.5 with days of exceedance of 210.

Pore Water: The pore water acute RQs for laboratory chemicals was less than one for the acute COC. The chronic RQ for laboratory chemicals was greater than one at 32. The corresponding days of exceedance for the chronic COC was 364 days.

Table 4-11. Environmental RQs by COU with Production Volumes of 2,500 lb/year for Aquatic Organisms with TCEP Surface Water Concentration (ppb) Modeled by VVWM-PSC

COU (Life Cycle Stage/Category/Subcategory)	OES	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Modeled Using VVWM-PSC				
					Max Day Average (ppb)[b]	COC Type	COC (ppb)	Days of Exceedance (days per year)	RQ
Manufacture/Import/Import	Import and repackaging	2,500	4	9.88	2,350	Acute	16,700	N/A	0.14
					315	Chronic	2.8	34	112.5
Processing/Incorporation into formulation, mixture, or reaction product/ Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	2,500	2	35.17	10,000	Acute	16,700	N/A	0.60
					684	Chronic	2.8	80	244.29
Processing/Incorporation into formulation, mixture, or reaction product/Paint and coating manufacturing	Incorporation into paints and coatings – 2-part reactive coatings	2,500	1	31.89	8,150	Acute	16,700	N/A	0.49
					310	Chronic	2.8	33	110.71
Commercial use/Paints and coatings/Paints and coatings	Use in paints and coatings – Spray application	2,500	2	23.25	5,500	Acute	16,700	NA	0.33
					371	Chronic	2.8	33	132.5
Processing/Incorporation into formulation, mixture, or reaction product/Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resins	2,500	1	31.53	9,040	Acute	16,700	N/A	0.54
					362	Chronic	2.8	52	129.28
Commercial use/Laboratory chemicals/ Laboratory chemicals	Laboratory chemicals	2,500	182	0.39	96.6	Acute	16,700	N/A	5.80E−03
					96.5	Chronic	2.8	210	34.46

N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs
[a] Production volume of 2,500 lb TCEP/year uses high-end estimates (95th percentile for all COUs except the laboratory chemicals COU uses the 1st percentile).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.

Table 4-12. Environmental RQs by COU with Production Volumes of 2,500 lb/year for Aquatic Organisms with TCEP Pore Water Concentration (ppb) Modeled by VVWM-PSC

COU (Life Cycle Stage/Category/Subcategory)	OES	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Benthic Pore Water Concentration (ppb)[b]	Benthic Pore Water			
						COC Type	COC (ppb)	Days of Exceedance	RQ
Manufacture/Import/Import	Import and repackaging	2,500	4	9.88	153	Acute	16,700	N/A	9.16E−03
					124	Chronic	2.8	252	44.28
Processing/Incorporation into formulation, mixture, or reaction product/Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	2,500	2	35.18	334	Acute	16,700	N/A	0.02
					271	Chronic	2.8	298	96.78
Processing/Incorporation into formulation, mixture, or reaction product/Paint and coating manufacturing	Incorporation into paints and coatings – 2-part reactive coatings	2,500	1	31.89	152	Acute	16,700	N/A	9.10E−03
					123	Chronic	2.8	251	43.92
Commercial use/Paints and coatings/Paints and coatings	Use in paints and coatings – Spray application	2,500	2	23.26	182	Acute	16,700	N/A	1.09E−02
					147	Chronic	2.8	260	52.5
Processing/Incorporation into formulation, mixture, or reaction product/Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resins	2,500	1	31.53	177	Acute	16,700	N/A	1.06E−02
					144	Chronic	2.8	262	51.4
Commercial use/Laboratory chemicals/Laboratory chemicals	Laboratory chemicals	2,500	182	0.40	90.5	Acute	16,700	N/A	5.42E−03
					89.6	Chronic	2.8	364	32

N/A = Days of Exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs

[a] Production volume of 2,500 lb TCEP/year uses high-end estimates (95th percentile for all COUs except the laboratory chemicals COU uses the 1st percentile).

[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.

EPA used surface water monitoring data from the WQP and published literature to characterize the risk of TCEP to aquatic organisms. These monitored surface water data reflect concentrations of TCEP in ambient water. WQP data show an average (± SEM) concentration for TCEP of 0.33 ± 0.02 ppb in surface water from 466 measurements taken throughout the United States between 2003 and 2022. The highest concentration recorded during this period was 7.66 ppb, which was recorded in August 2013 in Rochester, New York. Table 4-13 shows that RQ estimates were less than 1 for both acute and chronic COCs using the mean value from the WQP, however, the maximum recorded TCEP concentration within the WQP data show a RQ greater than 1.

Table 4-13. RQs Calculated Using Monitored Environmental Concentrations from WQX/WQP

Monitored Surface Water Concentrations (ppb) from 2003–2022	RQ Using Acute COC of 16,700 ppb	RQ Using Chronic COC of 2.8 ppb
Mean (Standard Error of the Mean): 0.33 (0.02) ppb	1.97E–05	0.11
Maximum: 7.66 ppb	4.58E–04	2.73

Five of the six studies from reasonably available published literature sampled waters within the United States, while one included sample sites from both U.S. and Canadian waters (Scott et al., 1996). All six studies from published literature are represented by general population surface water sampling where TCEP concentration are not associated with a specific facility. One study encompassed 85 sample sites for TCEP with study design placing sampling directly downstream from "intense urbanization and livestock production, detecting TCEP within 49 of the 85 samples and resulting in minimum and maximum TCEP concentrations of 0.02 and 0.54 ppb, respectively" (Kolpin et al., 2002). Across all studies a total of 185 samples resulted in 141 samples with TCEP detected and 44 non-detected samplings between 1994 and 2013. The mean (±SEM) for TCEP concentrations reported within surface water in the reasonably available published literature is 0.16 (±0.05) ppb with minimum and maximum concentrations of 0.0002 and 0.81 ppb, respectively. Table 4-14 shows RQs all estimates are less than one for both acute and chronic COCs.

Table 4-14. RQs Calculated Using TCEP in Surface Water from Monitored Environmental Concentrations from Published Literature

Monitored Surface Water Concentrations (ppb) from Published Literature	RQ Using Acute COC of 16,700 ppb	RQ Using Chronic COC of 2.8 ppb
Mean (Standard Error of the Mean): 0.16 (0.05) ppb	9.58E–06	5.71E–02
Maximum: 0.81 ppb	4.85E–05	0.29

4.3.3 Risk Characterization for Terrestrial Receptors

RQs were less than 1 for all relevant exposure scenarios when using the highest AERMOD predictions for air deposition to soil at 1,000 m. Table 4-15 presents soil concentration and chronic RQ values from the exposure scenario with the highest TCEP soil concentrations, indicating RQs below 1 for soil organisms based on modeling data. The highest soil concentration recorded from AERMOD predictions is 0.0055 mg/kg based on TCEP use in paints and coatings at job sites at 1,000 m. Soil concentrations and RQ values for all scenarios, production volumes, and meteorology models are presented within Table_Apx H-12.

Table 4-15. Calculated RQs Based on TCEP Soil Concentrations (mg/kg) as Calculated Using Modeled Data

OES	Production Volume (lb/year)[a]	Meteorological Model[b]	Soil Concentration (mg/kg) at 1,000 m[c]	Chronic RQ (Hazard Value: 612 mg/kg)
Use in paints and coatings at job sites	2,500	MetCT	3.97E–03	6.49E–06
		MetHIGH	5.58E–03	9.11E–06

[a] Production volume of 2,500 lb TCEP/yr uses high-end estimates (95th percentile).

[b] The ambient air modeled concentrations and deposition values are presented for two meteorology conditions (Sioux Falls, South Dakota, for central tendency meteorology; and Lake Charles, Louisiana, for higher-end meteorology).

[c] Estimated concentrations of TCEP (90th percentile) that could be in soil via air deposition at a community (1,000 m from the source) exposure scenario.

Risk characterization and trophic transfer for terrestrial receptors is based on modeled soil data from AERMOD because there are no published literature or monitoring databases with TCEP soil concentrations from U.S. sites and one comparative study from Germany (Mihajlovic and Fries, 2012). Transient increases in TCEP concentration have been observed with mean concentrations elevated from 0.008 to 0.023 mg/kg immediately following snowmelt conditions (Mihajlovic and Fries, 2012). RQs to soil invertebrates were below 1 for soil TCEP concentrations as reported for different sample periods from Mihajlovic and Fries (2012) (Table 4-16).

Table 4-16. RQs Calculated Using TCEP Soil Concentrations from Published Literature

Sample Collection Conditions	Mean TCEP Concentration in Soil (mg/kg)	Chronic RQ (Hazard Value: 612 mg/kg)	Reference (Overall Quality Determination)
Soil TCEP concentrations in January	5.89E–03	9.62E–06	(Mihajlovic and Fries, 2012) (High)
Soil TCEP concentration prior to snowmelt	7.67E–03	1.25E–05	
Soil TCEP concentration 24 hours after snowmelt	2.34E10–02	3.76E–05	

4.3.4 Risk Characterization Based on Trophic Transfer in the Environment

Trophic transfer of TCEP and potential risk to terrestrial animals was evaluated using a screening-level approach conducted as described in the EPA's *Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a). TCEP concentrations within biota and resulting RQ values for all six relevant COUs represented by seven OESs (Table 4-9), two production volume scenarios (2,500 and 25,000 lb/year), and two meteorological models for soil deposition are presented in Table_Apx H-13.
Table 4-17 presents biota concentrations and RQ values for the highest soil concentration via AERMOD (Paints and coatings) at the 2,500 production volumes. RQs were below 1 for all soil concentrations and COUs based on the chronic hazard threshold for terrestrial invertebrate identified within Section 4.2.4.2. The chronic TRV, calculated using empirical toxicity data with mice and rats, also resulted in RQs less than 1 for all modeled soil concentrations. The overall hazard confidence for the chronic mammalian assessment and terrestrial invertebrates reported within Section 4.2.6 as robust and moderate, respectively, providing increased confidence in the application of these ecologically relevant hazard thresholds.

Estimates of risk represented as RQ values were calculated using hazard thresholds with *in vivo* data measuring ecologically relevant endpoints such as mortality, reproduction, or growth. These RQ values are all below 1 for all species and corresponding trophic levels represented (Table 4-17). The earthworm and American kestrel are important tools in this screening-level trophic transfer analysis as they represent an animal with direct ingestion of soil (*i.e.*, the earthworm) and as a top avian predator (*i.e.*, the kestrel). Hazard values representing effects at the sub-organ level were identified for the earthworm (alterations in gastrointestinal tract) and American kestrel (alterations in plasma thyroid hormone levels). TCEP in biota calculated for the earthworm and American kestrel are at doses of 0.0055 and 0.0016 mg/kg/day, respectively, for the highest modeled soil TCEP concentration with a production volume of 2,500 lb/year. They did not equal or exceed these species hazard thresholds described within Section 4.2.4.2. The hazard value for the American kestrel (doses of 0.0025 mg/kg/day) did not result in any detectable impacts to ecologically relevant endpoints of body weight or food consumption from this 21-day dietary exposure study with TCEP (Fernie et al., 2015). One COU (*i.e.*, Use in paints and coatings at job sites) at the 25,000 lb/year production volume resulted in TCEP concentrations of 0.025 mg/kg/day; however, this production volume is believed to be an overestimate of current production volumes in the United States (see Section 1.1.1). In addition, the screening-level analysis used equation terms (*e.g.*, area use factor and the proportion of TCEP absorbed from prey and soil) all set to the most conservative values further emphasizing a cautious approach to risk to TCEP via trophic transfer.

Table 4-17. RQs for Screening-Level Trophic Transfer of Soil TCEP in Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs[a]

Organism	TCEP Concentration in Biota (mg/kg/day)[b]	Hazard Threshold (mg/kg-bw/day)	Reference for Hazard Value or TRV (Overall Quality Determination)	RQ
Nematode (*Caenorhabditis elegans*)	0.0055	612	(Xu et al., 2017) (High)	9.0E–06
Mammal	0.004	44	N/A[c]	9.8E–05
Woodcock (*Scolopax minor*)	0.005	N/A	N/A[d]	N/A

[a] Calculated using highest modeled soil TCEP concentrations with a production volume of 2,500 lb/year (0.0055 mg/kg); see also Equation 4-1.
[b] TCEP concentration represents the highest modeled soil concentration via AERMOD modeling with a production volume of 2,500 lb/year.
[c] Mammal TCEP TRV value calculated using several studies as per (U.S. EPA, 2007a).
[d] No TCEP hazard threshold value for this representative species is available.

Risk estimates were calculated for the highest releasing COUs at each production volume scenario for TCEP soil concentrations resulting from the application of biosolids (see Section 3.3.3.5). The high-end estimates with the 2,500 lb/year production volume scenario for Processing/ Incorporation into formulation, mixture, or reaction product/ Paint and coating manufacturing COU resulted in trophic transfer screening-level RQ values below 1 for both the soil invertebrate and mammal (Table 4-18). The same COU with central tendency estimates from the 25,000 lb/year production volume scenario also resulted in RQ values below 1 for the soil invertebrate and mammal (Table 4-18).

Table 4-18. RQs for Screening-Level Trophic Transfer of Biosolid TCEP in Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs[a]

Organism	TCEP Concentration in Biota (mg/kg/day)[b]	Hazard Threshold (mg/kg-bw/day)	Reference for Hazard Value or TRV (Overall Quality Determination)	RQ
Processing/Incorporation into formulation, mixture, or reaction product/ Paint and coating manufacturing COU, high-end estimate from 2,500 lb/year PV scenario				
Nematode (*Caenorhabditis elegans*)	0.141	612	(Xu et al., 2017) (High)	2.3E–04
Mammal	0.112	44	N/A[c]	2.5E–03
Woodcock (*Scolopax minor*)	0.140	N/A	N/A[d]	N/A
Processing/Incorporation into formulation, mixture, or reaction product/ Paint and coating manufacturing COU, central tendency estimate from 25,000 lb/year PV scenario				
Nematode (*Caenorhabditis elegans*)	0.529	612	(Xu et al., 2017) (High)	8.6E–04
Mammal	0.420	44	N/A[c]	9.5E–03
Woodcock (*Scolopax minor*)	0.527	N/A	N/A[d]	N/A

[a] Calculated using highest modeled biosolids TCEP concentrations with a production volumes of 2,500 lb/year (0.141 mg/kg) and 25,000 lb/year (0.529 mg/kg); see also Equation 4-1.
[b] TCEP concentration represents the highest modeled biosolid concentration as per Section 3.3.3.5.
[c] Mammal TCEP TRV value calculated using several studies as per (U.S. EPA, 2007a).
[d] No TCEP hazard threshold value for this representative species is available.

There are no reported studies within the pool of reasonably available published literature that quantify TCEP soil concentrations in the United States. A study with an overall quality determination of high monitored TCEP soil concentrations in the summer (August) and winter (January and February) months in Germany (Mihajlovic and Fries, 2012). The soil collection site was characterized as being located approximately 3 km from the city center of Osnabrueck and about 20 m from buildings constructed of reinforced concrete with facades predominately comprised of glass. Biota concentrations and RQ values were calculated using the same assumptions as described previously in Table 4-10, utilizing the highest TCEP soil concentration reported in Mihajlovic and Fries (2012). Note that this study should be considered to represent TCEP concentrations in soil from an ambient urban environment and is not directly comparable to scenarios detailed within the current risk evaluation. In a related study at the same site, the authors postulated that TCEP concentrations resulted from atmospheric deposition and potentially from cars, and emphasizing the importance of considering atmospheric deposition of chlorinated organophosphate esters (*e.g.*, TCEP) in future risk assessments (Mihajlović et al., 2011). The RQs are below 1 for all species and corresponding trophic level represented (Table 4-19). TCEP concentrations in biota calculated for the earthworm and American kestrel were 5.89×10^{-3} and 1.70×10^{-3} mg/kg/day, respectively, and do not equal or exceed these species hazard thresholds described in Section 4.2.4.2.

Table 4-19. RQs Calculated with Highest Mean TCEP Soil Concentration (5.89E–03 mg/kg) from Monitored Values in Published Literature for Screening-Level Trophic Transfer of TCEP in Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs[a]

Organism	TCEP Concentration in Biota (mg/kg/day)[b]	Hazard Threshold (mg/kg-bw/day)	Reference for Hazard Value or TRV (Overall Quality Determination)	RQ
Nematode (*Caenorhabditis elegans*)	5.89E–03	612	Xu et al. (2017) (High)	9.6E–06
Mammal	4.60E–03	44	N/A[c]	1.0E–04

[a] As reported in (Mihajlovic and Fries, 2012); see also Equation 4-1
[b] TCEP concentration represents the highest mean recorded soil concentration (5.89E−03 mg/kg) as reported in (Mihajlovic and Fries, 2012)
[c] Mammal TCEP TRV value calculated using several studies as detailed in (U.S. EPA, 2007a)
[d] No TCEP hazard threshold value for this representative species is available

RQs were below 1 for semi-aquatic terrestrial receptors via trophic transfer from fish and the resulting in the highest modeled TCEP surface water concentrations (Processing/ Incorporation into article/ Aerospace equipment and products and automotive articles and replacement parts containing TCEP, Table 4-20). RQ and biota concentration values for all COUs are presented within Table_Apx H-14. The hazard confidence for the chronic mammalian assessment was reported as robust within Section 4.2.6 and BCF values used to approximate TCEP concentrations within fish were from a high-quality study (Arukwe et al., 2018). The modeled TCEP concentrations within this analysis are five orders of magnitude greater than surface water concentrations identified from the WQP database and the published literature (Table 4-13 and Table 4-14). These results align with previous risk assessments that concluded that TCEP is not viewed as a bioaccumulative compound (U.S. EPA, 2015a; EC, 2009; ECB, 2009).

Table 4-20. Selected RQs (Highest Fish TCEP Concentrations) Based on Potential Trophic Transfer of TCEP from Fish to American Mink (*Mustela vison*) as a Model Aquatic Predator Using EPA's Wildlife Risk Model for Eco-SSLs[a]

COU	Production Volume (lb/year)	Release Distribution	SWC[a] (ppb)	Fish Concentration (mg/kg)	American Mink (*Mustela vison*)	
					TCEP in Biota (mg/kg/day)	RQ
Processing/Incorporation into article/Aerospace equipment and products and automotive articles and replacement parts containing TCEP	2,500	High-End	10,900	3.71	2.34	0.08

[a] See also Equation 4-1
[b] TCEP Surface Water Concentration (SWC) calculated using VVWM-PSC

4.3.5 Connections and Relevant Pathways from Exposure Media to Receptors

4.3.5.1 Aquatic Receptors

Surface Water, Benthic Porewater, and Sediment

Within the aquatic environment, a two-tiered modeling approach was employed to predict surface water, pore water, and sediment TCEP concentrations. If E-FAST 2014 predicted 7Q10 surface water concentrations were greater than the chronic or acute COCs, the VVWM-PSC model was then used to confirm whether the predicted surface water concentration days of exceedance as determined by the acute COC and chronic COC. For TCEP, all five applicable COUs (Table 4-9) modeled in E-FAST produced chronic RQ values greater or equal to 1, prompting the use of VVWM-PSC for greater ecological resolution on TCEP concentrations and days of exceedance within the water column and benthic compartments (see Section 4.3.1).

Air Deposition to Water and Sediment

EPA used IIOAC and AERMOD to estimate air deposition from hypothetical facility releases and to calculate pond water and sediment concentrations 1,000 m from the hypothetical facility. Pond water concentrations from air deposition were estimated for the COUs with air releases (Table 4-9). The highest estimated 95th percentile pond water concentration from annual deposition, across all exposure scenarios, was 0.49 ppb for the Commercial use of paints and coatings scenario at an annual production volume of 2,500 lb per year. This highest modeled concentration within a pond at 1,000 m from a point source was which is 4 orders of magnitude less than the lowest surface water concentration modeled using the model, VVWM-PSC (2,350 ppb as a maximum 1-day average concentration for the Manufacture/Import/Import COU at an annual production volume of 2,500 lb per year). Air deposition to sediment as reported in Section 3.3.2.10 indicated the highest annual deposition at 1,000 m was 125 ppb, which is about seven times lower than the lowest sediment TCEP value modeled with VVWM-PSC (Incorporation into paints and coatings – solvent borne at 893 ppb) and about 40 times lower than the highest PSC value for laboratory chemicals (5,040 ppb). Using VVWM-PSC, sediment concentrations from aquatic releases of TCEP ranged from 893 ppb to 5,040 ppb for the production volume of 2,500 lb/year, respectively, and represent a significant driver of TCEP deposition to sediment within flowing water systems. Although the IIOAC and AERMOD were applied to a generic farm pond setting to calculate concentrations of TCEP in pond surface water and pond sediment, these models do not account for media exchange of the chemical of interest as is the case for VVWM-PSC. It is not anticipated that air deposition to water significantly contributes to TCEP concentrations within flowing receiving waters.

TCEP Runoff from Biosolids

Due to its persistence, it is likely that dissolved TCEP will eventually reach surface water via runoff after the land application of biosolids. A review of reasonably available literature indicates that modeled surface water, pore water, and sediment concentrations are approximately half the highest concentrations and approximately 50 times greater than the mean values biosolid concentrations reported in Wang et al. (2019c). Direct exposure of TCEP to aquatic receptors via biosolids was not assessed quantitatively (see Section 3.3.3).

4.3.5.2 Terrestrial Receptors

Dermal Contact and Inhalation by Wildlife

Direct exposure of TCEP to terrestrial receptors via air was not assessed quantitatively because dietary exposure was determined to be the driver of exposure to wildlife. The contribution of exposure risk from inhalation relative to the ingestion exposure route is not expected to drive risk because of dilution associated environmental conditions and the deposition of TCEP from air to soil (U.S. EPA, 2003a, b). AERMOD results indicate a maximum ambient air concentration (95th percentile, MetHIGH) of

6.08×10^{-7} μg/m^3 at 1,000 m from a hypothetical facility for the Use of paints and coatings – spray application OES under the 2,500 lb/year production volume using the Suburban forest land category scenario (see Section 3.3.1.2). AERMOD results for the same conditions and COU for air deposition to soil indicate a TCEP concentration of 5.58 μg/kg at 1,000 m from a hypothetical facility (Table_Apx H-12). In addition, TCEP is not persistent in air due to short half-life in the atmosphere ($t_{1/2}$ = 5.8 hours) (U.S. EPA, 2017a) and because particle-bound TCEP is primarily removed from the atmosphere by wet or dry deposition (see Section 4.1.3.2).

Based on the *Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2003a, b) document, for terrestrial wildlife, relative exposures associated with inhalation and dermal exposure pathways are lower, even for volatile substances, compared to direct ingestion and ingestion of food (by approximately 1,000-fold). In addition, TCEP is not expected to bioaccumulate in tissues, and the screening-level trophic transfer analysis indicated that concentrations will not increase from prey to predator in either aquatic or terrestrial food webs (see Appendix F.2.6).

Biosolids

TCEP is released to the environment by various exposure pathways (Figure 2-1). The exposure pathway for terrestrial organisms is through soil. Deposition of TCEP from air to soil is the primary exposure pathway. A secondary source of TCEP contamination in soil is from the application of biosolids. Risk estimates were calculated for the highest releasing COUs at each production volume scenario for TCEP soil concentrations resulting from the application of biosolids (see Section 3.3.3.5) within a screening-level trophic transfer analysis. Results from screening the highest COUs at both the 2,500 lb/year and 25,000 lb/year production volume scenarios resulted in RQs below one for all organisms and both PV scenarios (see Section 4.1.4).

Air Deposition to Soil

As described in Section 3.3.3.2, IIOAC and subsequently AERMOD were used to assess the estimated release of TCEP via air deposition from specific exposure scenarios to soil (Table 4-9). Estimated concentrations of TCEP that could be deposited in soil via air deposition at the community level (1,000 m from the source) exposure scenarios have been calculated (see Section 4.3.1).

Soil in Diet

Following the basic equations as reported within Chapter 4 of EPA's *Guidance for Developing Ecological Soil Screening Levels*, wildlife receptors may be exposed to contaminants in soil by two main pathways: incidental ingestion of soil while feeding, and ingestion of food items that have become contaminated due to uptake from soil (U.S. EPA, 2005a). Within this model, incidental oral soil exposure is added to the dietary exposure resulting in total oral exposure greater than 100 percent (see Section 4.1.4).

Surface Water Ingestion in Wildlife

Because surface water sources for wildlife water ingestion are typically ephemeral, the trophic transfer analysis for terrestrial organisms assumed TCEP exposure concentration for wildlife water intake are equal to soil concentrations for each corresponding exposure scenario (see Section 4.1.4).

For semi-aquatic terrestrial species, the TRV was used with the American mink for the screening-level assessment (Table 4-10). Similar to the soil concentrations used as term $Soil_i$ in Equation 4-3, the highest surface water concentration modeled via VVWM-PSC was used as a surrogate for the TCEP concentration found in the American mink's diet (see Section 4.3.1.1).

Semi-aquatic Wildlife

The American mink was used as the representative species for semi-aquatic mammals. As a conservative assumption, 100 percent of the American mink's diet is predicted to come from fish. Fish concentration (mg/kg) was calculated using surface water concentrations of TCEP from VVWM-PSC assuming a BCF of 0.34 as reported for whole body values from 1 mg/L TCEP exposures under laboratory conditions (Arukwe et al., 2018). The conservative approach for calculated fish tissue concentrations presented in Section 4.1.2.2 was utilized for trophic transfer analysis to semi-aquatic mammals (see Section 4.3.1.1).

4.3.6 Summary of Environmental Risk Characterization

4.3.6.1 COUs/OESs with Quantitative Risk Estimates

EPA had uncertainty in the production volume; however, even at the realistic production volume of 2,500 lb/year, EPA found chronic RQs above 1 with more than 30 days of exceedance for aquatic receptors from both TCEP concentrations in surface water and pore water modeled with VVWM-PSC. Additionally, because of the physical and chemical and fate properties, EPA expects TCEP to partition between water and sediment and be persistent within the sediment compartment. Therefore, EPA has moderate confidence that there is risk to aquatic organisms for 5 out of 20 COUs.

The current environmental risk characterization on TCEP utilizes two alternate production volume assumptions for the calculation of RQ values. The 25,000 lb/year production volume is used as the high-end estimation. It is based on the reporting threshold for TCEP in CDR; however, given EPA's research, this is believed to be an overestimate of current production volumes in the United States. The 2,500 lb production volume is reflective of estimated current production volumes. In the current section, the analyses using 2,500 lb/year production volume are presented. Table 4-21 and Table 4-22 present RQ values for exposure scenarios with a production volume of 2,500 lb/year and corresponding environmental risk for aquatic and terrestrial receptors, respectively. Exposure data and corresponding RQ values produced with a production volume of 25,000 lb/year are presented within the Appendix H.

Exposure concentrations were modeled based on COU related releases to the aquatic environment and are represented by TCEP values within surface water and pore water. Confidence in aquatic exposure estimates is "moderate" with modeling parameters considering inputs from COUs and physical and chemical and fate parameters specific to TCEP. Surface water monitoring data were available from the WQP database and published literature. The overall exposure confidence for acute and chronic aquatic assessment were both rated as "moderate" (Table 4-24) with the inclusion of physical and chemical parameters represented within models performed with VVWM-PSC. The VVWM-PSC model identified substantial deposition of TCEP to the benthic compartment, which comprises sediment and benthic pore water. Physical and chemical properties including but not limited to K_{OC}, benthic half-life, and hydrolysis half-life within the VVWM-PSC model, aligns with the partitioning to organic carbon in sediment (see Appendix F.2.3.2) and persistence (see Appendix F.2.3.1). These parameters resulted in modeled data indicating TCEP concentrations residing within pore water over longer durations of time (days of exceedance) when compared to results from surface water concentrations for the chronic COC (2.8 ppb).

Within the aquatic environment, chronic RQs for aquatic receptors from TCEP exposure are elevated above one and have corresponding days of exceedance greater than 30 days within both surface water and pore water. Chronic RQ values for COUs with releases to water are presented with several scenarios for flow and production volume. Table 4-21 demonstrates RQ values from high-end production estimates from a 2,500 lb/year scenario with a 7Q10 low flow release representing the 50th percentile

stream flows of all facilities in a given industry sector, as defined by the SIC codes of the industry sector. Tables within Appendix H represent modeled surface water (Table_Apx H-2) and pore water (Table_Apx H-3) using central tendency production estimates from the 2,500 lb/year scenario with the same 50th percentile 7Q10 flows. Both central tendency and high-end estimate scenarios with the 2,500 lb/year production volume were found to have chronic RQs greater than 1 with days of exceedance greater than 30 for 4 COUs with quantified releases to water. The RQ value for the central tendency estimate for the paints and coatings COU with a P50 7Q10 low flow condition resulted in an RQ value of 6.7; however, the days of exceedance were 29. The COC for this RQ calculation was derived from a TCEP exposure of 30 days, thus the COC for this COU was 1 day short of exceeding that duration. Acute RQs for these production volume scenarios and under 1 for all five COUs with quantified releases to water.

When modeling TCEP concentrations in surface water and pore water with a 7Q10 low flow release representing the 90th percentile stream flows of all facilities in each industry sector, both the high-end and central tendency estimates resulted in chronic RQs less than 1 and days of exceedance less than 30 for all five COUs with quantified releases to water. Results for these scenarios with the 90th percentile 7Q10 stream flows and high-end estimates are presented in Table_Apx H-6 and Table_Apx H-7 for surface water and pore water, respectively. Results of the 90th percentile 7Q10 stream flows and central tendency estimates are presented in Table_Apx H-4 and Table_Apx H-5 for surface water and pore water, respectively. These results indicate the critical role of receiving water flow as an input in determining TCEP concentrations within both surface water and pore water.

For pore water, chronic RQs were greater than or equal to 1 with over 30 days of exceedance for all five relevant COUs (Table 4-21). Days of exceedance were greater in pore water (Table 4-12) than surface water (Table 4-11), indicating that TCEP is a more persistent hazard to benthic dwelling organisms with increased durations of exposure. All relevant COCs and relevant flow data for VVWM-PSC results for modeled pore water concentrations are available in Table 4-12. Concern for these RQs within pore water is the lasting effects on benthic biota and potential community-level impacts from chronic TCEP exposure within this aquatic compartment. Many benthic invertebrates are detritivores, meaning they feed on dead plant and animal material or contribute to the liberation of additional nutrient resources by further breaking down these materials. These detritivorous benthic invertebrates often serve as an important food source for many juvenile fishery and non-game resident species.

Chronic RQs were greater than one with over 30 days of exceedance for surface water and pore water TCEP modeled via VVWM-PSC at the 2,500 lb/year production volume for all five relevant COUs (Life Cycle Stage/Category/Subcategory/OES):

- Manufacturer/Import/Import/Repackaging;
- Processing/Incorporation into formulation, mixture, or reaction product/Paint and coating manufacturing/Incorporation into paints and coatings – 1-part coatings and 2-part reactive coatings
- Commercial use/Paints and coatings/Paints and coatings/Use in paints and coatings – Spray application
- Processing/Incorporation into formulation, mixture, or reaction product/Polymers used in aerospace equipment and products/Processing into 2-part resin article; and
- Commercial use/Laboratory chemicals/Laboratory chemicals/Lab chemical – Use of laboratory chemicals.

Table 4-21. Exposure Scenarios (Production Volume of 2,500 lb TCEP/year) and Corresponding Environmental Risk for Aquatic Receptors with TCEP in Surface Water and Pore Water

COU		OES[a b]	Aquatic Receptors[c]									
			Surface Water					Pore Water				
Life Cycle Stage/Category	Subcategory		Acute RQ[d]	Conf in Acute RQ Inputs[e]	Chronic RQ[f]	DoE[g]	Conf in Chronic RQ Inputs[e]	Acute RQ[d]	Conf in Acute RQ Inputs[e]	Chronic RQ[f]	DoE[g]	Conf in Chronic RQ Inputs[e]
Manufacture/Import	Import	Repackaging	0.14	Moderate	112.50	34	Moderate	9.16E−03	Moderate	44.28	252	Moderate
Processing/Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	0.60	Moderate	244.29	80	Moderate	0.02	Moderate	96.78	298	Moderate
Processing/Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 2-part reactive coatings	0.49	Moderate	110.71	33	Moderate	9.10E−03	Moderate	43.92	251	Moderate
Processing/Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resins	0.33	Moderate	132.50	33	Moderate	1.09E−02	Moderate	52.50	260	Moderate
Commercial use/Paints and coatings	Paints and coatings	Use in paints and coatings – Spray application	0.54	Moderate	129.28	52	Moderate	1.06E−02	Moderate	51.40	262	Moderate
Commercial use/Laboratory chemicals	Laboratory chemicals	Lab chemical – Use of laboratory chemicals	5.8E−02	Moderate	34.46	210	Moderate	5.42E−03	Moderate	32	364	Moderate

Modeled TCEP concentrations and RQ values for all relevant exposure scenarios are available in Table 4-11, and Table 4-12

[a] Production volume of 2,500 lb TCEP/yr uses high-end estimates (95th percentile for all COUs except the laboratory chemicals COU uses the 1st percentile)

[b] Risk assessed to aquatic receptors based on TCEP releases from wastewater, WQP database, and published literature

[c] All exposure values and Days of Exceedance (DoE) modeled using VVWM-PSC

[d] Acute RQ derived using a Concentration of Concern of 16,700 ppb

Conf = Confidence. Confidence in Acute RQ or Chronic RQ inputs is detailed in Section 4.3.7.2

[f] Chronic RQ derived using a Primary Concentration of Concern of 2.8 ppb

[g] Days of Exceedance (DoE) modeled using VVWM-PSC

Table 4-22. Exposure Scenarios (Production Volume of 2,500 lb TCEP/year) and Corresponding Environmental Risk for Terrestrial Receptors with TCEP in Soil (Invertebrates) and Trophic Transfer

COU		OES[a]	Meteorological Model[b]	Terrestrial Receptors[c]					
				Soil (Invertebrates)[d]		Trophic Transfer (Soil)[d]		Trophic Transfer (Water)[e]	
Life Cycle Stage/Category	Subcategory			RQ	Conf. in RQ Inputs[f]	Mammal RQ	Conf. in RQ Inputs[f]	American Mink RQ	Conf. in RQ Inputs[f]
Manufacture/Import	Import	Repackaging	MetCT	2.4E−06	Moderate	1.8E−06	Robust	0.02	Robust
			MetHI	3.1E−09		2.3E−06			
Processing/Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	MetCT	5.4E−08	Moderate	4.0E−05	Robust	0.08	Robust
			MetHI	9.3E−08		6.8E−05			
Processing/Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 2-part reactive coatings	MetCT	1.8E−08	Moderate	1.3E−05	Robust	0.07	Robust
			MetHI	3.9E−08		2.9E−05			
Processing/Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resins	MetCT	2.0E−08	Moderate	4.7E−05	Robust	0.08	Robust
			MetHI	4.2E−08		4.6E−05			
Processing/Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	MetCT	6.4E−08	Moderate	1.5E−05	Robust	NA	Robust
			MetHI	6.3E−08		3.1E−05			
Commercial Use/Paints and coatings	Paints and coatings	Use in paints and coatings at job sites	MetCT	6.5E−06	Moderate	0.005	Robust	0.04	Robust
			MetHI	9.1E−06		0.007			
Commercial Use/Laboratory chemicals	Laboratory chemicals	Lab chemical – Use of laboratory chemicals	MetCT	7.9E−08	Moderate	5.8E−05	Robust	7.0E−04	Robust
			MetHI	7.6E−08		5.6E−05			

[a] Production volume of 2,500 lb TCEP/year uses high-end estimates (95th percentile for all COUs except the laboratory chemicals COU uses the 1st percentile)
[b] The ambient air modeled concentrations and deposition values are presented for two meteorology conditions (MetCT: Sioux Falls, South Dakota, for central tendency meteorology; and MetHI: Lake Charles, Louisiana, for higher-end meteorology)
[c] Risk assessed to terrestrial receptors based on TCEP releases as fugitive air and stack air deposition to soil, trophic transfer, and published literature.
[d] Estimated concentrations of TCEP (90th percentile) that could be in soil via air deposition at a community (1,000 m from the source) exposure scenario
[e] Fish concentration (mg/kg) was calculated using surface water concentrations of TCEP from VVWM-PSC assuming a BCF of 0.34 as reported for whole body values from 1 mg/L TCEP exposures under laboratory conditions (Arukwe et al., 2018)
[f] Conf = Confidence; Confidence in RQ inputs are detailed in Section 4.3.7.2

RQs were less than 1 for all relevant COUs for air deposition to soil at 1,000 m (Table 4-22). The highest soil concentration from AERMOD predictions is 0.0055 mg/kg based on TCEP use in Paints and coatings at job sites at 1,000 m with the 2,500 lb/year production volume and higher-end meteorology condition. There are no published literature or monitoring databases with TCEP soil concentrations from U.S. sites and one comparative study from Germany (Mihajlovic and Fries, 2012). RQs for soil invertebrates were less than 1 with soil TCEP concentrations as reported for different sample periods from Mihajlovic and Fries (2012) (Table 4-16). This study should be considered to represent TCEP concentrations in soil from an ambient urban environment and is not directly comparable to scenarios detailed within the current risk evaluation. Mihajlović et al. (2011) emphasized the importance of atmospheric deposition of chlorinated organophosphate esters in risk assessments, which the current risk evaluation has taken into consideration for environmental risk characterization.

Trophic transfer of TCEP and potential risk to terrestrial animals was based on modeled soil data from AERMOD and concentrations reported within Mihajlovic and Fries (2012). A screening-level approach was conducted as described in EPA's *Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a). The two analyses performed represented: (1) trophic transfer for animals from exposures originating with TCEP soil concentrations and terrestrial prey items (Table 4-19), and (2) trophic transfer based for animals from exposures with TCEP water concentrations and aquatic prey items (Table 4-20). Table 4-22 demonstrates that RQs were less than 1 for any modeled soil concentrations and COUs based on the chronic hazard threshold for terrestrial invertebrate identified in Appendix H. The chronic TRV, calculated using empirical toxicity data with mice and rats, also demonstrated RQs less than 1 for all modeled soil concentrations (Table 4-22). In addition, RQs were less than 1 for all species represented within trophic levels using TCEP soil concentrations reported within Mihajlovic and Fries (2012) (Table 4-19). For semi-aquatic animals, RQs were also less than 1 for semi-aquatic terrestrial mammals via trophic transfer from fish and the highest modeled TCEP surface water concentrations (Table 4-20). The results of these screening-level trophic transfer analyses corroborate previous risk assessments indicating TCEP is not a bioaccumulative compound (U.S. EPA, 2015a; EC, 2009; ECB, 2009).

In the current environmental risk characterization for aquatic and terrestrial organisms, EPA considered aggregating exposure that a population would experience from being in close proximity to multiple facilities releasing TCEP to the environment. However, EPA did not aggregate across facilities for environmental exposures or risk because location information was not reasonably available for facilities releasing TCEP to the environment. Environmental media concentrations from monitoring data (*i.e.*, not associated with a specific exposure scenario or COU) were not aggregated with modeled environmental media concentrations associated with a specific exposure scenario or COU. TCEP from monitored surface water data reported within the WQP indicated a mean of 0.33 ± 0.02 ppb (see Section 4.3.2). Table 4-13 demonstrates that this mean surface water concentration for TCEP resulted in acute and chronic RQ values of 1.05×10^{-5} and 0.11, respectively. Similar database monitoring information were not reasonably available for sediment TCEP concentrations; however, the model used to predict surface water, sediment, and porewater TCEP concentrations was inclusive of physical and chemical properties (*i.e.*, K_{OW}, K_{OC}, water column half-life, photolysis half-life, hydrolysis half-life, and benthic half-life) known to contribute to TCEP's persistence within these media.

EPA also considered aggregating across pathways of exposure for aquatic and terrestrial organisms, but did not, because releases of TCEP to surface water and sediment were found to significantly contribute to these media when compared to deposition to water and/or sediment via air (see Section 4.3.5.1). Similarly, the most significant pathway for exposure to terrestrial receptors is via soil, which was modeled from air deposition (see Section 4.3.5.2). For aquatic organisms, surface water and sediment

pathways involve primary exposure routes such as epithelial uptake (skin, gills) and oral uptake. Aggregation of exposures via both surface water and dietary exposure was not conducted for aquatic organisms because TCEP is not expected to bioaccumulate except at very high concentrations that could result in risk directly from surface water (see Appendix F.2.6). The screening-level trophic transfer analysis performed included TCEP within prey in addition to soil ingestion for terrestrial receptors and water ingestion for semi-aquatic mammals (see Section 4.3.1.1).

4.3.6.2 COUs/OESs without Quantitative Risk Estimates

The following section represents a qualitative discussion of those remaining COUs and subsequent OESs lacking quantitative risk estimates.

Recycling and Distribution in Commerce

EPA did not have sufficient data to estimate releases to the environment for the following COUs:

- Processing – Recycling
- Distribution in commerce

EPA was not able to quantify releases of TCEP to the environment during the recycling of e-waste. E-waste recycling activities include receiving e-waste at the facility, dismantling or shredding the e-waste, and sorting the recycled articles and generated scrap materials (NIOSH, 2018; Yang et al., 2013; Sjödin et al., 2001). Only a subset of e-waste recycling facilities is expected to handle TCEP-containing products. The exact number of these facilities is unknown, and data were not reasonably available on the volume or source of TCEP contained in electronics processed at any of the facilities identified.

EPA did not find reasonably available data to quantify environmental releases of TCEP from e-waste recycling facilities. The total releases are expected to be low because TCEP is not typically used in electronics (Stapleton et al., 2011). Multiple studies show detections of TCEP at electronics and electrical equipment waste (e-waste) recycling facilities at air concentrations ranging from 1.0×10^{-7} to 1.1×10^{-3} mg/m^3, though the source of the TCEP at each facility is not specified (NCBI, 2020; Grimes et al., 2019; Stubbings et al., 2019; NIOSH, 2018; Yang et al., 2013; Sjödin et al., 2001). The low air concentration within facilities helps provide insight as to why one electronic recycling company categorized TCEP as "less commonly used in electronics now and in the past" with a detection percentage of 18 percent and range of "not detectable" to 10 ng/m^3 resulting in TCEP not being quantified in the NIOSH Report on Metals and Flame Retardants (Grimes et al., 2019). The concentrations at the site were based on personal air sampling for 19 participants over 2 days (Grimes et al., 2019). TCEP-containing materials from the recycling process are typically treated or disposed following the initial processing and not reprocessed or reused (Yang et al., 2013). EPA did not identify any reasonably available data for the weight fraction of TCEP in e-waste. This qualitative analysis indicates that releases of TCEP to the environment are potentially present from the recycling of e-waste. However, under similar exposure durations and exposure frequencies reported in the current section, releases from e-waste recycling facilities are expected to be lower relative to other quantified scenarios corresponding to environmental risk for terrestrial receptors (see Table 4-22) with the recycling COU expected to have lower risk than the quantified scenarios described within Section 4.3.6.1.

For purposes of assessment in this risk evaluation, distribution in commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP from import sites to downstream processing and use sites, or for final disposal of TCEP. The steps of loading and unloading that are assessed during other COUs/OESs consists of unloading TCEP into the formulation process and loading refers to packaging the finished product prior to shipment. Loading and unloading activities that occur during a distribution in commerce scenario would only refer to loading or unloading sealed containers

from a transport vehicle. EPA expects, under standard operating procedures, that environmental releases from sealed containers are not expected to occur.

Aerospace Equipment and Products
EPA does not expect significant releases to the environment for the following COUs/OESs:

- Industrial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP
    - OES: Installing article (containing 2-part resin) for aerospace applications (electronic potting)
- Commercial use – Other use – Aerospace equipment and products
    - OES: Installing article (containing 2-part resin) for aerospace applications

Specifically, EPA does not expect significant releases to occur during the installation of TCEP-containing aircraft and aerospace articles into or onto the relevant transportation equipment. The release assessment for these COUs/OESs are provided within Section 3.6.3 of the Engineering Supplemental file and summarized here. After TCEP-containing resins have cured, EPA expects TCEP release will be limited by the hardened polymer matrix. Releases may occur via the mechanism of "blooming" or volatilization from the cured resin surface during the service life of the aircraft or aerospace article, but EPA expects that releases via this mechanism during installation activities will be negligible (OECD, 2009; NICNAS, 2001). Furthermore, installation of aerospace equipment and products would be installed without any type of further processing of the article that would lead to potential releases (sanding, drilling, etc.). The Agency was not able to quantify environmental releases from blooming due to a lack of reasonably available information on the end use and service life of the product. Based on the finding of limited environmental releases from installation of TCEP-containing aircraft and aerospace articles into or onto the relevant transportation equipment, EPA determined that risk to the environment is not expected from releases of TCEP during the installation of these articles.

Commercial Uses (COUs) that TCEP is no longer actively incorporated into
The COUs listed below are only linked to end of service life disposal as manufacturing and processing is not ongoing:

- Commercial use – Furnishing, cleaning, treatment/care products – Fabric and textile products;
- Commercial use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products;
- Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation; and
- Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites.

EPA has confirmed from literature sources that TCEP was used for these purposes in past decades. However, these commercial uses were phased out beginning in the late 1980s or early 1990s and replaced by other flame retardants or flame-retardant formulations. EPA was unable to locate data to estimate the TCEP throughput used for these products, the amounts of these products that have already reached the end of their service life or amounts that have already been disposed of. The Agency assumes that products with TCEP that are still in use represents a fraction of the overall amount of TCEP previously used for these purposes and these types of products (*e.g.*, insulation and furniture) will result in a final deposition to landfills for disposal. TCEP releases to the environment from these commercial use COUs are expected to be lower relative to other quantified scenarios, as these commercial COUs have limited environmental release potential past end of service life disposal when compared to the quantified scenarios described within Section 4.3.6.1 The consumer assessment for these articles

resulted in no consumer risk for inhalation, ingestion dermal risk for adults for the COUs with moderate confidence. Due to the lack of reasonably available information on (1) the amount of TCEP used in these products, (2) the amounts of these products that have already reached the end of their service life, or (3), the amount of articles that remain in commercial environments, EPA is unable to quantify environmental releases for commercial COUs listed above.

Processing/Incorporation into Formulation, Mixture, or Reaction Product Processing/Incorporated into Article

EPA identified the following environmental releases via waste disposal; however, the Agency was unable to perform quantitative risk characterization of environmental releases related to waste disposal for the following COUs:

- Processing/Incorporation into formulation, mixture, or reaction product/Paint and coating manufacturing;
- Processing/Incorporation into formulation, mixture, or reaction product/Paint and coating manufacturing;
- Processing/Incorporation into formulation, mixture, or reaction product/Polymers used in aerospace equipment and products; and
- Processing/Incorporation into article/Aerospace equipment and products

EPA was able to perform quantitative risk characterization (Table 4-9) on the COUs listed above based on environmental releases to either fugitive or stack air and/or wastewater to on-site treatment or discharge to POTW, where applicable (Table 3-2). Waste disposal refers to either landfill or incineration and relies on inputs provided by the ESD or GSs. The proportion of the throughput that goes to either landfills or incinerators was not detailed within the ESD or GS. Details pertaining to the fate of disposal to these waste streams were unknown, a qualitative analysis of the disposal COU is presented below.

Consumer Uses

Although there is the possibility of environmental releases from consumer articles containing TCEP via off-gassing of consumer articles, down the drain release of TCEP from domestic laundry, the end-of-life disposal and demolitions of consumer articles, EPA was unable to quantify the environmental releases for the following COUs:

- Consumer use – Paints and coatings;
- Consumer use – Furnishing, cleaning, treatment/care products – Fabric and textile products;
- Consumer use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products;
- Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – insulation; and
- Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites

EPA was unable to quantify environmental exposures from consumer releases and disposal due to limited reasonably available information on source attribution of the consumer COUs. In previous assessments, EPA has considered down the drain analysis for consumer products for which a reasonably foreseen direct discharge exposure scenario can be assumed (*e.g.*, drain cleaner, lubricant, oils). TCEP containing dust present on consumer clothing may be released to the environment via domestic laundry; however, due to uncertainties in the source attribution of consumer COUs to dust, and the subsequent loading of dust on to clothing, EPA did not quantify environmental exposures for this scenario. Consumer releases to the environment are anticipated to be dispersed and less than direct TCEP releases

from COUs/OESs quantified for risk estimates for aquatic and terrestrial receptors detailed within Table 4-9.

Disposal

TCEP was among the 10 most frequently found compounds in a study that collected wastewater from multiple sites in the Research Triangle Park area of North Carolina between 2002 and 2005 (Giorgino et al., 2007). The study detected TCEP in 61.9 percent of wastewater samples, with a maximum concentration of 0.7 ppb. The maximum concentration from the USGS study (0.7 ppb) is similar to the maximum surface water TCEP concentration reported within published literature (0.81 ppb) used to calculate risks (see Section 4.3.2) and resulted in RQ values of less than one for both acute and chronic COCs (Table 4-14). The researchers indicated that flame retardants were measured primarily at sites downstream from municipal wastewater discharges and elevated concentrations were due to surface waters collected at a site downstream from an industrial fire.

Incineration of articles containing TCEP may create localized environmental releases. Aston et al. (1996) reported TCEP concentrations of up to 1.95 mg/kg in pine needles (Pinus ponderosa) in the Sierra Nevada foothills in the mid-1990s (Table 4-5). The source of the TCEP is unknown; however, authors suspected that these levels may have been due to aerial transport and deposition from nearby point sources such as incinerators.

The demolition and removal of commercial and consumer articles may result in environmental exposures to TCEP. Construction waste and old consumer products can be disposed of in municipal solid waste landfills and construction and demolition landfills. Section 3.3.3.8 models the resulting groundwater concentration that may occur from TCEP that leaches from landfills. Section 3.3.3.5 highlights suspected leaching of TCEP from nearby landfills (Norman Landfill, Himco Dump and Fort Devens, MA) (Buszka et al., 2009; Barnes et al., 2004; Hutchins et al., 1984). Groundwater from one well in Elkhart, Indiana, near the Himco Dump reported TCEP concentrations of 0.65 ppb to 0.74 ppb (Buszka et al., 2009). The Himco Dump is a closed, formerly unlicensed landfill that included a 4-acre construction debris area. EPA issued a notice in the Federal Register finalizing the deletion of part of the Himco Dump Superfund site from the National Priorities List (NPL) and the Indiana Department of Environmental Management (IDEM) formally concurred with EPA's proposal on January 26, 2022. EPA proposed the site for partial deletion in March 2022. Fort Devens is a former army installation established in 1917 and closed in 1996 and is also an EPA superfund site. Monitoring wells down-gradient of a land application facility near Fort Devens, Massachusetts, indicated TCEP concentrations from 0.28 ppb to 0.81 ppb (Hutchins et al., 1984). TCEP was detected throughout the entire length of a leachate plume near a municipal landfill (subtitle D) near Norman, Oklahoma (Barnes et al., 2004). TCEP concentration detected within the groundwater plume down-gradient of the Landfill in Norman, Oklahoma, ranged from 0.22 ppb to 0.74 ppb (Barnes et al., 2004). Leachate samples from landfill sites in Japan detected TCEP at ranges from 4.1×10^{6} to 5.4×10^{9} ppb with authors indicating that plastic wastes may serve as the origin (Yasuhara, 1995).

Without a full characterization of non-hazardous landfill (*e.g.,* Norman Landfill) conditions and historical wastes (*e.g.,* Himco Dump and Fort Devens) around the country, the data needed to produce quantitative risk estimates for disposal is not reasonably available. EPA does not have data representing municipal and managed landfills and is uncertain how often contaminant migration occurs given modern practices of non-hazardous landfill and historical site management. Source attribution of the consumer uses to the leaching concentration exhibited within Sections 3.3.3.7 and 3.3.3.8 are not reasonably available; therefore, it is unknown if these concentrations are the result of consumer and/or commercial disposal. The possibility of environmental exposure to TCEP after the release from disposal of consumer

wastes exists. The maximum TCEP concentrations recorded within groundwater at the Norman Landfill, Himco Dump, and Ft. Devens are 0.74 ppb, 0.81 ppb, and 0.74 ppb, respectively—which are similar to the to the maximum surface water concentrations reported within published literature (0.81 ppb) used to calculate risks (see Section 4.3.2) resulting in RQ values less than one for both acute and chronic COCs (Table 4-14).

For the commercial uses that have been phased out, any currently used products that contain TCEP are expected to be disposed of in landfills but will represent just a fraction of previous amounts from when TCEP was used more widely. Further data are lacking with which to estimate exposure and risk from disposal or waste treatment activities for these COUs and EPA has not quantified such risks. EPA's confidence in these exposures is indeterminate and cannot quantify risk for the disposal or waste treatment activities for these COUs. EPA acknowledges that while some releases and exposures could occur during the disposal of the wide variety of items that TCEP has found its way into, based on a review of the limited information on TCEP within groundwater at landfills and wastewater runoff presented in the section above, these are expected to be minimal and dispersed, and exposures are expected to be negligible.

4.3.7 Overall Confidence and Remaining Uncertainties Confidence in Environmental Risk Characterization

The overall confidence in the risk characterization combines the confidence from the environmental exposure, hazard threshold, and trophic transfer sections. This approach aligns with the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and TCEP Systematic Review Protocol (U.S. EPA, 2024p). The confidence from the trophic transfer section was completed in the same manner as the confidence in hazard threshold presented in Section 4.2.6 and Appendix G.2.3.1. For trophic transfer, EPA considers the evidence for chronic mammalian robust, the evidence for invertebrates moderate, and the evidence for chronic avian slight (Table 4-23). Synthesis of confidence for exposure, hazard, and trophic transfer (when applicable) resulted in the following confidence determinations for risk characterization RQ inputs: (1) robust for chronic mammalian evidence, (2) moderate for acute and chronic aquatic evidence, and (3) slight for chronic avian evidence (Table 4-24).

4.3.7.1 Trophic Transfer Confidence

Quality of the Database; Strength (Effect Magnitude) and Precision

Several conservative assumptions were applied across different representative organisms within trophic groups to represent a screening-level approach. For example, modeled TCEP concentrations within water (VVWM-PSC) and soil (via AERMOD) were applied to all COUs. TCEP concentrations obtained from these models were specific to each COU and production volume scenarios. Examination of potential risk from TCEP using this hazard value should be viewed as a conservative approach employed using both AERMOD modeled data and soil concentrations within published literature (Mihajlovic and Fries, 2012).

Trophic transfer analysis utilized American woodcock and American kestrel within the soil-based pathway to determine potential risk from TCEP. The hazard value for the raptor species is limited to a single study observing increased thyroid hormone production with no effects on body weight or food consumption from a 21-day feeding study (Fernie et al., 2015). No representative hazard data were available for the woodcock as an avian insectivore. RQ values were not calculated for the woodcock, which served as a prey item to the kestrel, representing uptake and transfer from a soil invertebrate to insectivore to carnivore.

Short-tailed shrew and American mink were employed as representative species using a mammalian TRV adjusted to their respective body weights. Mammalian hazard values for trophic transfer utilized ecologically relevant endpoints from high-quality studies originating from human health animal model investigations. The resulting TRV (Table 4-7) derived from mammal studies was used to calculate the hazard threshold in mg/kg-bw. Because the TRV is scaled by body weight, smaller representative species will have greater body burden from TCEP exposure than larger species.

For soil invertebrates, two high-quality soil invertebrate studies were available. Trophic transfer analysis used an ecologically relevant ChV from a nematode with endpoints related to reduced growth and shortened lifespan. The earthworm hazard value was also demonstrated in this analysis, although the earthworm did not have an ecologically relevant endpoint effect. The earthworm is still useful for assessing trophic transfer hazards because of its direct ingestion of soil. The earthworm also serves as a relevant prey item for all trophic levels (*i.e.*, short-tailed shrew, woodcock, and American kestrel).

Consistency

Inputs for soil and water TCEP concentrations displayed similarities among modeled and monitored concentrations. The highest soil concentrations modeled via AERMOD (Table 4-15) were within one order of magnitude to the highest soil concentrations reported within published literature (Table 4-16) (Mihajlovic and Fries, 2012). Concentrations of TCEP in whole fish reported within published literature (Guo et al., 2017b) represent concentrations two to three orders of magnitude lower than calculated fish TCEP concentrations (see Section 4.1.2). Any comparison to measured values reported within published literature should be viewed conservatively as organisms with direct proximity to the source of TCEP release and resulting surface water concentrations as calculated using VVWM-PSC.

Biological Relevance

The use of hazard values derived from singular studies for American kestrel, earthworm, and nematode are limiting in biological relevance; however, the application of conservative assumptions at each trophic level ensures a cautious approach to determining potential risk. For example, if the results of the trophic transfer show that exposure from TCEP is lower than the hazard threshold for thyroid effects, than a qualitative assertion can be made that the exposure levels from TCEP do not indicate risk. For avian species, only a single high-quality level study was available for the American kestrel with no hazard value for the avian insectivore within this analysis. The short-tailed shrew and American mink were selected as appropriate representative mammals for the soil- and aquatic-based trophic transfer analysis, respectively (U.S. EPA, 1993b). Overall, the use of exposure factors (*i.e.*, feed intake rate, water intake rate, the proportion of soil within the diet) from a consistent resource assisted in addressing species specific differences within the RQ equation (U.S. EPA, 1993b).

Physical and Chemical Relevance

The highest modeled TCEP concentrations for water and soil were used to investigate potential risk from trophic transfer. Assumptions within the trophic transfer equation (Equation 4-3) for this analysis have been considered to represent conservative screening values (U.S. EPA, 2005a) and those assumptions were applied similarly for each trophic level and representative species. Applications across representative species included assuming 100 percent TCEP bioavailability from both the soil (AF_{sj}) and biota representing prey (AF_{ij}). It is likely these considerations overrepresent TCEP's ability to transfer among trophic levels; however, it is a precaution built into the screening-level approach (U.S. EPA, 2005a).

Environmental Relevance

Although several aspects of the RQ equation were conservative and represented various species, there are still uncertainties associated with overall relevance of this model to fit all wildlife scenarios for potential TCEP risk. The current trophic transfer analysis investigated potential risk resulting from TCEP exposure in media such as soil and water. This analysis was extended to represent uptake from those media to soil invertebrates and fishes as a basis of trophic transfer from these prey to other higher trophic levels. Analysis included TCEP soil concentrations from published literature but ultimately relied on modeled TCEP water concentrations as the monitored TCEP values from WQP are three to five orders of magnitude less than modeled concentrations. The area use factor is the home range size relative to the contaminated area (*i.e.*, site/home range = AUF with the AUF within this screening-level analysis designated as 1 for all organisms). Application of this value in the RQ equation increases the conservative approach to trophic transfer analysis for larger animals such as mammals and birds assuming longer residence within an exposed area or a large exposure area.

Table 4-23. TCEP Evidence Table Summarizing Overall Confidence Derived for Trophic Transfer

Types of Evidence	Quality of the Database	Consistency	Strength and Precision	Biological Gradient/ Dose-Response	Relevance[a]	Trophic Transfer Confidence
Aquatic						
Acute Aquatic Assessment	N/A	N/A	N/A	N/A	N/A	N/A
Chronic Aquatic Assessment	N/A	N/A	N/A	N/A	N/A	N/A
Aquatic plants (vascular and algae)	N/A	N/A	N/A	N/A	N/A	N/A
Terrestrial						
Chronic Avian Assessment	+	++	+	N/A	+	Slight
Chronic Mammalian Assessment	+++	++	++	N/A	++	Moderate
Terrestrial invertebrates	++	++	++	N/A	++	Moderate

[a] Relevance includes biological, physical/chemical, and environmental relevance.
+ + + Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the hazard estimate.
+ + Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize hazard estimates.
+ Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.

4.3.7.2 Risk Characterization Confidence

Environmental risk characterization evaluated confidence from environmental exposures and environmental hazards. Hazard confidence was represented by evidence type as reported previously in Section 4.2.6. Trophic transfer confidence was represented by evidence type as reported in the preceding Section 4.3.7.1. Exposure confidence has been synthesized from Section 4.1.5.1 and is further detailed in the current section. The following confidence determinations for risk characterization RQ inputs are: (1) robust for chronic mammalian evidence, (2) moderate for acute and chronic aquatic evidence, and (3) slight for chronic avian evidence (Table 4-24).

Surface water concentration of TCEP were modeled initially using E-FAST and further refined using VVWM-PSC. Refined modeling with VVWM-PSC allowed estimates of TCEP pore water and sediment concentrations in addition to providing modeled days of exceedance for each compartment. Uncertainty associated with location-specific model inputs (*e.g.,* flow parameters and meteorological data) is present as no facility locations were identified for TCEP releases.

The modeled data represent estimated concentrations near hypothetical facilities that are actively releasing TCEP to surface water, while the reported measured concentrations represent sampled ambient water concentrations of TCEP. Differences in magnitude between modeled and measured concentrations may be due to measured concentrations not being geographically or temporally close to known releasers of TCEP. VVWM-PSC allowed for the application of a standard, conservative set of parameters and adjust for physical-chemical properties of TCEP. For example, stream reach was set to represent a waterway with a width of 8 m and depth of 2 m.

Physical and chemical properties including, but not limited to K_{OC}, benthic half-life and hydrolysis half-life appear to accurately represent TCEP's persistence; however, sensitivity analysis indicated that K_{OC} input parameters heavily influenced the role of sediment deposition to sediment. Maruya et al. (2016) represents an ambient environmental monitoring study within the published literature that made both surface water and sediment collections at the same sites and similar time periods within a watershed. Surface water collected in August and October 2013 and sediment samples collected in September 2013 were taken at 6 sites downstream of urban areas along the Santa Clara River in Southern California. TCEP sediment concentrations were consistently one order of magnitude higher than TCEP surface water concentrations across all sample sites. Specifically, mean (± SE) TCEP concentrations for surface water and sediment were 0.32 ± 0.04 ppb and 2.59 ± 0.75 ppb, respectively. Although a single study, Maruya et al. (2016) illustrates how TCEP within the water column of a flowing system can sorb to sediment to produce elevated concentrations. The WQP data and published literature on surface water TCEP concentrations is three to four orders of magnitude lower than modeled surface water concentrations. Confidence in the exposure components of the RQ inputs for benthic assessment is supported as studies within published literature are one to three orders of magnitude lower than results obtained from VVMW-PSC modeling. Confidence in exposure parameters for surface water have been rated "moderate" as the results are modeled from directly downstream from a hypothetical facility releasing TCEP.

Similar to aquatic exposures for TCEP, environmental exposures to soil invertebrates, mammals, and avian species relied on modeling air deposition to soil via AERMOD with supporting information from published literature. The AERMOD model included two meteorological conditions (Sioux Falls, South Dakota, for central tendency meteorology; and Lake Charles, Louisiana, for higher-end meteorology) in addition to different production volumes (2,500 and 25,000 lb/year) to characterize potential amounts of annual TCEP deposition to soil from air. One high-quality comparative study on TCEP soil

concentrations was identified within the published literature. TCEP fish tissue concentrations within the Great Lakes (Guo et al., 2017b) are two to three orders of magnitude lower than the TCEP tissue concentrations calculated using a whole organism BCF value from another high-quality study (Arukwe et al., 2018). Modeled soil concentrations were within one order of magnitude of a single study from published literature (Mihajlovic and Fries, 2012); however, it is important to note that similarity with a single study is not enough to build confidence in the relevance or accuracy of modeled results.

Table 4-24. TCEP Evidence Table Summarizing Overall Confidence for Environmental Risk Characterization

Types of Evidence	Exposure	Hazard	Trophic Transfer	Risk Characterization RQ Inputs
Aquatic				
Acute aquatic assessment	++	++	N/A	Moderate
Chronic aquatic assessment	++	++	N/A	Moderate
Terrestrial				
Chronic avian assessment	++	+	+	Slight
Chronic mammalian assessment	++	+++	++	Robust
Terrestrial invertebrates	++	++	++	Moderate

+ + + Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the hazard estimate.
+ + Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize hazard estimates.
+ Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.

5 HUMAN HEALTH RISK ASSESSMENT

EPA assessed human health risks of TCEP exposure to workers and ONUs, consumers, and the general population. Section 5.1 describes exposures to workers and ONUs via inhalation and oral routes; workers via dermal routes; consumers via inhalation, dermal, and oral routes; and the general population via oral, dermal, and inhalation routes. Human health hazards, including cancer and non-cancer endpoint identification and dose-response, are described in Section 5.2. Human health risk characterization is described in Section 5.2.9.

Updates to this section since the draft risk evaluation was released in December 2023 include the following:

1. Updated the evidence integration/causal descriptor for developmental toxicity from likely to suggestive;
2. Evaluated additional epidemiology studies for neurotoxicity, kidney toxicity, immune/hematological, thyroid, lung/respiratory, body weight, developmental toxicity, and cancer;
3. Revised analysis of the CEM Model for consumers to account for model updates and minor changes to parameter inputs;
4. Revised analysis for consumer paints and coatings COU;
5. Added sensitivity analysis for minimum weight fraction values; and
6. Revised dermal estimates and oral ingestion estimates from soils to include soil concentrations from BST analysis.

5.1 Human Exposures

Human Exposures (Section 5.1):
Key Points

EPA evaluated all reasonably available information for occupational, consumer, and general population exposure to TCEP, including consideration of the potential for increased susceptibility across PESS considerations (see also Section 5.3.3 and Appendix D). The key points are summarized below:

- Workers and ONUs can be exposed to TCEP via inhalation by dust or vapor.
  - However, large amounts of dust are not expected to be generated based on the types of activities that occur during the processing or use of TCEP-containing products or articles.
  - Workers can also be exposed to mists generated during the spray application of TCEP-containing paint products, but ONUs are not expected to be present during this use.
  - Workers will be exposed to TCEP via dermal exposure when processing liquid TCEP. However, once TCEP has been incorporated into an article the ability for appreciable amounts of TCEP to be absorbed through the skin will decrease significantly as there is little need for further processing of an article during installation.
- Chronic TCEP exposures from consumer articles to infants and children are the most relevant duration and populations of interest. Children's mouthing activity is an important factor when estimating exposure to TCEP in consumer products.
  - For consumer exposures, the inhalation route dominates exposure for building and construction materials such as roofing insulation, acoustic ceilings, and wood flooring. Exposures to infants and children for fabric and textiles, foam seating and bedding products, and wooden TV stands is dominated by the oral route.
  - Inhalation exposures are highest for building and construction products due to emission of vapors from consumer articles.
  - Dermal exposures are highest for wood resin products to children.
  - Ingestion exposures are highest for foam seating and bedding products for children.
- Fish ingestion is the most important exposure scenario for TCEP exposure to the general population. The bioaccumulation factor (BAF) and fish ingestion rate are sensitive parameters that influence these exposure estimates. Tribal populations for whom fish is important dietarily and culturally may have higher exposures than the general population and subsistence fishers.
- Fenceline communities may have elevated exposures from facilities that release TCEP. No site-specific information was available for TCEP, so EPA varied several inputs to show a range of possible exposures from a hypothetical facility.
- EPA identified several PESS groups: Infant exposure to TCEP via human milk was estimated by considering a maternal dose due to occupational, consumer, and general population exposures. Firefighters were identified as a PESS group through occupational exposure (see Section 5.3.3). Children and infants were identified as PESS through consumer exposure. Tribal communities, subsistence fishers, children, infants, and people living in fenceline communities near facilities that emit TCEP were identified as PESS through general population exposures.

5.1.1 Occupational Exposures

TCEP – Occupational Exposures (Section 5.1.1):
Key Points

EPA evaluated the reasonably available information for occupational exposures. The key points of the occupational exposure assessment are summarized below:

- Occupational exposure data available for TCEP:
  - EPA only identified monitoring data for dust occurring within an e-waste recycling facility; monitoring data for the remaining COUs/OESs were not found, most likely because TCEP does not have an assigned OSHA PEL (permissible exposure limit) and is therefore not typically tested for in the workplace.
  - For OESs that do not have data, EPA used relevant generic scenario and/or emission scenario documents to identify worker activities and exposure routes that are reasonably expected to occur. Exposure distributions were then created using Monte Carlo simulation with 100,000 iterations and the Latin Hypercube sampling method.
- The OES, use of paints and coatings – spray application, had the highest occupational exposure for inhalation and dermal exposure; this is due to mist being generated during application as well as a higher dermal loading value:
  - Inhalation exposure for use of paints and coatings – spray application ranges from 5.5 mg/m^3 (95th percentile, 8-hr TWA, resin-based paints) to 1.7×10^{-1} mg/m^3 (50th percentile, 8-hr TWA, water-based paints). EPA identified mist generation as the main driver of exposure but is not expected to occur during other COUs/OESs.
  - Dermal acute retained dose (mg/kg-day) ranges from 8.02 (95th percentile) to 1.48 (50th percentile).

The following subsections briefly describe EPA's approach to assessing occupational exposures and results for each condition of use assessed. For additional details on development of approaches and results refer to the *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Environmental Release and Occupational Exposure Assessment* (U.S. EPA, 2024n). As discussed in Section 3.1.1, EPA has mapped the industrial and commercial COUs to OESs in Table 3-1.

5.1.1.1 Approach and Methodology

As described in the *Final Scope of the Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) CASRN 115-96-8* (U.S. EPA, 2020b), for each COU, EPA distinguishes exposures for workers and ONUs. Normally, a primary difference between workers and ONUs is that workers may handle TCEP and have direct contact with the chemical, while ONUs are working in the general vicinity of workers but do not handle TCEP and do not have direct contact with it. Where possible, for each COU, EPA identified job types and categories for workers and ONUs.

As discussed in Section 3.1.1, EPA established OESs to assess the exposure scenarios more specifically within each COU. Table 3-1 provides a crosswalk between COUs and OESs. Figure 5-1 provides the approaches used by EPA to estimate exposures for the OESs included in this risk evaluation of TCEP. EPA did not identify any relevant inhalation exposure monitoring data to TCEP vapor for any of the OESs, because TCEP does not have an Occupational Safety and Health Act (OSHA) permissible exposure limit (PEL). For two OESs, monitoring data were available for TCEP in dust. The quality of

the monitoring data was evaluated using the data quality review evaluation metrics and the categorical ranking criteria described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Relevant data were assigned an overall quality determination of high, medium, low, or uninformative. In addition, EPA established an overall confidence for the data when integrated into the occupational exposure assessment. The Agency considered the assessment approach, the quality of the data and models, as well as uncertainties in assessment results to assign an overall confidence level of robust, moderate, or slight.

Where monitoring data were reasonably available, EPA used these data to characterize central tendency and high-end inhalation exposures. Where no inhalation monitoring data were identified, but inhalation exposure models were reasonably available, EPA estimated central tendency and high-end exposures using only modeling approaches. If both inhalation monitoring data and exposure models were reasonably available, where applicable, EPA presented central tendency and high-end exposures using both. EPA only identified measured dermal exposure estimates for dust generated at e-waste recycling facilities. Monitoring data were not reasonably available for any other COUs. EPA standard models, such as the EPA Mass Balance Inhalation Model and Fractional Absorption Model, were used to estimate high-end and central tendency inhalation and dermal exposures for workers in each OES.

For many cases, EPA did not have monitoring data to estimate inhalation exposure for ONUs. In some cases, this was addressed with the use of exposure models, when available. However, most OESs do not contain inhalation exposure estimates for ONUs. In general, EPA expects ONU exposures to be less than worker exposures. Dermal exposure for ONUs was not evaluated because these employees are not expected to be in direct contact with TCEP.

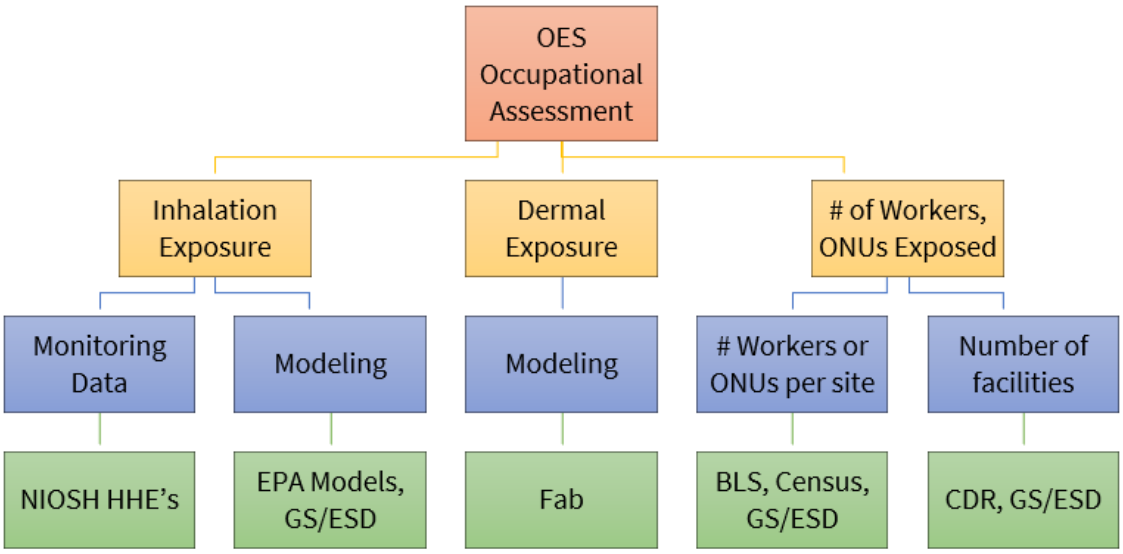


Figure 5-1. Approaches Used for Each Component of the Occupational Assessment for Each OES
CDR = Chemical Data Reporting; GS = Generic Scenario; ESD = Emission Scenario Document; BLS = Bureau of Labor Statistics; NIOSH (HHE) = National Institute for Occupational Safety and Health (Health Hazard Evaluations); Fab = Fractional Absorption Model

In Table 5-1, EPA provides a summary for each OES by indicating whether monitoring data were reasonably available; how many data points were identified, the quality of the data; EPA's overall confidence in the data; whether the data were used to estimate inhalation exposures for workers and ONUs; and whether EPA used modeling to estimate inhalation exposure to dust, vapors, or mist and dermal exposures for workers and ONUs.

Table 5-2 provides a summary of EPA estimates for the total number of potentially exposed workers and ONUs for each OES. To prepare these estimates, EPA first attempted to identify NAICS codes associated with each OES. For these NAICS codes, EPA then reviewed Standard Occupational Classification (SOC) codes from the Bureau of Labor Statistics (BLS) and classified relevant SOC codes as workers or ONUs. All other SOC codes were assumed to represent occupations where exposure is unlikely. EPA also estimated the total number of facilities associated with the NAICS codes previously identified based on data from the U.S. Census Bureau.

EPA then estimated the average number of workers and ONUs potentially exposed per generic site by dividing the total number of workers and ONUs by the total number of facilities. Finally, using EPA's estimates for the number of facilities using TCEP, the Agency was able to estimate the total number of workers and ONUs potentially exposed to TCEP for each OES. Additional details on EPA's approach and methodology for estimating the number of facilities using TCEP and the number of workers and ONUs potentially exposed to TCEP can be found in the *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Environmental Release and Occupational Exposure Assessment* (U.S. EPA, 2024n).

Table 5-1. Summary for Each OES

OES	Inhalation Exposure									Dermal Exposure				
	Monitoring					Modeling		Inhalation Exposure Confidence[a]		Monitoring		Modeling	Dermal Exposure Confidence[a]	
	Worker	# Data Points	ONU	# Data Points	Overall Quality Determ-ination	Worker	ONU	Worker	ONU	Worker	Overall Quality Determ-ination	Worker	Worker	ONU
Manufacture (import) – Repackaging	×	N/A	×	N/A	N/A	✓	×	Moderate	Slight	×	N/A	✓	Moderate	N/A
Processing – Incorporation into paints and coatings – 1-part coatings	×	N/A	×	N/A	N/A	✓	×	Moderate	Slight	×	N/A	✓	Moderate	N/A
Processing – Incorporation into paints and coatings – 2-part reactive coatings	×	N/A	×	N/A	N/A	✓	×	Moderate	Slight	×	N/A	✓	Moderate	N/A
Processing – Formulation of TCEP-containing reactive resins (for use in 2-part systems)	×	N/A	×	N/A	N/A	✓	×	Moderate	Slight	×	N/A	✓	Moderate	N/A
Processing – Processing into 2-part resin article	×	N/A	×	N/A	N/A	✓	×	Moderate	Slight	×	N/A	✓	Moderate	N/A
Processing – Recycling e-waste	✓	55	✓	21	High	×	×	Moderate	Moderate	×	N/A	✓	Moderate	N/A
Distribution – Distribution in commerce	Distribution in Commerce[b]													
Industrial use – Installation of article	✓	1 (Surrogate)	×	N/A	High	×	×	Slight	Slight	×	N/A	×	N/A	N/A
Commercial use – Use and/or maintenance of articles	✓	1 (Surrogate)	×	N/A	High	×	×	Slight	Slight	×	N/A	×	N/A	N/A
Commercial use – Use of paints and coatings – Spray application	✓	Surrogate Spray GS	×	N/A	High	×	×	Moderate	Slight	×	N/A	✓	Moderate	N/A
Commercial use – Lab chemical – Use of laboratory chemicals	×	N/A	×	N/A	N/A	✓	×	Robust	Moderate	×	N/A	✓	Moderate	N/A
Commercial uses:	×	N/A	×	N/A	N/A	×	×	N/A	N/A	×	N/A	×	N/A	N/A

OES	Inhalation Exposure									Dermal Exposure				
	Monitoring					Modeling		Inhalation Exposure Confidence[a]		Monitoring		Modeling	Dermal Exposure Confidence[a]	
	Worker	# Data Points	ONU	# Data Points	Overall Quality Determ-ination	Worker	ONU	Worker	ONU	Worker	Overall Quality Determ-ination	Worker	Worker	ONU
Furnishing, cleaning, treatment/care products fabric and textile products • Foam seating and bedding products Construction, paint, electrical, and metal products • Building/construction materials – insulation Building/construction materials – wood and engineered wood products – wood resin composites														
Disposal	Evaluated as part of each OES as opposed to a standalone OES													

Where EPA was not able to estimate ONU inhalation exposure from monitoring data or models, this was assumed equivalent to the central tendency experienced by workers for the corresponding OES; dermal exposure for ONUs was not evaluated because they are not expected to be in direct contact with TCEP.

[a] Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the hazard estimate.

Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize hazard estimates.

Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.

[b] Distribution in Commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP or sealed packages of TCEP containing products (Section 3.1.1).

5.1.1.2 Summary of Inhalation Exposure Assessment

Table 5-2 summarizes the number of facilities and total number of exposed workers for all OESs.

Table 5-2. Summary of Total Number of Workers and ONUs Potentially Exposed to TCEP for Each OES[a]

OES	Total Exposed Workers/Site	Total Exposed ONUs/Site	Total Exposed/Site (Exposure days/yr High-End – Central Tendency)	Number of Facilities[a]	Notes
Manufacture (import) – Repackaging	1	0	1 (7 – 4)	1 generic site	424690 – Other Chemical and Allied Products Merchant Wholesalers
Processing – Incorporation into paints and coatings – 1-part coatings	14	5	19 (38 – 6)	1 generic site	325510 – Paint and Coating Manufacturing
Processing – Incorporation into paints and coatings – 2-part reactive coatings	14	5	19 (2 – 1)	1 generic site	325510 – Paint and Coating Manufacturing
Processing – Formulation of TCEP-containing reactive resins (for use in 2-part systems)	27	12	39 (6 – 1)	1 generic site	325211 – Plastics Material and Resin Manufacturing
Processing – Processing into 2-part resin article	75	64	139 (250 – 72)	1 generic site	326400 – Aerospace Product and Parts Manufacturing
Processing – Recycling e-waste	2	2	4 (250 – 250)	Unknown	562920 – Materials Recovery Facilities
Distribution – Distribution in commerce			Distribution in commerce[b]		
Industrial use – Installation of article	75	64	139 (250 – 250)	1 generic site	326400 – Aerospace Product and Parts Manufacturing
Commercial use – Use and/or maintenance of articles	75	64	139 (250 – 250)	1 generic site	326400 – Aerospace Product and Parts Manufacturing
Commercial and Industrial use – Use of paints and coatings – Spray application	3	0	3	Sites vary based on multiple throughput scenarios; see Table 3-2	811121 – Automotive Body, Paint, and Interior Repair and Maintenance
	4	0	4 (Exposure days based on 1-, 2-, or 250-day scenarios)		238320 – Painting and Wall Covering Contractors

OES	Total Exposed Workers/Site	Total Exposed ONUs/Site	Total Exposed/Site (Exposure days/yr High-End – Central Tendency)	Number of Facilities[a]	Notes
Commercial Use – Lab chemical – Use of laboratory chemicals	3	3	6 (220 – 214)	13 sites (1st percentile) 6 sites (5th percentile)	541380 – Testing laboratories 541713 – Research and development in nanotechnology 541714 – Research and development in biotechnology (except nanobiotechnology) 541715 – Research and development in the physical, engineering, and life sciences (except nanotechnology and biotechnology) 621511 – Medical Laboratories
Commercial Uses – • Furnishing, cleaning, treatment/care products ○ Fabric and textile products ○ Foam seating and bedding products • Building/construction materials ○ Insulation ○ Wood resin composites		Manufacturing and processing for these COUs has ceased		N/A	
Disposal			Evaluated as part of each OES as opposed to a standalone OES		

[a] EPA's approach and methodology for estimating the number of facilities using TCEP and the number of workers and ONUs potentially exposed to TCEP can be found in the *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Environmental Release and Occupational Exposure Assessment* (U.S. EPA, 2024n).
[b] Distribution in Commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP or sealed packages of TCEP containing products (Section 3.1.1).

A summary of inhalation exposure results based on monitoring data and exposure modeling for each OES is presented for workers in Table 5-3 and Table 5-4, respectively. ONUs are presented in Table 5-5. These tables provide a summary of time-weighted average (TWA) inhalation exposure estimates as well as acute exposure concentrations (AC), average daily concentrations (ADC), lifetime average daily concentrations (LADC), and subchronic average daily concentration (SCADC). The ADC is used to characterize risks for chronic non-cancer health effects whereas the LADC is used for chronic cancer health effects. The SCADC represents repeated exposure for approximately 30 days and is used for intermediate exposure scenarios. Additional details regarding AC, ADC, LADC, and SCADC calculations along with EPA's approach and methodology for modeling inhalation exposure can be found in *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Environmental Release and Occupational Exposure Assessment* (U.S. EPA, 2024n).

Table 5-3. Summary of Inhalation Exposure Results for Workers Based on Monitoring Data for Each OES

OES	Inhalation Monitoring (Worker, ppm)									
	TWA		AC		ADC		LADC		SADC	
	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency
Processing – Recycling e-waste	9.68E−04	1.00E−07	6.6E−04	6.80E−08	4.51E−04	4.66E−08	2.31E−04	1.85E−08	4.83E−04	4.99E−08
Industrial use – Installation of article	1.3E−05	1.3E−05	8.8E−06	8.8E−06	6.5E−06	6.5E−06	3.1E−06	2.4E−06	6.5E−06	6.5E−06
Commercial use – Use and/or maintenance of articles	1.3E−05	1.3E−05	8.8E−06	8.8E−06	6.5E−06	6.5E−06	3.1E−06	2.4E−06	6.5E−06	6.5E−06

Table 5-4. Summary of Inhalation Exposure Results for Workers Based on Exposure Modeling for Each OES

OES	Inhalation Modeling (Worker, mg/m^3)									
	TWA (8-hr)		AC		ADC		LADC		SADC	
	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency
Manufacture (import) – Repackaging	4.1E−02	1.1E−02	2.8E−02	7.5E−03	3.1E−03	8.9E−05	1.2E−04	3.4E−05	3.7E−03	1.1E−03
Processing – Incorporation into paints and coatings – 1-part coatings	1.0E−01	1.7E−02	7.1E−02	1.1E−02	8.0E−04	1.9E−04	3.2E−04	7.3E−05	9.2E−03	2.2E−03
Processing – Incorporation into paints and coatings – 2-part reactive	4.0E−01	9.6E−02	2.7E−01	6.5E−02	7.9E−04	1.9E−04	3.1E−04	7.1E−05	9.6E−03	2.3E−03
Processing – Formulation of TCEP-containing reactive resins (for use in 2-part systems)	4.1E−01	7.4E−02	2.8E−01	5.1E−02	8.4E−04	1.8E−04	3.3E−04	6.9E−05	1.0E−02	2.2E−03
Processing – Processing into 2-part resin article	1.8E−02	3.4E−03	1.2E−02	2.3E−03	2.3E−03	3.9E−04	9.2E−04	1.5E−04	8.1E−03	1.6E−03

OES	Inhalation Modeling (Worker, mg/m^3)									
	TWA (8-hr)		AC		ADC		LADC		SADC	
	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency
Distribution – Distribution in commerce	Distribution in Commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP or sealed packages of TCEP containing products (Section 3.1.1).									
Commercial and Industrial use – Paints and coatings – Spray (1-part coatings, 1-day application) (OES #7)	1.1	1.7E−01	7.5E−01	1.1E−01	2.1E−03	3.1E−04	1.1E−03	1.3E−04	2.5E−02	3.8E−03
Commercial and Industrial use – Paints and coatings – Spray (1-part coatings, 2-day application)	1.1	1.7E−01	7.5E−01	1.1E−01	4.1E−03	6.3E−04	2.1E−03	1.37E−04	5.0E−02	7.7E−03
Commercial and Industrial use – Paints and coatings – Spray (1-part coatings, 250-day application)	1.1	1.7E−01	7.5E−01	1.1E−01	5.1E−01	7.9E−02	2.6E−01	3.1E−02	5.5E−01	8.4E−02
Commercial and Industrial use – Paints and coatings – Spray (2-part coatings, 1-day application)	5.5	8.5E−01	3.8	5.7E−01	1.0E−02	1.6E−03	5.3E−03	6.3E−04	1.3E−01	1.9E−02
Commercial and Industrial use – Paints and coatings – Spray (2-part coatings, 2-day application)	5.5	8.5E−01	3.8	5.7E−01	2.1E−02	3.1E−03	1.1E−02	1.3E−03	2.5E−01	3.8E−02
Commercial and Industrial use – Paints and coatings – Spray (2-part coatings, 250-day application)	5.5	8.5E−01	3.8	5.7E−01	2.6	3.9E−01	1.3	1.6E−01	2.8	4.2E−01
Commercial and Industrial use – Lab chemical – Use of laboratory chemicals	9.3E−04	5.8E−04	7.9E−04	5.1E−04	4.3E−04	2.7E−04	1.5E−04	8.8E−05	4.6E−04	2.9E−04
Disposal	Assessed as part of each OES and not as a stand-alone OES									

Table 5-5. Summary of Inhalation Exposure Results for ONUs Based on Monitoring Data and Exposure Modeling for Each OES

OES	Inhalation Monitoring (ONU, mg/m^3)									
	TWA		AC		ADC		LADC		SADC	
	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency
Recycling of e-waste	1.9E−04	1.0E−07	1.3E−04	6.8E−08	8.9E−05	4.7E−08	4.5E−05	1.9E−08	9.5E−05	5.0E−08
Note: For many cases, EPA was not able to estimate inhalation exposure for ONUs, but EPA expects these to be lower than inhalation exposure for workers.										

5.1.1.3 Summary of Dermal Exposure Assessment

Table 5-6 presents the estimated dermal acute retained dose for workers in various exposure scenarios. The exposure estimates are provided for each OES based on the maximum possible exposure concentration (Y_{derm}), which is the highest concentration level of TCEP that a worker handles throughout the process. The exposure concentration is determined based either on EPA's review of currently available products and formulations containing TCEP or the assumption that neat TCEP is handled to formulate these products.

The occupational dermal dose estimates assume one exposure event (applied dose) per workday and that absorption through and into the skin may occur for up to 8 hours as representative of a typical workday. Also, it is assumed that workers will thoroughly wash their hands with soap and water at the end of their shifts. Regarding material remaining in the skin post-washing, EPA considers the quantity of material remaining in the skin as potentially absorbable in accordance with OECD Guidance Document 156 (OECD, 2022). Therefore, overall occupational dermal exposure consists of the amount absorbed during the 8-hour workday plus the amount remaining in the skin after washing the hands at the end of the 8-hour workday.

In order to estimate occupational dermal exposures to TCEP, EPA relied on fractional absorption data from Abdallah et al. (2016). This study used a low concentration (≈0.005 wt percent in acetone) of TCEP for *in vitro* dermal absorption testing of a finite dose (*i.e.*, 500 ng/cm^2) over a 24-hour period. As mentioned above, the occupational exposure estimates are based on a typical 8-hour workday. Cumulative absorption data from Abdallah et al. (2016) show 82.69 ng/cm^2 absorbed after 8 hours of exposure and the fraction remaining in the skin is 0.068 after 24 hours of exposure. Because there were no data for the quantity remaining in the skin after 8 hours of exposure, EPA conservatively assumed that the quantity in the skin after 24 hours of exposure is representative of the amount remaining in the skin after 8 hours of exposure. EPA used the cumulative absorption data to determine the fraction absorbed after an 8-hour exposure period (0.165), and then conservatively added the fraction remaining in the skin at 24 hours (0.068). Therefore, the overall fractional absorption from an 8-hour exposure was calculated for a dilute solution containing TCEP as $F_{abs} = 0.165 + 0.068 = 0.233$.

Table 5-6. Summary of Dermal Retained Dose for Workers Based on Exposure Modeling for Each OES

OES	Max TCEP Weight Fraction (Max Y_{derm})	Non-occluded Worker Dermal Retained Dose	
		Dose (mg/day)	
		High-End	Central Tendency
Manufacture (import) – Repackaging	1.0	6.54	2.18
Processing – Incorporation into paints and coatings – 1-part coatings	1.0	6.54	2.18
Processing – Incorporation into paints and coatings – 2-part reactive coatings	1.0	6.54	2.18
Processing – Formulation of TCEP-containing reactive resins (for use in 2-part systems)	1.0	6.54	2.18
Processing – Processing into 2-part resin article	4.0E−01	2.62	8.73E−01
Processing – Recycling e-waste	1.40E−05	4.4E−05	1.8E−05
Distribution – Distribution in commerce	Distribution in commerce [a]		
Industrial use – Installation of article	N/A	N/A	N/A
Commercial use – Use and/or maintenance of articles	N/A	N/A	N/A
Commercial and Industrial use – Use of paints and coatings – Spray application OES	0.25	8.02	1.48
Commercial use – Lab chemical – Use of laboratory chemicals	1.0	6.54	2.18

Commercial uses:
• Furnishing, cleaning, treatment/care products
  ○ Fabric and textile products
  ○ Foam seating and bedding products
• Construction, paint, electrical, and metal products
  ○ Building/construction materials – insulation
  ○ Building/construction materials – Wood and engineered wood products – Wood resin composites
	N/A	N/A	N/A

Disposal	Evaluated as part of each OES as opposed to a standalone OES		
All dermal exposure scenarios are considered to be to a finite dose; therefore, no scenario is considered occluded. [a] Distribution in Commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP or sealed packages of TCEP containing products.			

5.1.1.4 Weight of Scientific Evidence Conclusions for Occupational Exposure

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S.

EPA, 2024n) provides a summary of EPA's overall confidence in its inhalation exposure estimates for each of the OESs assessed.

5.1.1.4.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the Occupational Exposure Assessment

Number of Workers

Several uncertainties surround the estimated number of workers potentially exposed to TCEP. Current CDR data reported in 2020 do not show production volumes that exceed the threshold of 25,000 pounds and therefore, information was not available to estimate the number of workers associated with manufacturing, processing, or use of TCEP.

There are inherent limitations to the use of CDR data as reported by manufacturers and importers of TCEP. Manufacturers and importers are only required to report if they manufactured or imported more than 25,000 lb of TCEP at a single site during any calendar year; as such, CDR may not capture all sites and workers associated with any given chemical because it is possible for entities to use less than the CDR threshold. Therefore, EPA assumes that any ongoing manufacturing, import, processing, or use of TCEP occurs using volumes below the CDR threshold.

There are also uncertainties with BLS data, which are used to estimate the number of workers for the remaining COUs. First, BLS' OES employment data for each industry/occupation combination are only available at the 3-, 4-, or 5-digit NAICS level, rather than the full 6-digit NAICS level. This lack of granularity could result in an overestimate of the number of exposed workers if some 6-digit NAICS are included in the less granular BLS estimates but are not likely to use TCEP for the assessed applications. EPA addressed this issue by refining the OES estimates using total employment data from the U.S. Census' Statistics of U.S. Businesses (SUSB). However, this approach assumes that the distribution of occupation types (SOC codes) in each 6-digit NAICS is equal to the distribution of occupation types at the parent 5-digit NAICS level. If the distribution of workers in occupations with TCEP exposure differs from the overall distribution of workers in each NAICS, then this approach will result in inaccuracy but would be unlikely to systematically either overestimate or underestimate the count of exposed workers.

Second, EPA's judgments about which industries (represented by NAICS codes) and occupations (represented by SOC codes) are associated with the uses assessed in this report are based on EPA's understanding of how TCEP is used in each industry. Designations of which industries and occupations have potential exposures is nevertheless subjective, and some industries/occupations with few exposures might erroneously be included, or some industries/occupations with exposures might erroneously be excluded. This would result in inaccuracy but would be unlikely to systematically either overestimate or underestimate the count of exposed workers.

Analysis of Exposure Monitoring Data

This risk evaluation uses existing worker exposure monitoring data to assess exposure to TCEP during some COUs, depending on availability of data. To analyze the exposure data, EPA categorized each data point as either "worker" or "occupational non-user." The categorizations are based on descriptions of worker job activity as provided in literature and EPA's judgment. In general, samples for employees that are expected to have the highest exposure from direct handling of TCEP are categorized as "worker" and samples for employees that are expected to have the lower exposure and do not directly handle TCEP are categorized as "occupational non-user."

Exposures for ONUs can vary substantially. Most data sources do not sufficiently describe the proximity of these employees to the TCEP exposure source. As such, exposure levels for the "occupational non-

user" category will have high variability depending on the specific work activity performed. It is possible that some employees categorized as "occupational non-user" have exposures similar to those in the "worker" category depending on their specific work activity pattern.

Some scenarios have limited exposure monitoring data in literature, if any. Where there are few data points available, it is unlikely the results will be representative of worker exposure across the industry. In cases where there was no exposure monitoring data, EPA used monitoring data from similar COUs as a surrogate. For example, EPA did not identify inhalation monitoring data for installation of articles based on the systematic review of literature sources. However, EPA estimated inhalation exposures for this OES using monitoring data for TCEP exposures during furniture manufacturing (Mäkinen et al., 2009). EPA expects that inhalation exposures during furniture manufacturing occur from handling or contacting TCEP-containing products, which is comparable to inhalation exposures expected during installation of TCEP-containing products for aircraft or aerospace applications as well as automotive parts and replacement parts. While these COUs have similar worker activities contributing to exposures, it is unknown that the results will be fully representative of worker exposure across different COUs.

Where sufficient data were reasonably available, the 95th and 50th percentile exposure concentrations were calculated using reasonably available data. The 95th percentile exposure concentration is intended to represent a high-end exposure level, while the 50th percentile exposure concentration represents a typical exposure level. The underlying distribution of the data, and the representativeness of the reasonably available data, are not known. Where discrete data were not reasonably available, EPA used reported statistics (*i.e.,* 50th and 95th percentile). Because EPA could not verify these values, there is an added level of uncertainty.

EPA calculated ADC and LADC values assuming workers and ONUs are regularly exposed during their entire working lifetime, which likely results in an overestimate. Individuals may change jobs during their career such that they are no longer exposed to TCEP, and actual ADC and LADC values would be lower than the estimates presented.

The following describe additional uncertainties and simplifying assumptions associated with use of this modeling approach for TCEP:

- *No OSHA PEL (Very Little Monitoring Data):* While EPA has confidence in the models used, it is possible that they may not account for variability of exact monitoring processes and practices at an individual site.
- *No 2020 CDR Reporters and Only One 2016 CDR Reporter (with No Downstream Details Provided):* Assumptions of an ongoing production volume of 2,500 and 25,000 lb per site-year could overestimate actual amount of TCEP handled at a given site, thus overestimating actual exposures and releases. Release and exposure information using the 25,000 lb per site-year is provided in the Engineering Supplemental file.

Modeled Dermal Exposures

The Fractional Absorption Model is used to estimate dermal exposure to TCEP in occupational settings. The model assumes a fixed fractional absorption of the applied dose; however, fractional absorption may be dependent on skin loading conditions. The model also assumes a single exposure event per day based on existing framework of the EPA/OPPT 2-Hand Dermal Exposure to Liquids Model and does not address variability in exposure duration and frequency. Additionally, the studies used to obtain the underlying values of the quantity remaining on the skin (Q_u) did not take into consideration the fact that liquid retention on the skin may vary with individuals and techniques of application on and removal from the hands. Also, the data used were developed from three kinds of oils; therefore, the data may not

be applicable to other liquids. Based on the uncertainties described above, EPA has a moderate level of confidence in the assessed baseline exposure (see Table 5-1).

5.1.2 Consumer Exposures

TCEP – Consumer Exposures (Section 5.1.2):
Key Points

EPA evaluated the reasonably available information for the following consumer exposures, the key points of which are summarized below:

- Limited information is available on TCEP in consumer products.
  - There are no current safety data sheets.
  - Weight fraction estimates in some cases were derived from literature values that were over 20 years old and from maximum values reported in Washington State databases.
- The highest exposure estimates were from inhalation of the roofing insulation scenario (1.42 mg/kg/d) and the wood flooring scenario (1.24 mg/kg/day). However, EPA's confidence in these estimates is low. Of the scenarios with moderate or robust confidence, the highest inhalation and oral exposure estimates were from the textile for children's outdoor play structures scenario (0.0604 mg/kg/day, 0.185 mg/kg/day, respectively).
- Inhalation is the driver for exposure to building and construction materials (*e.g.*, roofing insulation, acoustic ceiling) and wood flooring for adults.
- Oral ingestion is the driver for exposure for fabric and textile products, foam seating and bedding products, and wooden television stands for children and infants.

5.1.2.1 Approach and Methodology

The migration of additive flame retardants from indoor sources such as building materials, fabrics, textiles, and wood articles (from either ongoing COUs or in service products/articles at the end of their life cycle) appear to be a likely source of flame retardants found in indoor dust, suspended particles, and indoor air (Dodson et al., 2012; Weschler and Nazaroff, 2010). However, the relative contribution of different sources of TCEP in these matrices is not well characterized. For example, building insulation, textiles, and paints and coatings that contain TCEP have differing magnitudes of emissions that depend on a variety of differing conditions.

Modeling was conducted to estimate exposure from the identified consumer COUs. Exposures via inhalation, oral, and dermal routes to TCEP-containing consumer products were estimated using EPA's Consumer Exposure Model (CEM), Version 3.2 (U.S. EPA, 2023). Figure 5-2 below displays the embedded models within CEM 3.2.

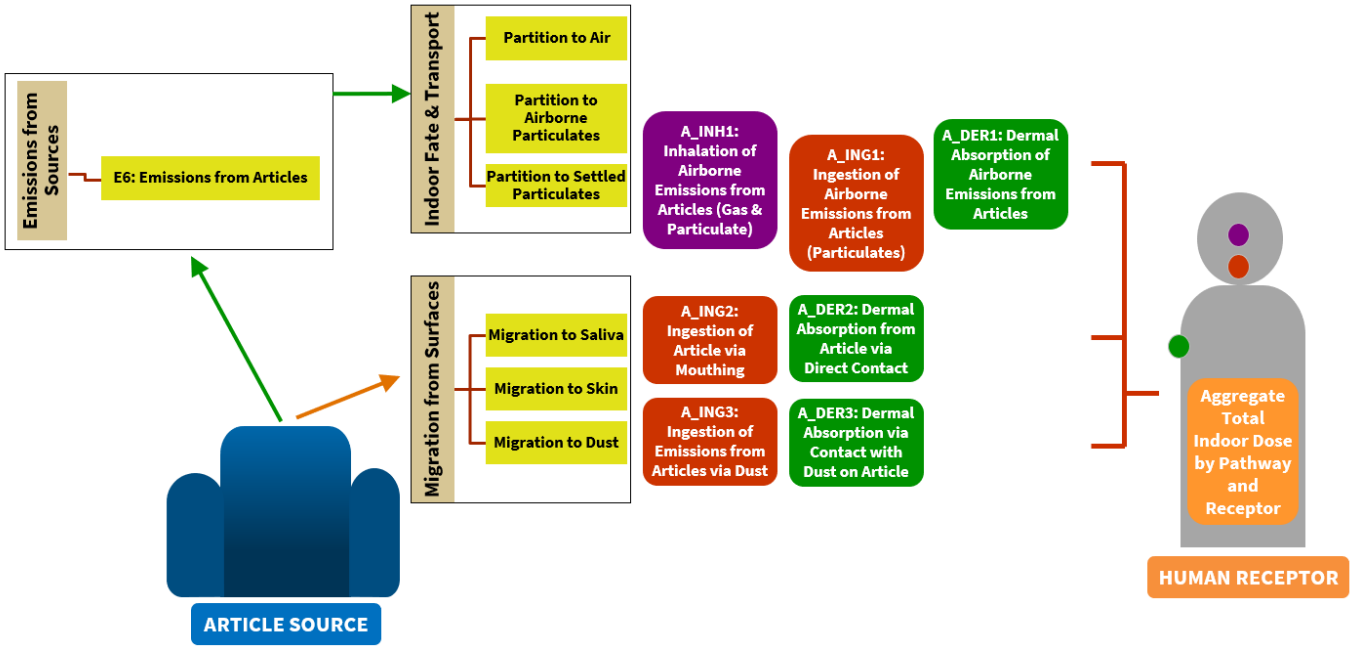


Figure 5-2. Consumer Pathways and Routes Evaluated in this Assessment

CEM 3.2 estimates acute dose rates and chronic average daily doses for inhalation, ingestion, and dermal exposures of consumer products and articles. CEM 3.2 gives exposure estimates for various lifestages, including the following:

- Adult (≥21 years)
- Youth 2 (16–20 years)
- Youth 1 (11–15 years)
- Child 2 (6–10 years)
- Child 1 (3–5 years)
- Infant 2 (1–2 years)
- Infant 1 (<1 year)
- Lifetime LADD/LADC (lifetime average daily dose/lifetime average daily concentration)

Exposure inputs for these various lifestages are provided in EPA's CEM, Version 3.0 Appendices (U.S. EPA, 2019d). CEM, 3.2 acute exposures are for an exposure duration of 1 day, and chronic exposures are for an exposure duration of 1 year. For more information on specific use patterns, and exposure inputs for populations, please see Appendix J A summary of key parameters used for the various consumer exposures scenarios are provided in Table 5-10.

5.1.2.2 Consumer COUs and Exposure Scenarios

Table 5-7. Summary of Consumer COUs, Exposure Scenarios, and Exposure Routes

Life Cycle Stage	Category	Subcategory	Consumer Use and Exposure Scenario	Form(s)	Routes Evaluated		
					Consumer User		
					Oral	Inhalation	Dermal
Consumer Use	Paints and coatings	Paints and coatings	N/A	Liquid			Q
				Vapor		Q	
				Mist			Q
Consumer Use	Furnishing, cleaning, treatment/care products	Fabric and textile products	Direct contact through use of products/articles containing TCEP	Air/Particulate		✓	
				Dust	✓		✓
				Article/Product Contact/Mouthing	✓		✓
Consumer Use	Furnishing, cleaning, treatment/care products	Foam seating and bedding products	Direct contact through use of products/articles containing TCEP	Air/Particulate		✓	
				Dust	✓		✓
				Article/Product Contact/Mouthing	✓		✓
Consumer Use	Construction, paint, electrical, and metal products	Building/construction materials – insulation	Direct contact through use of building/construction materials made containing TCEP	Air/Particulate		✓	
				Dust	✓		✓
				Article/Product Contact[a]			
		Building/construction materials – Wood and engineered wood products – Wood resin composites	Direct contact through use of wood and wood products made containing TCEP	Air/Particulate		✓	
				Dust	✓		✓
				Article/Product Contact/Mouthing	✓		✓
Disposal	Wastewater, liquid wastes, and solid wastes	Wastewater, liquid wastes, and solid wastes	Direct contact through use of products/articles containing TCEP	Article/Product Contact			Q
				Dust			Q
				Air/Particulate		Q	
			Long-term emission/mass-transfer through use of products containing TCEP	Dust			Q
				Air/Particulate		Q	

✓= Quantitatively assessed; Q = Qualitatively assessed

[a] Contact with the product is not expected (see Section 5.1.2.2.1).

Paints and Coatings – Including Those Found on Automotive Articles and Replacement Parts
Consumers are no longer able to purchase paints and coatings containing TCEP because their domestic retail production and manufacturing has ceased. It is possible that old paint cannisters stored in basements, crawlspaces, and/or garages may result in exposure to TCEP from off-gassing or during use by consumers.

Furthermore, the exposure to paints and coatings containing TCEP may occur via an article scenario in which the paint and coating has already been applied. There is a higher likelihood that older buildings and vehicles may have used TCEP-containing paints and coatings when the use of TCEP in consumer paints and coatings was more common. This dried scenario is like the acoustic ceilings/drywall scenario that was assessed for the building/construction materials COU. The exposure scenario of dried paints and coatings present in the indoor environment is qualitatively assessed and described in Section 5.1.2.2.5.

Due to limited reasonably available information regarding the use of paints and coatings and the uncertainties surrounding the weight fraction, activity and use patterns, and duration of use, EPA did not quantitatively assess the use of paints and coatings- including those found on automotive articles and replacement parts containing TCEP. See Section 5.3.2.2.2 for a qualitative assessment of consumer use of paints and coatings.

Fabric and Textile Products
In a study of the CHAMACOS cohort in California, Castorina et al. (2017) indicates that TCEP levels in dust are significantly associated with the presence of extremely worn carpets. Crowding, poor housing quality, and lack of maintenance by landlords can result in "extremely worn" carpets, warranting replacement. This suggests that individuals who are lower socioeconomic status may have increased exposure to TCEP due to the inability to replace extremely worn carpets.

Ionas et al. (2014) measured TCEP concentrations in different types (*e.g.*, hard plastic, soft plastic and rubber, wood and foam and textile) of children's toys in Antwerp, Belgium. This study reported a median TCEP concentration of 3 µg/g, mean of 10 µg/g, and maximum of 45 µg/g of TCEP in 36 percent in 25 foam and textile products sampled. For soft plastics and rubber products, a detection frequency of 42 percent in 31 toys with a median of 5 µg/g, mean of 10 µg/g, and maximum of 65 µg/g was reported. For hard plastic toys, the study author reported a detection frequency of 14 percent in 50 toys with a median of 2 µg/g, mean of 10 µg/g, and maximum of 25 µg/g. These mean concentrations correspond to a weight fraction of 0.001 percent.

EPA searched the Ecology Washington database (WSDE, 2023) in August 2022 and retrieved various information for fabric and textile products containing TCEP. The Ecology Washington database sampled for fabric and textile products that are likely to be mouthed or used by children under the age of three. The database had 67 products classified as textiles (synthetic fibers and blends), there were 2 detects at 0.01 percent and 1.3 percent. The 1.3 percent weight fraction was detected in the surface textile of a children's mini chair. The database indicated four detects of TCEP in carpet padding and rug mats. The weight fractions for these carpet products ranged from 0.01 to 0.02 percent.

Little additional information was found in the literature search on the percentages of TCEP in carpet back coating. A European patent has suggested that flame retardants may be generally used in carpet back coating at between 5 to 30 percent (Herrlich et al., 2013).

Two scenarios were modeled for the fabric, textile, and leather products not covered elsewhere—one for an outdoor children's play structure and one for carpet back coating. The CEM 3.2 scenario used for both scenarios were Fabrics: curtains, rugs, wall coverings (see Table 5-9). Values of 1.3 percent for fabric in children's play structure and 0.02 percent for the carpet back coating were selected for weight fractions for consumer modeling as these values are believed to be more representative of products readily available in the United States.

Foam Seating and Bedding Products

Various studies have reported the use of TCEP in furniture, automotive, and bedding foams (Maddela et al., 2020). In the early 2000s, Ingerowski et al. (2001) recorded TCEP in mattresses at 890 mg/kg (0.09%) in Germany. Ali et al. (2012) reported much lower concentrations of TCEP on mattresses surfaces (0.11 µg/g) in New Zealand. Two different case reports reported the acute death of dogs (one report on two pit bulls and one report on a German shepherd and a rottweiler) after chewing old automobile foams. The case studies found significant amounts (>2 ppm) of TCEP in their stomach contents (Lehner et al., 2010).

Fang et al. (2013) has measured another flame retardant (V6) at levels of 3.63 percent in couch foam and 5.3 percent in auto foams. TCEP has been reported to be an impurity in V6 of up to 14 percent. V6 is the dimer of TCEP, and it would be expected that TCEP would be an impurity of a V6 mixture. Hence, the product of these two values suggests TCEP is available in couch foams at 0.51 percent and in auto foams at 0.74 percent (Fang et al., 2013).

Hoehn et al. (2024) sampled foam seats from 51 vehicles (model year of 2015 or newer) across the United States. Only one sample detected of TCEP in auto foam however the authors did not report the weight fraction or amount detected. TCEP was detected in silicone samplers up to 391 mg/kg in the winter in 14 percent of vehicles, and up to 4,981 mg/kg in the summer in 44 percent of vehicles suggesting that seasonal variations and increases in temperature may lead to more expulsion of TCEP from materials inside a vehicle cabin (Hoehn et al., 2024).

Ingerowski et al. (2001) recorded TCEP in polyurethane soft foam at 19,800 mg/kg (1.98%), values from Fang et al. (2013) were selected for this furniture foam and auto foam scenarios as they were thought to be more current and representative of the U.S. population.

Bradman et al. (2014) sampled indoor dust concentrations between childcare facilities with and without foam napping equipment in 2011 in California. Median TCEP concentrations were significantly higher in rooms with foam napping equipment (median of 642.9 ng/g) vs. rooms without foam napping equipment (median of 260.9 ng/g).

For the foam toy block scenario, a weight fraction of 0.64 percent was calculated using information from Fang et al. (2013). This was based on the knowledge of 4.6 percent of V6 in polyurethane foam with an understanding that TCEP has been reported to be an impurity in V6 of up to 14 percent. Ionas et al. (2014) reports a lower weight fraction (0.001%) of TCEP in 25 foam and textile toys.

Building/Construction Materials – Insulation

TCEP has been reportedly used in building materials, including wood preservations coatings, glass fiber wallpapers, and acoustic ceilings (Maddela et al., 2020). High TCEP concentrations in dust (94 mg/kg) at a Swedish library were suggested to have been due the use of TCEP in the acoustic ceiling (Marklund et al., 2003).

Ingerowski et al. (2001) reported TCEP in polyurethane soft foam at 19,800 mg/kg (1.98%), and 68,000 mg/kg (6.8%) in acoustic ceilings. Kajiwara et al. (2011) recorded concentrations of TCEP in insulation boards of up to 10 ng/g in products purchased in Japan.

To assess the building/construction materials scenario, two exposure scenarios were run in CEM 3.2: roofing insulation (under the Plastic articles – foam insulation scenario) and acoustic ceiling (under the Drywall scenario). The weight fractions used for this modeling were 1.98 and 6.8 percent, respectively. These exposures scenarios measured the chronic release of TCEP from the roofing insulation and acoustic ceiling to the indoor air and indoor dust. They did not consider do-it-yourself (DIY) scenarios of a consumer installing these articles because they are no longer commercially available.

Wood and Engineered Wood Products

A case study reported neurotoxic signs (muscular weakness) experienced by a 5-year-old child after exposure to TCEP. It was postulated that the exposure was due to wood paneling that had been treated with a wood preserver coating containing 3 percent TCEP. However, TCEP in dust was not quantified. The study reported 600 mg/kg (0.06%) of TCEP in wood as cited in SCHER (2012). Ionas et al. (2014) reported a detection frequency of 25 percent in 8 wooden toys with a median of 4 µg/g, mean of 4 µg/g, and maximum of 5 µg/g, which corresponds to a mean weight fraction of 0.0004 percent. The products sampled in Ionas et al. (2014) were around 2007, with around half of the products coming from China.

Anecdotally, TCEP concentrations have been reported to be present in imported wooden TV stands. The photo below lists TCEP on a California Proposition 65 label on a wooden TV stand product imported to the United States from Malaysia (Figure 5-3).

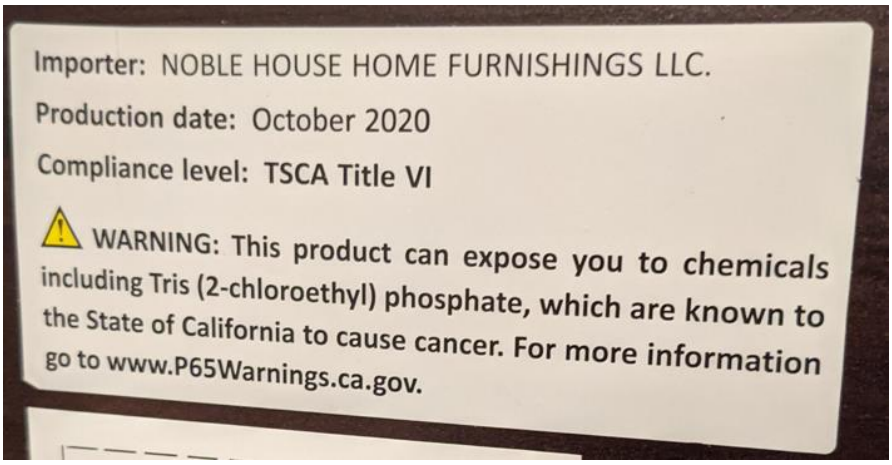


Figure 5-3. Photo of TCEP Label on Wooden Television Stand
Source: Photo by Yousuf Ahmad, U.S. EPA.

To assess the wood and engineered wood products scenario, two exposure scenarios for wood products (exposure from wood flooring and wooden TV stand) was run in CEM 3.2 utilizing the wood articles: hardwood floors, furniture predefined scenario with a weight fraction of 3 percent.

Wastewater, Liquid Wastes, and Solid Wastes

Consumers may be exposed to articles containing TCEP during the handling of disposal and waste. The removal of articles in DIY renovation scenarios may lead to direct contact with articles and the dust generated from the articles leading to consumer exposure. Due to the difficulties in quantifying consumer disposal of products containing TCEP, consumer disposal of TCEP was not quantitatively

assessed for this risk evaluation. Section 5.1.2.2.5 discusses the qualitative assessment for consumer disposals including the landfilling of building products and articles that contain TCEP.

5.1.2.2.1 Consumer Exposure Routes Evaluated

The COUs that were evaluated for TCEP were all articles. As such, the relevant underlying models utilized for TCEP included those listed in Table 5-8 below.

Table 5-8. CEM 3.2 Model Codes and Descriptions

Model Code	Description
E6	Emission from article placed in environment
A_INH1	Inhalation from article placed in environment
A_ING1	Ingestion after inhalation
A_ING2	Ingestion of article mouthed
A_ING3	Incidental ingestion of dust
A_DER1	Direct transfer from vapor phase to skin
A_DER2	Dermal dose from article where skin contact occurs
A_DER3	Dermal dose from skin contact with dust

CEM 3.2 contains 73 specific product and article categories and several generic categories that can be user-defined for any product and article. Table 5-9 presents a crosswalk between the COU subcategories with these predefined scenarios. In some cases, one COU mapped to multiple scenarios, and in other cases one scenario mapped to multiple COUs.

Table 5-9. Crosswalk of COU Subcategories, CEM 3.2 Scenarios, and Relevant CEM 3.2 Models Used for Consumer Modeling

TCEP COU Subcategory	Exposure Scenario	CEM 3.2 Scenario	E6	A_INH1	A_ING1	A_ING2	A_ING3	A_DER1	A_DER2	A_DER3
Fabric and textile products	Carpet back coating	Fabrics: curtains, rugs, wall coverings	●	●	●	●	●	●	●	●
	Textile for outdoor children's outdoor play structures	Fabrics: curtains, rugs, wall coverings	●	●	●	●	●	●	●	●
Foam seating and bedding product	Foam used in automobiles, foam used in living room furniture	Plastic articles: furniture (sofa, chairs, tables)	●	●	●	●	●	●	●	●
	Mattress	Plastic articles: mattresses	●	●	●	●	●	●	●	●
	Other foam objects (toy blocks)	Plastic articles: other objects with potential for routine contact (toys, foam blocks, tents)	●	●	●	●	●	●	●	●

TCEP COU Subcategory	Exposure Scenario	CEM 3.2 Scenario	E6	A_INH1	A_ING1	A_ING2	A_ING3	A_DER1	A_DER2	A_DER3
Building/construction materials – insulation	Insulation	Plastic articles: foam insulation	●	●	●		●	●		●
	Acoustic ceiling	Drywall (acoustic ceiling)	●	●	●		●	●		●
Building/construction materials – wood and engineered wood products – wood resin composites	Wood flooring	Wood articles: hardwood floors, furniture	●	●	●	●	●	●	●	●
	Wooden TV stand	Wood articles: hardwood floors, furniture	●	●	●	●	●	●	●	●

In total, the four COUs for TCEP were mapped to nine CEM 3.2 scenarios. Relevant consumer behavioral pattern data (*i.e.*, use patterns) and product-specific characteristics were applied to each of the scenarios. For more information on specific use patterns and product-specific characteristics please see Appendix J.

Inhalation, oral and dermal routes were evaluated for each of the article COUs. The article model Ingestion of article mouthed (A_ING2) was only evaluated for the COUs where it was anticipated that mouthing of the product would occur. For example, it is unlikely that a child will mouth roofing insulation or an acoustic ceiling, hence the A_ING2 Model was deemed inappropriate for estimating exposure for these COUs. The A_DER2 Model (dermal dose from article where skin contact occurs) was not used for estimating dermal exposure to roofing insulation and acoustic ceilings because dermal contact is not expected to occur for these articles.

The chronic and lifetime exposure estimates are the most relevant durations for consumer articles. Furnishings, building materials, and foam seating and bedding products are typically used over a longer time frame than other types of consumer products with direct applications (*e.g.*, household cleaners, solvents). The exposure scenario of relevance for consumers for building and construction materials, fabric and textile products, and foam seating and bedding products is that of a repeated exposure over a chronic duration. As such, the exposure estimates presented in the successive sections focus on the chronic average daily doses rather than the acute estimates. A summary of the acute, chronic, and lifetime exposure estimates are presented in Section 5.1.2.3 and further discussed in Appendix I.5.6.

The CEM, Version 3.2 was selected for the consumer exposure modeling as the most appropriate model to use based on the type of input data available for TCEP-containing consumer products. The advantages of using CEM to assess exposures to consumers and bystanders are as follows:

- CEM model has been peer-reviewed;
- CEM accommodates the distinct inputs available for the products containing TCEP; and
- CEM uses the same calculation engine to compute indoor air concentrations from a source as the higher-tier Multi-Chamber Concentration and Exposure Model (MCCEM) but does not require measured chamber emission values (which are not available for TCEP).

Consumer modeled exposure estimates were compared to the reported monitoring and reported modeled estimates for indoor air and indoor dust. Residential indoor air, indoor dust, and personal breathing zone data were identified and evaluated during systematic review (U.S. EPA, 2024r, x). Sections 3.4.1 and 3.4.2 provide a summary of the reported monitoring and reported modeled data in indoor air and indoor dust. A challenge in comparing EPA modeled exposures estimates with the reported monitoring and

modeled data in the literature is that EPA's modeled exposure estimates are by COU, whereas reported information in the literature are not typically specified by COU. For a characterization of model sensitivity and full exposure results, see Appendix J.

Table 5-10. Summary of Key Parameters for Article Modeling in CEM 3.2[a]

Consumer Exposure Scenarios	Initial Concentration of SVOC in Article (mg/cm^3)	Weight Fraction of Chemical (%)	Density Product/Article (g/cm^3)	Duration of Article Contact (min)	Frequency of Article Contact (Events/Day)	Surface Area of Article (m^2)	Thickness of Article Surface Layer (m)	Interzone Ventilation Rate (m^3/h)	Use Environment Volume (m^3)
Textile-outdoor play structures	4.03	1.30	0.31	180	1	17.8608	0.055	1E−30	492
Carpet back coating	4.00E−02	0.02	0.2	1,140	5	1.6	0.5	1E−30	492
Foam living room	2.22E01	0.74	0.03	600	10	0.4225	0.01	88.6092	50
Foam auto	2.22E01	0.74	0.03	600	1	0.4225	0.01	9.4872	2.4
Mattresses	2.67E−02	0.09	0.03	600	1	3.097	0.5	107.01	36
Other foam objects	1.92E−01	0.64	0.03	3.8	40	0.6606	0.01	108.978	50
Roofing insulation	5.94E−01	1.98	0.03	0	1	158	0.5	1E−30	492
Wood flooring	3.00E01	3.00	1	1,140	10	211	0.1	88.6092	50
Wood TV stand	3.00E01	3.00	1	120	10	1.38	0.1	88.6092	50
Acoustic ceiling	1.12E01	6.80	0.165	0	1	12.6	0.5	107.01	36

[a] For detailed information on selection of parameters refer to *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Consumer Exposure Modeling Inputs* (U.S. EPA, 2024e).

5.1.2.2.2 Inhalation Exposure Assessment

Due to its vapor pressure of 0.0613 mm Hg at 25 °C, it is expected that under non-heated conditions TCEP concentrations in air would be negligible. However, research has indicated that inhalation exposure of TCEP can be higher than dermal exposure (Ortiz Carrizales, 2018). In addition, concentrations of TCEP in the indoor air have been shown to be higher than ambient air concentrations (Wong et al., 2018). In general, concentrations of organophosphate flame retardants increase both indoors and outdoors during warmer seasons (Wang et al., 2019a).

Generally, TCEP release is higher at higher temperatures. However, the material to air coefficient (K_{MA}) values for TCEP have been shown to be similar at 35 and 55 °C. This implies that after reaching a certain temperature, TCEP emission rates increase in a K_{MA}-independent manner with further increase in temperature. The K_{MA} value at 23 °C for polyisocyanurate (PIR) foam was 7.76×10^{6} and for polyurethane foam (PUF) was 3.87×10^{6} (Maddela et al., 2020).

Due to its presence in particulates both less than and greater than 2.5 µm, and its presence in the gaseous phase, EPA expects both inhalation pathways (<2.5 µm deposits in lung and <0.1 µm deposits in alveolar region) and ingestion pathways (>2.5 µm deposits in mouth) to be contributors to TCEP exposure. See Section 3.3.1.2.1 for more details regarding the particle vs. gas phase distribution of TCEP. Consumer inhalation exposure to TCEP is expected through the direct inhalation of indoor air and dust. Table 5-11 below illustrates the steady state SVOC concentrations and respirable particle (RP) concentrations resulting from consumer exposure to articles containing TCEP.

Table 5-11. Steady State Air Concentrations and Respirable Particle of TCEP from Consumer Modeling (CEM 3.2)

COU Subcategory	Consumer Scenario	Air SVOC (mg/m^3)	Respirable Particles (µg/mg)
Fabric and textile products	Carpet back coating	3.06E–02	3.79E–02
	Textile-outdoor play structures	3.97	4.81
Foam seating and bedding product	Foam auto	1.04E–04	2.43E–05
	Foam living room	9.34E–06	3.33E–06
	Mattresses	4.45E–04	1.33E–04
	Other foam objects	1.26E–05	4.50E–06
Building/construction materials – insulation	Roofing insulation	9.32	1.13E01
	Acoustic ceiling	7.52E–01	2.25E–01
Building/construction materials – wood and engineered wood products – wood resin composites	Wood flooring	6.59	2.34
	Wood TV stand	4.31E–02	1.53E–02

The insulation scenario followed by the wood-resin scenario had the highest TCEP air concentrations (9.32 and 6.59 mg/m^3, respectively).

Exposures doses (chronic average daily inhalation doses [CADDs]) for all of the COU subcategories were estimated for the inhalation pathway via the following formulae) (A_INH1):

Equation 5-1.

$$CADD_{Air} = \frac{C_{gas\_avg} \times FracTime \times InhalAfter \times CF_1}{BW \times CF_2}$$

Equation 5-2.

$$CADD_{Particulate} = \frac{SVOCRP_{air\_avg} \times RP_{air\_avg} \times (1 - IF_{RP}) FracTime \times InhalAfter \times CF_1}{BW \times CF_2}$$

Equation 5-3.

$$CADD_{total} = CADD_{Air} + CADD_{Particulate}$$

Where:

$CADD_{Air}$	=	Potential Chronic Average Daily Dose, air (mg/kg-day)
$CADD_{Particulate}$	=	Potential Chronic Average Daily Dose, particulate (mg/kg-day)
$CADD_{total}$	=	Potential Chronic Average Daily Dose, total (mg/kg-day)
C_{gas_avg}	=	Average gas phase concentration (µg/m^3)
$SVOCRP_{air\_avg}$	=	Average SVOC in RP concentration, air (µg/mg)
RP_{air_avg}	=	Average RP concentration, air (mg/m^3)
IF_{RP}	=	RP ingestion fraction (unitless)
$FracTime$	=	Fraction of time in environment (unitless)
$InhalAfter$	=	Inhalation rate after use (m^3/hr)
CF_1	=	Conversion factor (24 hr/day)
BW	=	Body weight (kg)
CF_2	=	Conversion factor (1,000 µg/mg)

Exposures doses (Acute Dose rate ADRs) for all of the COU subcategories were estimated for the inhalation pathway via the following formulae (A_INH1):

Equation 5-4.

$$ADR_{Air} = \frac{C_{gas\_max} \times FracTime \times InhalAfter \times CF_1}{BW \times CF_2}$$

Equation 5-5.

$$ADR_{Particulate} = \frac{SVOCRP_{air\_max} \times RP_{air\_avg} \times FracTime \times InhalAfter \times CF_1}{BW \times CF_2}$$

Equation 5-6.

$$ADR_{total} = ADR_{Air} + ADR_{Particulate}$$

Where:

ADR_{Air}	=	Potential Acute Dose Rate, air (mg/kg-day)
$ADR_{Particulate}$	=	Potential Acute Dose Rate, particulate (mg/kg-day)
ADR_{total}	=	Potential Acute Dose Rate, total (mg/kg-day)

C_{gas_max}	=	Maximum gas phase concentration (µg/m^3)
$SVOCRP_{air\_max}$	=	Maximum SVOC in RP concentration, air (µg/mg)
RP_{air_max}	=	Maximum RP concentration, air (mg/m^3)
$FracTime$	=	Fraction of time in environment (unitless)
$InhalAfter$	=	Inhalation rate after use (m^3/hr)
CF_1	=	Conversion factor (24 hr/day)
BW	=	Body weight (kg)
CF_2	=	Conversion factor (1,000 µg/mg)

The ADR and CADD equations (Equation 5-1, Equation 5-2, Equation 5-3, Equation 5-4, Equation 5-5, and Equation 5-6) for A_INH1 consider both contributions from air and particulates. The average gas phase concentration is considered for CADDair, and the maximum gas phase concentration is considered for ADRair. The average SVOC in the RP concentration is considered for CADDparticulate, and the maximum SVOC in the RP concentration is considered for ADRparticulate. CADDair and CADDparticulate are summed to obtain CADDtotal. Similarly, ADRair and ADRparticulate are summed to get ADRtotal. The SVOC in the RP concentration is given in µg/mg and is multiplied by an average RP concentration (in mg/m^3).

Although the inhalation exposures to consumer articles containing TCEP are dominated by gas phase air concentrations vs. the SVOC in RP concentrations, EPA decided to include both in the inhalation exposure estimates. Therefore, EPA presented consumer inhalation values as doses (mg/kg-day), rather than concentrations (mg/m^3), because the dose values expressed as mg/kg-day included contributions from both the gas and particulate phases.

CEM 3.2 outputs include inhalation doses for all lifestages. Inhalation doses are calculated for lifestages by varying the BW and inhalation rate for the various population groups. These inhalation dose calculations are simplified and do not take into consideration lifestages differences in ventilation, anatomy, and metabolism. This risk evaluation presents one inhalation value (the adult value) by COU (see Table 5-15 and Table 5-16). Appendix J.1.3 presents the reported CEM inhalation doses with breathing weight and body weight adjustments for all lifestages.

A summary of the acute, chronic, and lifetime inhalation doses are presented in Section 5.1.2.3. Table 5-10 presents a summary of the key parameters used for consumer modeling with CEM 3.2. For more information on CEM 3.2, input parameters, sensitivity analysis, and assumptions used for consumer modeling please see Appendix J.

5.1.2.2.3 Dermal Exposure Assessment

Consumers may be dermally exposed to TCEP via skin contact with consumer articles, skin contact with dust generated from consumer articles, or the deposition of vapor generated from articles onto the skin. CEM 3.2 contains dermal modeling components that estimate absorbed dermal doses resulting from dermal contact with chemicals found in consumer products: Direct transfer from vapor phase to skin (A_DER1), dermal dose from article where skin contact occurs (A_DER2), and dermal dose from skin contact with dust (A_DER3). All three models were used to estimate exposure to articles containing TCEP, except for A_DER2, which was not used for the Building/construction materials – insulation COU because direct article contact with skin was not expected.

Contact of skin with articles drives the dermal exposure estimate in cases where contact is expected. Otherwise, skin contact with dust is the driver of dermal exposure. The following equation was used to calculate CADD for A_DER2:

Equation 5-7.

$$CADD = \frac{C_{art} \times \frac{SA}{Bw} \times l \times FR_{abs\_art} \times ED_{cr}}{AT_{cr}}$$

Where:

$CADD$	=	Potential Chronic Average Daily Dose (mg/kg-day)
C_{art}	=	Chemical concentration in article (mg/cm^3)
SA/BW	=	Surface area to body weight ratio (cm^2/kg)
$FRabs\_art$	=	Fraction absorbed (unitless)
$EDcr$	=	Exposure duration, chronic (years)
$ATcr$	=	Averaging time, chronic (years)
l	=	Average distance a diffusing molecule travels per contact (cm/day)

Many of these parameters are calculated within CEM 3.2. The parameter *l* is a function of duration of article contact (min/day). A_DER3 has a similar formula:

Equation 5-8.

$$CADD = \frac{Dust_{cr\_wgt} \times \frac{SA}{BW} \times AF \times FA \times EvD \times ED_{cr}}{CF_1 \times AT_{cr}}$$

Where:

$Dust_{cr\_wgt}$	=	Chronic weighted dust concentration (µg/mg)
AF	=	Adherence factor of dust to hand (mg/cm^2-event)
FA	=	Fraction absorbed (unitless)
EvD	=	Frequency of article contact per day (events/day)
CF_1	=	Conversion factor (insert value)

Compared to A_DER2, this formula substitutes a chronic weighted dust concentration for the chemical concentration and replaces the *l* term with an adherence factor (*AF)* and frequency of article contact (*EvD).*

A key parameter in estimating results for A_DER2 and A_DER3 is fraction absorbed (F_{abs}). While the duration of interaction with materials that contain TCEP may be shorter than the duration that was tested in the dermal absorption study (*i.e.,* a 24-hour exposure), EPA cannot assume that consumers would immediately wash their hands following contact with treated objects (*e.g.,* carpets). Therefore, the dose that is deposited on the skin during an activity would be expected to remain on the skin until the skin is eventually washed. As a result, EPA applied a 24-hour value for fraction absorbed (35.1%) from Abdallah et al. (2016) to all consumer dermal exposures scenarios.

Table 5-12 provides the chronic dermal doses from each of the underlying models in CEM 3.2 and for adults and children 3 to 6 years of age. All lifestages were analyzed. For more information on the consumer dermal exposure inputs, equations, results (for all lifestages) and sensitivity analysis please see Appendix J and EPA's CEM 3.0 Appendices (U.S. EPA, 2019d).

Table 5-12. Chronic Dermal Average Daily Doses (mg/kg-day) of TCEP from Consumer Article Modeling for Adults and Children 3 to 6 Years of Age (CEM 3.2)

COU Subcategory	Consumer Scenario	Lifestage	A_DER1 Vapor to Skin	A_DER2 Skin Contact	A_DER3 Skin Contact with Dust	Total Chronic Dermal ADD
Fabric and textile products	Carpet back coating	Adult	2.29E–07	3.16E–04	3.45E–05	3.50E–04
		Child	3.68E–07	5.07E–04	5.53E–05	5.63E–04
	Textile – outdoor play structures	Adult	2.97E–06	1.26E–02	8.41E–04	1.35E–02
		Child	4.78E–06	2.03E–02	1.35E–03	2.17E–02
Foam seating and bedding product	Foam auto	Adult	3.88E–10	5.65E–03	1.78E–08	5.65E–03
		Child	6.44E–10	9.38E–03	2.96E–08	9.38E–03
	Foam living room	Adult	6.95E–10	1.26E–02	2.16E–08	1.26E–02
		Child	1.16E–09	2.10E–02	3.60E–08	2.10E–02
	Mattresses	Adult	3.32E–08	1.54E–03	3.99E–07	1.54E–03
		Child	5.51E–08	2.55E–03	6.64E–07	2.55E–03
	Other foam objects	Adult	9.40E–10	8.69E–04	1.16E–07	8.69E–04
		Child	1.56E–09	1.44E–03	1.92E–07	1.44E–03
Building/construction materials – insulation	Roofing insulation	Adult	3.49E–05	0.00	9.98E–04	1.03E–03
		Child	5.61E–05	0.00	1.61E–03	1.66E–03
	Acoustic ceiling	Adult	5.64E–06	0.00	6.79E–05	7.36E–05
		Child	9.05E–06	0.00	1.09E–04	1.18E–04
Building/construction materials – wood and engineered wood products – wood resin composites	Wood flooring	Adult	4.94E–05	2.37E–01	4.05E–03	2.41E–01
		Child	7.93E–05	3.80E–01	6.51E–03	3.87E–01
	Wood TV stand	Adult	3.23E–07	7.68E–02	2.65E–05	7.69E–02
		Child	5.19E–07	1.23E–01	4.26E–05	1.23E–01

5.1.2.2.4 Oral Exposure Assessment

Consumers may be exposed to TCEP via transfer from hand to mouth, ingestion after inhalation, mouthing of articles, and the incidental ingestion of dust generated from consumer articles. CEM 3.2 contains an ingestion modeling component that estimates ingestion doses resulting from consumer products: ingestion after inhalation (A_ING1), ingestion of article mouthed (A_ING2), and incidental ingestion from dust (A_ING3). All three models were used to estimate exposure to articles containing TCEP, except for A_ING2, which was not used for the building/construction materials COU as mouthing of the article was not expected.

Mouthing is a particular important route for estimating exposure to children and infants who may have higher exposures to toys and children's products. CEM 3.2 has four choices for mouthing scenarios: 0, 1 (low), 10 (medium), and 50 (high) cm^2. The high mouthing input was selected for outdoor play structures and other foams (toy blocks) because these are children's products. The medium values were selected for carpet back coating, wood flooring, wooden TV stand, foam furniture in the living room, foam seat in an automobile, and the mattress scenarios.

The following equation was used to calculate CADD for A_ING2:

Equation 5-9.

$$CADD = \frac{MR \times CA \times D_m \times ED_{cr} \times CF_1}{BW \times AT_{cr} \times CF_2}$$

Where:

$CADD$	=	Potential Chronic Average Daily Dose (mg/kg-day)
MR	=	Migration rate of chemical from article to saliva (mg/cm^2/hr)
CA	=	SA/BW= Surface area to body weight ratio (cm^2/kg)
D_m	=	Duration of mouthing (min/hr)
ED_{cr}	=	Exposure duration, chronic (years)
CF_1	=	Conversion factor (24 hr/day)
CF_2	=	Conversion factor (60 min/hr)
AT_{cr}	=	Averaging time, chronic (years)
BW	=	Body weight (kg) = Conversion factor (60 min/hr)

The following equation was used to calculate CADD for A_ING3:

Equation 5-10.

$$CADD = \frac{Dust_{cr\_wgt} \times FracTime \times DustIng}{BW \times CF}$$

Where:

$CADD$	=	Potential Chronic Average Daily Dose (mg/kg-day)
$Dustcr\_wgt$	=	Chronic weighted dust concentration (µg/mg)
$FracTime$	=	Fraction of time in environment (unitless)
$DustIng$	=	Dust ingestion rate (mg/day)
BW	=	Body weight (kg)
CF	=	Conversion factor (1,000 µg/mg)

The chronic weighted dust concentration was weighted between the dust available from the respirable portion, floor dust, and abraded particles.

Table 5-13 presents the chronic ingestion doses from each of the underlying models in CEM 3.2 and for adults and infants 1 to 2 years of age. All lifestages were analyzed. For more information on the consumer dermal exposure inputs, equations, results (for all lifestages) and sensitivity analysis please see Appendix J and EPA's CEM 3.0 Appendices (U.S. EPA, 2019d).

Table 5-13. Chronic Ingestion Average Daily Doses (mg/kg-day) of TCEP from Consumer Article Modeling for Adults and Infants 1 to 2 Years of Age (CEM 3.2)

COU Subcategory	Consumer Scenario	Lifestage	A_ING1 Ingestion after Inhalation	A_ING2 Mouthing	A_ING3 Ingestion of Dust	Total Chronic Ingestion ADD
Fabric and textile products	Carpet back coating	Adult	2.23E–08	0	2.48E–05	1.43E–03
		Infant	8.12E–08	2.70E–01	3.15E–04	1.82E–02
	Textile – outdoor play structures	Adult	2.80E–06	0	3.02E–04	1.44E–02
		Infant	1.02E–05	2.70E–01	3.84E–03	1.83E–01
	Foam auto	Adult	5.78E–10	0	3.22E–10	3.97E–04

COU Subcategory	Consumer Scenario	Lifestage	A_ING1 Ingestion after Inhalation	A_ING2 Mouthing	A_ING3 Ingestion of Dust	Total Chronic Ingestion ADD
Foam seating and bedding product		Infant	2.11E–09	2.70E–01	4.09E–09	5.04E–03
	Foam living room	Adult	6.90E–12	0	7.84E–10	7.94E–03
		Infant	2.51E–11	2.70E–01	9.95E–09	1.01E–01
	Mattresses	Adult	3.05E–10	0	1.45E–07	9.66E–04
		Infant	1.11E–09	2.70E–01	1.84E–06	1.23E–02
	Other foam objects	Adult	8.81E–12	0	1.05E–09	6.87E–03
		Infant	3.21E–11	1.35	1.33E–08	8.72E–02
Building/construction materials – insulation	Roofing insulation	Adult	6.61E–06	0	7.19E–03	2.12E–02
		Infant	2.41E–05	0	9.13E–02	2.70E–01
	Acoustic ceiling	Adult	5.01E–07	0	2.44E–04	4.01E–01
		Infant	1.83E–06	0	3.10E–03	5.10
Building/construction materials – wood and engineered wood products – wood resin composites	Wood flooring	Adult	4.24E–06	0	1.46E–03	1.07
		Infant	1.55E–05	2.70E–01	1.85E–02	1.36E01
	Wood TV stand	Adult	2.78E–08	0	9.53E–06	1.07
		Infant	1.01E–07	2.70E–01	1.21E–04	1.36E01

For children and infants, mouthing was the dominant route of exposure. For teenagers and adults, ingestion of dust was the dominant route of exposure as no mouthing of the consumer articles are expected.

Sensitivity analyses indicated that "Area of article mouthed" was the driver for the mouthing estimates. The area of article mouthed was more important for the ingestion estimate compared to the initial concentration of the SVOC in the article, the density of the article, the surface area of the article, and the duration of article contact.

For more information on the consumer ingestion exposure inputs, equations, results (for all lifestages) and sensitivity analysis please see Appendix J and EPA's CEM, Version 3.2 User Guide and Appendices (U.S. EPA, 2023).

5.1.2.2.5 Qualitative Exposure Assessment

Paints and Coatings

A review of literature reporting TCEP used outside the US from the early 2000s provides some evidence of the use of TCEP in paints and coatings. Ingerowski et al. (2001) detected TCEP in 85 percent of 983 household products in Germany and reported TCEP in wood preservation coatings at a concentration of 10,000 mg/kg (1.0%). Haumann and Thumulla (2002) detected TCEP in paints at a maximum of 840 mg/kg (0.084%) in Germany prior to 2002 (TERA, 2013).

Table 5-14 is a summary of the information gathered for the commercial use of paints and coatings COU. This data indicate TCEP is used at a high-end of 25 percent in commercial paints and coatings.

Table 5-14. Summary of Commercial Paints and Coatings Concentrations and Density of TCEP

Paint Products	TCEP Concentration (Mass Fraction)		Product Density (kg/m^3)	
	Low-End	High-End	Low-End	High-End
7 Industrial and commercial paints and coatings	0.1%	25%	1,000	1,490

Consumer exposures to articles (including automotive articles and replacements parts) that have been coated with TCEP-containing paints and coatings will mimic consumer exposures from the article scenarios (*e.g.*, acoustic ceilings, wood resin products). The paints and coatings scenario within CEM 3.2 is for the active application of paints and coatings in a product scenario. Thus, for this risk evaluation, the dried paints and coatings scenario can be considered a part of the quantitatively assessed articles scenarios.

The maximum weight fractions (25%) presented in Table 5-14. are up to 4 times higher than the weight fractions available for consumer articles (6.8%). This suggests that commercial and industrial products contain higher levels of TCEP than products and articles available for the consumer market. Although it is possible for consumers to obtain commercial paints and coatings for applications, it is not probable, and EPA has determined that it is not likely for consumers to obtain TCEP containing paints and coatings products that are available for commercial applications.

The dermal route is the most important route to consider for exposures to paints and coatings containing TCEP. The occupational dermal exposure estimates for workers using TCEP-containing paints and coatings are presented in Section 5.1.1.3. The commercial use of paints and coatings results in a high-end exposure estimate of 8.02 mg/day and a central tendency estimate of 1.48 mg/day (see Table 5-6). This scenario is based on a spray application scenario under working conditions for non-occluded scenarios.

Differences in the occupational and consumer exposure scenarios of paints and coatings provide context to this qualitative assessment. Products available for the industrial and commercial market are formulated differently than for consumers. Moreover, workers work with industrial grade formulations that have higher concentrations of TCEP and may be exposed to paints and coatings containing TCEP under exposures scenarios that result in higher exposures (*e.g.*, spray application vs. typical domestic painting).

Wastewater, Liquid Wastes, and Solid Wastes

At the end of their life cycles, consumer articles may be disposed of in municipal solid waste landfills, construction, and demolition landfills, or undergo incineration. Groundwater monitoring data in Section 3.3.3.5 suggests that TCEP can migrate from municipal unlined landfills to groundwater via landfill leachate. Water discharges from laundered clothing that picks up TCEP may also be a potential source of TCEP to surface waters. The successive sections attempt to describe TCEP exposures that may be a result of the disposal, demolition and removal of household articles and dust containing TCEP. Due to the difficulties in source attribution, EPA was unable to relate consumer COUs to these TCEP exposures. However, they are qualitatively discussed to capture additional ways individuals may be exposed to TCEP via consumer articles.

Wastewater: Section 3.3.2.7 states that laundry wastewater may contribute to elevated environmental surface water concentrations of TCEP. Clothing has been hypothesized to act as a sink for TCEP to transfer organophosphate esters from the indoor environment to surface waters via wastewater from domestic and commercial laundry sources (Schreder and La Guardia, 2014). A study investigating the relationship between the fate of phthalates and flame retardants transferring from clothing to laundry wastewater found that chemicals with a log K_{OW} less than 4 showed a greater than 80 percent release to laundry water, whereas chemicals with a log K_{OW} greater than 6 only showed less than 10 percent release to laundry wastewater (Saini et al., 2016). Furthermore, these findings also suggest that dermal exposure to TCEP may be enhanced from clothing to sweat (Saini et al., 2016).

TCEP was among the 10 most frequently found compounds, detected at 61.9 percent in wastewater samples (maximum of 0.7 µg/L), in a study that collected wastewater from multiple sites in Research Triangle Park area of North Carolina between 2002 and 2005 (Giorgino et al., 2007). Flame retardants were measured primarily at sites downstream from municipal wastewater discharges and at a site downstream from an industrial fire. TCEP samples were detected in four of eight sites, and at three of three sites that had major upstream wastewater discharges. A possible explanation for TCEP detection at the one other site (without an upstream wastewater discharge) was that a fire at an industrial cleaning-supply warehouse occurred upstream a few months before the sampling event. It is believed that water applied to control the fire had entered the nearby tributary. In addition, two of these sites near wastewater discharges are also located near state recreation areas where public facilities, campgrounds, dump stations, swimming beaches and boating access are available (Giorgino et al., 2007).

Solid Wastes: A CDC NIOSH report evaluated the occupational exposure to flame retardants at four gymnastics studios in the mid-2010s. The researchers sampled old foam blocks, mats, padded equipment and employees via hand wipe samples before and after work. TCEP was detected at 343 ng/ft^2 at one of the gymnastics studios in June 2014, but was not detected in April 2015 after the replacement of new foam blocks (Broadwater et al., 2017). A similar study measured 1.6 to 1.9 µg/g dry weight of TCEP in polyurethane foam blocks in a Seattle gym. TCEP was detected at a mean concentration of 1.18 µg/g dry weight in gym dust concentrations across four gyms. Dust samples were collected from the homes of four gym instructors. TCEP was found at a mean concentration of 2.5 µg/g dry weight at the instructors' residences (La Guardia and Hale, 2015).

A study from the Sierra Nevada foothills suggests that the presence of TCEP on the surfaces of ponderosa pine needles can be explained by the aerial transport and deposition from nearby point sources where chemicals were released during the incineration of plastic waste articles (Aston et al., 1996).

Recycling: TCEP is not typically used in electronics (Stapleton et al., 2011). A CDC NIOSH report assessed employee exposure to flame retardants at an electronics recycler in November 2016 and February 2017. TCEP was detected in surface wipe samples at the disassembly workstation at 154 ng/100 cm^2. The report indicated the workers were incorrectly wearing N95 respirators and were dry sweeping. To prevent exposure to airborne TCEP dust particles, the report recommends prohibiting dry sweeping to clean work areas (Grimes et al., 2019).

Landfills: The demolition and removal of consumer articles may result in exposures to TCEP. Construction waste and old consumer products can be disposed of in municipal solid waste landfills and construction and demolition landfills. Section 3.3.3.8 models the resulting groundwater concentration that may occur from leaching of TCEP from landfills. Section 3.3.3.63.3.3.8 highlights suspected leaching of TCEP from nearby landfills (Norman Landfill, Himco Dump, and Fort Devens) (Buszka et

al., 2009; Barnes et al., 2004; Hutchins et al., 1984). The Himco Dump is a closed unlicensed landfill that included a 4-acre construction debris area. EPA issued a notice in the Federal Register finalizing the deletion of part of the Himco Dump Superfund site from the National Priorities List (NPL). The Indiana Department of Environmental Management (IDEM) formally concurred with EPA's proposal on January 26, 2022, and EPA proposed the site for partial deletion in March, 2022. Fort Devens is also an EPA superfund site, a former army instillation site that was established in 1917 and closed in 1996, is also a closed superfund sites. TCEP was detected throughout the entire length of a leachate plume near a municipal landfill (subtitle D) near Norman, Oklahoma (Barnes et al., 2004). Leachate samples from landfill sites in Japan detected TCEP at ranges from 4.1 to 5430 mg/mL. This study suggested that the origin may be due to plastic wastes (Yasuhara, 1995).

Moran et al. (2023) described that open-air landfills and local waste disposal practices, rather than former defense sites, may be an important source of atmospheric TCEP in the Arctic. A study around Troutman Lake, AK indicated higher deposition values on the north side of the lake, which corresponded with locations of a landfill (900 m from the NW sampling site and 1,400 m from the NE sampling site). The north-west side of Troutman Lake had the highest deposition with a magnitude of 1,300 ng/m^2/day. Troutman lake lies directly south of the village of Gambell on the NW corner of Sivuqaq. The island is home to the Sivuqaq Yupik people who practice a traditional subsistence lifestyle. Although the lake was a former chemical disposal site used by the military, Moran et al. (2023) suggests that the military site was closed (1950) prior to the use of TCEP.

Without a full characterization of non-hazardous landfill (*e.g.*, Norman Landfill) conditions and historical wastes (*e.g.*, Himco dump and Ft. Devens) around the country, EPA is uncertain how often contaminant migration occurs given modern practices of non-hazardous landfill and historical site management. However, the possibility of exposure to TCEP after the release from disposal of consumer wastes exists.

5.1.2.3 Summary of Consumer Exposure Assessment

Table 5-15. Summary of Acute Daily Rate of Consumer Articles Modeled with CEM 3.2

COU Subcategory	Consumer Exposure Scenario	Lifestage	Exposure Dose (mg/kg/day)		
			Oral	Inhalation	Dermal
Fabric and textile products	Carpet back coating	Adult	2.43E–04	5.12E–02	6.56E–04
		Children	4.00E–02	N/A	1.05E–03
	Textile for children's outdoor play structures	Adult	3.84E–03	1.06	2.33E–02
		Children	9.12E–02	N/A	3.73E–02
Foam seating and bedding product	Foam automobile	Adult	2.82E–07	2.90E–04	5.65E–03
		Children	3.66E–02	N/A	9.39E–03
	Foam living room	Adult	1.86E–07	5.19E–04	1.26E–02
		Children	3.66E–02	N/A	2.10E–02
	Mattress	Adult	3.50E–06	3.15E–03	1.55E–03
		Children	3.66E–02	N/A	2.57E–03
	Foam – other (toy block)	Adult	2.47E–07	7.02E–04	8.96E–04
		Children	1.83E–01	N/A	1.49E–03
Building/construction materials − insulation	Roofing insulation	Adult	8.87E–02	2.32E01	1.29E–02
		Children	1.27	N/A	2.07E–02
	Acoustic ceiling	Adult	5.91E–03	5.31	1.90E–03
		Children	8.45E–02	N/A	3.05E–03
Building/construction materials − wood and engineered wood products − wood resin composites	Wood flooring	Adult	1.07E–01	1.80E02	5.42E–01
		Children	1.57	N/A	8.71E–01
	Wooden TV stand	Adult	7.03E–04	1.18	7.88E–02
		Children	4.66E–02	N/A	1.27E–01

Table 5-16. Summary of Chronic Average Daily Doses of Consumer Articles Modeled with CEM 3.2

COU Subcategory	Consumer Exposure Scenario	Lifestage	Exposure Dose (mg/kg/day)		
			Oral	Inhalation	Dermal
Fabric and textile products	Carpet back coating	Adult	2.48E–05	4.67E–03	3.50E–04
		Children	3.69E–02	N/A	5.63E–04
	Textile for outdoor children's outdoor play structures	Adult	3.05E–04	6.05E–02	1.35E–02
		Children	4.09E–02	N/A	2.17E–02
Foam Seating and Bedding Product	Foam automobile	Adult	9.01E–10	7.95E–07	5.65E–03
		Children	3.66E–02	N/A	9.38E–03
	Foam living room	Adult	7.90E–10	1.42E–06	1.26E–02
		Children	3.66E–02	N/A	2.10E–02
	Mattress	Adult	1.45E–07	6.79E–05	1.54E–03
		Children	3.66E–02	N/A	2.55E–03
	Foam – other (toy block)	Adult	1.05E–09	1.93E–06	8.69E–04
		Children	1.83E–01	N/A	1.44E–03
Building/construction materials − insulation	Roofing insulation	Adult	7.20E–03	1.42	1.03E–03
		Children	1.03E–01	N/A	1.66E–03
	Acoustic ceiling	Adult	2.45E–04	1.15E–01	7.36E–05
		Children	3.50E–03	N/A	1.18E–04
Building/construction materials − wood and engineered wood products − wood resin composites	Wood flooring	Adult	1.46E–03	1.01	2.41E–01
		Children	5.75E–02	N/A	3.87E–01
	Wooden TV stand	Adult	9.56E–06	6.57E–03	7.69E–02
		Children	3.67E–02	N/A	1.23E–01

Table 5-17. Summary of Lifetime Average Daily Doses of Consumer Articles Modeled with CEM 3.2

COU Subcategory	Consumer Exposure Scenario	Exposure Dose (mg/kg/day)		
		Oral	Inhalation	Dermal
Fabric and textile products	Carpet back coating	1.63E–02	6.04E–03	3.77E–04
	Textile for outdoor children's outdoor play structures	1.70E–02	7.83E–02	1.45E–02
Foam seating and bedding product	Foam automobile	1.63E–02	1.03E–06	6.19E–03
	Foam living room	1.63E–02	1.84E–06	1.38E–02
	Mattress	1.63E–02	8.78E–05	1.68E–03
	Foam – other (toy block)	8.14E–02	2.49E–06	9.53E–04
Building/construction materials − insulation	Roofing insulation	5.26E–03	1.04	7.55E–04
	Acoustic ceiling	5.83E–04	1.48E–01	7.92E–05
Building/construction materials − wood and engineered wood products − wood resin composites	Wood flooring	1.98E–02	1.30	2.59E–01
	Wooden TV stand	1.63E–02	8.50E–03	8.27E–02

5.1.2.4 Weight of Scientific Evidence Confidence for Consumer Exposure

The overall exposure confidence for the various consumer scenarios ranged from slight to moderate. Slight confidence in the exposure estimates were mainly due to data uncertainties. Information on article weight fraction was sparse, and it was unclear whether many of the literature values were still relevant for articles used today. EPA considered a worst-case approach to consumer weight fraction and varied this parameter in the sensitivity analysis as reported in Appendix J (Consumer Exposure). Information on exposure scenarios (*e.g.*, mouthing durations, use durations, frequency of contacts per day) were also limited. Furthermore, limited monitoring data were available to corroborate the modeled consumer exposure estimates and validate current use of TCEP in consumer articles. In addition, there are uncertainties related to CEM 3.2 modeling approaches (*e.g.*, deterministic vs. stochastic approaches, background concentrations, assumptions for dermal absorption parameters).

Table 5-18. Weight of Scientific Evidence Confidence for Chronic Consumer Exposure Modeling Scenarios

Consumer COU			Confidence in Model Used[a]	Confidence in Model Default Values[b]	Confidence in User-Selected Varied Inputs[c]					Monitoring Data	Overall Exposure Confidence[i]
Category	Subcategory	Form			Density Used[d]	Use Duration[e]	Weight Fraction[f]	Room of Use[g]	Dermal K_p, F_{abs}, Mouthing[h]		
Fabric and textile products	Carpet back coating	Article	++	+++	++	+++	++	+++	+	Limited	Moderate
	Textile for outdoor children's outdoor play structures	Article	+++	+	++	++	++	++	++	Limited	Moderate
Building/ construction materials – insulation	Roofing insulation	Article	++	++	+	N/A	+	+++	+	None	Slight
	Acoustic ceiling	Article	+	++	+	N/A	+	++	+	Limited	Slight
Foam seating and bedding product	Foam automobile	Article	+++	+++	++	++	++	+++	+	Limited	Moderate
	Foam living room	Article	+++	+++	++	+++	++	+++	++	Limited	Moderate
	Mattress	Article	+++	+++	++	+++	+	+++	+	None	Slight
	Foam – other (toy block)	Article	+++	+++	++	++	+	+++	++	None	Slight
Building/ construction materials – wood and engineered wood products – wood resin composites	Wood flooring	Article	+++	+++	++	+++	+	+++	+	None	Slight
	Wooden TV stand	Article	+++	+++	++	++	+	+++	+	Limited	Moderate

Consumer COU			Confidence in Model Used[a]	Confidence in Model Default Values[b]	Confidence in User-Selected Varied Inputs[c]					Monitoring Data	Overall Exposure Confidence[i]
Category	Subcategory	Form			Density Used[d]	Use Duration[e]	Weight Fraction[f]	Room of Use[g]	Dermal K_p, F_{abs}, Mouthing[h]		

[a] Confidence in Model Used considers whether model has been peer reviewed, as well as whether it is being applied in a manner appropriate to its design and objective. The model used (CEM 3.2) has been peer reviewed, is publicly available, and has been applied in a manner intended, to exposures associated with uses of household products and/or articles. Medium was selected for the carpet-back coating scenario and a roofing insulation scenario because of uncertainties surrounding the barrier layers. Low was selected for acoustic ceiling because the related CEM scenario was Drywall, and these products have different product characteristics.

[b] Confidence in Model Default Values considers default value data source(s) such as building and room volumes, interzonal ventilation rates, and air exchange rates. These default values are all central tendency values (*i.e.,* mean or median values) sourced from EPA's *Exposure Factors Handbook* (U.S. EPA, 2017d, 2011a). Low was selected for outdoor play structures, as there were uncertainties on the area volumes related to this scenario.

[c] Confidence in User-Selected Varied Inputs considers the quality of their data sources, as well as relevance of the inputs for the selected consumer condition of use.

[d] Density Used was primarily available for product descriptions. (Westat, 1987)

[e] Use Duration is primarily sourced from the EPA's *Exposure Factors Handbook* and by the judgment of the exposure assessor.

[f] Weight fraction of TCEP in articles was sourced from the available literature and database values.

[g] Room of use (zone 1 in modeling) is informed by professional judgment of the exposure assessor based on the article scenario. The reasonableness of these judgments is considered in the reported confidence ratings.

[h] The dermal permeability coefficient (K_p) used (0.022 cm/hr) and fraction absorbed (F_{abs}) used (35.1%) was derived from a study of TCEP tested on human ex vivo skin (Abdallah et al., 2016). Frequency of mouthing (Low, Medium, High) was estimated using the assessors' judgment when considering the exposure scenario. Literature values override (Poet et al., 2000) CEM 3.2 default values for fraction absorbed.

[i] + + + Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the exposure estimate.

+ + Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize exposure estimates.

+ Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.

5.1.2.4.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the Consumer Exposure Assessment

EPA recognizes the need to include an uncertainty analysis. One important distinction for such an analysis is variability vs. uncertainty—both aspects need to be addressed. Variability refers to the inherent heterogeneity or diversity of data in an assessment. It is a quantitative description of the range or spread of a set of values and is often expressed through statistical metrics, such as variance or standard deviation, which reflect the underlying variability of the data. Uncertainty refers to a lack of data or an incomplete understanding of the context of the risk evaluation decision.

Variability cannot be reduced but can be better characterized. Uncertainty can be reduced by collecting more or better data. Quantitative methods to address uncertainty include non-probabilistic approaches such as sensitivity analysis and probabilistic or stochastic methods. Uncertainty can also be addressed qualitatively by including a discussion of factors such as data gaps and subjective decisions or instances where professional judgment was used.

Uncertainties associated with approaches and data used in the evaluation of consumer exposures are described below. A sensitivity analysis was conducted for the following COUs to understand the drivers for the inhalation, ingestion, and dermal estimates (Table 5-19).

Table 5-19. Sensitivity Analysis for Chronic Consumer Exposure Modeling Scenarios

Consumer COU		User-Selected Varied Inputs[a]				Results
Subcategory	**Consumer Exposure Scenario**	**Initial SVOC Concentration in Article (mg/cm^3)[b]**	**Mouthing Duration (min)[c]**	**Surface Area of Article (m^2)**	**Events per day (n)**	
Fabric and textile products	Textile for outdoor children's play structures	4.03 0.93 0.30	High (8.4/7/10) Low (2.3/3.65/5)	–	–	Mouthing duration is a driver of ingestion exposures.
Building/ construction materials – insulation	Roofing insulation	0.594 0.180 0.06	–	–	–	SVOC concentration is a driver of inhalation exposures.
Building/ construction materials – wood and engineered wood products – wood resin composites	Wood flooring	30 12	–	211 105	10 5	SVOC concentration is a driver of dermal exposures. Surface area of the article and Events per day (n) influence the dermal exposure estimates

[a] User selected inputs were varied for each of the listed consumer exposure scenarios.

[b] Initial SVOC concentration in article is a function of the product weight fraction and article density.

[c] The high mouthing duration defaults in CEM 3.0 were 10 min/event for an infant (< 1 year of age), 7 min/event for an infant aged 1–2 years, and 8.4 min/event for a child 3–5 years. EPA modified the mouthing durations to 5 min/event for infants < 1 years, 3.65 min/event for 1–2 years, and 2.3 min/event for children 3–5 years to test the sensitivity of this parameter.

A clear finding of the sensitivity analysis indicated that the initial SVOC concentration (a product of the density and weight fraction) was a significant driver in the inhalation and dermal exposure estimates for all scenarios. The initial SVOC concentration was also relevant for the ingestion estimate for the insulation scenario, likely because there was no estimate for direct mouthing of this COU. Mouthing duration is an important driver of ingestion exposures for children's play structures. For full results on the sensitivity analysis please refer to Appendix J (Consumer Exposures).

In the absence of parameter information from the literature, EPA used scientific judgement to select parameters for consumer modeling. There are uncertainties associated with any scientific judgment. The *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Consumer Exposure Modeling Inputs* (U.S. EPA, 2024e) provides a full list of parameters and description of rationale as to why certain parameter values were selected.

Weight Fraction

The key uncertainty in the consumer exposures assessment was the availability of relevant article weight fractions data. The Ecology Washington database was the main source of weight fraction information for the fabric, textile, and leather products scenarios. The 1.3 percent weight fraction for textiles in outdoor play structures was based on a value from the Washington State Database where the maximum weight fraction of 67 articles was 1.3 percent (WSDE, 2023). Of the 67 articles, there were only 2 that contained TCEP. The other article had a level of TCEP of 0.5 percent. Additionally, the database indicated four detects of TCEP in carpet padding and rug mats (ranged from 0.01 to 0.02%). This illustrates the limited data availability of weight fraction information for the fabric and textile products scenario.

The building and construction products scenario (*e.g.*, insulation, acoustic ceiling, wood resin products) relied on old, foreign literature values from Ingerowski et al. (2001) as cited in SCHER (2012). Anecdotal information from the literature suggested TCEP is present in these products but did not have specific information on weight fraction and article concentrations.

Values from Fang et al. (2013) were used to estimate weight fractions for foam seating and bedding products. There are uncertainties in these estimates because concentrations of V6 (a dimer of TCEP) were utilized in determining a TCEP weight fraction. This study measured TCEP at 14 percent as an impurity in V6, and hence this proportion was used to estimate weight fractions of foam seating and bedding products (Fang et al., 2013). There are uncertainties associated with how much TCEP is present as an impurity in V6.

TCEP in articles are not captured in CDR or Datamyne databases, as Datamyne does not include articles/products containing the chemical unless the chemical name is included in the description. Based on descriptions provided on the bills of lading, Figure 1-3 provides an estimate of the volume of TCEP imported as the chemical (not in an identified product or article) from 2012 to 2020. This limitation further illustrates the difficulty in obtaining current concentrations and weight fractions of TCEP in consumer products.

Duration and Frequency of Contact and Mouthing

For the carpet back coating scenario and wood flooring scenario, a literature value indicated that children under 12 years old spend 19 hours per day indoors (EFH 2011). It was assumed that the frequency of contact per day was 5 events for carpet and 10 events for flooring, and that the area mouthed was 10 cm^2. It should be noted that these values are conservative assumptions for duration and

frequency of contact (*i.e.,* typical frequency may be less than these estimates). The dermal exposure estimates are sensitive to the frequency of events per day parameter.

A further limitation for the carpet back coating and insulation scenario is the presence of a boundary layer (*e.g.,* top of the carpet, drywall in between insulation and living space) between the TCEP containing material and the potentially exposed human (*e.g.,* infant, child, adult). CEM 3.2 uses an overall mass transfer coefficient that is empirically estimated from an equation based on the AMEM guidance (the complexity of individual phase mass transfer is subsumed into an overall mass transfer coefficient that is either measured or estimated from a regression equation based on assorted chemical measurements). Although CEM 3.2 does not explicitly consider a boundary layer in its modeling, this does not mean that the model does not attempt to capture this complexity. Nevertheless, it is an uncertainty associated with the consumer modeling for the scenarios where a boundary layer would be expected. The modeling as conducted suggests that the TCEP would migrate to the surface of the carpet from the back coating components, or the dust particles would migrate from the insulation behind the drywall to the living area.

Oral ingestion estimates are driven by mouthing of articles for infants and children. A sensitive parameter driving these estimates is the duration of mouthing parameters. The recommended estimates from CEM 3.2 are 8.4 min/hr, 7 min/hr, and 10 min/hr for young children (aged 3–5 years), infants (1–2 years), and infants (<1 year), respectively.

Trends and Monitoring Data

The paucity of monitoring information related to the consumer COUs makes it difficult for EPA to have confidence in whether the consumer articles are nationally representative. Moreover, the decreasing trend of TCEP use, seen in the production volume data and environmental monitoring data, coupled with the understanding that many manufactures have replaced TCEP with alternatives in their products, build more uncertainty about the relevance of the consumer modeling to current consumers.

A systematic review revealed that there is limited information related to weight fractions of TCEP in consumer articles. No SDS were available for TCEP in consumer products. For the limited monitoring and experimental literature that was available, it is unclear how relevant the concentrations of TCEP at the time of sampling is related to consumer articles that are produced today.

In 2013, the State of California amended Technical Bulletin 117, a residential upholstered furniture flammability standard that was first implemented in 1975. The original TB 117 required interior filling materials of upholstered furniture to withstand exposure to a 12 second small open flame (the small flame impingement test, a one second flame, and the open flame test). This was replaced with a smolder resistance test, which tests a lighted cigarette on the fabric outside of the foam in 2013. TB 117-2013 is of significance to consumer articles, particularly fabric and textiles, and foam seating and bedding products, as article manufacturers no longer are required to meet the stringent flame standards of TB 117. Flame retardant concentrations in these articles are expected to decrease following this change. The available monitoring and experimental data on TCEP used in this consumer assessment was gathered pre-2013 (Table 5-20).

Table 5-20. Summary of Sampling Date for TCEP Weight Fraction Data

COU Subcategory	Weight Fraction Selected	Source	Sampling Date
Fabric and textile products	• 0.02% carpet back coating • 1.3% fabric in children's play structures	Ecology Washington database WSDE (2023)	2012
Foam seating and bedding products	• 0.51% furniture foam • 0.74% auto foam • 0.64% toy foam blocks	Fang et al. (2013)	2009–2011
Building/construction materials – insulation	• 1.98% insulation • 6.8% acoustic ceiling	Ingerowski et al. (2001)	<2001
Building/construction materials – wood and engineered wood products – wood resin composites	• 3% hardwood floors, wooden TV stand	SCHER (2012)	1997[a]

[a] Ionas et al. (2014) did provide more recent (2007) data on TCEP in wood toys at 0.0004%. However, due to the recent evidence suggesting TCEP use in wooden TV stands, and because TB 117-2013 is relevant for upholstered foam and furniture materials, EPA selected a weight fraction of 3% for consumer modeling.

Due to the limited information available on article weight fractions, EPA was unable to select a range of weight fraction for each of the COUs, and rather proceeded to assess consumer exposures to TCEP containing articles with a single discrete weight fraction value per article scenario. Additional sensitivity analysis varying the initial SVOC concentration in the article was conducted to help characterize the results (Table 5-19).

Ionas et al. (2014) stratified their data on TCEP in toys by time of manufacture (before and after 2007 when the REACH regulation went into force). Pre-2007, TCEP was detected in 32 percent of 63 children's toys whereas post-2007 TCEP was detected in 22 percent of 51 children's toys. Nevertheless, consumer modeling was conducted with possible weight fractions to understand the potential exposure of such products in furnishings and the built consumer environment.

Table 5-21 summarizes the indoor air and indoor dust monitoring data that was available in the United States. For a description of statistical methods, methodology of data integration, and treatment of non-detects and outliers used to generate these estimates, please see the *Supplemental Information File: Environmental Monitoring Concentrations Reported by Media Type* (U.S. EPA, 2024i).

Table 5-21. Summary of Indoor Monitoring Data of TCEP from U.S. Studies

Matrices	Location Type	Count of Estimates from Studies Containing U.S. Data	Unit	Fraction	Average of Arithmetic Estimates	Average of 90th Percentile Estimates
Indoor Air	Public spaces	1	ng/m^3	Particulate	2.0	4.6
	Residential	1	ng/m^3	Vapor/gas	9.5	2.1E01
Indoor Dust	Public spaces	1	ng/g	Dry	8.2E02	1.9E03
	Residential	9	ng/g	Dry	1.1E03	2.2E03
	Vehicles	1	ng/g	Dry	4.2E03	8.9E03

The maximum SVOC air concentration of 9.32 mg/m^3 for the insulation condition of use is five orders of magnitude higher than the 90th percentile estimate of indoor residential air concentrations found in one U.S. study (2.1×10^{-5} mg/m^3) (Dodson et al., 2017). The maximum respirable portion dust concentration of 11.13 µg/mg (1.1×10^7 ng/g) is four orders of magnitude higher than the 90th percentile estimate of residential indoor dust concentrations among nine U.S. studies (2.2×10^3 ng/g).

Modeling Approach Uncertainties

CEM 3.2 is a deterministic model where the outputs are fully determined by the choices of parameter values and initial conditions. Stochastic approaches feature inherent randomness, such that a given set of parameter values and initial conditions can lead to an ensemble of different model outputs. The overall approach to the CEM modeling is intended to capture a range of low- to high-intensity user exposure estimates by varying only a limited number of key parameters that represent the range of consumer product and use patterns for each scenario. A limited set of parameters were varied in the sensitivity analysis described in Table 5-19. Because not all parameters were varied, there is uncertainty regarding the full range of possible exposure estimates. Although these estimates are thought to reflect the range of exposure estimates for the suite of possible exposures based on the varied parameters, the scenarios presented are not considered bounding or "worst-case," as there are unvaried parameters that are also identified as sensitive inputs held constant at a central tendency value. Because EPA's largely deterministic approach involves choices regarding highly influential factors such as weight fraction and mouthing duration, it likely captures the range of potential exposure levels although it does not necessarily enable characterization of the full probabilistic distribution of all possible outcomes.

CEM 3.2 has a set of predefined consumer exposure scenarios that do not always line up with the conditions of use. For example, the CEM scenario utilized for consumer exposure to carpet back coating was Fabrics: curtains, rugs, wall coverings. There are uncertainties on how TCEP migrates from carpet back coatings to the surface of carpets and rugs. The literature describes that triphosphate esters such as TCEP have "blooming potential," which refers to the ability for the chemical to diffuse from a rubber or plastic material to the outer surface after curing (SCHER, 2012). Furthermore, the study from Castorina et al. (2017) has indicated that TCEP levels in dust are significantly associated with the presence of extremely worn carpets, suggesting that TCEP can be sampled in the dust from carpets and make it to the surface.

Background levels of TCEP in indoor air and indoor dust are not considered or aggregated in this assessment; therefore, there is potential for underestimating consumer exposures. Furthermore, consumer exposures were evaluated on a COU specific basis and are based on the use of a single consumer article, not multiple articles in the indoor environment.

There are uncertainties regarding the use of the 35.1 percent dermal fraction absorption (F_{abs}) parameter for the consumer dermal exposure estimates. This is the 24-hour value for fraction absorbed from Abdallah et al. (2016). EPA cannot assume that consumers would immediately wash their hands following contact with consumer articles. Therefore, it was assumed that the dose that deposited on the skin during exposure to a consumer article would remain on the skin until the skin was eventually washed. While the duration of interaction with materials that contain TCEP may be shorter than the duration that was tested in the dermal absorption study (*i.e.,* a 24-hour exposure), EPA decided to use the 35.1 percent fraction absorption value from Abdallah et al. (2016), due to uncertainties related to consumer hand-washing behaviors.

5.1.3 General Population Exposures

TCEP – General Population Exposures (Section 5.1.3):
Key Points

EPA evaluated the reasonably available information for the following general population exposures, the key points of which are summarized below:

- Oral ingestion for tribal fishers had the highest exposure estimates (0.008 to 219 mg/kg-day) among all routes. The highest tribal exposure estimates were for the formulation of TCEP containing reactive resin OES.
- The hypothetical scenario of a child playing in mud near a facility releasing TCEP to the ambient air resulted in the highest dermal exposures at a maximum of 7.97 mg/kg-day for use of paints and coatings at job sites OES. Estimates for a child conducting activities with soil (2.12×10^{-3} mg/kg-day) and incidental soil ingestion (1.08×10^{-1} mg/kg-day) were calculated. Paints and coatings were the only OES for the children playing in mud scenario with margin of exposures (MOEs) below the benchmark for non-cancer as described in Section 5.3.2.3.
- The highest inhalation exposure concentrations were for the use of paints and coatings at job sites OES at a central tendency estimate of 3.36×10^{-5} and a 95th percentile of 8.21×10^{-5} µg/m^3.
- Exposure estimates for drinking water non-dilute from surface water (1.46×10^{-4} mg/kg-day) were highest for the formulation of TCEP containing reactive resins OES.
- Children in fenceline communities and subsistence fishers are PESS who may have elevated exposures to TCEP compared to the rest of the general population due to industrial and commercial environmental releases.

General population exposures occur when TCEP is released into the environment and the environmental media is then a pathway for exposure. Section 3.3 provides a summary of the monitoring, database, and modeled data on concentrations of TCEP in the environment. Figure 5-4 below provides a graphic representation of where and in which media TCEP is estimated to be found and the corresponding route of exposure.

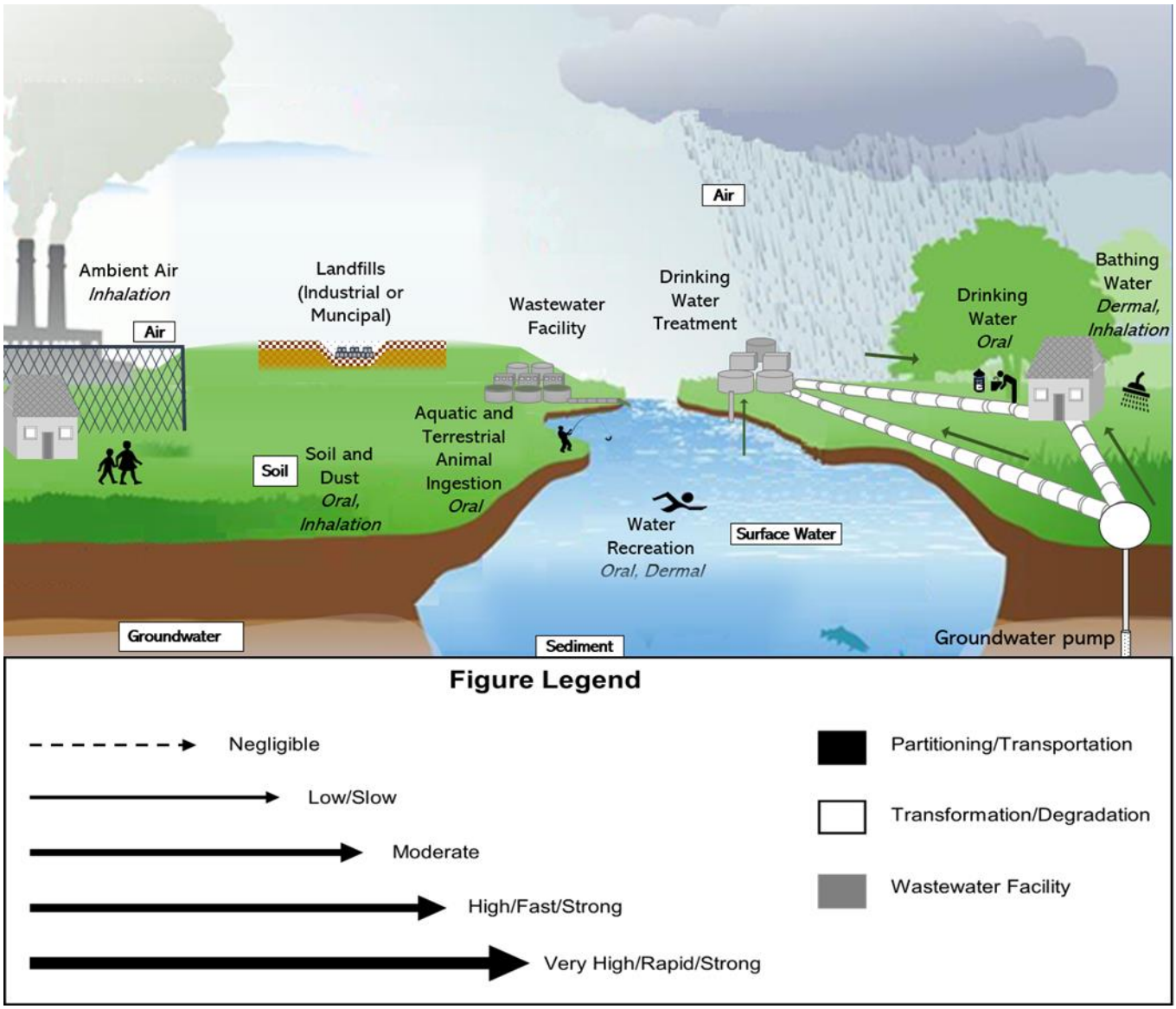


Figure 5-4. Potential Human Exposure Pathways to TCEP for the General Population[a]

[a] The diagram presents the media (white text boxes) and routes of exposure (italics for oral, inhalation, or dermal) for the general population. Sources of drinking water from surface or water pipes is depicted with grey arrows.

This diagram pairs with Figure 2-1 depicting the fate and transport of the subject chemical in the environment.

5.1.3.1 Approach and Methodology

TCEP is used primarily as an additive flame retardant in a variety of materials. TCEP has been detected in the indoor and outdoor environment and in human biomonitoring indicating that some amount of exposure is occurring in some individuals, although exposures likely vary across the general population. See Section 3.3 and *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Extraction Information for General Population, Consumer, and Environmental Exposure* (U.S. EPA, 2024r) for a summary of environmental and biomonitoring studies where TCEP has been detected.

Releases of TCEP are likely to occur through the following mechanisms: diffusion from sources, gas-phase, and particle-phase mass-transfer, abrasion of materials to form small particulates through routine use, and direct transfer from articles to dust adhered to the article surface. Releases of flame retardants to the outdoor environment may occur through direct releases to water, land, and air as well as indirect releases from the indoor environment.

For a more detailed discussion about indoor SVOC exposure, fate, and transport in the indoor environment, please see Section 2.2.2.

Exposure to the general population was estimated for the industrial and commercial releases per OES. Table 3-3 illustrates how the industrial and commercial releases to the environmental media varies by OES.

Modeled air concentrations (see Section 3.3.1.2) were utilized to estimate inhalation exposures (see Section 5.1.3.2) to the general population at various distances from a hypothetical facility. Modeled surface water concentrations (see Section 3.3.2.5) were utilized to estimate oral drinking water exposures, oral fish ingestions exposures, incidental oral exposures (see Section 5.1.3.4), and incidental dermal exposures (see Section 5.1.3.3) for the general population. Modeled groundwater concentrations (see Section 3.3.3.8), were also used to estimate oral drinking water exposures (see Section 5.1.3.4) to the general population. Modeled soil concentrations (see Section 3.3.3.2) via deposition were used to estimate dermal and oral exposures (see Sections 5.1.3.3 and 5.1.3.4) to children who play in mud and other activities with soil.

Exposures estimates from industrial and commercial releases of TCEP were compared to exposure estimates from non-scenario specific monitoring data to ground truth the results (*e.g.*, indoor dust exposures). Table 5-22 summarizes the environmental media monitoring data that was available in the United States. For a description of statistical methods, methodology of data integration and treatment of non-detects and outliers used to generate these estimates please see the *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Environmental Monitoring Concentrations Reported by Media Type* (U.S. EPA, 2024i).

Table 5-22. Summary of Environmental Monitoring Data of TCEP from the Literature for U.S. Studies

Matrices	Location Type	Count of Estimates from Studies Containing U.S. Data	Unit	Fraction	Average of Arithmetic Estimates	Average of 90th Percentile Estimates
Environmental media						
Ambient Air	General Population	6	ng/m^3	Any	1.3E−01	2.5E−01
Drinking Water	General Population	1	ng/L	Any	4.9	9.5
Sediment	General Population	1	ng/g	Dry	2.3	4.1
Surface Water	General Population	5	ng/L	Any	1.3E02	2.5E02
Wastewater	Treated Effluent	2	ng/g	Wet	2.1E01	4.3E01
	Treated Effluent	4	ng/L	Wet	8.1E02	1.2E03
Ecological media						

Matrices	Location Type	Count of Estimates from Studies Containing U.S. Data	Unit	Fraction	Average of Arithmetic Estimates	Average of 90th Percentile Estimates
Aquatic Fish	General Population	1	ng/g	Lipid	1.0E01	2.5E01
Terrestrial Birds	General Population	2	ng/g	Wet	5.3	9.7
Terrestrial Plants	Remote	1	ng/g	Wet	1.3E02	2.2E02
Human biomonitoring						
Human Hair	General Population	2	ng/g	Dry	2.7E02	4.2E02
Human Nails	General Population	1	ng/g	Dry	6.3E02	1.4E03

Figure 5-5 depicts the direct and indirect methods EPA used to estimate general population exposures. The direct assessment used environmental release estimates that were related to the industrial and commercial OES (see Section 3.2). Release estimates were used to model ambient air concentrations (see Section 3.3.1.2), surface water concentrations (see Section 3.3.2.5), soil concentrations (see Section 3.3.3.2), and groundwater concentrations as a result of landfill leachate (see Section 3.3.3.8). EPA modeled estimates for the environmental media were used to estimate inhalation, dermal and ingestion doses for various anticipated scenarios (*e.g.,* children's dermal exposure to soil, fish ingestion for the general population, drinking water ingestion exposure). Further information on the assessed exposure scenarios is presented in the individual sections below. In addition, EPA estimated exposure doses using an indirect estimation method via reverse dosimetry (see Section 5.1.3.5). Furthermore, to help "ground truth" the results, the reported environmental monitoring and reported modeled data (*i.e.,* TCEP concentration and doses in dietary sources, dust, soil, ambient air, indoor air, and surface water) were compared against the exposure estimates calculated from the direct assessment patterns.

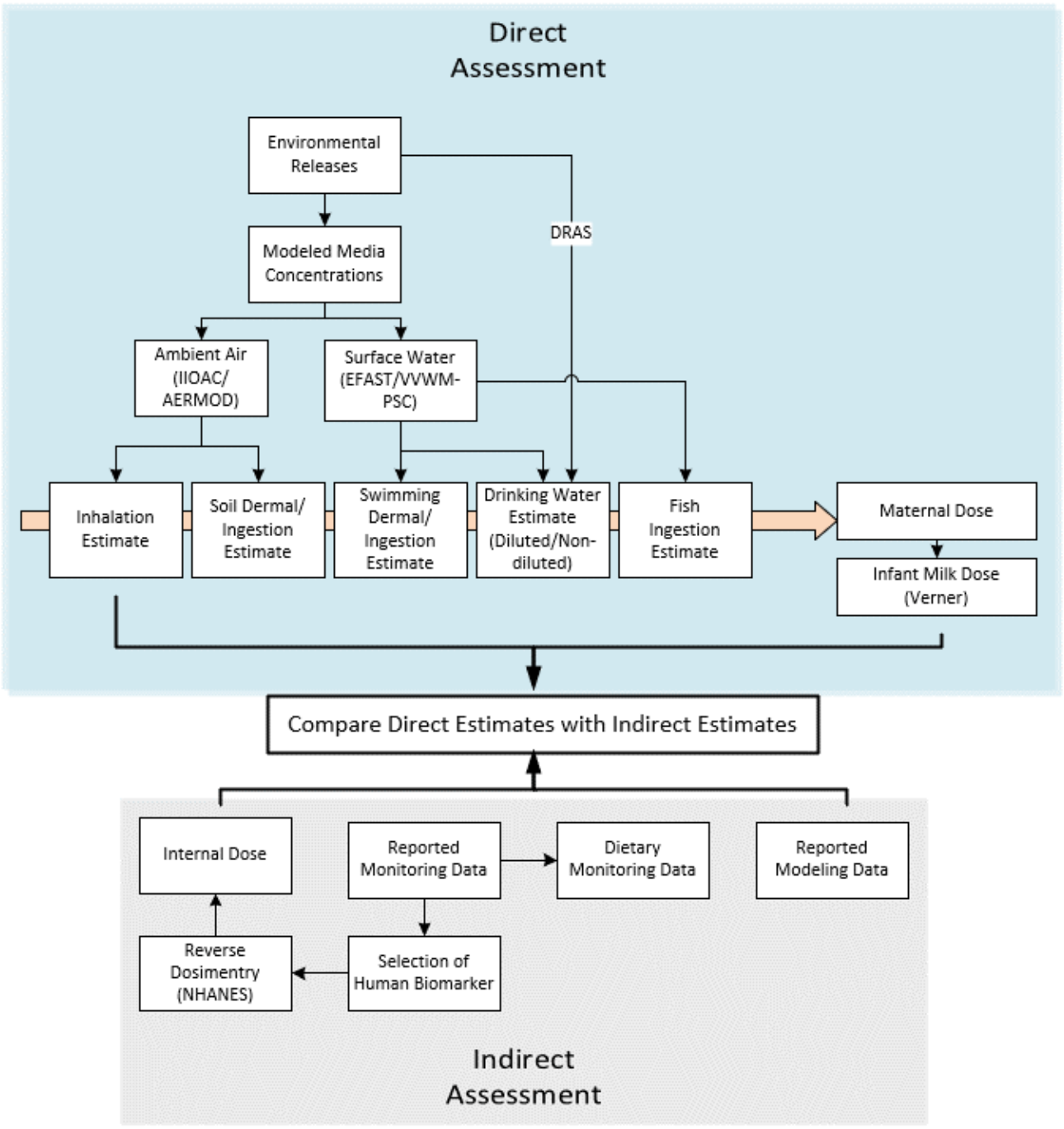


Figure 5-5. Direct and Indirect Exposure Assessment Approaches Used to Estimate General Population Exposure to TCEP

For each exposure pathway, central tendency and high-end exposures were estimated. *EPA's Guidelines for Human Exposure Assessment* defined central tendency exposures as "an estimate of individuals in the middle of the distribution." It is anticipated that these estimates apply to most individuals in the United States. High-end exposure estimates are defined as "plausible estimate of individual exposure for those individuals at the upper end of an exposure distribution, the intent of which is to convey an estimate of exposure in the upper range of the distribution while avoiding estimates that are beyond the true distribution." It is anticipated that these estimates apply to some individuals, particularly those who may live near facilities with elevated concentrations.

5.1.3.1.1 General Population Exposure Scenarios

Figure 5-6 provides an illustration of the exposure scenarios considered for general population exposure.

Ambient Air Exposure Scenarios

The Ambient Air Methodology utilizing AERMOD evaluated exposures to human populations at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and one area distance (100 to 1,000 m) from a hypothetical releasing facility for each OES. Human populations for each of the eight finite distances were placed in a polar grid every 22.5 degrees around the respective distance ring. This results in a total of 16 modeled exposure points around each finite distance ring for which exposures are modeled. Figure 5-6 provides a visual depiction of the placement of exposure points around a finite distance ring. Although the visual depiction only shows exposure point locations around a single finite distance ring, the same placement occurred for all eight finite distance rings.

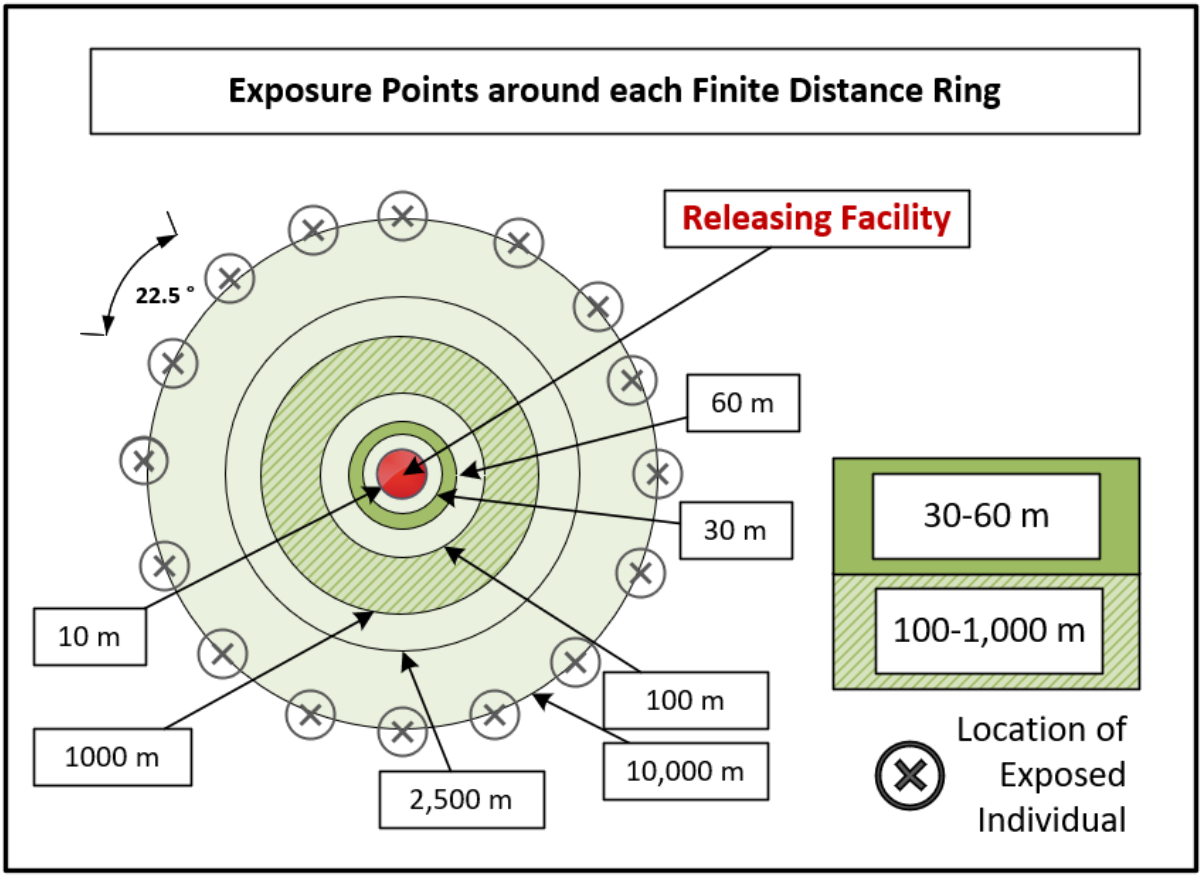


Figure 5-6. Modeled Exposure Points for Finite Distance Rings for Ambient Air Modeling (AERMOD)

Modeled exposure points for the area distance evaluated were placed in a cartesian grid at equal distances between 200 and 900 m around each releasing facility (or generic facility for alternative release estimates). Exposure points were placed at 100-meter increments. This results in a total of 456 points for which exposures are modeled. Figure 5-6 provides a visual depiction of the placement of these exposure points (each dot) around the area distance ring.

Although the ambient air is a minor pathway for TCEP, the general population may be exposed to ambient air concentrations and air deposition because of TCEP releases. Relevant exposures scenarios considered in this risk evaluation include ambient air inhalation for populations living nearby releasing

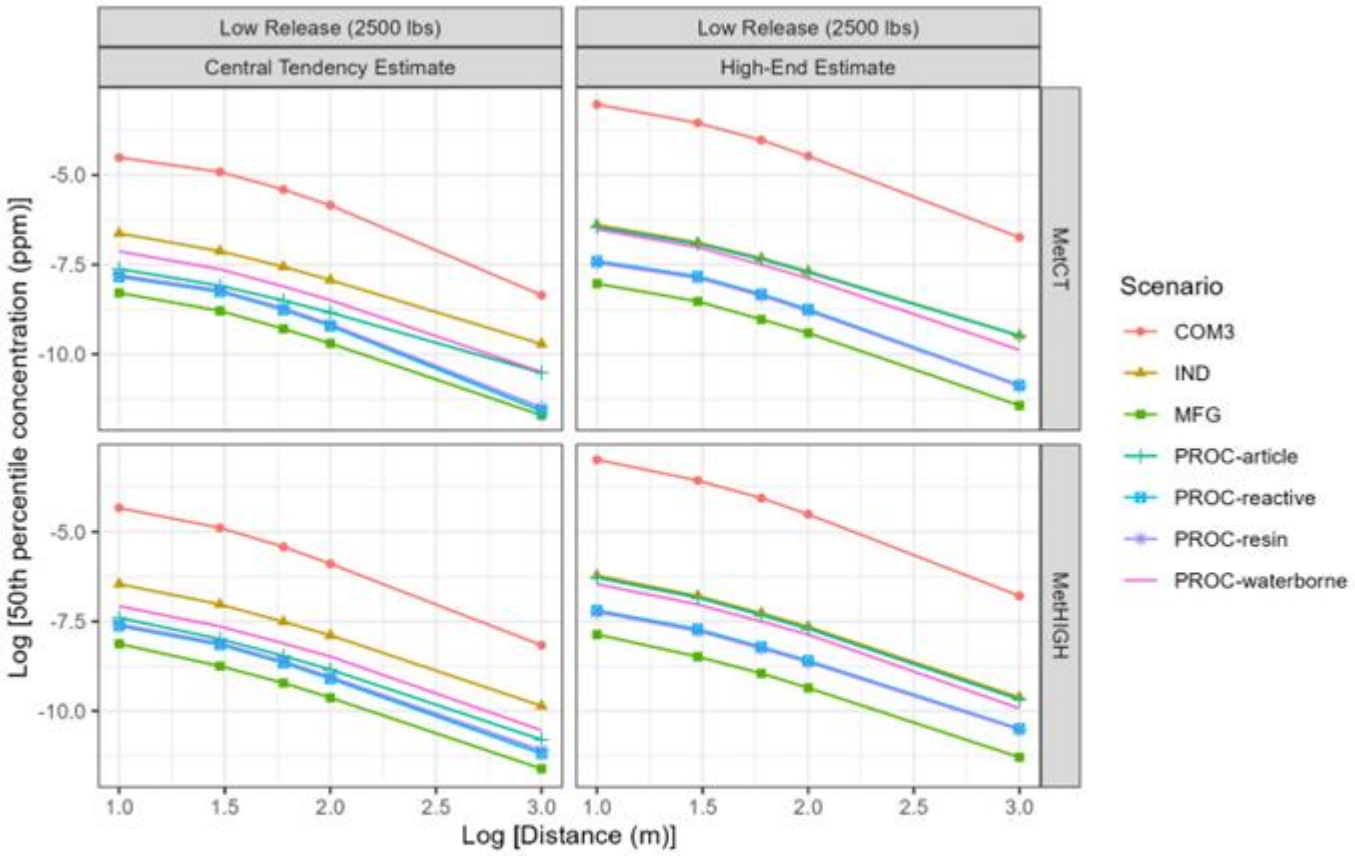
facilities, and ingestion and dermal exposure of soil to children result of ambient air deposition from a nearby facility.

Soil Exposure Scenarios
Air deposition fluxes from AERMOD were used to estimate soil concentrations at various distances from the hypothetical facility for each OES (see Section 3.3.3.2). Oral ingestion and dermal absorption exposure estimates of soil were calculated for children aged 3 to 6 years. Ingestion estimates were calculated for a central tendency and high intake rate. Dermal absorption estimates were calculated for two exposure scenarios: a child playing in mud, and a child performing activities with soil.

Water Exposure Scenarios
TCEP is expected to be found predominantly in water or soil. Section 3.3.2.5 provides modeled estimates of TCEP in surface water due to release of TCEP to water. Section 3.3.2.6 provides model estimates of TCEP in surface water due to air deposition to surface waters. Section 3.3.3.8 provides modeled estimates of TCEP in groundwater due to estimated migration from landfill leachate. Each of these estimates were used to calculate an exposure dose from drinking water for the general population. Additionally, modeled surface water concentrations (see Section 3.3.2.5) were used to calculate a dermal exposure estimate from swimming, incidental ingestion estimates from swimming, fish ingestion exposure.

5.1.3.2 Summary of Inhalation Exposure Assessment

Modeled ambient air concentrations for various distances from a hypothetical facility for each COU are presented in Section 3.3.1.2. Figure 5-7 below is a graph of the inhalation concentration by distances for the low production volume (2,500 lb/year) low-end and high-end estimates by the central tendency and high meteorology data. The x-axis is in log scale of distances in meters and the y-axis is in log scale of the 50th percentile concentrations in ppm.



COM3 refers to Use in paints and coatings at job sites.
IND refers to Use of Lab Chemicals.
MFG refers to Repackaging of Import Containers.
PROC-article refers to Processing into 2-part resin article.
PROC-resin refers to Incorporation into paints and coatings - resins/solvent-borne.
PROC-waterborne refers to Incorporation into paints and coatings - waterborne coatings.
PROC-reactive refers to Formulation of TCEP containing reactive resin.

Figure 5-7. General Population Inhalation Concentrations (ppm) by Distance (m) in Log Scale

Table 5-23 below indicates the ambient air concentrations at one distance (100 m) for each of the OES. For a full set data for all distances please see Appendix I.3.

Table 5-23. Excerpt of Ambient Air Modeled Concentrations for the 2,500 lb Production Volume, High-End Release Estimate for all COUs at 100 m, Suburban Forest Land Category Scenario

OES[a]	Meteorology	Source	Concentration (ppm) by Percentile		
			10th	50th	95th
Use in paints and coatings at job sites	MetCT	FUG_U	1.15E−05	3.36E−05	6.45E−05
	MetHIGH	FUG_U	8.77E−06	3.08E−05	8.21E−05
Use of laboratory chemicals	MetCT	ALL	1.51E−08	2.04E−08	3.33E−08
	MetHIGH	ALL	1.16E−08	2.24E−08	3.32E−08
Repackaging of import containers	MetCT	ALL	1.50E−10	3.88E−10	9.12E−10
	MetHIGH	ALL	2.34E−10	4.39E−10	1.12E−09
Processing into 2-part resin article	MetCT	ALL	1.48E−08	1.93E−08	2.70E−08
	MetHIGH	ALL	9.46E−09	1.96E−08	2.72E−08
Incorporation into paints and coatings – 2-part reactive coatings	MetCT	ALL	2.60E−11	1.60E−09	1.14E−08
	MetHIGH	ALL	3.46E−10	2.29E−09	1.11E−08
Incorporation into paints and coatings – 1-part coatings	MetCT	ALL	4.80E−09	1.31E−08	2.87E−08
	MetHIGH	ALL	4.00E−09	1.35E−08	3.51E−08
Formulation of TCEP containing reactive resin	MetCT	ALL	2.72E−11	1.78E−09	1.26E−08
	MetHIGH	ALL	3.73E−10	2.52E−09	1.21E−08
[a] Table 3-3 provides a crosswalk of industrial and commercial COUs to OESs					

5.1.3.3 Summary of Dermal Exposure Assessment

5.1.3.3.1 Incidental Dermal from Swimming

The general population may swim in surface waters (streams and lakes) that are affected by TCEP contamination. Modeled surface water concentrations from EFAST 2014 were used to estimate acute doses and average daily doses because of dermal exposure while swimming.

The following equations were used to calculate incidental dermal (swimming) doses for all COUs, for adults, youth, and children:

Equation 5-11.

$$ADR = \frac{SWC \times K_p \times SA \times ET \times CF1 \times CF2}{BW}$$

Equation 5-12.

$$ADD = \frac{SWC \times K_p \times SA \times ET \times RD \times ET \times CF1 \times CF2}{BW \times AT \times CF3}$$

Where:

ADR = Acute Dose Rate (mg/kg-day)
ADD = Average Daily Dose (mg/kg-day)
SWC = Chemical concentration in water (µg/L)

K_p = Permeability coefficient (cm/h)
SA = Skin surface area exposed (cm^2)
ET = Exposure time (h/day)
RD = Release days (days/year)
ED = Exposure duration (years)
BW = Body weight (kg)
AT = Averaging time (years)
$CF1$ = Conversion factor (1.0×10^{-3} mg/µg)
$CF2$ = Conversion factor (1.0×10^{-3} L/cm^3)
$CF3$ = Conversion factor (365 days/year)

A summary of inputs utilized for these exposure estimates are provided in Appendix I.

EPA used the dermal permeability coefficient (K_p) (0.022 cm/h) derived by Abdallah et al. (2016) from their *in vitro* study that measured TCEP absorption through excised human skin.

Table 5-24. Modeled Incidental Dermal (Swimming) Doses for all COUs for Adults, Youths, and Children, for the 2,500 lb High-End Release Estimate

OES[a]	Surface Water Concentration		Adult (≥ 21 years)		Youth (11–15 years)		Child (6–10 years)	
	30Q5 Conc. (µg/L)	Harmonic Mean Conc. (µg/L)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)
Repackaging of import containers	862.129	1,366.528	1.39E−03	6.02E−06	1.06E−03	4.61E−06	6.44E−04	2.80E−06
Incorporation into paints and coatings – 1-part coatings	3,819.444	5,912.114	6.14E−03	2.61E−05	4.70E−03	2.00E−05	2.85E−03	1.21E−05
Incorporation into paints and coatings – 2-part reactive coatings	3,462.800	5,360.066	5.57E−03	2.36E−05	4.27E−03	1.81E−05	2.59E−03	1.10E−05
Use in paints and coatings at job sites	2,029.305	3,216.574	3.26E−03	1.42E−05	2.50E−03	1.09E−05	1.52E−03	6.58E−06
Formulation of TCEP containing reactive resin	4,844.722	6,245.374	7.79E−03	2.75E−05	5.97E−03	2.11E−05	3.62E−03	1.28E−05
Use of laboratory chemicals	34.555	54.722	5.59E−05	2.41E−07	4.26E−05	1.85E−07	2.58E−05	1.12E−07

[a] Table 3-3 provides a crosswalk of industrial and commercial COUs to OES.

5.1.3.3.2 Incidental Dermal Intake from Soil

Dermal absorbed doses (DAD) were calculated for TCEP using the following formula:

Equation 5-13.

$$DAD = \frac{C_{soil} \times CF \times AF \times ABS_d \times SA_{skin} \times EV}{BW \times AT}$$

Where:

AF	=	Adherence factor of soil to skin (mg/cm^2-event)
ABS$_d$	=	Dermal absorption fraction
Csoil	=	Concentration in soil
CF	=	Conversion Factor
SAskin	=	Skin surface area (cm^2)
EV	=	Events per day
BW	=	Body weight (kg)
AT	=	Averaging time

Modeled soil concentrations were calculated from 95th percentile air deposition (see Section 3.3.3.2) for 100 and 1,000 m. These calculations were conducted for the COM-paints-use scenario (LOW PV – 2,500 lb, HE-95th percentile release).

Soil concentrations of 141.2 ng/g were modeled for the 2,500 lb production volume, high-end release estimate for the Incorporation into paints and coatings – 1 part coatings OES using BST (see Section 3.3.3.5).

The dermal absorption fraction (*ABS$_d$*) used was 35.1 percent (Abdallah et al., 2016). The skin surface area for the arms (0.106 m^2), hands (0.037 m^2), legs (0.195 m^2) and feet (0.049 m^2), and body weight (18.6 kg) of a 3- to 6-year-old was used from the *Exposure Factors Handbook* (U.S. EPA, 2017d). EPA used two different scenarios for the adherence factor of soil to skin: 96 mg/cm^2 for a child playing in mud and 0.467 mg/cm^2 for children's activity with soil. With an assumption of one event per day and an averaging time of 2 days, the dermal exposure estimates for the different scenarios were as follows:

Table 5-25. Modeled Soil Dermal Doses for the Commercial Use of Paints and Coatings COU, for Children

OES	Exposure Scenario	Distance (m)	Soil Concentration (ng/g)	Dermal Absorbed Dose (mg/kg-day)
Use in paints and coatings at job sites[a]	Activities with soil	100	1.14E04	3.89E–04
		1,000	8.65E01	2.95E–06
	Playing in mud	100	1.14E04	7.99E–02
		1,000	8.65E01	6.06E–04
Incorporation into paints and coatings – 1-part coatings	Activities with soil	N/A	1.41E02	4.82E–06
	Playing in mud	N/A	1.41E02	9.90E–04
[a] 95th percentile estimates				

5.1.3.4 Summary of Oral Exposures Assessment

5.1.3.4.1 Drinking Water Exposure

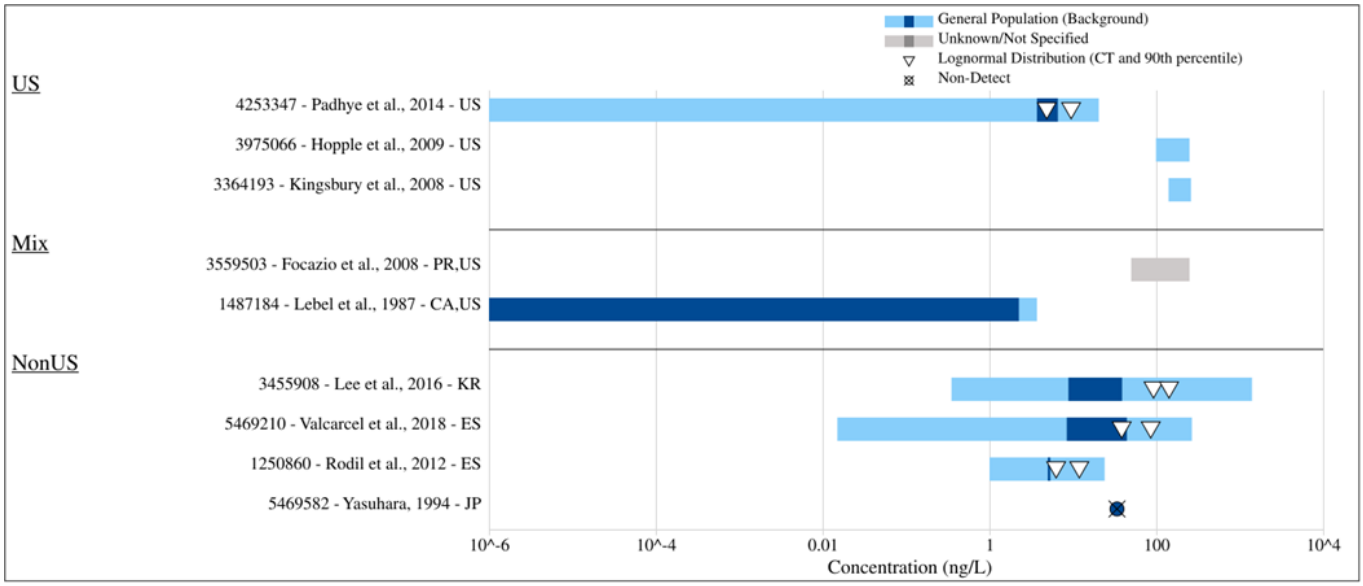


Figure 5-8. Concentrations of TCEP (ng/L) in Drinking Water from 1982 to 2014

A study of drinking water systems in the United States indicated a maximum of 470 ng/L and a median of 120 ng/L of TCEP in finished water, and a maximum of 200 ng/L and a median of 140 ng/L in distributed waters in 6 out of 19 drinking water systems. The drinking water systems collected samples from 19 drinking water treatment plants (DWTPs) across the United States, representing drinking water for more than 28 million Americans (Benotti et al., 2009).

TCEP has been detected in tap water in Korea at a mean of 39.5 and a maximum of 87.4 ng/L as recently as 2017 (Park et al., 2018). Because the OPFR concentrations were correlated with the distance of the pipes (both from the water intake source to the drinking water treatment facility and the drinking water treatment facility to the sampling site), this study has suggested that a possible source of OPFRs in tap water were pipes. Pipe materials are known to promote the formation of disinfection by products or biofilms (Park et al., 2018).

Drinking Water Intake Estimates via Modeled Surface Water Concentrations

Modeled surface water concentrations (see Sections 3.3.2.5 and 3.3.2.6) were used to estimate drinking water exposures. A 0 percent drinking water treatment removal efficiency was used for the purposes of this exposure estimation.

Drinking water intakes were calculated using the following formulae:

Equation 5-14.

$$ADR_{POT} = \frac{SWC \times \left(1 - \frac{DWT}{100}\right) \times IR_{dw} \times RD \times CF1}{BW \times AT}$$

Equation 5-15.

$$ADD_{POT} = \frac{SWC \times \left(1 - \frac{DWT}{100}\right) \times IR_{dw} \times ED \times RD \times CF1}{BW \times AT \times CF2}$$

Equation 5-16.

$$LADD_{POT} = \frac{SWC \times \left(1 - \frac{DWT}{100}\right) \times IR_{dw} \times ED \times RD \times CF1}{BW \times AT \times CF2}$$

Equation 5-17.

$$LADC_{POT} = \frac{SWC \times \left(1 - \frac{DWT}{100}\right) \times ED \times RD \times CF1}{AT \times CF2}$$

Where:

ADR_{POT}	=	Potential Acute Dose Rate (mg/kg/day)
ADD_{POT}	=	Potential Average Daily Dose (mg/kg/day)
$LADD_{POT}$	=	Potential Lifetime Average Daily Dose (mg/kg/day)
$LADC_{POT}$	=	Potential Lifetime Average Daily Concentration in drinking water (mg/L)
SWC	=	Surface water concentration (ppb or µg/L; 30Q5 conc for ADR, harmonic mean for ADD, LADD, LADC)
DWT	=	Removal during drinking water treatment (%)
IR_{dw}	=	Drinking water intake rate (L/day)
RD	=	Release days (days/yr for ADD, LADD and LADC; 1 day for ADR)
ED	=	Exposure duration (years for ADD, LADD and LADC; 1 day for ADR)
BW	=	Body weight (kg)
AT	=	Exposure duration (years for ADD, LADD and LADC; 1 day for ADR)
CF1	=	Conversion factor (1.0×10^{-3} mg/µg)
CF2	=	Conversion factor (365 days/year)

A method was derived to incorporate a dilution factor to estimate TCEP concentrations at drinking water locations downstream from surface water release points. Because no location information was available for facilities releasing TCEP, a dilution factor and distances to drinking water intake was estimated for each relevant SIC code. Table 5-26 provides the 50th quantile distances and 50th quantile dilution factors for the 30Q5 and harmonic mean flow for the relevant SIC codes.

Table 5-26. 50th Quantile Distances and 30Q5 and Harmonic Mean 50th Quantile Dilution Factors for Relevant TCEP SIC

SIC Codes	n	50th Quantile Distance (km)	50th Quantile Dilution Factor (30Q5)	50th Quantile Dilution Factor (Harmonic Mean)
Adhesives, Sealants, Plastics, Resins, Rubber Manufacturing	516	113.82	432.36	528.47
Paint Formulation	374	107.03	1,603.6	1,854.89
POTWs – All facilities	567	129.57	1,233.87	1,557.91
30Q5 = The lowest 30-day average flow that occurs (on average) once every 5 years				

To calculate the diluted water concentrations the surface water concentrations from E-FAST 2014 modeling were divided by the dilution factors presented in Table 5-26. Table 5-27 presents the diluted drinking water concentrations for adults for all industrial and commercial COUs.

Table 5-27. Modeled Drinking Water Ingestion Estimates for Diluted Surface Water Concentrations for Adults for All Industrial and Commercial COUs for the 2,500 lb High-End Release Estimate

OES[a]	Diluted Water Concentration		Adult (≥ 21 years)			
	Harmonic Mean Concentration (µg/L)	30Q5 Concentration (µg/L)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	$LADD_{POT}$ (mg/kg-day)	$LADC_{POT}$ (mg/L)
Repackaging of import containers	0.553	1.108	4.46E−05	1.67E−08	7.05E−09	6.41E−07
Incorporation into paints and coatings – 1-part coatings	2.059	3.687	1.48E−04	6.20E−08	2.62E−08	2.39E−06
Incorporation into paints and coatings – 2-part reactive coatings	1.867	3.343	1.35E−04	5.62E−08	2.38E−08	2.16E−06
Use in paints and coatings at job sites	1.303	2.607	1.05E−04	3.92E−08	1.66E−08	1.51E−06
Formulation of TCEP containing reactive resin	9.167	14.445	5.81E−04	2.76E−07	1.17E−07	1.06E−05
Use of laboratory chemicals	0.022	0.044	1.79E−06	6.68E−10	2.83E−10	2.57E−08
[a] See Table 3-3 for a crosswalk of industrial and commercial COUs to OESs.						

Table 5-28 provides the non-diluted drinking water intake estimates. In this case, it is assumed that the surface water outfall is located very close (within a few km) to the population. The dilution factor reduces the acute, chronic, and lifetime exposure estimates by a factor of three.

Table 5-28. Modeled Drinking Water Ingestion Estimates for Surface Water Concentrations for Adults for All Industrial and Commercial COUs for the 2,500 lb High-End Release Estimate

OES[a]	Water Concentration		Adult (≥ 21 years)			
	Harmonic Mean Concentration (µg/L)	30Q5 Concentration (µg/L)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	$LADD_{POT}$ (mg/kg-day)	$LADC_{POT}$ (mg/L)
Repackaging of import containers	862.129	1,366.528	5.4992E−02	2.60E−05	1.10E−05	9.99E−04
Incorporation into paints and coatings – 1-part coatings	3,819.444	5,912.114	2.3792E−01	1.15E−04	4.87E−05	4.43E−03
Incorporation into paints and coatings – 2-part reactive coatings	3,462.800	5,360.066	2.1570E−01	1.04E−04	4.41E−05	4.01E−03
Use in paints and coatings at job sites	2,029.305	3,216.574	1.2944E−01	6.11E−05	2.59E−05	2.35E−03
Formulation of TCEP containing reactive resin	4,844.722	6,245.374	2.5133E−01	1.46E−04	6.17E−05	5.62E−03
Use of laboratory chemicals	34.555	54.772	2.20E−03	1.04E−06	4.40E−07	4.01E−05
[a] Table 3-3 provides a crosswalk of industrial and commercial COUs to OES.						

A summary of inputs utilized for these exposure estimates is presented in Appendix I.2Appendix I.

Drinking Water via Leaching of Landfills to Groundwater

Groundwater concentrations from leaching from landfills was estimated for the 2,500 and 25,000 lb production volume scenarios (see Table 3-8. in Section 3.3.3.8). The relevant COU/OES that may be relevant for groundwater migration from landfill leachate are the Incorporation into paints and coatings – 1-part coatings, and Processing into formulation of TCEP-containing reactive resin. These OESs result in the following releases to landfill presented in Table 5-29. In addition, consumer articles could be disposed to municipal solid waste landfills and construction and demolition landfills.

Table 5-29. Landfill Releases of TCEP from Two Commercial and Industrial OESs

OES	Number of Release Days	Annual Release Per Site (kg-site-yr)	Daily Release (kg/site-day)
Incorporation into paints and coatings – 1-part coatings	2	2.15E01	9.27
Formulation of TCEP-containing reactive resin	17	4.29E01	2.49

Section 3.3.3.8 estimates a range of groundwater concentrations because of industrial and commercial releases. The range of concentrations varies due to leachate concentrations to be between 1.08×10^{-3} and 1.08×10^{1} µg/L. Using the same formulae for drinking water ingestion above, adult drinking water estimates because of landfill leachate contamination are presented in Table 5-30.

Table 5-30. Estimated Average Daily Doses and Lifetime Average Daily Concentrations for Adults from Groundwater Concentrations by DRAS

DRAS	Groundwater Concentration	Adult (≥21 years)	
		ADD (mg/kg-day)	$LADC_{POT}$ (mg/L)
Low Estimate: Low Leachate Concentration – 2,500 lb Production Volume	1.08E−03	3.3E−11	3.0E−09
High Estimate: High Leachate Concentration – 2,500 lb Production Volume	1.08E01	3.3E−07	3.0E−05

These results would be further lowered if dilution was incorporated to these drinking water estimates. Due to uncertainties in distance from drinking water intake location to the groundwater contamination site the dilution was not estimated.

The complete set of drinking water exposure estimates can be referenced in *Supplemental Information File: E-FAST Modeling Results* (U.S. EPA, 2024g).

5.1.3.4.2 Fish Ingestion Exposure

Surface water concentrations for TCEP associated with a particular COU were modeled using E-FAST 2014 as described in Section 3.3.2.5. Surface water concentrations based on harmonic mean surface water flows representing the 50th percentile stream flows of all facilities in each industry sector were used to estimate exposure from fish ingestion. The harmonic mean flow was used to estimate the concentration of TCEP in fish tissue because it represents long-term average flow conditions. As it takes time for chemical concentrations to accumulate in fish, a harmonic mean flow is more appropriate than a low streamflow value (*e.g.*, 7Q10) that occurs infrequently. Furthermore, dilutions of surface water concentrations of TCEP further downstream of a facility's outfall was not considered, as fish presumably reside within stream reaches receiving direct releases from a facility. This approach takes into account that people often harvest fishes originating from various locations regardless of known or unknown releases to the environment at that location; thus, it is more conservative because it estimates higher concentrations of TCEP in fish.

General population exposure estimates from fish consumption are provided for only adults 16 years or older to allow for comparison with subsistence and Tribal fishers, which also only estimated exposure for adults. However, as shown in Table_Apx I-2, the highest fish ingestion rate per kilogram of body weight for the general population is for a young toddler between 1 and 2 years old. While results are not shown, the exposure estimates for a young toddler are similar to an adult (*i.e.*, within the same magnitude). The 50th percentile (central tendency) and 90th percentile ingestion rate (IR) for adults is 5.04 g/day and 22.2 g/day, respectively. The ADRs were calculated using the 90th percentile IR. EPA typically uses the central tendency for chronic exposure estimates. However, EPA considers both the central tendency and 90th percentile IRs to be reasonable for the general population. The 90th percentile IR can also capture individuals within the general population that may have higher chronic exposures but not as high as the subsistence fisher. As a result, EPA used both fish ingestion rates to estimate an ADD and LADD. Exposure estimates via fish ingestion were calculated according to the following equation:

Equation 5-18.

$$ADR\ or\ ADD = \frac{SWC \times BAF \times IR \times CF1 \times CF2 \times ED}{AT \times BW}$$

Where:

ADR	=	Acute Dose Rate (mg/kg/day)
ADD	=	Average Daily Dose (mg/kg/day)
SWC	=	Surface water (dissolved) concentration (µg/L)
BAF	=	Bioaccumulation factor (L/kg wet weight)
IR	=	Fish ingestion rate (g/day)
$CF1$	=	Conversion factor (0.001 mg/µg)
$CF2$	=	Conversion factor for kg/g (0.001 kg/g)
ED	=	Exposure duration (year)
AT	=	Averaging time (year)
BW	=	Body weight (80 kg)

The years within an age group (*i.e.,* 62 years for adults) was used for the exposure duration and averaging time to characterize non-cancer risks. For cancer, the years within an age group was also used for the exposure duration while the averaging time is 78 years (*i.e.*, lifetime).

A BAF is preferred in estimating exposure because it considers the animal's uptake of a chemical from both diet and the water column. For TCEP, there are multiple wet weight BAF values reported for whole fish collected from water bodies that contained TCEP (Table 2-2). The modeled surface water concentrations were converted to fish tissue concentrations using the upper and lower bound of the BAFs reported in literature: 2,198 L/kg wet weight for walleye (*Sander vitreus*) collected from the U.S. Great Lakes (Guo et al., 2017b) and 109 L/kg wet weight for mud carp collected from an e-waste polluted pond in China (Liu et al., 2019a). While Guo et al. (2017b) is the only U.S. study that measured TCEP concentrations in fish samples and is presumably more representative of subsistence fishers in the United States, EPA considered BAF values from non-U.S. studies because of uncertainties with walleye's BAF and subsistence fishers consume more than just one fish species. As a result, BAF from non-U.S. studies were considered.

Table 5-31 compares the fish tissue concentration calculated from the scenario-specific modeled surface water concentrations using the two BAFs with measured fish tissue concentrations obtained from literature. For comparison, Table 5-31 also includes fish tissue concentrations presented in Table 4-1 that were derived from a BCF. The overall range for scenario-specific fish concentrations based on modeled concentrations is for wet weight, and monitoring studies reported both wet and lipid weight. While the lipid content was not available to convert from lipid to wet weight, measured fish tissue concentrations are still several orders of magnitude lower than that derived from modeled surface water concentrations and BAF or BCF.

Table 5-31. Fish Tissue Concentrations Calculated from Modeled Surface Water Concentrations and Monitoring Data

Data Approach	Data Description	Surface Water Concentration (µg/L)	Fish Tissue Concentration (µg/kg)
Modeled Surface Water Concentration	BAF (2,198) and the maximum 1-day average dissolved water concentrations from PSC under harmonic mean flow conditions	Overall range 3.4E01 to 4.8E03	Overall range 7.6E04 to 1.06E07, ww
	BAF (109) and the maximum 1-day average dissolved water concentrations from PSC under harmonic mean flow conditions	Overall range 3.4E01 to 4.8E03	Overall range 3.8E03 to 5.3E05, ww
	BCF and the maximum 1-day average dissolved water concentrations from PSC under 7Q10 flow conditions	Overall range 9.6E01 to 1.0E04	Overall range 3.2E01 to 3.4E03, ww
Fish Tissue Monitoring Data (Wild-Caught)	7 studies with over 200 fish tissue samples collected from 7 countries, including one U.S. study by Guo et al. (2017b)	Only one non-U.S. study collected water samples from the same waterbody and at the same time as the fish tissue samples. Surface water concentrations for that study ranged from 1.5E−02 to 2.34E−01	Central tendency range for U.S. study 6.55 to 3.56E01, lw Overall range among non-U.S. studies ND to 2.96, ww ND to 1.87E02, lw

Table 5-32 presents the exposure estimates for adult general population fish ingestion doses. These doses were calculated using the modeled scenario-specific surface water concentrations based on the 50th percentile stream flows and two BAFs.

Table 5-32. Adult General Population Fish Ingestion Doses by Scenario Based on a Production Volume of 2,500 lb/year, High-End Release Distribution, and Modeled Surface Water Concentrations Based on 50th Percentile Flow of Harmonic Mean

Scenario Name	SWC[a] (µg/L)	ADR[b] (mg/kg-day)		ADD[b] (mg/kg-day)				LADD[b] (mg/kg-day)			
		BAF 2,198	BAF 109	BAF 2,198		BAF 109		BAF 2,198		BAF 109	
		HE		CT	HE	CT	HE	CT	HE	CT	HE
Import and Repackaging	8.62E02	5.25E−01	2.60E−02	1.19E−01	5.25E−01	5.92E−03	2.60E−02	9.49E−02	4.17E−01	4.71E−03	2.07E−02
Incorporation into Paints and Coatings – 1-Part Coatings	3.82E03	2.33	1.15E−01	5.29E−01	2.33	2.62E−02	1.15E−01	4.20E−01	1.85	2.08E−02	9.17E−02
Incorporation into Paints and Coatings – 2-Part Reactive Coatings	3.46E03	2.11	1.05E−01	4.80E−01	2.11	2.38E−02	1.05E−01	3.81E−01	1.68	1.89E−02	8.31E−02
Use in Paints and Coatings at Job Sites	2.03E03	1.24	6.13E−02	2.81E−01	1.24	1.39E−02	6.13E−02	2.23E−01	9.82E−01	1.11E−02	4.87E−02
Formulation of TCEP Containing Reactive Resin	4.84E03	2.95	1.46E−01	6.71E−01	2.95	3.33E−02	1.46E−01	5.33E−01	2.34	2.64E−02	1.16E−01
Laboratory Chemicals	3.46E01	2.10E−02	1.04E−03	4.78E−03	2.10E−02	2.37E−04	1.04E−03	3.80E−03	1.67E−02	1.89E−04	8.29E−04

[a] Surface water concentrations based on 50th percentile flow of harmonic mean flow conditions.
[b] ADR calculated using the 90th percentile fish ingestion rate (22.2 g/day). ADD and LADD were calculated using both the mean (CT) and 90th percentile (HE) fish ingestion rates, 5.04 g/day and 22.2 g/day respectively. An ADD based on the 90th percentile ingestion rate is the same as an ADR.

5.1.3.4.3 Subsistence Fish Ingestion Exposure

Subsistence fishers represent a PESS group for TCEP due to their greatly increased exposure via fish ingestion (142.4 g/day compared to a 90th percentile of 22.2 g/day for the general population) (U.S. EPA, 2000b). The ingestion rate for subsistence fishers apply to only adults aged 16 to < 70 years. EPA calculated exposure for subsistence fishers using Equation 5-18 and the same inputs as the non-subsistence fisher except for the ingestion rate. Furthermore, unlike the general population fish ingestion rates, there is no central tendency or 90th percentile IR for the subsistence fisher. The same value was used to estimate both the ADD and ADR.

EPA is unable to determine subsistence fisher exposure estimates specific to younger lifestages based on lack of reasonably available information. The exposure estimates for an adult subsistence fisher in Table 5-33 were calculated using the array of modeled scenario-specific surface water concentrations and BAF.

Table 5-33. Adult Subsistence Fisher Doses by Scenario Based on a Production Volume of 2,500 lb/year, High-End Release Distribution, and Modeled Surface Water Concentrations Based on 50th Percentile Flow of Harmonic Mean

Scenario Name	SWC[a] (µg/L)	ADD, ADR (mg/kg-day) BAF 2,198	ADD, ADR (mg/kg-day) BAF 109	LADD (mg/kg-day) BAF 2,198	LADD (mg/kg-day) BAF 109
Import and repackaging	8.62E02	3.37	1.67E−01	2.68	1.33E−01
Incorporation into paints and coatings – 1-part reactive coatings	3.82E03	1.49E01	7.41E−01	1.19E01	5.89E−01
Incorporation into paints and coatings – 2-part reactive coatings	3.46E03	1.35E01	6.72E−01	1.08E01	5.34E−01
Use in paints and coatings at job sites	2.03E03	7.94	3.94E−01	6.31	3.13E−01
Formulation of TCEP containing reactive resin	4.84E03	1.90E01	9.40E−01	1.51E01	7.47E−01
Laboratory chemicals	3.46E01	1.35E−01	6.70E−03	1.07E−01	5.33E−03

[a] Surface water concentrations based on 50th percentile flow of harmonic mean flow conditions.

5.1.3.4.4 Tribal Fish Ingestion Exposure

Tribal populations represent another PESS group. In the United States there are a total of 574 federally recognized American Indian Tribes and Alaska Native Villages and 63 state recognized Tribes. Many Tribal cultures are essentially synonymous with and inseparable from their lands and resources (Harper et al., 2007). The relationship that Tribal people have towards the land is not one-dimensional and includes but is not limited to hunting and fishing, food gathering, livestock, commerce and economy, art, education, health care, and social systems (Harris and Harper, 2011). This relationship forms the basis of *Tamanwit* (natural law). Such an intricate connection to the land and the distinctive lifeways and cultures between individual Tribes create many unique exposure scenarios that can expose Tribal members to higher doses of contaminants in the environment. However, EPA quantitatively evaluated only the Tribal fish ingestion pathway for TCEP because of data limitations and recognizes that this overlooks many other unique exposure scenarios.

U.S. EPA (2011a) (Chapter 10, Table 10-6) summarizes relevant studies on Tribal-specific fish IRs that covered 11 Tribes and 94 Alaskan communities. EPA used the highest mean IR per kilogram of body

weight reported in a 1997 survey of adult members (16 years and older) of the Suquamish Tribe in Washington. Adults reported a mean IR of 2.7 g/kg-day, or 216 g/day assuming an adult body weight of 80 kg. In comparison, the IRs for the adult subsistence fisher and general population are 142.2 and 22.2 g/day, respectively. A total of 92 adults responded to the survey funded by ATSDR through a grant to the Washington State Department of Health, of which 44 percent reported consuming less fish/seafood today compared to 20 years ago. One reason for the decline is restricted harvesting caused by increased pollution and habitat degradation (Duncan, 2000).

Because current fish consumption rates are suppressed by contamination, degradation, or loss of access, EPA reviewed existing literature for IRs that reflect heritage rates. Heritage rates refer to those that existed prior to non-indigenous settlement on Tribal fisheries resources, as well as changes in culture and lifeways (U.S. EPA, 2016b). Heritage IRs were identified for four Tribes, all located in the Pacific Northwest region, among available literature. The highest heritage IR was reported for the Kootenai Tribe in Idaho at 1,646 g/day (Ridolfi, 2016) (that study was funded through an EPA contract). The authors conducted a comprehensive review and evaluation of ethnographic literature, historical accounts, harvest records, archaeological and ecological information, as well as other studies of heritage consumption. The heritage IR is estimated for Kootenai members living in the vicinity of Kootenay Lake in British Columbia, Canada; the Kootenai Tribe once occupied territories in parts of Montana, Idaho, and British Columbia. It is based on a 2,500 calorie per day diet, assuming 75 percent of the total caloric intake comes from fish and using the average caloric value for fish. Notably, the authors acknowledged that assuming 75 percent of caloric intake comes from fish may overestimate fish intake.

EPA calculated exposure via fish consumption for Tribes using Equation 5-18 and the same inputs as the general population except for the IR. Two IRs were used: 216 g/day for current consumption and 1,646 g/day for heritage consumption. Similar to the subsistence fisher, EPA used the same IR to estimate both the ADD and ADR. The heritage IR is assumed to be applicable to adults. For current IR, U.S. EPA (2011a) provides values specific to younger lifestages, but adults still consume higher amounts of fish per kilogram of body weight. An exception is for the Squaxin Island Tribe in Washington that reported an IR of 2.9 g/kg-day for children under 5 years old. That IR for children is nearly the same as the adult IR of 2.7 g/kg-day for the Suquamish Tribe. As a result, exposure estimates based on current IRs focused on adults (Table 5-34).

Table 5-34. Adult Tribal Fish Ingestion Doses by Scenario Based on a Production Volume of 2,500 lb/year, High-End Release Distribution, Modeled Surface Water Concentrations Based on 50th Percentile Flow, and Two Fish Ingestion Rates

Scenario Name	SWC[a] (µg/L)	ADD, ADR (mg/kg-day) BAF 2,198	ADD, ADR (mg/kg-day) BAF 109	LADD (mg/kg-day) BAF 2,198	LADD (mg/kg-day) BAF 109
Current mean fish ingestion rate reported by the Suquamish Tribe (216 g/day)					
Import and repackaging	8.62E02	5.12	2.54E−01	4.07	2.02E−01
Incorporation into paints and coatings – 1-part reactive coatings	3.82E03	2.27E01	1.12	1.80E01	8.93E−01
Incorporation into paints and coatings – 2-part reactive coatings	3.46E03	2.06E01	1.02	1.63E01	8.10E−01
Use in paints and coatings at job sites	2.03E03	1.20E01	5.97E−01	9.57	4.75E−01
Formulation of TCEP containing reactive resin	4.84E03	2.88E01	1.43	2.29E01	1.13

Scenario Name	SWC[a] (µg/L)	ADD, ADR (mg/kg-day) BAF 2,198	ADD, ADR (mg/kg-day) BAF 109	LADD (mg/kg-day) BAF 2,198	LADD (mg/kg-day) BAF 109
Laboratory chemicals	3.46E01	2.05E−01	1.02E−02	1.63E−01	8.08E−03
Heritage fish ingestion rate (1,646 g/day)					
Import and repackaging	8.62E02	3.90E01	1.93	3.10E01	1.54
Incorporation into paints and coatings – 1-part reactive coatings	3.82E03	1.73E02	8.57	1.37E02	6.81
Incorporation into paints and coatings – 2-part reactive coatings	3.46E03	1.57E02	7.77	1.25E02	6.17
Use in paints and coatings at job sites	2.03E03	9.18E01	4.55	7.30E01	3.62
Formulation of TCEP containing reactive resin	4.84E03	2.19E02	1.09E01	1.74E02	8.64
Laboratory chemicals	3.46E01	1.56	7.75E−02	1.24	6.16E−02
[a] Surface water concentrations based 50th percentile flow of harmonic mean flow conditions.					

5.1.3.4.5 Incidental Oral Ingestion from Soil

Average Daily Doses (ADD) were calculated for TCEP ingestion using the following formula:

Equation 5-19.

$$ADD = \frac{C \times IR \times EF \times ED \times CF}{BW \times AT}$$

Where:

ADD = Average Daily Dose (mg/kg/d)
C = Soil Concentration (mg/kg)
IR = Intake Rate of contaminated soil (mg/d)
EF = Exposure Frequency (d)
CF = Conversion Factor (10×10^{-6} kg/mg)
BW = Body Weight (kg)
AT = Averaging Time (non-cancer: ED × EF; cancer: 78 years × EF)

Modeled soil concentrations were calculated from 95th percentile air deposition (see Section 3.3.3.2) concentrations for 100 m and 1,000 m from a hypothetical facility. These calculations were conducted for the COM-Paints-USE scenario (LOW PV − 2,500 lb, HE-95th percentile release).

Soil concentrations of 141.2 ng/g were modeled for 2,500 lb production volume, high-end release estimate for the Incorporation into paints and coatings – 1 part coatings OES using BST (Section 3.3.3.5).

The mean intake rate for children aged 3 to 6 years varied; 41 mg/d was selected for the mean intake rate and 175.6 was selected for the 95th percentile intake rate (U.S. EPA, 2017d). Body weight (18.6 kg) of a 3- to 6-year-old was estimated from the *Exposure Factors Handbook* (U.S. EPA, 2017d).

Table 5-35. Modeled Soil Oral Doses for Soil Concentrations Estimated from Air Deposition and Biosolids Application for the 2,500 lb High-End Release Estimates

OES	Distance (m)	Soil Concentration (ng/g)	Average Daily Dose (Mean Intake) (mg/kg-day)	Average Daily Dose (95th Intake) (mg/kg-day)
Use in paints and coatings at job sites[a]	100	1.14E04	2.51E–05	1.08E–04
	1,000	8.65E01	1.91E–07	8.17E–07
Incorporation into paints and coatings – 1-part coatings	N/A	1.41E02	3.11E–07	1.33E–06
[a] 95th percentile estimates				

5.1.3.4.6 Incidental Oral Ingestion from Swimming

The general population may swim in surfaces waters (streams and lakes) that are affected by TCEP contamination. Modeled surface water concentrations from EFAST 2014 were used to estimate acute doses and average daily doses due to ingestion exposure while swimming.

The following equations were used to calculate incidental oral (swimming) doses for all COUs, for adults, youth, and children:

Equation 5-20.

$$ADR = \frac{SWC \times IR \times CF1}{BW}$$

Equation 5-21.

$$ADD = \frac{SWC \times IR \times ED \times RD \times CF1}{BW \times AT \times CF2}$$

Where:

ADR = Acute Dose Rate (mg/kg/day)
ADD = Average Daily Dose (mg/kg/day)
SWC = Surface water concentration (ppb or µg/L)
IR = Daily ingestion rate (L/day)
RD = Release days (days/yr)
ED = Exposure duration (years)
BW = Body weight (kg)
AT = Averaging time (years)
$CF1$ = Conversion factor (1.0×10^{-3} mg/µg)
$CF2$ = Conversion factor (365 days/year)

A summary of inputs utilized for these estimates are present in Appendix II.2.

Table 5-36. Modeled Incidental Oral (Swimming) Doses for All COUs, for Adults, Youth and Children, for the 2,500 lb High-End Release Estimate

OES[a]	Surface Water Concentration		Adult (≥ 21 yrs)		Youth (11−15 yrs)		Child (6−10 yrs)	
	30Q5 Concentration (µg/L)	Harmonic Mean Concentration (µg/L)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)
Repackaging of import containers	862.129	1,366.528	2.97E−03	1.29E−05	4.61E−03	2.00E−05	2.60E−03	1.13E−05
Incorporation into paints and coatings – 1-part coatings	3,819.444	5,912.114	1.32E−02	5.59E−05	2.04E−02	8.67E−05	1.15E−02	4.89E−05
Incorporation into paints and coatings – 2-part reactive coatings	3,462.800	5,360.066	1.19E−02	5.07E−05	1.85E−02	7.86E−05	1.05E−02	4.43E−05
Use in paints and coatings at job sites	2,029.305	3,216.574	7.00E−03	3.04E−05	1.09E−02	4.72E−05	6.13E−03	2.66E−05
Formulation of TCEP containing reactive resin	4,844.722	6,245.374	1.67E−02	5.90E−05	2.59E−02	9.16E−05	1.46E−02	5.17E−05
Use of laboratory chemicals	34.555	54.772	1.19E−04	5.18E−07	1.85E−04	8.03E−07	1.04E−04	4.53E−07

[a] Table 3-3 provides a crosswalk of industrial and commercial COUs to OES.

5.1.3.4.7 Human Milk Exposure

Infants are a potentially susceptible subpopulation for various reasons including their higher exposure per body weight, immature metabolic systems, and the potential for chemical toxicants to disrupt sensitive developmental processes. To determine whether a quantitative analysis of infant exposure to TCEP via human milk could be informative, EPA considered available exposure and hazard information for TCEP. Based on its slight lipophilicity and small mass, TCEP has the potential to accumulate in milk. In fact, available biomonitoring studies demonstrated the presence of TCEP in human milk.

One U.S. study measured a mean wet weight concentration of 0.036 ng/mL among 100 samples collected from across the country. The detection frequency was 37 percent, and the range was n.d. to 0.8 ng/mL (Ma et al., 2019). Among non-U.S. studies, the highest concentrations were observed by Kim et al. (2014), in which TCEP was measured in 89 milk samples collected in three Asian countries (Philippines, Japan, and Vietnam), ranging from non-detect to 512 ng/g lipid weight, with an average of 0.14 to 42 ng/g. Another study by Sundkvist et al. (2010) collected milk samples from 286 mothers in Sweden, where concentrations ranged from 2.1 to 8.2 ng/g lipid weight, with a median of 4.9 ng/g. One study by He et al. (2018a) collected three milk samples in Australia, and concentrations ranged from n.d. to 0.47 ng/mL wet weight.

The hazard endpoints identified for TCEP (neurotoxicity for acute scenarios; reproductive toxicity for intermediate/chronic exposure scenarios as well as carcinogenicity) are relevant for the milk pathway and are protective of effects that may occur in infants as described in Section 5.2. Because TCEP can transfer to human milk and infants may be particularly susceptible to its health effects, EPA further evaluated infant exposures through the milk pathway for specific COUs.

EPA considered all maternal groups—occupational, consumer, and general population—when modeling TCEP concentrations in milk. Maternal doses are presented in Section 5.1 for occupational, Section 5.1.2.3 for consumer, and Section 5.1.3 for general population.

TCEP concentrations in milk were estimated based on the maternal doses using a multi-compartment physiologically-based pharmacokinetic (PBPK) model identified by EPA as the best available model (Verner et al., 2009; Verner et al., 2008), hereafter referred to as the Verner model. Only chronic (not acute) maternal doses were considered because the model is designed to estimate only continuous maternal exposure. For more information on the Verner model, including modeled compartments, data input requirements, and its system of differential equations, refer to Appendix I.5.

The Verner model requires all maternal doses to be entered as oral doses. For consumers, CEM 3.2 already provides inhalation estimates as an oral dose. The only adjustment for maternal consumer doses was to account for body weight differences. CEM 3.2 assumes a body weight of 80 kg, which is less representative of women of reproductive age because it combines males and females. To derive a dose representative of women of reproductive age, EPA applied an adjustment factor of 1.21 based on a body weight of 65.9 kg (80 kg/65.9 kg) (U.S. EPA, 2011a). The body weight of 65.9 kg is for women 16 to 21 years of age. Body weight increases with age for women of childbearing age, thus reducing overall exposure estimates. As a result, 65.9 kg is the most health protective. Furthermore, only chronic maternal doses from consumer scenarios were considered because TCEP is primarily found in consumer articles that are typically used over a long-time frame.

For OESs, high-end inhalation concentrations were converted to oral equivalent doses using the following equation:

Equation 5-22.

$$Oral\ Equivalent\ Dose = \frac{Inhalation\ Conc \times ED \times IR}{BW}$$

Where:

Oral Equivalent Dose	=	in mg/kg-day
Inhalation Conc	=	Inhalation concentration (mg/m^3)
ED	=	8-hour TWA (high-end) for workers
IR	=	Inhalation rate 1.25 m^3/hr for workers
BW	=	Body weight (65.9 kg)

For workers, maternal dermal doses include both chronic (ADD) and subchronic (SCADD). The SCADD represents repeated exposure for 30 days or more and is used with intermediate exposure scenarios. Dermal ADD and SCADD from high-end exposure levels for workers without personal protective equipment (PPE) (*i.e.*, gloves) were used to estimate infant exposure. These values are presented in Section 5.1 and adjusted by body weight. Inhalation ADD and SCADD were calculated using Equation 5-23.

Equation 5-23.

$$ADD\ or\ SCADC = \frac{D \times EF \times EY}{AT_{ED} \times AT_{EF} \times AT_{EY}}$$

Where:

D	=	Oral-equivalent inhalation dose from Equation 5-22 (mg/kg-day)
EF	=	Exposure frequency (days/yr) (22 days/year for SCADD; 250 days/year for ADD)
EY	=	Working years (1 year for SCADD; 40 years for ADD)
AT_{EF}	=	Averaging time for exposure frequency (30 days for SCADD; 365 days for ADD)
AT_{EY}	=	Averaging time for exposure years (1 year for SCADD; 40 years for ADD)

For consumers and workers, maternal doses were combined across all exposure routes for each COU: inhalation (using the oral equivalent dose calculated with Equation 5-22 and Equation 5-23), dermal, and/or oral routes. For general population, maternal doses were not combined because certain exposure pathways (*i.e.*, fish ingestion and undiluted drinking water) demonstrated significantly higher doses than others and will likely be the main driver of risk. EPA focused on these sentinel exposure pathways.

EPA used 30 years as the age of pregnancy throughout the human milk pathway. This parameter is applicable to chemicals that accumulate over time. TCEP, being only slightly lipophilic and having a half-life of less than 24 hours, is not expected to accumulate. Initial model simulations that varied the age of pregnancy confirmed this expectation. A sensitivity analysis also showed that maternal age had a negligible effect (see Appendix I.5).

Infant doses are calculated using the modeled TCEP concentrations in milk and milk intake rates described in the Agency's *Exposure Factors Handbook* (U.S. EPA, 2011a) for multiple age groups within the first year of life. The handbook presents a mean and upper (95th percentile) milk intake rate for each age group, and infant doses were calculated using both ingestion rates. The model estimated an average dose for each age group and each milk ingestion rate.

Appendix I.5.4 presents the average infant doses via the human milk pathway for all COUs within each maternal group, as well as the range of modeled TCEP concentrations in milk.

5.1.3.4.8 Dietary Exposure (Non-TSCA)

For general population exposure, literature values indicate dietary exposure from all food groups based on monitoring data (Table 5-37). The exposure dose associated with ingesting food can be derived by multiplying the concentration of chemical in food by the ingestion rate for that food and dividing by body weight (U.S. EPA, 1992). Within this overall framework, exposures could be estimated by grouping all foods and liquids together and using a generic overall exposure factor, disaggregating discrete food groups, and using food group specific exposure factors, or estimating exposures for unique food items.

Other EPA programs such as the Office of Pesticides (OPP) estimates exposure from food from using two distinct pieces of information: (1) the amount of a pesticide residue that is present in and on food (*i.e.*, residue level), and (2) the types and amounts of foods that people eat (*i.e.*, food consumption). Residue levels are primarily developed via crop field trials, monitoring programs, use information including the percent of crop treated, and commercial and consumer practices such as washing, cooking, and peeling practices. Various sources provide food consumption data, including the USDA's continuing survey of Food Intake by Individuals (CSFII), the National Health and Nutrition Examination Survey (NHANES), What We Eat in America (WWEIA). OPP uses the Dietary Exposure Evaluation Model–Food Commodity Intake Database (DEEM-FCID) model to estimate dietary exposures. (EPA-HQ-OPP-2007-0780-0001; DEEM-FCID).

For this risk evaluation, EPA used available monitoring data to estimate central tendency and high-end concentrations of TCEP in specific food groups. Figure 5-9 provides the monitoring concentrations of TCEP in various food groups.

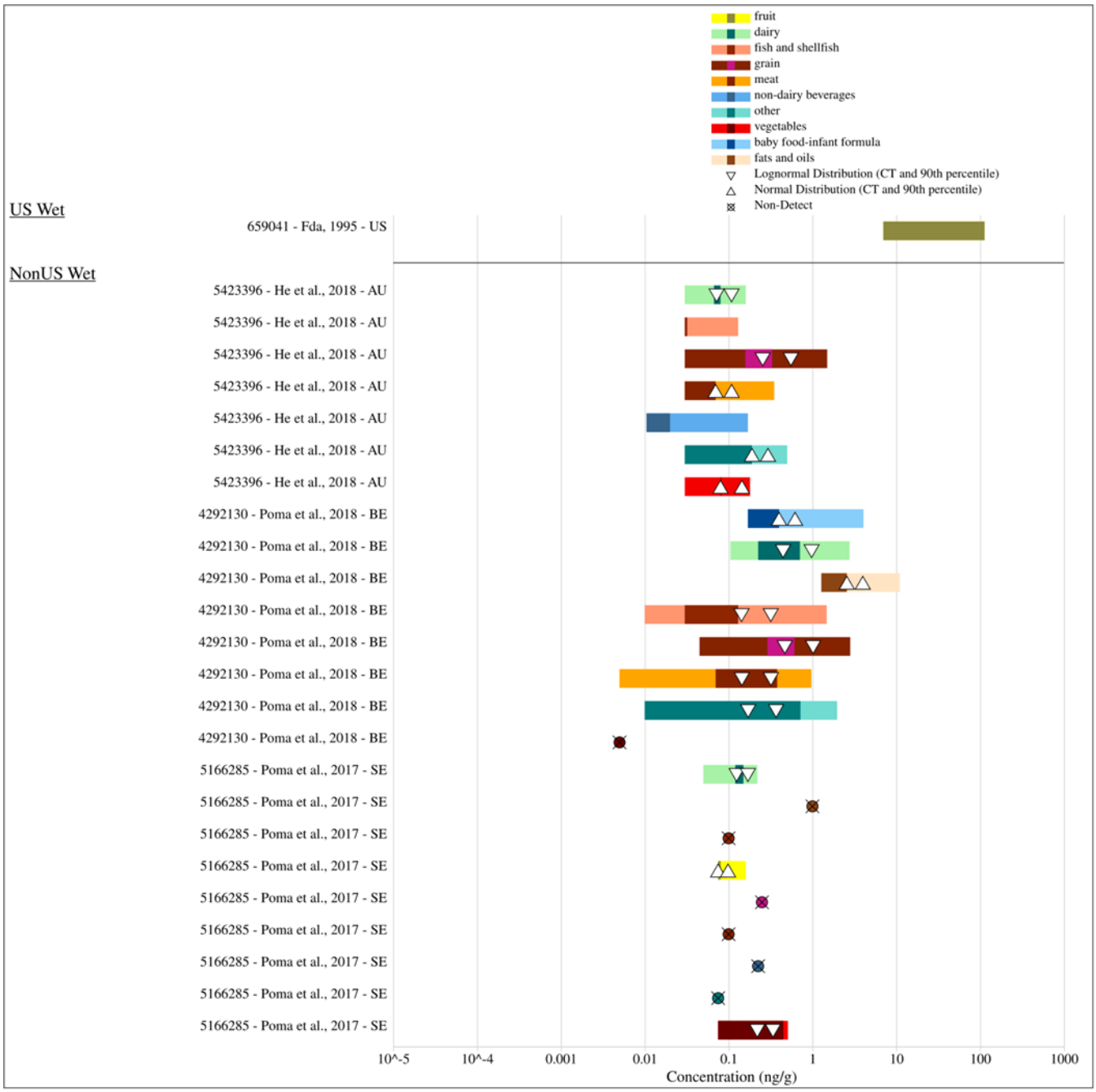


Figure 5-9. Concentrations of TCEP (ng/g) in the Wet Fraction of Dietary from 1982 to 2018

Table 5-37. Concentrations of Foods Found in the Monitoring Literature in ng/g

Food Type	Count of Estimates from All Studies (n)	Average of Arithmetic Mean Estimates for All Data	Average of 90th Percentile Estimates for All Data
Baby food/formula	1 (17)	4.0E−01	6.2E−01
Dairy	3 (45)	8.7E−02	1.3E−01
Fats and oils	1 (10)	2.6	4.0
Fish and shellfish	1 (53)	1.4E−01	3.2E−01
Fruit	1 (5)	7.5E−02	9.8E−02
Grain	2 (19)	2.3E−01	4.9E−01
Meat	2 (50)	3.0E−02	4.7E−02
Vegetables	2 (24)	1.4E−01	4.8E−01
Other	2 (14)	1.9E−01	2.9E−01

Equations

The equation used to calculate the chronic dose for each age group due to dietary exposure of fruits, grains, vegetables, meat, dairy, fats, and seafood is presented in Equation 5-24 below.

Equation 5-24.

$$ADD = \frac{FC \times IR \times ED}{AT}$$

Where:

ADD = Average Daily Dose used for chronic non-cancer risk calculations due to ingestion food group (mg/kg-day)
FC = TCEP concentration in food group (mg/g)
IR = Food group ingestion rate by age group (g/kg bw-day)
ED = Exposure Duration
AT = Averaging Time

An Australian study indicated that more than 75 percent of the estimated daily intake of TCEP came from dietary ingestion (4.1 out of 4.9 ng/kg bw/day). This study reported that grains (oatmeal, pasta, bread) contributed 39 percent and non-alcoholic beverages contributed 32 percent of total TCEP intake (He et al., 2018b). Poma et al. (2018) measured TCEP in different food groups in Belgium. In total they found food intake of TCEP to be 207 ng/d and 2.8 ng/kg/day. TCEP was most concentrated in fats (49 ng/d) and grains (49 ng/d), followed by milk (31 ng/d), meat (23 ng/d), and cheese (23 ng/d). Poma et al. (2018) suggests that the dietary intake was dominated by fats food group because of the inclusion of the fish oil supplement fat food group, for which a total of 19 g/d was estimated.

5.1.3.5 Exposure Reconstruction Using Human Biomonitoring Data and Reverse Dosimetry

EPA describes the approach used to estimate doses based on biomonitoring below. TCEP has been quantified in human samples in hair, nails (Liu et al., 2016; Liu et al., 2015), blood serum, plasma (Zhao et al., 2017), urine (Figure 5-10), and human milk (see Section 5.1.3.4.7).

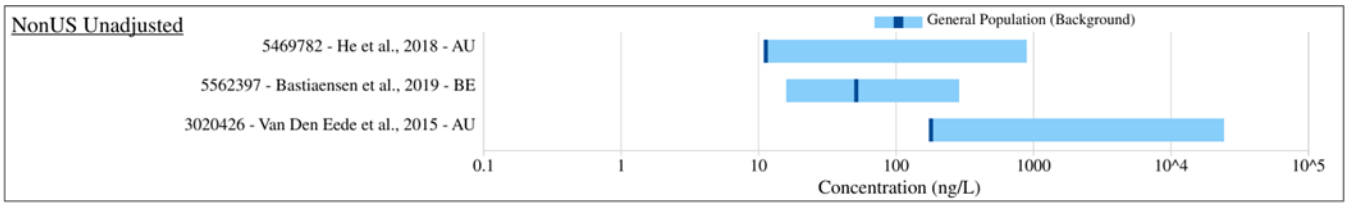


Figure 5-10. Concentrations of TCEP (ng/L) in the Unadjusted Urine from 2015 to 2019

BCEP, a metabolite of TCEP, has been reported in the 2011 to 2014 NHANES data (CDC, 2013), as well as the peer-reviewed literature (Wang et al., 2019d; He et al., 2018a; Dodson et al., 2014) (Figure 5-11, Figure 5-12).

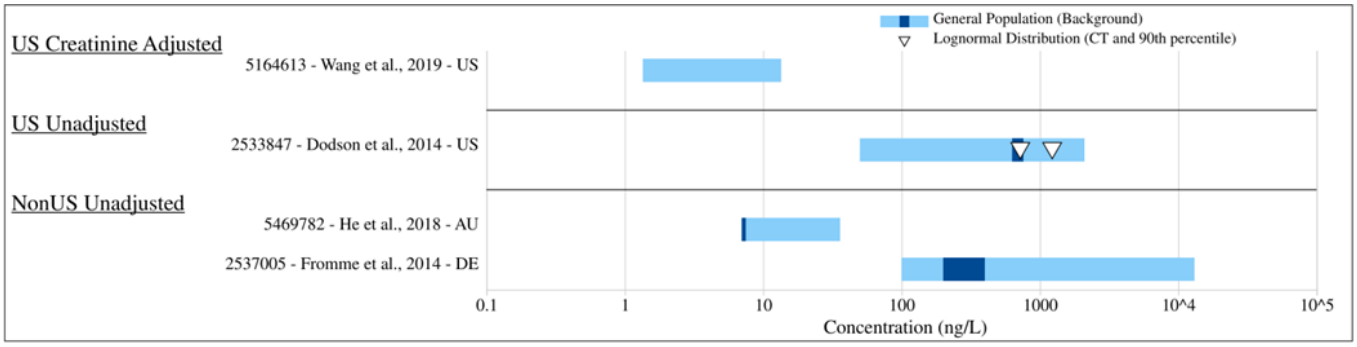


Figure 5-11. Concentrations of BCEP (ng/L) in the Creatinine-Adjusted Urine from 2014 to 2019

Urinary Bis(2-chloroethyl) phosphate (BCEtP) (creatinine corrected) (2011 - 2014)

CAS Number 3040-56-0

Metabolite of Tris(2-chloroethyl) phosphate (TCEtP)

Geometric mean and selected percentiles of urine concentrations (in µg/g of creatinine) for the U.S. population from the National Health and Nutrition Examination Survey.

Demographic Categories	Survey (Years)	Geometric Mean (95% CI)	50th Percentile (95% CI)	75th Percentile (95% CI)	90th Percentile (95% CI)	95th Percentile (95% CI)	Sample Size
Total population	11-12	0.491 (.443-.545)	.498 (.441-.558)	.969 (.811-1.11)	2.11 (1.92-2.35)	3.39 (2.96-3.79)	2409
Total population	13-14	0.447 (.396-.505)	.388 (.337-.444)	.856 (.743-.981)	2.03 (1.72-2.38)	3.94 (2.74-5.13)	2649
Age 6-11 years	11-12	0.968 (.806-1.16)	.865 (.724-1.13)	1.88 (1.51-2.14)	4.22 (2.93-5.44)	6.77 (4.22-15.6)	394
Age 6-11 years	13-14	0.855 (.720-1.02)	.833 (.676-.981)	1.60 (1.18-2.12)	4.25 (3.39-5.43)	6.83 (4.97-8.99)	418
Age 12-19 years	11-12	0.574 (.433-.760)	.537 (.404-.690)	1.23 (.788-1.90)	3.11 (1.90-5.15)	5.15 (2.74-9.05)	386
Age 12-19 years	13-14	0.516 (.429-.620)	.442 (.350-.568)	1.06 (.768-1.38)	2.33 (1.70-3.03)	4.48 (2.42-6.77)	423
Age 20+ years	11-12	0.445 (.396-.501)	.457 (.398-.524)	.855 (.748-1.01)	1.87 (1.60-2.09)	2.89 (2.29-3.49)	1629
Age 20+ years	13-14	0.408 (.362-.460)	.349 (.313-.393)	.742 (.632-.875)	1.87 (1.42-2.31)	3.12 (2.38-4.69)	1808
Males	11-12	0.449 (.413-.489)	.449 (.400-.506)	.865 (.779-1.02)	2.07 (1.77-2.43)	3.28 (2.89-4.15)	1217
Males	13-14	0.42 (.370-.476)	.373 (.322-.406)	.826 (.725-.954)	2.01 (1.50-2.43)	3.70 (2.44-5.50)	1336
Females	11-12	0.534 (.466-.612)	.534 (.464-.621)	1.04 (.879-1.22)	2.14 (1.92-2.46)	3.41 (2.76-4.48)	1192
Females	13-14	0.476 (.417-.543)	.407 (.350-.467)	.909 (.742-1.04)	2.06 (1.75-2.41)	3.99 (2.61-5.26)	1313
Mexican Americans	11-12	0.482 (.347-.669)	.509 (.381-.666)	1.05 (.673-1.61)	2.18 (1.46-3.12)	3.12 (1.97-6.71)	286
Mexican Americans	13-14	0.515 (.394-.672)	.477 (.343-.637)	1.01 (.665-1.47)	2.35 (1.57-3.03)	3.19 (2.43-6.34)	426
Non-Hispanic Blacks	11-12	0.537 (.480-.599)	.517 (.469-.595)	1.10 (.927-1.29)	2.43 (1.97-2.98)	3.79 (3.08-6.23)	666
Non-Hispanic Blacks	13-14	0.374 (.321-.435)	.328 (.267-.450)	.732 (.630-.867)	1.56 (1.18-1.80)	2.41 (1.86-3.17)	578
Non-Hispanic Whites	11-12	0.466 (.407-.535)	.481 (.399-.563)	.900 (.767-1.09)	1.92 (1.61-2.34)	2.99 (2.41-3.72)	776
Non-Hispanic Whites	13-14	0.446 (.393-.506)	.379 (.333-.437)	.857 (.731-1.00)	2.03 (1.64-2.44)	4.68 (2.51-5.58)	1012
All Hispanics	11-12	0.529 (.446-.626)	.523 (.450-.613)	1.09 (.819-1.41)	2.45 (1.97-2.94)	3.43 (2.52-5.21)	552
All Hispanics	13-14	0.495 (.406-.604)	.472 (.371-.585)	.980 (.736-1.36)	2.27 (1.69-2.75)	3.14 (2.53-3.94)	666
Asians	11-12	0.606 (.512-.716)	.587 (.473-.732)	1.29 (1.07-1.58)	2.77 (2.11-3.62)	4.78 (2.77-7.50)	327
Asians	13-14	0.477 (.412-.553)	.442 (.371-.500)	.792 (.606-1.28)	2.33 (1.51-3.46)	4.18 (2.76-9.34)	291

Figure 5-12. Concentrations of BCEP from NHANES data for the U.S. Population from 2011 to 2014

TCEP has also been detected in personal hand wipes and wristbands (Figure 5-13, Figure 5-14). Xu et al. (2016) calculated dermal absorption daily doses at a mean of 0.088 ng/kg/day.

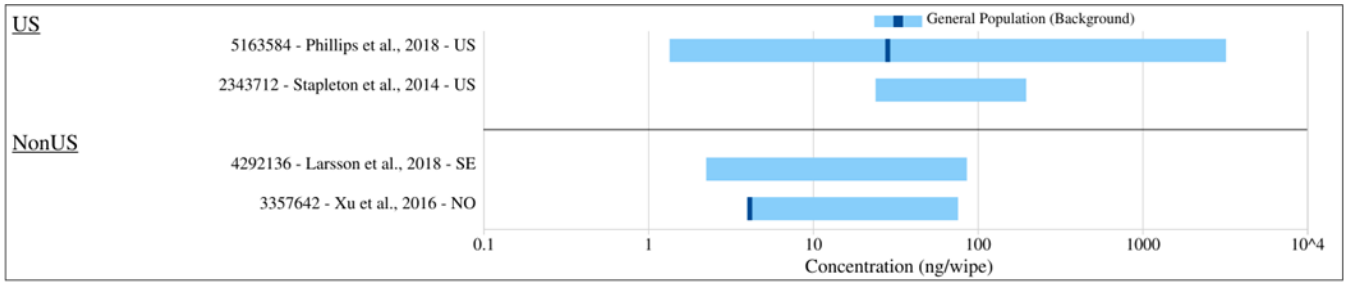


Figure 5-13. Concentrations of TCEP (ng/wipe) in Surface Wipes from 2014 to 2018

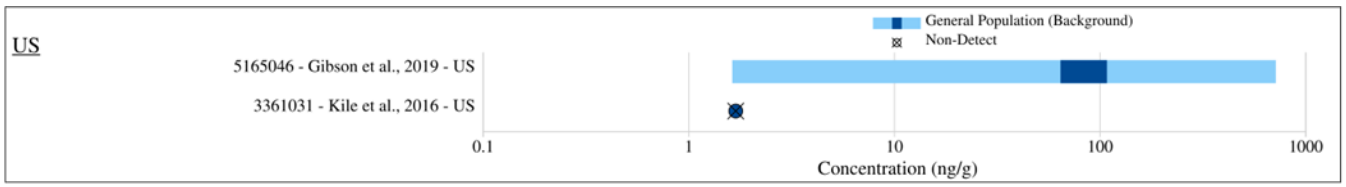


Figure 5-14. Concentrations of TCEP (ng/wipe) in Silicone Wristbands from 2012 to 2015

TCEP human biomonitoring data were previously extracted from peer-reviewed studies and curated to produce one set of summary statistics per study. A total of two peer-reviewed studies, resulting in six

datasets with sampling years from 2014 to 2018, reported TCEP data in human hair, human nails, and human urine for the U.S. general population. Additional data are available for occupational workers and highly exposed populations (Mayer et al., 2021; Shen et al., 2018; Jayatilaka et al., 2017). Researchers from the CDC measured urine samples for BCEP in 76 members of the general population and 146 firefighters who performed structure firefighting while wearing full protective clothing and respirators. BCEP was detected in 10 percent of the general population, but the median concentration was too low to quantify with acceptable repeatability and accuracy. For firefighters, BCEP was detected in 90 percent of firefighters at a median of 0.86 ng/mL (Jayatilaka et al., 2017). Table 5-38 provides the number of datasets for the general population and media type in the United States.

Table 5-38. Human TCEP/BCEP U.S. Biomonitoring Datasets by Population, Type, and Number

Population	Media Type	# of Datasets
General Population	Human Hair	2
General Population	Human Nails	1
General Population (BCEP)	Human Urine	3

Urinary BCEP was selected as a biomarker of exposure for TCEP. Urinary BCEP is a recommended target for biomonitoring of TCEP (Dodson et al., 2014). Furthermore, the robust dataset provided by the NHANES survey that varies results across demographics, age groups, and time and allows for more confidence in the values calculated by the exposure reconstruction.

Urinary volume and flow can vary between individuals due to differences in hydration status. One approach to account for this variability is by taking creatinine-adjusted values for urinary concentration. The NHANES data already provides creatinine adjusted values and more information on this adjustment can be referenced in their fourth report (CDC, 2013).

Equation 5-25.

$$DI = \frac{C_{Cr} * Cr_e}{BW * F_{ue}}$$

Where:

DI	=	Daily intake of the parent compound (mg/kg-day)
C_{Cr}	=	Creatinine adjusted concentration of analyte in urine (mg biomarker/g creatinine)
Cr_e	=	Creatinine excretion rate (g creatinine/day)
BW	=	Body weight (kg)
F_{ue}	=	Urinary excretion fraction (mg biomarker excreted/mg parent compound intake)

Kinetic data on the metabolism of TCEP is limited. Literature values have suggested a F_{ue} of 0.07 based on *in vitro* human liver microsomes (HLM) experiment, and a value of 0.13 based on *in vitro* human liver S9 fraction experiment (Van den Eede et al., 2013).

The creatinine excretion rate was normalized by body weight (in units of mg creatinine per kg bodyweight per day). Cr_e can be estimated from the urinary creatinine values reported in biomonitoring studies (*i.e.*, NHANES) using the equations of Mage et al. (2008). Assessments from Health Canada and U.S. Consumer Product Safety Commission (CPSC) have used similar approaches to quantifying creatinine excretion rate (Health Canada, 2020; CHAP, 2014).

To simplify this analysis, a few excretion rates were selected for various age groups (250 mg/day at 3 years old and 1,750 mg/day for a 20-year-old adult male) from the literature (Mage et al., 2008). The 2013 to 2014 urinary BCEP concentrations were selected as the most recent and representative concentrations for the U.S. population. Using the geometric mean and the 95th percentile concentrations from the 2013 to 2014 NHANES data, the daily intakes are estimated in Table 5-39.

Table 5-39. Reconstructed Daily Intakes from Creatinine Adjusted Urinary BCEP Concentrations from NHANES (2013–2014).

Statistic	*Fue*	3-year-old Intake (mg/kg-day)[a]	20-year-old Intake (mg/kg-day)[b]
Geomean	0.13	0.119	0.069
95th Percentile	0.13	0.952	0.525
Geomean	0.07	0.221	0.128
95th Percentile	0.07	1.768	0.975

[a] 3-year-old has a BW of 13.8 kg, and Cr_e of 250 mg/d. Used 6–11 year data for NHANES value (0.855 µg/g geomean and 6.83 µg/g 95th percentile) because no data for younger lifestages were available.
[b] 20-year-old has a BW of 80 kg, and Cr_e of 1,750 mg/d. Used Adult data for NHANES value (0.408 µg/g geomean and 3.12 µg/g 95th percentile).

Wang et al. (2019d) similarly calculated exposure doses of 19 volunteers from Albany, NY of the parent TCEP using creatinine-adjusted urinary concentrations of BCEP. Wang et al. (2019d) found TCEP doses to range 11.9 (50th percentile) to 38.6 ng/kg-bw/day. Parameters used by Wang et al. (2019d) included a 0.63 value for *Fue* based on literature values for bis(1,3-dichloro-2-propyl) phosphate (BDCIPP), and daily urine excretion values of 20 mL/kg-bw/day and 22.2 mL/kg-bw/day for children. Nevertheless, Wang et al. (2019d) stratified TCEP exposure doses by gender, ethnicity and age, and indicated that females (7.82 ng/kg-bw/day) had higher doses than males (4.35 ng/kg-bw/day), Caucasians (8.52 ng/kg-bw/day) had higher doses than Asians (4.59 ng/kg-bw/day), and individuals aged 40 and above (9.61 ng/kg-bw/day) had higher doses than lower age groups.

5.1.3.6 Summary of General Population Exposure Assessment

The general population can be exposed to TCEP from inhalation of air; dermal absorption from soils and surface waters; and oral ingestion of TCEP in drinking water, fish, and soils. Infants can also be exposed to TCEP via human milk. The sentinel exposure scenario for general population exposures was fish consumption. Oral ingestion estimates of fish consumption are provided for the general population and subsistence fishing populations, as well as Tribal populations, with high-end and central tendency BAF in Table 5-41.

5.1.3.6.1 General Population Exposure Results

Table 5-40 provides a summary of the acute oral exposure estimates for non-diluted and diluted drinking water. See Section 5.1.3.4.1 for information on dilution factors used to estimate TCEP concentrations at drinking water. Table 5-41 provides a summary of the chronic oral exposure estimates for non-diluted and diluted drinking water; drinking water estimates based on landfill leaching to groundwater; incidental ingestion of ambient waters during swimming general population and subsistence fisherman fish ingestion estimates; and 50th and 95th percentile soil intakes at 100 and 1,000 m from hypothetical facilities. Table 5-42 provides a summary of acute and chronic dermal exposures estimates of dermal exposure to surface water when swimming and exposure estimates of dermal exposure to chronic concentration of TCEP in soils. Table 5-43 below provide a summary of the relevant acute, chronic, and lifetime exposures. These summary tables present oral, dermal, and inhalation exposures as a result environmental releases (air, water, and disposal releases) for the applicable OES.

Table 5-40. General Population Acute Oral Ingestion Estimates for Drinking Water Summary Table

Acute Oral Exposure Estimates (mg/kg day)												
OES[a]	Drinking Water						Drinking Water (Diluted[b])					
	Adult (≥21 Years)	Infant (Birth to <1 Year)	Youth (16–20 Years)	Youth (11–15 Years)	Child (6–10 Years)	Toddler (1–5 Years)	Adult (≥21 Years)	Infant (Birth to <1 Year)	Youth (16-20 Years)	Youth (11–15 Years)	Child (6–10 Years)	Toddler (1–5 Years)
Import	5.5E−02	1.9E−01	4.2E−02	4.2E−02	5.4E−02	6.9E−02	4.5E−05	1.6E−04	3.4E−05	3.4E−05	4.4E−05	5.6E−05
Incorporation into paints and coatings – 1-part coatings	2.4E−01	8.3E−01	1.8E−01	1.8E−01	2.3E−01	3.0E−01	1.5E−04	5.2E−04	1.1E−04	1.1E−04	1.5E−04	1.9E−04
Incorporation into paints and coatings - 2-part reactive coatings	2.2E−01	7.6E−01	1.7E−01	1.7E−01	2.1E−01	2.7E−01	1.3E−04	4.7E−04	1.0E−04	1.0E−04	1.3E−04	1.7E−04
Use in paints and coatings at job sites	1.3E−01	4.5E−01	9.9E−02	1.0E−01	1.3E−01	1.6E−01	1.0E−04	3.7E−04	8.1E−05	8.1E−05	1.0E−04	1.3E−04
Formulation of TCEP containing reactive resin	2.5E−01	8.8E−01	1.9E−01	1.9E−01	2.5E−01	3.1E−01	5.8E−04	2.0E−03	4.5E−04	4.5E−04	5.7E−04	7.3E−04
Use of laboratory chemicals	2.2E−03	7.7E−03	1.7E−03	1.7E−03	2.2E−03	2.8E−03	1.8E−06	6.3E−06	1.4E−06	1.4E−06	1.8E−06	2.2E−06

[a] Table 3-3 provides a crosswalk of industrial and commercial COUs to OES

[b] A method was derived to incorporate a dilution factor to estimate TCEP concentrations at drinking water locations downstream from surface water release points. Because no location information was available for facilities releasing TCEP, a dilution factor and distances to drinking water intake was estimated for each relevant SIC code. Table 5-26 provides the 50th quantile distances and 50th quantile dilution factors for the relevant SIC codes.

Table 5-41. Summary of General Population Chronic Oral Exposures

Oral (mg/kg/day)								
OES[a]	**Drinking Water (Diluted[d])**	**Drinking Water**	**Drinking Water (via Leaching to Groundwater)**	**Ambient Water (Incidental ingestion)**	**Soil Intake (50th) at 100 m**	**Soil Intake (95th) at 100 m**	**Soil Intake (50th) at 1,000 m**	**Soil Intake (95th) at 1,000 m**
Repackaging of import containers	1.67E−08	2.60E−05	N/A	1.29E−05	1.24E−10	5.30E−10	1.58E−12	6.78E−12
Incorporation into paints and coatings – 1-part coatings	6.20E−08	1.15E−04	1.29E−06	5.59E−05	3.89E−09	1.67E−08	3.44E−11	1.47E−10
Incorporation into paints and coatings – 2-part reactive coatings	5.62E−08	1.04E−04	N/A	5.07E−05	5.63E−10	2.41E−09	7.42E−12	3.18E−11
Use in paints and coatings at job sites	3.92E−08	6.11E−05	N/A	3.04E−05	9.15E−06	3.92E−05	4.77E−08	2.04E−07
Formulation of TCEP containing reactive resin	2.76E−07	1.46E−04	N/A	5.90E−05	6.19E−10	2.65E−09	7.90E−12	3.38E−11
Processing into 2-part resin article	N/A	N/A	1.29E−06	N/A	5.30E−09	2.27E−08	5.41E−11	2.32E−10
Use of laboratory chemicals	6.68E−10	1.04E−06	N/A	5.20E−07	5.94E−09	2.54E−08	6.50E−11	2.78E−10
OES	**General Population (GP)**		**Subsistence Fisher (SF)**		**Tribes (Current[b])**		**Tribes (Heritage[c])**	
	BAF 2,198	**BAF 109**	**BAF 2,198**	**BAF 109**	**BAF 2,198**	**BAF 109**	**BAF 2,198**	**BAF 109**
Import	5.25E−01	2.60E−02	3.37	1.67E−01	1.89E01	9.40E−01	2.95E01	1.46
Incorporation into paints and coatings – 1-part coatings	2.33	1.15E−01	1.49E01	7.41E−01	8.40E01	4.16	1.31E02	6.47
Incorporation into paints and coatings – 2-part reactive coatings	2.11	1.05E−01	1.35E01	6.72E−01	1.18E02	3.77	1.18E02	5.87
Use in paints and coatings at job sites	1.24	6.13E−02	7.94	3.94E−01	6.94E01	2.21	6.94E01	3.44
Formulation of TCEP containing reactive resin	2.95	1.46E−01	1.90E01	9.40E−01	1.66E02	5.28	1.66E02	8.21
Processing into 2-part resin article	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Use of laboratory chemicals	2.10E−02	1.04E−03	1.35E−01	6.70E−03	1.18	3.77E−02	1.18	5.86E−02

[a] Table 3-3 provides a crosswalk of industrial and commercial COUs to OES.
[b] Current fish consumption rate at 216 g/day based on survey of Suquamish Indian Tribe in Washington (see Section 5.1.3.4.4).
[c] Heritage fish consumption rate at 1,646 g/day based on study of Kootenai Tribe in Idaho (see Section 5.1.3.4.4).
[d] A method was derived to incorporate a dilution factor to estimate TCEP concentrations at drinking water locations downstream from surface water release points. Because no location information was available for facilities releasing TCEP, a dilution factor and distances to drinking water intake was estimated for each relevant SIC code. Table 5-26 provides the 50th quantile distances and 50th quantile dilution factors for the relevant SIC codes.

Table 5-42. Summary Acute and Chronic General Population Dermal Exposures

Dermal (mg/kg/day)					
OES[a]	**Surface Water (Swimming)**	**Soil Mud at 100 m**	**Soil Activity at 100 m**	**Soil Mud at 1,000 m**	**Soil Activity at 1,000 m**
Repackaging of import containers	6.00E−06	3.93E−07	1.91E−09	5.02E−09	2.44E−11
Incorporation into paints and coatings – 1-part coatings	2.60E−05	1.23E−05	6.00E−08	1.09E−07	5.30E−10
Incorporation into paints and coatings – 2-part reactive coatings	2.40E−05	1.78E−06	8.68E−09	2.35E−08	1.14E−10
Use in paints and coatings at job sites	1.40E−05	2.90E−02	1.41E−04	1.51E−04	7.36E−07
Formulation of TCEP containing reactive resin	2.80E−05	1.96E−06	9.54E−09	2.50E−08	1.22E−10
Processing into 2-part resin article	N/A	1.68E−05	8.18E−08	1.71E−07	8.34E−10
Use of laboratory chemicals	2.41E−07	1.88E−05	9.16E−08	2.06E−07	1.00E−09

[a] Table 3-3 provides a crosswalk of industrial and commercial COUs to OES.

Table 5-43. Summary of General Population Inhalation Exposures

Inhalation (μg/m^3)		
OES[a]	**Ambient Air (50th)**	**Ambient Air (95th)**
Repackaging of import containers	4.39E−10	1.12E−09
Incorporation into paints and coatings – 1-part coatings	1.35E−08	3.51E−08
Incorporation into paints and coatings – 2-part reactive coatings	2.29E−09	1.11E−08
Use in paints and coatings at job sites	3.36E−05	8.21E−05
Formulation of TCEP containing reactive resin	2.52E−09	1.21E−08
Processing into 2-part resin article	1.96E−08	2.72E−08
Use of laboratory chemicals	2.24E−08	3.33E−08

[a] Table 3-3 provides a crosswalk of industrial and commercial COUs to OES.

5.1.3.7 Weight of Scientific Evidence Conclusions for General Population Exposure

Sections 5.1.3.2, 5.1.3.3, 5.1.3.4, and 5.1.3.5 summarize the direct and indirect exposure assessment approaches taken to estimate general population exposures. A judgment on the weight of scientific evidence supporting the exposure estimate is decided based on the strengths, limitations, and uncertainties associated with the exposure estimates. The judgment is summarized using confidence descriptors: robust, moderate, slight, or indeterminate confidence descriptors.

EPA used general considerations (*i.e.*, relevance, data quality, representativeness, consistency, variability, uncertainties) as well as chemical-specific considerations for its weight of scientific evidence conclusions.

EPA modeled three routes of exposure: (1) inhalation from ambient air; (2) oral ingestion from drinking water, fish ingestion, soil intake, and human milk intake; and (3) dermal exposures from surface water and soil. Within each of these modeled pathways, EPA considered multiple variations in its analyses (*i.e.*, multiple distances for inhalation exposures, diluted vs. non-diluted conditions for drinking water exposures, high vs. low BAF for fish ingestion) to help characterize the general population exposure estimates and to explore potential variability. The resulting exposure estimates were a combination of central tendency and high-end inputs for the various exposure scenarios. Modeled estimates were compared with monitoring data to evaluate overlap, magnitude, and trends. Table 5-44 indicates the confidence EPA has in their general population exposure estimates for each scenario.

Table 5-44. Overall Confidence for General Population Exposure Scenarios

Route	General Population Exposure Scenario	Confidence (+ Slight, ++ Moderate, +++ Robust)
Oral	Drinking Water (diluted)	+++
Oral	Drinking Water	++
Oral	Drinking Water (via Leaching to Groundwater)	++
Oral	Surface Water (incidental ingestion)	++
Oral	Fish Ingestion (SF-HighBAF)	+
Oral	Fish Ingestion (GP-HighBAF)	+
Oral	Fish Ingestion (Tribal-HighBAF, Current or Heritage Ingestion Rate)	+
Oral	Fish Ingestion (SF-LowBAF)	++
Oral	Fish Ingestion (GP-LowBAF)	++
Oral	Fish Ingestion (Tribal-LowBAF, Current or Heritage Ingestion Rate)	++
Oral	Children's Soil Intake (50th) at 100 m	+
Oral	Children's Soil Intake (95th) at 100 m	+
Oral	Children's Soil Intake (50th) at 1,000 m	++
Oral	Children's Soil Intake (95th) at 1,000 m	++
Oral	Human Milk Intake	+
Dermal	Surface Water (swimming)	++
Dermal	Children playing in Mud at 100 m	+
Dermal	Children activities with Soil at 100 m	+

Route	General Population Exposure Scenario	Confidence (+ Slight, ++ Moderate, +++ Robust)
Dermal	Children playing in Mud at 1,000 m	++
Dermal	Children activities with Soil at 1,000 m	++
Inhalation	Inhalation 100 m – MetCT	++
Inhalation	Inhalation 1,000 m – MetCT	+++
Inhalation	Inhalation 100 m – MetHIGH	++
Inhalation	Inhalation 1,000 m – MetHIGH	+++

5.1.3.7.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the General Population Exposure Assessment

No site-specific information was reasonably available when estimating release of TCEP to the environment. Release estimates were provided for hypothetical sites. As such, there is considerable uncertainty in the production volume estimate (2,500 lb), and the resulting environmental release estimates. In addition, there is uncertainty in the relevancy of the monitoring data to the modeled estimates presented in this evaluation. Manufacturers have begun to phase out the use of TCEP as demonstrated by the declining production volumes and the introduction of new regulations (*e.g.*, California TB 117-2013) that have shifted the use away from TCEP and other organophosphate flame retardants. For each release scenario, due to the lack of reasonably available information on the distribution of TCEP across industry sectors, it was assumed that the full production volume of 2,500 lb was released for each COU. This conservative assumption further contributes to the uncertainty when characterizing the resulting modeled exposure estimates.

Drinking Water Estimates

Exposure estimates for the diluted drinking water estimates ranged from 0.022 to 9.167 ug/L which is one to two orders of magnitude greater than the estimates found in the monitoring literature in the US: average of 4.9 ng/L and 90th percentile of 9.5 ng/L. The modeled estimates are more in line with a study of drinking water systems from 19 drinking water systems across the US, where the median measured concentrations of TCEP in finished water was 0.12 ug/L (Benotti et al., 2009). There is uncertainty surrounding the distance between release sites and drinking water intake locations. Nevertheless, the assessment conducted analyses for diluted and undiluted drinking water estimates to account for this uncertainty. Only 5 percent of surface water samples detected TCEP in the Water Quality Portal (see Section 3.3.2.4).

The systematic review resulted in only a few cases demonstrating migration of TCEP to groundwater from suspected landfill leachate (Buszka et al., 2009; Barnes et al., 2004; Hutchins et al., 1984). Furthermore, there are inherent uncertainties associated with estimating exposures from the transport of chemicals through various media (*e.g.*, landfill disposal to groundwater to drinking water). In addition, TCEP was detected in only 2 percent of groundwater samples in the WQP (see Section 3.3.3.7).

EPA has robust confidence in the diluted drinking water estimate, whereas EPA has moderate confidence in the non-diluted drinking water estimates. EPA has slight confidence in the drinking water estimates as a result of leaching from landfills to groundwater and subsequent migration to drinking water wells.

Fish Ingestion Estimates
To account for the variability in fish consumption across the United States, fish intake estimates were considered for both subsistence fishing populations and the general population. In estimating fish concentrations, diluted surface water concentrations were not considered. It is unclear what level of dilution may occur between the surface water at the facility outfall and habitats where fish reside. A considerable source of uncertainty in the fish ingestion estimates was the selection of a BAF. Two BAFs were considered (109 and 2,198 L/kg wet weight) due to uncertainties with the high-end BAF value and to account for various fish species. No monitoring data were available indicating the consumption of fish containing TCEP. EPA did find very limited monitoring data indicating TCEP concentrations in fish tissue. The reported wet weight fish tissue concentrations in the monitoring data are several magnitudes lower than the modeled estimates with either the low or high BAF.

Soil and Swimming Ingestion/Dermal Estimates
Two scenarios (children playing in mud and children conducting activities with soil) captured a wider range of potential exposures to TCEP containing soils. EPA's *Exposure Factors Handbook* provided detailed information on the child skin surface areas and event per day of the various scenarios (U.S. EPA, 2017d). It is unclear how relevant dermal and ingestion estimates from soil exposure are as TCEP is expected to migrate from surface soils to groundwater. Furthermore, there are inherent uncertainties associated with estimating exposures from the transport of chemicals through various media (*e.g.,* air to land and subsequent soil ingestion and dermal absorption).

There are no recorded values of TCEP in soils in the United States. A study in Germany reported highest concentrations of TCEP in soil, 1 day after snow melt at 23.48 ng/g (Mihajlovic and Fries, 2012). The 95th percentile estimated modeled concentrations of soil because of air deposition for the use of paints and coatings at job sites scenario was 1.14×10^{4} ng/g at 100 m and 8.65×10^{1} ng/g at 1000 m. The foreign monitoring data is within range of the modeled soil estimates via air deposition. The child playing in mud scenario assumes that the child will be exposed all over the arms, hands, legs, and feet. Furthermore, there are uncertainties regarding the relevance of the selected dermal absorption fraction of 35.1 percent as discussed in Section 5.1.2.4.1.

Non-diluted surface water concentrations were used when estimating dermal exposures to adults and youth swimming in streams and lakes. TCEP concentrations will dilute when released to surface waters, but it is unclear what level of dilution will occur when the general population swims in waters with TCEP releases.

Inhalation
Modeled inhalation estimates are provided for a range of general population scenarios: various distances from the emitting facility (10, 30, 60, 100, 1,000, 2,500, 10,000 m), two meteorology conditions (Sioux Falls, South Dakota, for central tendency meteorology and Lake Charles, Louisiana, for higher-end meteorology), central tendency and high-end release estimates for the low production volume (2,500 lb), and 10th, 50th and 95th percentile exposure concentrations. Because no site-specific information for TCEP release is available, EPA was unable to identify specific meteorological conditions that were relevant to the air release.

Furthermore, EPA did not consider indoor to outdoor transfer of TCEP for general population inhalation exposures. As discussed in Section 3.3.1.2.1, there are uncertainties surrounding the particle vs. gas phase distribution of TCEP. It is unclear how sensitive this parameter is to the final inhalation and deposition results. Use of paints and coatings at jobs sites was the OES with the highest modeled exposure estimates (8.21×10^{-5} ppm or 960 ng/m^{3}) which is four orders of magnitude higher than the

average 90th percentile estimates for US data (3.1×10^{-1} ng/m^3). Where information was unavailable, EPA relied on AERMOD defaults when estimating inhalation exposures.

Reverse Dosimetry

Exposure estimates via reverse dosimetry provide an estimate of exposure based on biomonitoring concentrations. Although NHANES provides nationally representative biomonitoring estimates, there is no way to attribute the sources of TCEP to these biomonitoring estimates. NHANES only provided urinary BCEP concentrations for the years 2011–2014. It is anticipated that these concentrations have likely decreased due to the decrease in production volume and phase-out of TCEP and accompanying shift to other alternatives. In addition, there are modeling uncertainties associated with the reverse dosimetry calculation of estimating internal TCEP doses from BCEP metabolite concentrations. Uncertainties include creatinine adjustment and the accuracy of urinary excretion fraction. NHANES biomonitoring estimates do not differentiate between TSCA and non-TSCA exposures. Hence, the reverse dosimetry estimates will be an overestimate of the actual exposure levels due to TSCA COUs. The 95th percentile estimate for TCEP intakes from reverse dosimetry is 1.8 mg/kg/day for children three years of age and 0.98 mg/kg/d for adults 20 years of age. These reverse dosimetry estimates of TCEP were within an order of magnitude of the highest general population, low BAF, oral fish intake estimates (0.33 mg/kg/day for formulation of TCEP containing reactive resins OES). This corroboration builds confidence in the plausibility of the general population fishing exposure estimates.

Key Variables, Parameters for General Population Assessment

Table 5-45 provides a list of key variables and parameters that influence the general population exposure assessment. This table presents the sources of uncertainties and variabilities of key parameters for the different exposure scenarios. For more detail on a comprehensive set of parameters used in the general population exposure assessment, please see Appendix I.

Table 5-45. Qualitative Assessment of the Uncertainty and Variability Associated with General Population Assessment

Variable Name	Relevant Section(s) in Risk Evaluation	Data Source(s)	Confidence (Robust, Moderate, Slight)
General population exposure assessment			
Environmental release estimates	3.2	EPA Modeled	+
Environmental monitoring data	3.3.1.1, 3.3.2.2, 3.3.2.3, 3.3.2.7, 3.3.2.8, 3.3.3.1, 3.3.3.3, 3.3.3.6, 3.4.1.1, 3.4.1.2, 3.4.2.1	Extracted and evaluated data (all) plus key studies	++
Fish intake rate	5.1.3.4.2	(U.S. EPA, 2014a), (U.S. EPA, 2011a), (Ridolfi, 2016)	++
Exposure factors and activity patterns	Appendix I	*Exposure Factors Handbook* (U.S. EPA, 2017d)	+++
Key parameters for modeling environmental concentrations			
Water modeling defaults: river flow, dimensions, characteristics	3.3.2.5, Appendix I	EFAST/VVWM – PSC defaults	++

Variable Name	Relevant Section(s) in Risk Evaluation	Data Source(s)	Confidence (Robust, Moderate, Slight)
General population exposure assessment			
Air modeling defaults: meteorological data, indoor/outdoor transfer,	3.3.1.2, Appendix I	IIOAC/AERMOD defaults	++
Landfill leachate concentrations and landfill loading rates	3.3.3.8	DRAS defaults, (Masoner et al., 2016; Masoner et al., 2014b)	+
Drinking water treatment and wastewater treatment removal	2.2.2, F.2.5.2, F.2.5.3	(Life Sciences Research Ltd, 1990b, c) (Padhye et al., 2014; Benotti et al., 2009; Snyder et al., 2006; Westerhoff et al., 2005; Stackelberg et al., 2004).	++
BAF	2.2, 5.1.3.4.2	(Guo et al., 2017b) and (Liu et al., 2019a).	+ (high BAF) ++ (low BAF)
Gas phase vs. particulate phase distribution, particle size	3.3.1.2.1, Appendix I	(Okeme, 2018), (Wolschke et al., 2016).	++
Human biomonitoring and reverse dosimetry parameters			
Biomonitoring data	5.1.3.5	Extracted and evaluated data (all) plus key studies	++
Fraction of urinary excretion	5.1.3.5	(Van den Eede et al., 2013).	++
Half-life in the body	Appendix I	https://comptox.epa.gov/dashboard/chemical/adme-ivive-subtab/DTXSID5021411	++

Finally, EPA did not consider all possible exposure pathways, but rather focused on pathways that were within the scope of its conceptual model and most likely to lead to exposures for the general population. This may result in a potential underestimation of exposure in some cases. Examples of exposure pathways that were not considered include incidental ingestion of suspended sediment and surface water during recreational swimming and ingestion of non-fish seafood such as aquatic invertebrates or marine mammals. However, EPA expects these exposures to be less than those that were included in the overall assessment for the general population. As such, their impact will likely be minimal and would be unlikely to influence the overall magnitude of the results.

5.1.3.7.2 Strengths, Limitations, and Key Sources of Uncertainty for the Human Milk Pathway

Strengths of the Milk Model and Overall Approach

The Verner model integrates critical physiological parameters that includes pre- and postpartum changes in maternal physiology, lactation, and infant growth. In addition, EPA implemented the Verner Model in "R" to readily enable adjustments tailored to risk evaluation needs. For example, risk assessors can tailor model inputs such as maternal doses to be more representative of women of reproductive age, thus reducing the potential for underestimating infant doses. The overall approach to analyze infant exposure

through human milk also considers a wide range of data sources. It incorporates (1) available biomonitoring data (see Section 5.1.3.4.7) on TCEP's potential transfer to human milk and its effects on infants or development, (2) chemical properties influencing TCEP excretion in human milk, and (3) the best available quantitative approaches for exposure. The half-life for TCEP was estimated using high-throughput toxicokinetics, which predicts *in vivo* behavior based on i*n vitro* measures from human hepatocytes and plasma using simple toxicokinetics model (Wambaugh et al., 2019). These considerations were integrated into EPA's decision to proceed with a quantitative exposure analysis.

Uncertainty Associated with Predicting Accumulation in Milk

Well established criteria exist for predicting passive transport of chemicals across cell membranes, including size, lipophilicity, water solubility, acid/base properties, and ionization. Nevertheless, predictions of chemical accumulation via passive transport may be confounded by the pH gradient between plasma and milk. The pH of human milk (7.08) is lower than plasma (7.42). Chemicals that are weak acids or bases may accumulate to higher levels in milk than predicted based on passive diffusion due to the pH gradient. For chemicals, the pH change can modify the molecular structure in a manner that retards diffusion into the plasma medium that is more basic (Alonso-Amelot, 2018; Wang and Needham, 2007). It is not known if TCEP is subjected to ionization trapping because of the pH gradient. Furthermore, it is not known whether TCEP is a substrate for active transporters in mammary epithelial cells. These gaps in could introduce uncertainties in how much TCEP accumulates in milk, and thus an infant's level of exposure.

Uncertainty in the Multi-compartment PBPK Model Inputs and Outputs

The uncertainties associated with deriving maternal doses for workers, consumer, and general population scenarios are described in Sections 5.1.1.4.1, 5.1.2.4.1, 5.1.3.7.1, respectively. Furthermore, the model requires oral maternal doses. However, exposure can occur through oral, dermal, and inhalation pathways for workers, consumers, and the general population. While an inhalation-to-oral extrapolation of exposures was performed for TCEP to run the model, differences in absorption potential and/or surface area between the lungs and gastrointestinal tract can introduce uncertainties into the modeled TCEP concentrations in milk. Also, enzymes involved in xenobiotic metabolism are variably expressed across many organs and tissues, including sites of absorption such as the gastrointestinal tract, lung, and skin (Bonifas and Blomeke, 2015; Lipworth, 1996). However, the liver has the highest detoxification capacity in mammals (Schenk et al., 2017). After oral administration, xenobiotic chemicals absorbed from the gastrointestinal tract first pass through the liver before reaching the systemic circulation. This "first-pass effect" may result in lower systemic bioavailability for chemicals absorbed via the oral route compared to dermal and inhalation routes (Mehvar, 2018). Therefore, route-to-route extrapolations may result in underestimating TCEP concentrations in milk. For TCEP, however, the effect on TCEP concentrations in milk is expected to be small given its relatively slow clearance rate (*i.e.*, TCEP can partition to other parts of the body because it is not rapidly metabolized by the liver). The toxicity values used to estimate risks from TCEP exposure are also all based on oral studies (see Section 5.2.7), and EPA assumed absorption for the oral route is 100 percent (see Section 5.2.5).

Finally, a TCEP-specific source of uncertainty may derive from calculated rather than measured half-life values and partition coefficients. See Table_Apx I-18 in Appendix I.5.1 for more information. The calculated partition coefficients derive from K_{OW} values, lipid and water fractions of blood and tissue, and previously reported tissue compositions (Verner et al., 2008; Price et al., 2003). The lack of quantifiable uncertainty in these calculated values precludes a robust analysis of their contribution to overall model uncertainty. However, a sensitivity analysis was conducted for TCEP to evaluate certain chemical parameters' effects on model estimates. Overall, the model is sensitive to half-life where an increase or decrease leads to a near equivalent change in the infant milk dose. K_{OW}, which is used to

calculate partition coefficients, has a modest effect on the predicted infant dose. Infant doses are also insensitive to alterations in milk lipid fraction. Appendix I.5.1 describes the results of the sensitivity analysis in greater details.

Uncertainty and Variability Associated with Infant Exposure Dose: The Verner Model assumes exclusive milk intake for the infant until the end of lactation for up to 12 months. It does not include a weaning period where formula and/or solid foods are gradually introduced. Therefore, the model may overestimate infant intake during periods of transition between human milk and formula or solid food intake.

Weight of Scientific Evidence for Human Milk Pathway

The weight of scientific evidence judgement integrates various considerations to determine confidence in the evaluation of infant's exposure to TCEP via human milk. The strengths of the Verner PBPK Model are that it is peer-reviewed and well-documented (Verner et al., 2009; Verner et al., 2008). However, the model was not validated for TCEP because data were unavailable. It was validated using data on persistent organic pollutants, which are more lipophilic and have much longer half-lives than TCEP (*i.e.*, 6–27 years vs. <24 hours) measured in mothers and infants from a Northern Quebec Inuit population. Furthermore, it is unclear how uncertainties in model inputs like partition coefficients affect modeled TCEP concentrations in milk. Despite these uncertainties, the Verner PBPK model reflects best available data identified by EPA, and as such, EPA relied on it to evaluate the human milk pathway. Biomonitoring data, albeit limited and discussed in Section 5.1.3.4.7, are available to ground truth modeled concentrations against measured data. Some of the lowest modeled TCEP concentrations in milk are below measured concentrations, but it is important to note that biomonitoring data does not distinguish between exposure routes nor allow for source apportionment (*i.e.*, exposure from TSCA COUs cannot be isolated). EPA has slight confidence in the maternal doses as inputs to the Verner model human milk. The slight confidence is a result of applying various conservative assumptions in the absence of site-specific information, route-to-route extrapolation, and other reasons described in Sections 5.1.1.4.1, 5.1.2.4.1, and 5.1.3.7.1. The infant MOEs based on the modeled concentrations are still 1 to 2 orders of magnitudes higher than the mothers, in addition to modeled TCEP concentrations in human milk being higher than measured data for most COUs. Therefore, EPA has moderate overall confidence that the exposed mothers are more sensitive than infants exposed to TCEP through the human milk pathway, and therefore protecting the mother is protective of the infant.

5.1.4 Aggregate Exposure Scenarios

EPA has defined aggregate exposure as "the combined exposures from a chemical substance across multiple routes and across multiple pathways (40 CFR 702.33)." The fenceline methodology (U.S. EPA, 2022b), (Draft Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities Version 1.0) aggregated inhalation estimates and drinking water estimates from co-located facilities. Due to the lack of reasonably available site-specific data for TCEP, EPA was unable to employ this approach.

Source attribution is a key challenge when attempting to characterize an aggregate exposure scenario. When considering pathway specific estimates and aggregate exposures, there is uncertainty associated with which pathways co-occur in each population group. Further, there is variability within a given exposure pathway. For the same exposure scenarios, central tendency estimates are more likely to occur than high-end estimates.

Aggregate Exposure across Routes

EPA presents total acute and chronic exposure estimates in the consumer assessment (see Section 5.1.2.3 and Appendix J.1.3). Generally, exposure estimates to consumer articles are dominated by a single route (*i.e.*, mouthing by infants and children). However, there are cases where aggregate exposures across routes are important to consider when inhalation, dermal and ingestion estimates are within similar ranges, and estimating risks from one route of exposure may underestimate the risk to a consumer COU. The includes Figure 5-15 that aggregates the consumer exposure estimates by route (inhalation, dermal, ingestion) for each COU, lifestage combination (U.S. EPA, 2024b).

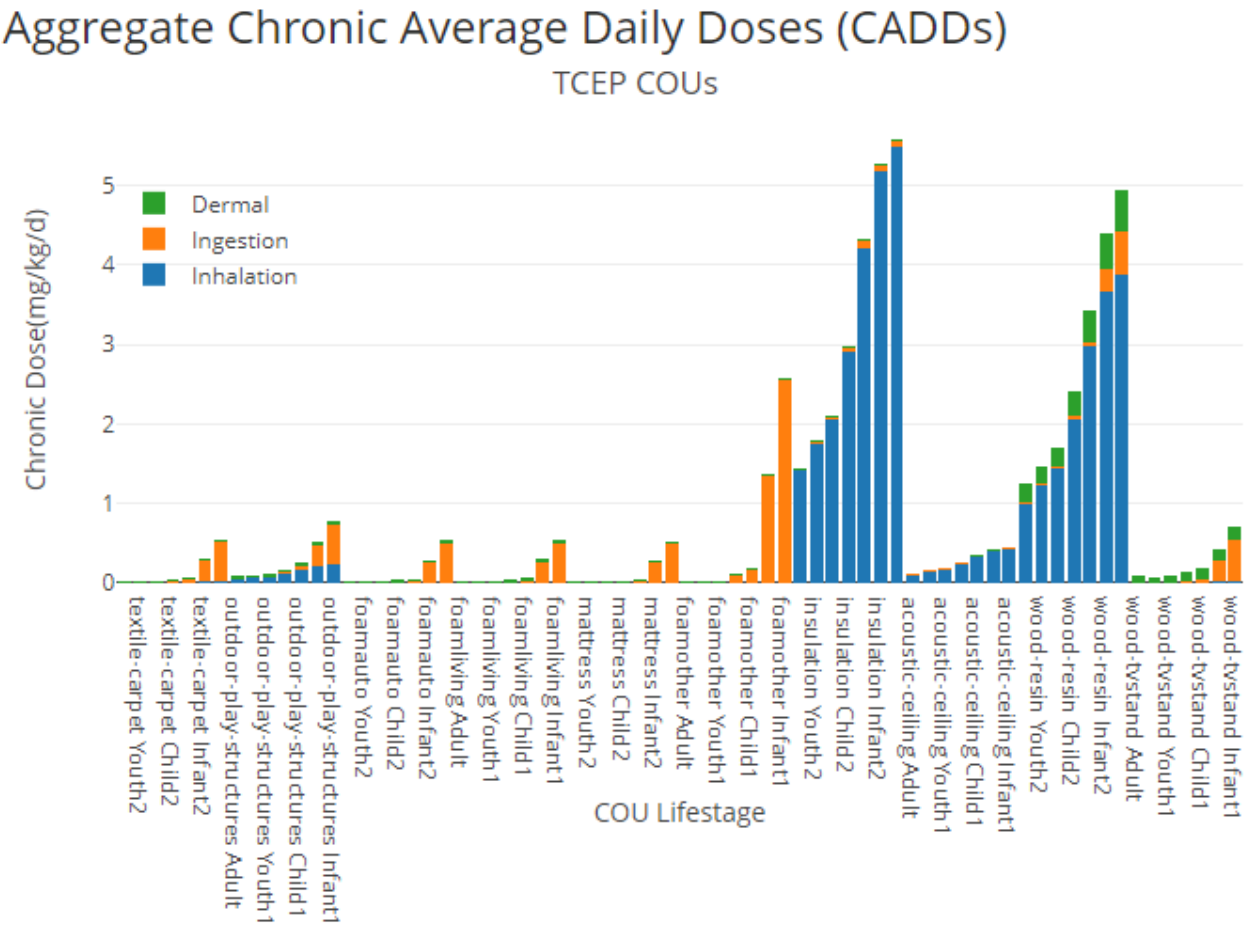


Figure 5-15. Aggregate CADDs for Each Consumer COU, Lifestage

Figure 5-15 demonstrates that for certain consumer products (outdoor play structures, wood resin and wooden TV stand), exposure is not dominated by a single route and that it is important to consider multiple routes of exposure. Section 5.3.4 further discusses the aggregate risk characterization of these COUs and the relevant lifestages.

Aggregate Exposure across COUs

A worker may be involved in multiple activities that use TCEP that have varying multiple OESs. Consumers may have multiple articles at home that contain TCEP. For example, a consumer could hypothetically have insulation with TCEP and have wooden articles containing TCEP in the home. No evidence was found suggesting that a single consumer is exposed through multiple consumer COUs. Due to lack of reasonably available data indicating co-exposures of multiple TCEP containing activities

or products in the occupational and indoor environment, EPA did not assess aggregate exposure across consumer, commercial, or industrial COUs.

Aggregate Exposure across Exposure Scenarios

A child in the general population may be exposed to TCEP via soil ingestion and drinking water. In the case of the general population exposure estimates, a production volume of 2,500 lb used to estimate releases for each individual OES. EPA did not aggregate exposure estimates to the general population because exposure estimates were based on release estimates assuming a production volume of 2,500 lb per OES, and an aggregation would double count the production volume. Thus, in the example above the soil ingestion estimates were based on 2,500 lb per OES, and the drinking water estimate was based on 2,500 lb per OES. Thus, it could be misleading to aggregate these exposure estimates.

Furthermore, a child may be exposed to TCEP via mouthing of consumer articles as well as via drinking water, fish ingestion, or inhalation of ambient air. The source of consumer exposure is via the consumer purchase of finished articles containing TCEP, whereas the source of environmental exposure from soil is due to the environmental release from a nearby hypothetical facility. EPA did not quantitively assess aggregate exposure across exposure scenarios because no data was available indicating the co-exposure of TCEP from multiple exposure scenarios.

5.1.5 Sentinel Exposures

EPA defines sentinel exposure as "the exposure from a chemical substance that represents the plausible upper bound of exposure relative to all other exposures within a broad category of similar or related exposures (40 CFR 702.33)." In terms of this risk evaluation, EPA considered sentinel exposures by considering risks to populations who may have upper bound exposures; for example, workers and ONUs who perform activities with higher exposure potential, or consumers who have higher exposure potential or certain physical factors like body weight or skin surface area exposed. EPA characterized high-end exposures in evaluating exposure using both monitoring data and modeling approaches. Where statistical data are available, EPA typically uses the 95th percentile value of the available dataset to characterize high-end exposure for a given condition of use. For general population and consumer exposures, EPA occasionally characterized sentinel exposure through a "high-intensity use" category based on elevated consumption rates, breathing rates, or user-specific factors.

EPA varied the general population exposure scenarios to help characterize the risk estimates. Risk estimates were calculated for diluted and non-diluted drinking water conditions, soil intakes for children's activities with soil and playing in mud scenario, and inhalation estimates at various distances from a hypothetical facility. Furthermore, fish ingestion intakes were estimated using a high and low BAF value for both subsistence fisherman and the general population. The sentinel exposure for these general population exposure scenarios was fish ingestion for subsistence fisherman and fishers who are members of Tribes.

The sentinel exposure for the consumer assessments by route were inhalation from building and construction materials (roofing insulation) for consumers, oral ingestion of TCEP from children's mouthing of foam seating and bedding products (foam toy blocks), and children's dermal absorption of TCEP from wood resin products (wood flooring).

5.2 Human Health Hazard

Human Health Hazards (Section 5.2):
Key Points

EPA evaluated the reasonably available information for human health hazards, including consideration of the potential for increased susceptibility across PESS factors and acute, intermediate, and chronic exposures to TCEP (see also Section 5.3.3 and Appendix D). The key points of the human health hazard assessment are summarized below:

- EPA concluded that TCEP is likely to cause neurotoxicity, kidney toxicity, and reproductive toxicity based on consideration of evidence across epidemiology studies, animal toxicity studies with apical outcomes, and mechanistic information.
- EPA concluded that TCEP is likely to be carcinogenic when considering human epidemiological studies, a 2-year rodent bioassay, and mechanistic evidence.
- Laboratory animal studies identified possible susceptible sex/lifestages (1) males for reproductive toxicity (with adolescents as potentially most susceptible) (moderate evidence); (2) neurotoxicity, with greater sensitivity among females (and potentially during pregnancy) (robust evidence); and (3) reproductive/developmental targets resulting in decreased fertility and viability of offspring (slight to moderate evidence).
- Human epidemiological data show slight evidence for possible effects in susceptible subpopulations including developmental effects on growth and gestational age in children of exposed mothers as well as decreases in IQ among children in with lower socioeconomic status. There are possible sex differences for some developmental outcomes.
- The acute non-cancer endpoint for TCEP was derived from tremors in pregnant female rats in a developmental neurotoxicity study with a NOAEL of 40 mg/kg-day.
    - Human equivalent dose (HED) (daily) = 9.46 mg/kg-day
    - Human equivalent concentration (HEC) (continuous) = 51.5 mg/m^3 (4.41 ppm), extrapolated from oral data
    - Benchmark margin of exposure (MOE) = 30, based on 10× intraspecies uncertainty factor (UF) and 3× interspecies UFs
- The intermediate/chronic endpoint for TCEP was derived from reproductive organ effects (decreases in seminiferous tubule numbers in adolescent male mice) in a 35-day oral feeding study with a BMDL of 21 mg/kg-day.
    - HED (daily) = 2.73 mg/kg-day
    - HEC (continuous) = 14.9 mg/m^3 (1.27 ppm), extrapolated from oral data
    - Benchmark MOE = 30, based on 10× intraspecies and 3× interspecies UFs
- The cancer endpoint for TCEP is based on the observation of kidney adenomas or carcinomas in male rats from a 2-year oral gavage study.

5.2.1 Approach and Methodology

EPA used the approach described in Figure 5-16 to evaluate, extract, and integrate evidence for TCEP human health hazard and conduct dose-response modeling. This approach is based on the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), updates to the systematic review processes presented in the TCEP Systematic Review Protocol (U.S. EPA, 2024p), and the *Framework for Human Health Risk Assessment to Inform Decision Making* (U.S. EPA, 2014b).

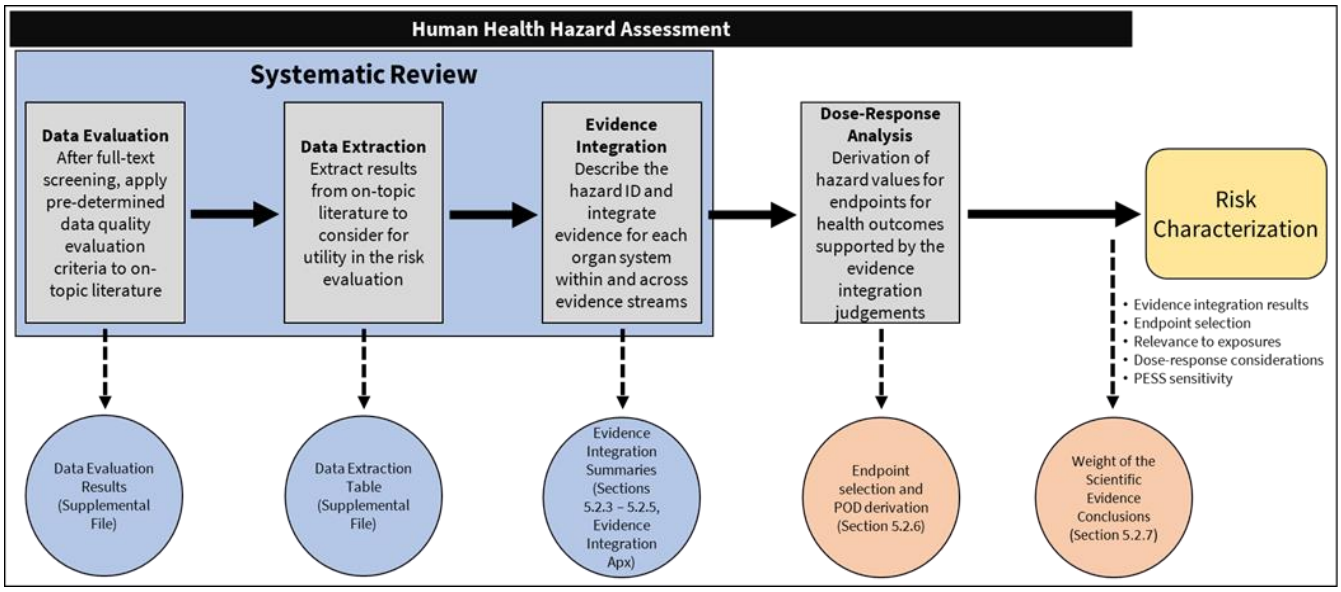


Figure 5-16. EPA Approach to Hazard Identification, Data Integration, and Dose-Response Analysis for TCEP

For the human health hazard assessment, EPA systematically reviewed data sources identified in the literature search conducted in 2019. The literature search was updated in 2024 to search specifically for human epidemiological and inhalation animal toxicity studies to address limited data for these endpoints; EPA found several epidemiological studies but no new inhalation animal toxicity studies. EPA first screened titles and abstracts and then full texts for relevancy using population, exposure, comparator, and outcome (PECO) screening criteria. Studies that met the PECO criteria were then evaluated for data quality using pre-established quality criteria and metrics. Although EPA used data quality criteria for many studies, EPA has not developed such criteria for toxicokinetics data other than dermal absorption studies. EPA also did not formally evaluate mechanistic studies for data qualtiy but did consider whether selected genotoxicity studies followed existing guidelines. Following data quality evaluation, EPA extracted the toxicological information from each evaluated study, including studies with uninformative quality determinations. The results of data quality evaluation and extraction of key study information for dermal absorption studies as well as human and animal phenotypic toxicity studies are presented in supplemental files (U.S. EPA, 2024q, s, y, z).

EPA considered studies that received low, medium, or high overall quality determinations for hazard identification, evidence integration, and dose-response analysis; only one part of the dermal absorption study was low quality. Information from studies of uninformative quality were only discussed on a case-by-case basis for hazard identification and evidence integration and were not considered for dose-response analysis. For example, if an uninformative study identified a significantly different outcome compared with high- or medium-quality studies and the uninformative rating was not expected to influence the specific results being discussed, EPA considered the uninformative study for the hazard outcome being considered.

After evaluating individual studies for data quality, EPA summarized hazard information by hazard outcome and considered the strengths and limitations of individual evidence streams (*i.e.*, human studies of apical (phenotypic) endpoints if available, animal toxicity studies with phenotypic endpoints, and supplemental mechanistic information). The Agency integrated data from these evidence streams to arrive at an overall evidence integration conclusion for each health outcome category (*e.g.*, reproductive toxicity). When weighing and integrating evidence to estimate the potential that TCEP may cause a

given human health hazard outcome, EPA uses several factors adapted from Sir Bradford Hill (Hill, 1965). These elements include consistency, dose-response relationship, strength of the association, temporal relationship, biological plausibility, and coherence, among other considerations. Sections 5.2.3, 5.2.3.2.9, and 5.2.4 discuss hazard identification and evidence integration conclusions for non-cancer hazard outcomes, genotoxicity information, and cancer, respectively. Section 5.2.4 also presents an MOA analysis for cancer.

EPA conducted dose-response analysis for the health outcome categories that received a judgment of *likely* ("evidence indicates that TCEP exposure likely causes [health effect]") during evidence integration. The Agency also conducted dose-response analysis for health outcomes that resulted in *suggestive* evidence and compared the PODs (*i.e.*, human equivalent concentrations [HECs] or human equivalent doses [HEDs] divided by UFs for non-cancer effects; IURs or cancer slope factors [CSFs] for cancer effects) for both *likely* and *suggestive* evidence integration conclusions (U.S. EPA, 2024k). However, EPA only considered the health outcomes and associated specific health effects from the *likely* evidence integration judgments to use as toxicity values when estimating risks from exposure to TCEP.

If supported by statistically and/or biologically significant results and if the dose-response data could be reasonably modeled, EPA conducted benchmark dose (BMD) modeling. The dose-response assessment, including selection of studies and chosen PODs, is discussed in Section 5.2.5.

Finally, EPA assigns confidence ratings for each human health hazard outcome chosen for acute, intermediate, and chronic exposure scenarios. These ratings consider the evidence integration conclusions as well as additional factors such as relevance of the health outcome (and associated health effect[s]) to the exposure scenario (acute, intermediate, or chronic) and PESS sensitivity. This overall weight of scientific evidence analysis is presented in Section 5.2.6.

Throughout each of these human health hazard analysis steps, EPA considered results of previous analyses, including EPA's *Provisional Peer-Reviewed Toxicity Values for Tris(2-chloroethyl)phosphate* (U.S. EPA, 2009) and the 2009 *European Union Risk Assessment Report* (ECB, 2009).

5.2.2 Toxicokinetics Summary

This section describes the absorption, distribution, metabolism, and elimination (ADME) data available for TCEP. For full details on toxicokinetics see Appendix K.1. The PBPK model used to estimate doses to infants ingesting human milk is described in Section 5.1.3.4.7, with details presented in Appendix I.5.

In Vivo ADME Information

EPA did not identify *in vivo* human studies that evaluated ADME information for TCEP by any route of exposure. However, *in vivo* ADME studies in rats and mice found that radiolabeled TCEP is rapidly and extensively absorbed following oral dosing (Burka et al., 1991; Herr et al., 1991). TCEP is primarily eliminated in the urine, with more than 75 percent of a dose of 175 mg/kg eliminated within 24 hours for both rats and mice (Burka et al., 1991). TCEP distributes widely throughout the body. Herr et al. (1991) found radioactivity in blood, liver, and brain (including cerebellum, brainstem, caudate, hypothalamus, cortex, hippocampus, and midbrain) in male and female rats. There was no significant difference in the amount of TCEP present in blood and all brain regions after 24 hours of exposure (Herr et al., 1991).

TCEP is predominantly metabolized in the liver in both rats and mice. Metabolites reported by Burka et al. (1991) were bis(2-chloroethyl) hydrogen phosphate (BCHP, also identified as BCEP); the

glucuronide of bis(2-chloroethyl) 2-hydroxyethyl phosphate (BCGP); and bis(2-chloroethyl) carboxymethyl phosphate (BCCP).

Minegishi et al. (1988) is an ADME study that tested TCEP (which is identified as TMCEP in the study) in 5-week-old male Wistar rats. ^{14}C-labeled TCEP concentrations were measured in urine, feces, expired air, and body after exposure to a single oral dose. Almost 100 percent of the 50 µmol/kg radioactive dose was recovered –93.48 percent in urine, 5.64 percent in feces, 1.66 percent in expired air, and 0.76 percent in carcass (body). Biliary excretion for TCEP was nearly 25 percent in 48 hours, and the highest radioactivity was found in the liver of rats treated with TCEP at 168 hours after administration. The longest half-life (second phase) was observed in the adipose tissue of rats treated with TCEP (87 hours) (Minegishi et al., 1988).

In Vitro Dermal Absorption

Although no dermal *in vivo* toxicokinetic studies are available, EPA identified Abdallah et al. (2016), which measured dermal absorption using excised human skin in multiple *in vitro* experiments conducted according to OECD TG 428, *Skin Absorption: In Vitro Method.* The experiments used exposures of either 24 or 6 hours; acetone or 20 percent Tween 80 (polyoxyethylenesorbitan monooleate) in water as the vehicle; 500 or 1,000 ng/cm^2 application to skin; and finite (depletable) or infinite dose. EPA gave each of the finite dose experiments overall quality determinations of medium. For the experiment that claimed to investigate an infinite dose, EPA assigned a low overall quality determination scenario, because conditions for infinite dosing (use of neat or large body of material) were not met and the results did not reflect steady-state flux throughout the experiment (*e.g.*, applied dose was depletable).

EPA used the 500 ng/cm^2 24-hour finite dose application in acetone (0.005 percent solution) to estimate absorption for workers because this was the only experiment for which the authors reported absorption at multiple time points. Because EPA assumes workers wash their hands after an 8-hour shift, EPA used the value of 16.5 percent, which is the amount of TCEP absorbed at 8 hours. In accordance with OECD Guidance Document 156 (OECD, 2022), EPA also added the quantity of material remaining in the skin (6.8%) at the end of the experiment as potentially absorbable.[4] Therefore, EPA assumes workers absorb 23.3 percent TCEP through skin and used this value to calculate risks for workers (see Section 5.1.1.3).

For consumer exposures and exposure to soil scenarios that assume hand washing does not occur for 24 hours, EPA used the value at 24 hours (28.3%) plus the amount remaining in skin (6.8%) from the same experiment used for workers (500 ng/cm^2 24-hour finite dose application in acetone); total absorption was 35.1 percent absorption and was used to calculate risks (see Sections 5.1.2.2.3 and 5.1.3.3.2).

The estimates identified above apply to finite exposure scenarios for which the TCEP dose is depleted over time. For exposure scenarios such as swimming in which a maximum absorption rate is expected to be maintained (*i.e.*, the dose is not depletable during the exposure duration), EPA used the dermal permeability coefficient (K_p) of 2.2×10^{-2} cm/h derived by Abdallah et al. (2016) from the experiment that used the 24-hour 1,000 ng/cm^2 TCEP skin application to calculate risks (see Section 5.1.3.3.1). U.S. EPA (2024s) presents quality determinations for individual experiments conducted by Abdallah et al. (2016), with EPA comments for each of the data quality metrics. Data extraction tables with details on methods and results of the experiments are also presented in U.S. EPA (2024s).

[4] EPA used 6.8 percent (the total amount remaining in skin after washing) because the authors did not conduct tape stripping.

5.2.3 Non-cancer Hazard Identification and Evidence Integration

The sections below describe adverse outcome and mechanistic data available as well as evidence integration conclusions for each human health hazard outcome (*e.g.*, reproductive toxicity) that has been examined and/or observed in TCEP human epidemiological and animal toxicity studies. EPA identified multiple epidemiological studies relevant to non-cancer endpoints, and laboratory animal toxicity studies with apical endpoints (primarily oral studies), and some mechanistic studies depending on the hazard outcome.

Section 5.2.3.1 describes the key adverse outcomes with the most robust findings for TCEP that EPA considered for POD development (*i.e.*, those with *likely* evidence integration conclusions). Section 5.2.3.2 presents hazard identification and evidence integration for adverse outcome with weaker evidence.

Appendix L provides more information on the evidence integration conclusions for the TCEP hazard outcomes. The 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) describes the general process of evidence evaluation and integration, with relevant updates to the process presented in the TCEP Systematic Review Protocol (U.S. EPA, 2024p).

5.2.3.1 Key Human Health Hazard Outcomes

The sections below focus on hazard identification and evidence integration of neurotoxicity, reproductive toxicity, and kidney toxicity, which are the most sensitive key human health hazard outcomes associated with TCEP. These hazard outcome categories received *likely* evidence integration conclusions, and sensitive health effects were identified for these hazard outcomes.

In the risk evaluation, neurotoxicity forms the basis of the POD used for acute exposure scenarios and reproductive toxicity is the basis of the POD used for intermediate and chronic exposure scenarios.

5.2.3.1.1 Neurotoxicity

Humans

EPA identified several epidemiological studies that measured BCEP in urine (Hernandez-Castro et al., 2023a; Percy et al., 2022; Percy et al., 2021), and a study that measured TCEP in dust (Foster et al., 2024). In a longitudinal pregnancy and birth cohort, Percy et al. (2021) examined whether prenatal exposure of suspected neurotoxicants was associated with any changes in child cognition measures. Pregnant women at 16- and 26-weeks of gestation and at delivery provided urine samples to measure concentrations of BCEP; the authors then assessed children's cognition at 8 years. Maternal urinary BCEP was associated with a modest increase in child full scale IQ (FSIQ) with borderline significance ($\beta = 0.81$ per a ln-unit BCEP increase; 95% CI = 0.00, 1.61). Results were adjusted for maternal race, income, body mass index, and maternal IQ. Child sex did not significantly modify the association between maternal urinary metabolite concentration and child intelligence in most of the models (Percy et al., 2021). Percy et al. (2021) found no adverse association between BCEP concentration and child cognitive changes at 8 years of age. Quantitative dose-response requires a significant association; furthermore, an increase in IQ is not considered to be adverse. EPA assigned an overall quality determination of high to this study.

Another longitudinal cohort study assessed urinary BCEP concentrations at ages 1–5 years and association with cognitive abilities at 8 years (Percy et al., 2022). The BCEP concentration association with child FSIQ was small and positive ($\beta = 0.48$; 95% CI = –0.27, 1.24) but not statistically significant. In addition, the four IQ Index Scores (perceptual reasoning, verbal comprehension, working memory, and processing speed, ($\beta = 0.37$; 95% CI = –0.36, 1.11, $\beta = 0.15$; 95% CI = –0.66, 0.95, $\beta = 0.65$; 95%

CI = –0.19, 1.49, β = 0.47; 95% CI = –0.37, 1.30, respectively) had 95 percent confidence intervals which all included the null. Effect modification by measures of socioeconomic status (SES) like maternal education, race/ethnicity, household income, and neighborhood deprivation were evaluated. Children with lower SES experienced decreases in FSIQ for every natural log-unit increase in BCEP concentrations across multiple measures of SES, such as the effect modification of maternal education (β = –0.93; 95% CI = –1.77, –0.08, P = 0.01) (Percy et al., 2022). Although the relationships reported by Percy et al. (2022) mostly were not statistically significant, the authors observed adverse effects of OPEs on cognitive abilities for children using more than one measure of low SES. Given the inconsistencies in the outcomes of this study, it does not support a quantitative dose response analysis. EPA assigned a high overall quality determination to this study.

Foster et al. (2024) examined whether exposure to organophosphate esters (OPEs) that also included TCEP in home dust was associated with higher depression and stress levels across prenatal and postpartum time periods. A large nested prospective cohort of 718 mothers in the CHILD cohort study measured maternal depression and stress at 18–36 weeks gestation and 6 months and 1 year postpartum. For TCEP, the authors observed some increases in maternal perceived stress levels after adjusting for covariates when using two different analyses, but the 95th confidence limits all included the null. EPA assigned a medium overall quality determination to this study.

Hernandez-Castro et al. (2023a) evaluated associations of prenatal exposures to OPEs that included TCEP and child neurobehavior. The authors did not find an association between urinary BCEP levels and neurobehavioral outcomes. EPA assigned a medium overall quality determination to this study.

The epidemiological results mostly did not show significant changes in cognitive functions in children or maternal depression and stress during gestation were associated with maternal urine concentrations of BCEP as well as TCEP in home dust. Only Percy et al. (2022) showed children with low SES have reduced IQ. These inconsistencies and lack of association in most results limit any potential dose-response analysis.

Laboratory Animals

A review of high-quality acute, subchronic, and chronic studies in both rats and mice demonstrated neurotoxic effects in both sexes following TCEP exposure.

Effects in Adults: Dosing from one to a few days in multiple studies resulted in several signs of neurotoxicity. Female Fisher-344 rats administered 275 mg/kg of TCEP via oral gavage in a 1-day toxicity study exhibited increased brain lesions, seizures, and behavior effects (Tilson et al., 1990). NTP (1991b) reported that B6C3F1 mice administered the two highest doses (350 or 700 mg/kg-day) in a 16-day study exhibited ataxia and convulsive movements during the first three days of dosing. Moser et al. (2015) identified very slight to moderate tremors within three days of dosing at 125 mg/kg-day in 13 pregnant rats. Finally, pregnant mice administered 940 mg/kg-day TCEP via oral gavage were languid, prostrate, and exhibited jerking movements during GDs 7 through 14 (Hazleton Laboratories, 1983). Longer-term studies also resulted in multiple neurotoxic effects. NTP (1991b) administered 0, 22, 44, 88, 175, or 350 mg/kg-day TCEP to rats for 16 weeks. Females exhibited greater sensitivity than males. During week 4, the highest two doses were accidentally doubled, and female rats showed ataxia, excessive salivation, gasping, convulsions, as well as occasional hyperactivity. Rats exhibited necrosis of hippocampal neurons with increased dose-response (8 of 10 females at 175 mg/kg-day; 10 of 10 females at 175 and 350 mg/kg-day; and 2 of 10 ales at 350 mg/kg-day); females also showed changes in the thalamus. Mice did not exhibit neurotoxicity up to 700 mg/kg-day after 16 weeks exposure to TCEP (NTP, 1991b).

Female SD rats were administered 0, 50, 100, or 250 mg/kg-day TCEP via oral gavage for 60 days (Yang et al., 2018a) and exhibited occasional periods of hyperactivity and periodic convulsions at the highest dose, as well as learning impairment in the acquisition of the water maze tasks at particularly at 100 and 250 mg/kg-day. Histopathological changes in the hippocampus were observed at the two highest doses that included apoptosis and necrosis as well as invading inflammatory cells and calcified or ossified foci in the brain cortex at the highest dose (Yang et al., 2018a).
In a 2-year high-quality study in which rats were administered 0, 44, or 88 mg/kg-day TCEP via oral gavage, more than 40 percent of 88 mg/kg-day females exhibited histopathological changes such as focal gliosis, hemorrhage, mineralization, pigmentation, and hemosiderin in the brain stem and cerebellum (NTP, 1991b). Similar effects were not seen in male rats (only a six percent incidence of hemorrhage in the pons vs. none in controls). Male mice exhibited some increase in mineralization of the thalamus (56 and 52 percent at 175 and 350 mg/kg-day compared with 34 percent in controls) with no T3nges in brain histology in F0 adult CD-1 mice dosed with 700 mg/kg-day TCEP via gavage for several weeks during a cross-over mating study.

Developmental Neurotoxicity: Moser et al. (2015) assessed neurobehavioral effects and related hormonal responses in a non-guideline study after dosing pregnant Long-Evans rats from GD 10 through PND 22 via oral gavage of 0, 12, 40, and 90 mg/kg-day.[5] The authors measured brain acetylcholinesterase (AChE) activity, T3 and T4 levels, as well as brain and liver weights in offspring at PND 6 and 22. Serum AChE was measured in pups at PND22 (after inhibiting butyl cholinesterase activity). Liver weight, serum AChE, T3, and T4 of dams were measured when they were sacrificed at PND22. No changes were observed for these measures except an increase in liver weight relative to body weight of less than 10 percent in dams.

Multiple neurobehavioral tests were conducted. Using an elevated zero maze to measure anxiety-like behavior, no variables attained statistical significance for offspring of exposed dams when evaluated at PNDs 35 to 36 or PND 70 to 71. However, the data were highly variable, which could have precluded detection of effects (Moser et al., 2015).

In the functional observational battery (FOB) of the offspring, hindlimb grip strength (PND 29 to 30) and habituation (PND 29 to 30 and 78 to 79) did not differ from controls. The only significant FOB domain in rats treated with TCEP was activity (sex by-dose-by-day) ($p < 0.03$), with only the vertical activity counts in PND 29 to 30 males showing a dose effect ($p < 0.01$); post-hoc analysis showed no differences (Moser et al., 2015).

Offspring were then evaluated as adults (PND 83–101) and were tested for multiple outcomes in the Morris water maze. In the spatial training portion, TCEP did not result in changes in learning the platform position (latency, path length, path ratio); swim speed; or working memory (match-to-place). However, during the memory test, TCEP showed statistically significant dose-response effects for time in the correct quadrant and proximity score ($p < 0.05$), although rats in the 40 and 90 mg/kg-day groups had a greater preference for the target compared to controls. Testing with a visual platform revealed no differences in swim speed or latency. The authors observed a few differences in tests of spatial search pattern, although these apparently did not influence the direct learning and memory measurements.

During the righting reflex evaluated from PND 2 to 4, offspring of high-dose TCEP-treated rats showed a statistically significant sex-by-day interaction on PND 4 ($p < 0.05$), but there was no statistically significant overall sex-by-day-by dose interaction. TCEP exposure was not associated with changes in

[5] The highest dose was decreased from 125 to 90 mg/kg-day after 5 days.

locomotion using a motor activity ontogeny (on PNDs 13, 17, and 21) or tests that included a light transition component (PNDs 27 to 28 and 76 to 77) (Moser et al., 2015). Overall, Moser et al. (2015) notes that the behavioral changes do not suggest biologically relevant adverse outcomes or developmental toxicity. Other than tremors in dams early in the study, no TCEP-related adverse effects were observed in this study.

In a medium quality prenatal study, Kawashima et al. (1983) evaluated effects of TCEP exposure on neurodevelopment in pregnant Wistar rats gavaged with 0, 50, 100, or 200 mg/kg-day from GD 7 through 15. Twenty-three percent of the dams (7 of 30) at 200 mg/kg-day died between GD 10 and 14, with no obvious signs of intoxication after death except slight standing hair and weakness. These dams ate less than controls at GD 10 and 13. Dams at 50 and 100 mg/kg-day did not exhibit similar effects, nor did they show decreased body weight/weight gain.

Offspring 6 to 7 weeks old were examined for spontaneous behavior, coordinated movements, pain perception, hearing, and learning ability (Kawashima et al., 1983). At 200 mg/kg-day, male offspring exhibited a statistically significant decrease in numbers of rearing (9.8 vs. 19.3 in controls; $p < 0.01$), which is one of three measures of spontaneous behavior. The other measures (numbers of ambulation and fecal bowls) were not affected. The 200 mg/kg-day male offspring also took longer during the learning ability test (water maze performance) in the last of four trials ($p < 0.05$); the previous three trials were not statistically significantly different from controls but males at 200 mg/kg-day took consistently longer than controls; male offspring did not exhibit an increase in the number of errors in the water maze. Female offspring did not exhibit any statistically significant effects in these tests, and no effects were observed in the other neurological tests in either sex. The limited effects in male offspring were only seen at the highest dose, which resulted in excessive maternal toxicity.

Mechanistic Information

In a 1-day toxicity study, ICR male mice were administered via intraperitoneal injection a single dose at concentrations of 0, 50, 100, and 200 mg/kg for 2 hours to evaluate the pharmacological effects of TCEP. Combined administration of TCEP with psychoactive drugs; stimulants and depressants were used to analyze the neurochemical mechanism involved in the increased ambulatory activity. Data revealed that significantly high ambulatory activity was seen after the beginning of the measurement and decreased gradually after the administration of 200 mg/kg of TCEP. The authors note that these results suggest TCEP acts as a g-amino butyric acid (GABA) antagonist and not as a cholinergic agonist, and that TCEP increases ambulatory activity in ICR mice through a GABAergic mechanism (Umezu et al., 1998). The Umezu et al. (1998) study was not considered for dose-response analysis because it is not a relevant route of exposure, but it adds support to the potential neurotoxic nature of TCEP.

(Yang et al., 2018a) also conducted an analysis to identify possible biochemical processes and metabolic pathways affected after chronic exposure to TCEP but found low levels of GABA in TCEP-treated groups.

The metabolic pathway corresponding to GABA and other compounds provide a hypothesis to explore the possible neurotoxicity mechanisms. These findings have not been further elucidated by additional studies and thus are not conclusive regarding a mechanism for neurotoxicity.

Serum cholinesterase activity in female rats was 75 and 59 percent of controls ($p \leq 0.01$) at 175 or 350 mg/kg-day, respectively after 16-weeks repeated exposure.[6] Serum cholinesterase activity was not reduced in male rats or in either sex of mice after 16 weeks (NTP, 1991b). Moser et al. (2015) did not identify changes in brain or serum AChE of offspring after developmental exposure. Although serum cholinesterase activity may be associated with brain activity, The science policy of OPP (U.S. EPA, 2000d) concluded that the overall weight of scientific evidence for serum cholinesterase activity is the weakest link for brain cholinesterase.

Evidence Integration Summary

EPA identified four epidemiological studies that found changes in cognitive functions in children associated with maternal urine concentrations of BCEP and TCEP in home dust. Thus, the human evidence is slight for neurotoxicity because although TCEP was associated with some changes in outcomes, although there was inconsistency among the studies.

The evidence in animals is robust based on the magnitude and severity of histological changes in the hippocampus and other regions of the brain, clinical signs of toxicity, and behavioral changes in female rats. Results across available animal toxicological studies showed changes at the highest dose or increases in a dose-response manner. Effects in offspring did not show greater effects than adults.

The mechanistic data qualitatively support the evidence of hazard for TCEP; however, the data are indeterminate for the specific mechanism of TCEP hazard and are not able to be used for dose response. EPA considers the mechanistic evidence to be indeterminate.

Overall, EPA concluded that evidence indicates that TCEP likely causes neurotoxicity in humans under relevant exposure circumstances. This conclusion is based on effects from oral studies in rats and mice with dose levels between 22 and 700 mg/kg-day. Compared with exposure in adults, neurotoxicity is not expected to be increased after developmental exposure based on a lack of effects in a prenatal/postnatal study with doses up to 90 mg/kg-day (Table_Apx L-1).

5.2.3.1.2 Reproductive Toxicity

EPA guidance defines reproductive toxicity as a range of possible hazard outcomes that may occur after treatment periods of adequate duration to detect such effects on reproductive systems (U.S. EPA, 1996). Although reproductive toxicity is often associated with developmental toxicity and cannot be easily separated, this section describes male and female reproductive system toxicity (*e.g.*, effects on sperm, hormones) as well as effects on mating and fertility in a mouse continuous breeding study. Other offspring effects from the continuous breeding study (*e.g.*, decreases in live pups per litter) are described in Section 5.2.3.2.9. Neurodevelopmental investigations are described more fully in Section 5.2.3.1.1.

Humans

EPA did not identify epidemiological or human dosing studies that directly evaluated reproductive effects from TCEP exposure in the literature search conducted in 2019. Developmental effects from epidemiological studies are described in Section 5.2.3.2.9.

Laboratory Animals

Animal toxicity studies that evaluated reproductive effects after TCEP exposure consist of one reproductive assessment by continuous breeding (RACB) in mice (NTP, 1991a) and several repeated-

[6] After 16 days, serum cholinesterase activities in female rats receiving 175 or 350 mg/kg-day were 79.7 and 81.8 percent of controls, respectively; however, this study received an overall uninformative quality determination due to a viral infection.

dose studies that evaluated reproductive organs and hormones in adult and adolescent mice and in adult rats (Chen et al., 2015a; NTP, 1991b; Matthews et al., 1990).

A high-quality RACB study (NTP, 1991a) dosed F0 male and female CD-1 mice with 0, 175, 350, or 700 mg/kg-day TCEP for 1 week prior to cohabitation, 14 weeks cohabitation, and 3 weeks in a holding period; F0 mice were allowed to produce up to 5 litters per breeding pair. After weaning of final litters, the F0 male and female 700 mg/kg-day groups were crossbred with controls of the opposite sex to determine influence of sex on reproductive outcomes. F1 animals in the final litters of the continuous breeding phase received TCEP at the same doses as their parents for approximately 14 weeks (from weaning through 74 days of age, during a 1-week cohabitation phase, and during gestation and lactation). The F1 animals were then evaluated for reproductive outcomes.[7] Because F0 breeding pairs produced no litters at 700 mg/kg-day, F1 dose groups were limited to 0, 175, and 350 mg/kg-day. F0 control and high dose (700 mg/kg-day) and F1 adult mice were examined for changes in reproductive organs, sperm parameters, and estrous cyclicity.

Reproductive organs[8] of F344 rats and $B6C3F_1$ mice were evaluated in NTP 16-day, 16 to 18 week,[9] and 2-year studies (NTP, 1991b) that received overall high-quality determinations, except the 16-day rat study, which was uninformative due to a viral infection. Matthews et al. (1990) reported results of additional reproductive measurements (*e.g.*, sperm counts) from the 16- to 18-week NTP studies and received a medium quality determination for the reported endpoints. Chen et al. (2015a), a high-quality study, evaluated the male reproductive system at 0, 100, and 300 mg/kg-day TCEP for 35 days in an oral feeding study of 5-week-old adolescent male ICR mice. U.S. EPA (2024q) presents details extracted from these studies.

Reproductive Outcomes from RACB: The F0 continuous breeding phase of NTP (1991a), resulted in decreased fertility;[10] values of 72 percent fertility in the fifth litter per breeding pair at 350 mg/kg-day and 67 to 0 percent in the second through fifth litters at 700 mg/kg-day ($p < 0.05$) contrasted with F0 control fertility of 97 percent. The 700 mg/kg-day dose also resulted in 25 or more cumulative days to litter [11] vs. controls beginning in the second litter ($p < 0.05$).

During crossbreeding of F0 mice, the 700 mg/kg-day male × control female group resulted in lower pregnancy[12] and fertility indices ($p < 0.05$) but not when treated females were bred with untreated males.[13,14] F1 breeding (both sexes dosed) resulted in decreased fertility at 350 mg/kg-day (highest dose; $p < 0.05$).

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[7] The exposure duration was not clearly stated in NTP (1991a) for the F1 generation but Heindel et al. (1989) states that the continuous breeding protocol specifies that dosing of the F1 generation begins just after weaning.
[8] Gross necropsy and histopathology: *Males* – epididymis, preputial gland, prostate, seminal vesicles, testis; *Females* – clitoral gland, mammary glands, ovaries, uterus.
[9] NTP (1991b) stated that male rats were dosed for 18 weeks but Matthews et al. (1990) identified the studies as 16-week studies (vs. an 18-week study for male rats), even though they are the same studies described in NTP (1991b).
[10] The percent of mated females with copulatory plugs that got pregnant.
[11] This appears to be a measure of the number of days from start of cohabitation of the breeding pairs to the day when pups were born.
[12] Number of fertile pairs of the total number of cohabiting pairs.
[13] The number of breeding pairs examined ranged from 18 to 20 among dose groups.
[14] NTP (1991a) cited an inhalation study (Shepel'skaia and Dyshginevich, 1981) that administered TCEP at 0, 0.5, and 1.5 mg/m^3 to male rats continuously for four months and then mated with unexposed females. Similar to the RACB results, dams had significantly decreased litter size and also exhibited increased pre- and post-implantation loss at 1.5 mg/m^3. Shepel'skaia and Dyshginevich (1981) appears to be an abstract in Russian; EPA could not obtain this study or evaluate its quality.

Decreased fertility appeared earlier in the second generation (*i.e.,* in the single litters produced according to protocol) than in the first generation in which only in the second or subsequent litters from each of the breeding F0 pairs were affected.

Male Reproductive Toxicity: In males, effects on reproductive organs and hormone levels were identified but differed by study and dose. In adolescent mice, Chen et al. (2015a) found 22 and 41 percent decreases in seminiferous tubule numbers at 100 and 300 mg/kg-day, respectively ($p < 0.05$) as well as decreases in Leydig, Sertoli, and spermatogenic cells. The 300 mg/kg-day group also resulted in a testis weight decrease of 13.6 percent and testicular testosterone decrease of 18 percent ($p < 0.05$) as well as "absolute" disintegration of seminiferous tubules.

The RACB study (NTP, 1991a) identified a 34 percent decrease in epididymal sperm density, more than 3.4-fold increase in abnormal sperm, 45 percent fewer motile sperm, and a 30 percent decrease in testis weight ($p < 0.001$) for the only tested dose (700 mg/kg-day) in the F0 adult CD-1 mice. The treated F0 mice also exhibited minimal to mild testes hyperplasia (3/10 vs. 0/10 in controls). F1 male mice did not exhibit effects on sperm or reproductive organs at either 175 or 350 mg/kg-day (NTP, 1991a).

In the 16-week repeated dose study $B6C3F_1$ mice at 700 mg/kg-day exhibited decreases in absolute and relative testes weights ($p < 0.01$) (NTP, 1991b). Matthews et al. (1990) reported that the 700 mg/kg-day mice in this study had slightly reduced sperm counts ($p = 0.05$). Neither effect was observed at 175 mg/kg-day or lower. No changes in testes weights were observed in male rats up to 175 mg/kg-day after 16 weeks (NTP, 1991b), and sperm morphology could not be conducted on the F344 rats in the 16-week study due to technical difficulties (Matthews et al., 1990).[15,16] There were no changes in gross necropsy or histopathology in the 16-day or 16-week NTP studies as identified in the text, or in the 2-year NTP study as identified in incidence tables (NTP, 1991b).

The crossbreeding results described earlier suggest offspring effects are greater from treated males vs. treated females.

Female Reproductive Organ and Hormone-Related Effects: Adult F0 females administered 700 mg/kg-day TCEP in the RACB study exhibited decreased postnatal dam weights but no changes in estrous cyclicity. Lower doses were not examined, but the treated F1 female adults (175 or 350 mg/kg-day) also exhibited no estrous cycle changes. Two of ten F1 females at 350 mg/kg-day had ovarian cysts, whereas none of the ten controls exhibited cysts, although the authors did not suggest this to be a TCEP related effect.[17]; lower doses were not evaluated. As noted earlier, even though the RACB identified effects

[15] NTP (1991a) provided more details of the sperm morphology and vaginal cytology examinations (SMVCE) from the 16-week NTP study, citing an unpublished report (Gulati and Russell, 1985) and partly described by Matthews et al. (1990): The doses evaluated for mice were 0, 44, 175, and 700 mg/kg-day. The 700 mg/kg-day $B6C3F_1$ mice exhibited a 28 percent decrease in epididymal sperm density; more than a doubling of abnormal sperm; a 22 percent decrease in testicular weight; and decreased epididymis weights. Rats were evaluated at 0, 22, 88, and 175 mg/kg-day and Gulati and Russell (1985) stated that rats did not exhibit changes in epididymis and cauda epididymis weights or in percent abnormal epididymal sperm. Sperm density was reported as being increased and motility was decreased in rats at 175 mg/kg-day even though Matthews et al. (1990) did not report the results due to technical difficulties. Gulati and Russell (1985) was not readily available; therefore, EPA did not evaluate it for data quality.

[16] In (Shepel'skaia and Dyshginevich, 1981), cited by NTP (1991a), male rats exposed continuously to air concentrations of TCEP for four months exhibited effects on meiosis, post meiotic growth, and maturity of spermatozoids upon histopathological examination of males. Shepel'skaia and Dyshginevich (1981) appears to be an abstract in Russian; EPA could not obtain this study or evaluate its quality.

[17] In the F0 700 mg/kg-day dose group, two of 13 females also had ovarian cysts (one minimal, one mild) compared with none among 12 controls. However, one instance of lymphoma associated with the ovary and one instance of oophoritis was seen in the controls.

from treated female mice bred with untreated males, effects were less pronounced than those resulting from treated males crossbred with untreated females (NTP, 1991a).

There were no changes in gross necropsy or histopathology in females in the 16-day or 16-week NTP studies as noted in the text. No statistically or biologically noteworthy non-cancer effects were seen in the 2-year study. Although adenocarcinomas occurred in three mice at 350 mg/kg-day ($p < 0.05$ in the trend test), a fibroadenoma occurred in control mice; the trend for the combined tumor types was not statistically significant, and the incidence of adenocarcinoma was within the range of historical controls (NTP, 1991b).

Mechanistic Information

In vitro studies provide some supporting mechanistic evidence of reproductive effects. Chen et al. (2015b) identified several effects when mouse Leydig (TM3) cells were exposed to TCEP. At 100 µg/mL TCEP, which did not result in significant cytotoxicity, effects included large decreases in one gene associated with testosterone synthesis after all timepoints (6, 12, and 24 hours) and a second gene at 24 hours. After stimulation of testosterone synthesis genes with human chorionic gonadotropin (hCG), 100 µg/mL TCEP still significantly decreased mRNA levels compared with controls or hCG. Also at 100 µg/mL and 24 hours exposure, testosterone secretion was decreased by about 50 percent with TCEP alone and by about 39.9 percent (vs. hCG) after stimulation with hCG. TCEP exposure was also associated with increased transcription of genes for antioxidant proteins.

Exposure to 300 µg/mL TCEP (mostly after 24 hours) yielded generally greater changes in transcriptional levels of genes associated with testosterone synthesis (mostly decreased); increased transcription of genes encoding antioxidant proteins; increased activities of antioxidants; and decreased secretion of testosterone. This concentration resulted in 31.4 percent lower viability of cells than controls; thus, effects at this concentration may be at least partly secondary to cytotoxicity (Chen et al., 2015b). Overall, although some effects may have been due to general cytotoxicity, others are specific to male reproductive toxicity (Chen et al., 2015b).

TCEP exposure was not associated with estrogenic or anti-estrogenic effects using either a recombinant yeast reporter gene assay or by inducing alkaline phosphatase in human endometrial cancer Ishikawa cells (Follmann and Wober, 2006). Reers et al. (2016) also found no TCEP-related changes in endogenous androgen receptor (AR) mediated gene expression in metastatic prostate cancer cells (LNCaP) or in estrogen receptor α (ERα) and the aryl hydrocarbon receptor (AhR) target gene activation using ECC-1 cells (endometrial carcinoma cells). Krivoshiev et al. (2016) reported that 1,000 µM TCEP did not exhibit estrogenic activity in a cell proliferation assay using the breast adenocarcinoma cell line (MCF-7) but did show anti-estrogenic activity when co-treated with 17β-estradiol (E2), yielding a 32 percent relative inhibitory effect. Viability of TCEP to MCF-7 cells was 93 percent of viability in controls, and results are not expected to be overly influenced by cytotoxicity.

Evidence Integration Summary

There were no human epidemiological studies available for TCEP for reproductive outcomes (although see Section 5.2.3.2.9 for developmental outcomes), and the human evidence is indeterminate for reproductive effects.

For the animal studies, which primarily received high or medium overall quality determinations, biological gradients were seen for fertility index, number of litters per pair, and number of live pups per litter, which were decreased in a dose-related manner the F0 generation (NTP, 1991a) and for testes

histopathology in mice (Chen et al., 2015a), which exhibited increased magnitude and severity with increasing dose.

Consistent findings included decreased numbers of live pups per litter observed at the same dose in F0 and F1 mice in the RACB, with increasing severity in the second generation (NTP, 1991a), and decreased testes weights in mice at 300 mg/kg-day and higher (Chen et al., 2015a; NTP, 1991a, b). Decreases in testosterone and related effects were observed *in vivo* and *in vitro* (Chen et al., 2015a; Chen et al., 2015b), with related decreases in gene expression *in vitro* (Chen et al., 2015b).

Within and among animal studies, coherent changes were seen between related types of effects. Decreased testosterone in Chen et al. (2015a) and Chen et al. (2015b) support observed effects on testes and sperm in other studies. Also, in the first generation of the RACB study (NTP, 1991a), male reproductive effects were observed along with effects on fertility and live pups per litter.

Some effects differed among studies. Histopathological changes in the testes were also not routinely identified. Chen et al. (2015a) observed changes in seminiferous tubules in adolescent ICR mice that were not identified in other studies, including the F1 males in the RACB study that were dosed beginning at weaning (NTP, 1991a). These differences lend uncertainty regarding the association of this specific effect with TCEP exposure. However, studies differed in use of species or mouse strains and in use of gavage vs. feeding. Chen et al. (2015a) was also conducted more than 20 years after the other studies and differences in assessment methods could possibly explain the differences in results.

Effects on sperm were not identified in the F1 animals even though effects on live pups/litter and fertility were observed in the RACB study (NTP, 1991a). However, *in vitro* studies suggest other mechanisms (*e.g.*, oxidative stress, as suggested by Chen et al. (2015b)) might be operating and could contribute to the observed reproductive effects.

Overall, evidence in humans is indeterminate based on the lack of available studies. Evidence in animals is moderate based on studies with decreased testes weight, sperm effects, and/or reduced fertility, and some support from histopathological changes in testes. EPA considers the mechanistic evidence to be slight based on decreases in testosterone and gene expression but no direct estrogenic or androgenic agonism or antagonism. Overall, EPA concluded that evidence indicates that TCEP likely causes reproductive toxicity in humans under relevant exposure circumstances. This conclusion is based on effects primarily related to fertility in the RACB study and male reproductive toxicity and is based on oral studies in rats and mice with dose levels between 22 and 700 mg/kg-day (Table_Apx L-2). EPA guidelines for reproductive toxicity risk assessment (U.S. EPA, 1996) state that findings in animals are considered relevant to humans in the absence of evidence to the contrary.

5.2.3.1.3 Kidney Toxicity

Human

EPA identified an epidemiological study that measured BCEP in urine. Kang et al. (2019) examined whether exposure to OPEs, which also included TCEP, was associated with chronic kidney disease. A large cohort of 1578 adults who were not currently pregnant had their urine data evaluated for two chronic kidney disease related parameters which included the estimated glomerular filtration rate (eGFR) and albumin-to-creatine ratio (ACR). After adjusting for sex, age, race/ethnicity, poverty income ratio, smoking history, physical activity, and BMI, the association between concentration of a novel measurement of creatinine-adjusted BCEP levels (ng/mL) and eGFR was negative ($\beta = -1.26$) and statistically significant ($p < 0.05$); the association was not statistically significant when using the traditional creatinine adjustment (ng/mg). Urinary BCEP levels showed a marginally significant

association with ACR (β = 0.06; p = 0.08) for both the traditional and novel approaches. The study authors showed that the methods of urinary dilution adjustment can demonstrate significance and direction of association. The results show a likely association between TCEP exposure and chronic kidney disease after adjusting with a novel measurement of creatine. However, the strength of the association is weak. There is also a lack of consistency in the epidemiology data for the relationship of TCEP exposure and chronic kidney disease. This is also a cross-sectional study making identification of causality difficult. Therefore, the results do not support a dose-response analysis. EPA assigned a medium overall quality determination to this study.

Laboratory Animals

A review of the available animal toxicity studies for rats and mice identified the kidney as the target organ in both sexes following TCEP exposure. In a short-term (28-day) repeated oral toxicity study, male Fisher-344 rats were given a daily TCEP dose level of 350 mg/kg-day. Results showed signs of scattered proximal tubular regeneration in the cortex and outer stripe of the outer medulla (Taniai et al., 2012a). Other findings after short-term exposure included increased absolute and relative kidney weights in male rats at 175 and 350 mg/kg-day after 16-day oral repeated exposures.

Some effects were also observed after longer-term dosing. After 16 weeks of oral dosing, male rats had increased absolute and relative kidney weights at high-dose only (350 mg/kg-day) and female rats exhibited increased absolute and relative weights from 44 to 350 mg/kg-day (NTP, 1991b). Both F0 males and female mice exhibited cytomegaly of renal tubule cells decreased kidney weights and after dosing of 700 mg/kg-day TCEP for several weeks in a continuous breeding study (NTP, 1991a). In the 16-week study, male mice receiving 700 mg/kg-day had significantly reduced absolute kidney weights, decreased by 19.4 percent compared to the controls. Relative-to-body kidney weights were decreased at 175, 350, and 700 mg/kg-day by 13.3 percent, 16.0 percent, and 14.1 percent compared to controls. Tubule epithelial cells with enlarged nuclei (cytomegaly and karyomegaly) were observed in the kidneys of high-dose (700 mg/kg) male and female mice. These lesions were mostly observed in the proximal convoluted tubules of the inner cortex and outer stripe of the outer medulla.

In the 2-year bioassay, both sexes of rats and mice exhibited histopathological lesions in the kidney, including renal tubule hyperplasia and in male and female rats and epithelial cytomegaly and karyomegaly in both male and female mice (NTP, 1991b).

In the 2-year study, karyomegaly was observed in 32 percent and 78 percent of male mice dosed at 175 and 350 mg/kg-day, respectively, compared to 4 percent of control animals. Karyomegaly was also observed in 10 percent and 88 percent of female mice dosed at 175 and 350 mg/kg/day, respectively. Hyperplasia of the renal tubule epithelium was observed in 6 percent and 4 percent of male and female mice, respectively at 350 mg/kg-day compared to 2 percent and 0 percent of control male and female mice (NTP, 1991b). High-dose male rats (88 mg/kg-day) exhibited 48 percent incidences of hyperplasia of the renal tubule epithelium vs. 0 percent in controls. High dose female rats also exhibited increased incidence of focal hyperplasia of the renal tubule epithelium, by a 32 percent vs. 0 percent in controls (NTP, 1991b). The authors reported no changes blood urea nitrogen or creatinine in rats or mice.

As noted in Section 5.2.4.2, male rats after two years also exhibited dose-related increased incidence of renal tubule adenomas vs. control rats (48 vs. 2%); one control and one high dose male developed renal tubule carcinoma. High-dose female rats exhibited an increased incidence of renal tubule adenomas, but to a lesser extent than male rats (10 vs. 0 in controls). Eight percent of high-dose male mice had either renal tubule adenomas or adenocarcinomas compared with two percent in controls.

Mechanistic Information
Mechanistic data also supported the conclusion that TCEP targets the kidney. In a 28-day gavage study, markers for cell proliferation and apoptosis were increased in the kidneys (OSOM and cortex) of rats (Taniai et al., 2012b). *In vitro* exposure of primary rabbit renal proximal tubule cells (PTCs) resulted in reduced DNA synthesis, altered expression of cell cycle regulatory proteins, cytotoxicity, inhibition of ion- and non-ion-transport functions, and there was increased expression of pro-apoptotic regulatory proteins and decreased expression of proteins that inhibit apoptosis were also observed (Ren et al., 2012; Ren et al., 2009, 2008).

Evidence Integration Summary
EPA identified an epidemiological study that found changes in chronic kidney disease parameters, estimated glomerular filtration rate (eGFR) and albumin-to-creatine ratio (ACR) associated with urinary BCEP levels in U.S. adult population.

There was only one human epidemiological study available for TCEP with minimal effects between a TCEP metabolite and chronic kidney disease parameters and therefore, there is slight human evidence.

The evidence in laboratory animals is moderate based on incidences of kidney histopathology findings that increased with dose in rats and mice of both sexes. Increased incidences of kidney histopathological lesions were observed in rats and mice of both sexes following chronic exposures. Although less consistent, changes in kidney weights were also observed in multiple species. EPA considers the mechanistic evidence to be slight based on markers of cell proliferation and apoptosis in kidneys of rats after 28-day gavage treatment and supporting *in vitro* evidence.

Overall, evidence indicates that TCEP exposure likely causes non-cancer kidney effects in humans under relevant exposure circumstances based on oral studies with doses ranging from 22 to 700 mg/kg-day in rats and mice (Table_Apx L-4).

5.2.3.2 Other Human Health Hazard Outcomes
This section describes hazard identification and evidence integration for additional non-cancer health outcome categories not considered to be critical to this risk evaluation based on the results of evidence integration that identified evidence for these outcomes as *suggestive* or *inadequate* to assess effects. These hazard outcomes are as follows: Skin and eye irritation, mortality, hepatic, immune/ hematological, thyroid, endocrine (other effects), lung/respiratory, and body weight.

5.2.3.2.1 Skin and Eye Irritation
Laboratory Animals
In a medium-quality study (Confidential, 1973), rabbits dermally exposed to 0.5 mL (approximately 279 mg/kg[18]) TCEP for four hours did not show irritation through 48 hours at either the intact or abraded skin sites. However, 0.4 mL/kg TCEP (equivalent to 556 mg/kg) was administered to shaved dorsal skin of rabbits and repeated for four days, resulting in corrosivity and fissuring (FDRL, 1972). This study received an uninformative overall quality determination based on lack of information on statistical analysis, and it is not clear how long TCEP was in contact with skin each day or when corrosivity and fissuring first appeared.

[18] According to the accompanying protocol, the dose was 0.5 mL TCEP (equivalent to 695 mg) and some sites were abraded. Assuming 2.5 kg body weight of rabbits (2 to 3 kg was identified in the accompanying protocol), the dose was approximately 279 mg/kg-bw.

TCEP was not irritating to eyes of rabbits when administered at 0.1 mL and observed for 72 hours (Confidential, 1973) in a medium-quality study.

Evidence Integration Summary
The human evidence is indeterminate for skin and eye irritation. The two readily available dermal irritation studies in animals showed inconsistent results and the single eye irritation study of medium quality showed that TCEP is not irritating; these studies are indeterminate. Although one study was uninformative, EPA considered that these results are not affected by the lack of statistical analysis. Overall, the currently available evidence is inadequate to assess whether TCEP causes irritation in humans (see Appendix L.2).

5.2.3.2.2 Mortality

Laboratory Animals
EPA identified multiple oral studies and two dermal studies. In short-term oral mouse studies, no female CD-1 mice died at 940 mg/kg-day after dosing from GD 7 to 14 (Hazleton Laboratories, 1983). Seven of thirty pregnant female Wistar rats died in a prenatal study (Kawashima et al., 1983). In a 16-day repeated-dose study, no mice died at doses up to 350 mg/kg-day (NTP, 1991b).[19] At higher doses, 13 to 20 percent female mice died at 1,000 mg/kg-day and all mice died at 3,000 mg/kg-day after 8 to 14 days of exposure (NTP, 1991a; Hazleton Laboratories, 1983).

In longer-term studies, adult mortality was observed at lower doses in rats compared with mice. In 16- to 18-week subchronic studies that received medium-quality determinations for mortality, male and female rats exhibited decreased survival as low as 175 and 350 mg/kg-day, respectively, but both groups accidentally received double doses during week four; no mice died at doses up to 700 mg/kg-day after 16 weeks (Matthews et al., 1990).[20] No deaths occurred in rats or mice at lower doses (250 to 300 mg/kg-day) for 35 or 60 days (Yang et al., 2018a; Chen et al., 2015a); both studies received overall high-quality determinations. In a high-quality 2-year study, rats exhibited decreased survival (by 27 to 29%) at 88 mg/kg-day, but mice did not exhibit differences in survival up to 350 mg/kg-day (NTP, 1991b).

In a medium-quality dermal irritation study, four of six rabbits died after a four-hour exposure to approximately 279 mg/kg TCEP (Confidential, 1973).[21] These rabbits exhibited narcosis and paralysis before death. However, FDRL (1972) did not report any deaths in rabbits dermally exposed to approximately 556 mg/kg for 4 days. This study received an uninformative overall quality determination based on lack of information on statistical analysis.

Decreases in numbers of live born animals after parental exposure are described in Section 5.2.3.1.2.

Evidence Integration Summary
Human evidence is indeterminate for mortality because there are no human epidemiological studies that assessed this endpoint. There is modest evidence in animal studies that shows higher mortality in rats than mice on oral studies and uncertain potential for mortality via the dermal route given conflicting

[19] No rats died in a short-term study at doses up to 700 mg/kg-day (NTP, 1991b) that received an uninformative overall data quality determination due to a viral infection.
[20] NTP (1991b) reported that 9 of 10 male rats survived at 175 mg/kg-day in the 16-week study compared with 4 of 10 reported by Matthews et al. (1990), which is a report of the same study.
[21] The 2009 *European Union Risk Assessment Report* (ECB, 2009) reported results of an acute dermal study not readily available to EPA in which four rabbits were each exposed dermally to 2,150 mg/kg for 24 hours, using occlusive patches. No deaths, apparent signs of toxicity, or cholinesterase depression were observed in any of the rabbits 72 hours after treatment.

results. Overall, evidence suggests but is not sufficient to conclude that TCEP exposure causes mortality in humans under relevant exposure circumstances. This conclusion is based on oral studies in rats and mice that assessed dose levels between 12 and 700 mg/kg-day and dermal studies in rabbits at approximately 279 and 556 mg/kg-day (see Appendix L.2).

5.2.3.2.3 Liver

Laboratory Animals

EPA identified multiple high-quality animal studies that reported liver weight, histopathological changes, and one study measured enzyme changes. Liver weights were statistically increased in multiple oral gavage rodent studies. In 16- or 18-week studies, rats and mice exhibited absolute increases ranging from 10 to 84 percent and relative-to-body weight increases ranging from less than 10 to 51 percent, with the largest increases in female rats at the highest dose of 350 mg/kg-day (NTP, 1991b).[22] At the 66-week sacrifice in the chronic bioassay, male rat absolute and relative liver weights were increased by 20 and 19 percent, respectively, at 88 mg/kg-day (the highest dose) but female rats did not exhibit similar changes. Liver weight was not reported for mice in the chronic bioassay (NTP, 1991b).[23] F0 male mice (but not females) given 700 mg/kg-day TCEP for 18 weeks in a continuous breeding study via oral gavage exhibited increases in relative and absolute liver weight of 20 and 15 percent, respectively, with no accompanying body weight changes (NTP, 1991a). No liver weight changes were seen after 350 mg/kg-day in the F0 or F1 generation in the same study. Only the 16-day mouse study reported a decrease in (relative) liver weight in males (by 18%), but the change was seen only at 44 mg/kg-day without a dose-response (NTP, 1991b).[24]

In the 2-year oral gavage bioassay, male mice had 6 and 16 percent incidence of eosinophilic liver foci at 175 and 350 mg/kg-day compared with zero incidence in controls. EPA conducted a Fischer's exact test and identified the incidence at the highest dose to be statistically significant ($p < 0.01$). The foci are believed to be precursors to hepatocellular neoplasms (NTP, 1991b). Because these foci were not accompanied by increased basophilic and clear cell foci, which are considered part of the continuum with hepatocellular adenomas, NTP (1991b) states that it is uncertain whether eosinophilic foci were associated with TCEP exposure. Adenomas and carcinomas are discussed in Section 5.2.4.2. At 700 mg/kg-day in the continuous breeding study, F0 male mice exhibited cytomegaly (10/12) and hepatitis (4/12) vs. 0/10 per effect in controls; no other doses were evaluated in the F0 generation. F1 mice exhibited minimal or mild changes in liver histology at 350 mg/kg-day (NTP, 1991a).

Liver enzyme activity was measured only at the 66-week sacrifice in the 2-year bioassay (NTP, 1991b). Female rats at 88 mg/kg-day exhibited significantly decreased mean serum alkaline phosphatase (ALP) and alanine transferase (ALT) values with no change in aspartate transaminase (AST). No information was provided on the magnitude of change, and no differences were reported for male rats or mice of either sex (NTP, 1991b). Although increases in liver enzyme activity are typically associated with liver injury, decreases are harder to interpret. Decreases in serum ALT could occur after initial increases resulting from liver injury and has been associated with decreased levels of vitamin B_6 (Giannini et al., 2005). ALP is also present in bone and intestines and decreases have been associated with chronic

[22] The 350 mg/kg-day female rats also had increased body weight (by 20%) compared with controls (NTP, 1991b).
[23] In the 16-day rat study, females exhibited statistically significant increases in absolute and relative liver weights (by 17 and 14 percent, respectively) at 350 mg/kg-day but the study was uninformative due to a viral infection.
[24] Chen et al. (2015a) found that male mice had decreases of 17.3 and 18.1 percent in absolute liver weight at 100 and 300 mg/kg-day, respectively after 35 days of dosing in an oral feeding study. Body weights were also decreased by 13.5 and 14.8 percent at 100 and 300 mg/kg-day respectively (estimated from graphs using GrabIt!™ Copyright Datatrend Software, 1998–2001. https://download.cnet.com/Grab-It-XP/3000-2053_4-41084.html). EPA calculated decreased liver weights relative to body weights for male mice of 3.5 and 3.6 percent at 100 and 300 mg/kg-day, respectively (Chen et al., 2015a); therefore, the changes were within 10 percent and not considered adverse.

myelogenous leukemia, anemias, severe enteritis, and other conditions (Sharma et al., 2014; Giannini et al., 2005).

Due to uncertainty and lack of reasonably available information, EPA has not determined the decreased enzyme activities to be adverse. Furthermore, except for the liver weight changes identified in the reproductive and continuous breeding protocol in male mice at 700 mg/kg-day that were accompanied by histopathological changes, the increased liver weights in other studies are not clearly adverse due to the lack of histopathological changes and lack of increased enzyme activity.

Mechanistic Information

EPA identified mechanistic studies in liver and liver cells from both *in vivo* and *in vitro* studies. Limited mechanistic data indicate that TCEP may increase oxidative stress (based on increased hepatic antioxidant enzyme activities and accompanying gene expression) in the livers of male ICR mice after 35 days of dietary TCEP exposure (Chen et al., 2015a). *In vitro* studies show that TCEP induced oxidative stress, altered cellular energetics, and influenced cell signaling related to proliferation, growth, and cell survival in the liver (Mennillo et al., 2019; Zhang et al., 2017b; Zhang et al., 2017a; Zhang et al., 2016c; Zhang et al., 2016b).

Evidence Integration Summary

There are no epidemiology studies that investigated liver effects, and human evidence is indeterminate.

Male mice exhibited a dose-related increase in eosinophilic foci after two years (as well as an increase in hepatocellular adenoma) in a high-quality study (NTP, 1991b). Increases in absolute and relative liver weights in male and female rats occurred at lower doses as duration increased from 16 days to 16 weeks (NTP, 1991b). Absolute and relative liver weights also generally increased dose-dependently in female rats and female mice at 16 weeks and in male rats at 66 weeks (NTP, 1991b). Only at a higher dose (700 mg/kg-day) was concordance observed between increased absolute and relative liver weight and histopathological changes (NTP, 1991a).

However, NTP (1991b) suggests an uncertain association between TCEP exposure and eosinophilic foci. Also, there were no histopathology findings in rats or female mice, including no hypertrophy associated with liver weight increases. Liver weight increases were seen in female rats after 16 days and 16 weeks, but not 66 weeks of exposure. Increased liver weight was not seen in the 35-day study (Chen et al., 2015a). No biologically relevant changes in serum enzymes were seen in the 2-year bioassay and were not measured in shorter studies. Therefore, EPA determined that the animal evidence for adverse effects on the liver based on these data are slight for the association between TCEP and adverse liver effects.

Mechanistic information shows biological gradients for the induction of hepatic oxidative stress occurring earlier than apical endpoints. Also, across the *in vitro* studies, dose-related changes in viability, oxidative stress, and impaired mitochondrial functioning were observed. Oxidative stress is a plausible mechanism for eosinophilic foci (and tumor formation) that is relevant to humans. However, few potential mechanisms were investigated in available studies and oxidative stress was demonstrated *in vivo* at higher doses than those associated with liver lesions in the chronic study. This information suggests mechanistic evidence for liver effects is slight.

Based on the indeterminate human evidence, slight animal evidence showing increased liver weights in in the absence of relevant clinical chemistry findings or statistically significant histopathology changes, EPA concluded that evidence suggests but is not sufficient to conclude that TCEP exposure causes

hepatic toxicity in humans under relevant exposure circumstances. This conclusion is based on studies of mice and rats that assessed dose levels between 44 and 700 mg/kg-day (see Table_Apx L-5).

5.2.3.2.4 Immune/Hematological

Humans

In a high-quality study, Mendy et al. (2024) studied associations between multiple exposure measures (TCEP in dust as concentrations and loadings and BCEP in urine of mothers during 16 and 26 weeks of gestation and at delivery) for 342 mother-infant pairs in Cincinnati, Ohio and the risk of wheeze, respiratory infections, and hay fever/allergies. Mothers were given a questionnaire asking about respiratory symptoms in their children every six months up to five years of age. At five years, spirometry was assessed in the children, including forced expiratory volume in 1 second (FEV_1) and peak expiratory flow (PEF). FEV_1 was compared to a referent population of children.

TCEP and BCEP concentrations were treated as continuous variables when comparing with lung function. Another model investigated changes through time by dichotomizing the concentrations as less than or greater than the geometric mean.

Respiratory Infections: Urinary BCEP levels at 16 weeks (but not 26 weeks or at delivery) were associated with respiratory infections (relative risk (RR): 1.43; 95% CI: 1.08–1.90) in a model adjusted for child's sex, child's race/ethnicity, birth weight, gestational term, family income, and child receiving breast milk. When dichotomizing concentrations (greater or less than the geometric mean), high BCEP concentrations at 16 and 26 weeks were associated with respiratory infections (RR: 1.78; 95% CI: 1.04–3.07). TCEP in dust was not associated with increases in respiratory infections (Mendy et al., 2024).

Wheeze: When dichotomizing the concentrations, high BCEP concentrations in urine at gestational weeks 16 and 26 were associated with higher risk of wheeze (RR: 1.63; 95% CI: 1.12–2.36). Similarly, high BCEP at week 16 of gestation and at delivery was associated with higher risk of wheeze (RR: 1.88; 95% CI: 1.18–2.99) (Mendy et al., 2024).

Hay Fever/Allergies: High BCEP at week 16 of gestation and at delivery was associated with higher risk hay fever/allergies (RR: 1.65; 95% CI: 1.08–2.51). TCEP dust concentrations at 20 weeks gestation were associated with a higher risk of hay fever/allergies (RR: 1.11; 95% CI: 1.01–1.21), with a similar relationship between dust loadings and hay fever/allergies; both dust models were adjusted for child's sex, child's race/ethnicity, birth weight, gestational term, family income, and child receiving breast milk. When examined for each sex, TCEP dust concentrations and dust loadings were associated with higher risk of hay fever/allergies in females (RR: 1.29; 95% CI: 1.16–1.44, RR: 1.33; 95% CI: 1.21–1.47, respectively), and sex differences were statistically significant for both dust loadings and concentrations ($P_{interaction} < 0.001$). Significant associations between TCEP in dust and hay fever/allergies were not seen when males were examined separately (Mendy et al., 2024).

Lung Function: Prenatal dust TCEP loadings were associated with lower FEV_1 at age 5 years (β: –6.17; 95% CI: –11.09, –1.25). In contrast, urinary BCEP at 26 weeks of gestation was associated with a higher FEV_1 (β: 4.88; 95% CI: 0.04–9.73), which would seemingly suggest a possible beneficial association (Mendy et al., 2024).

Navaranjan et al. (2021) used a case-cohort design nested in a Canadian cohort study to investigate the association between 29 OPEs in dust vacuumed from children's homes (including sleeping quarters) when they were 3 to 4 months old and childhood asthma at 5 years old or recurrent wheeze at ages 2 to 5 years. For each chemical, odds ratios (ORs) were determined between each of the three higher exposure

quartiles compared with the lowest quartile. Models were adjusted for study site, household income, child sex, and parental history of asthma. TCEP was not associated with asthma or wheeze. Pearson correlations showed that TCEP was not significantly associated with other measured OPEs (range: 0.00–0.13). EPA assigned a medium overall quality determination to this study.

Araki et al. (2020) studied the association between flame retardants and phthalate chemicals of 128 elementary school-aged children in Japan with wheeze and allergic symptoms (*i.e.*, rhinoconjunctivitis, eczema) as determined by questionnaire; urine samples were taken during the home visit when the questionnaire was administered. The participation rate was low (only 2.9%).

TCEP in urine (standardized for creatinine content) was not significantly associated with wheeze, rhinoconjunctivitis, or eczema when comparing the second or third tertile of exposure with the first tertile. The chemical mixture was associated with increased rhinoconjunctivitis (OR: 2.60; 95% CI: 1.38–5.14). Models were adjusted for sex, grade, household income, and dampness index. Spearman correlation coefficients between TCEP and other related chemicals were not statistically significant and ranged from –0.122 and 0.094. Araki et al. (2020) received a high overall quality determination.

Liao et al. (2023) assessed the association between serum TCEP concentrations and Sjögren's syndrome (SjS) in a study of 138 SjS patients and 145 controls in Hangzhou, China. The only statistically significant associations found between TCEP serum levels and SjS were for an analysis using exposure quartiles in which the observed relationship with SjS was non-linear. Compared to the lowest TCEP exposure quartile (first quartile), the odds of SjS were significantly lower in the second and third quartile and higher in the fourth quartile in the crude analyses. After adjusting for potential confounders, the inverse associations observed for the second and third quartile remained statistically significant, but the positive association for the fourth quartile was attenuated and no longer statistically significant ($p = 0.143$). Despite some issues with the terminology used, the methods described in the paper are generally consistent with a case-control design, which would be appropriate because SjS is a rare disease. Logistic regression was used, which is appropriate for a case-control study. SjS is substantially more common in women than men and may be associated with age. Although controls were matched to cases based on gender during participant selection, the statistical analyses didn't account for this matching, and some but not all analyses were stratified based on gender. Controls and cases weren't matched by age during participant selection, but the analyses adjusted for age as well as other potential confounders (smoking status and drinking habits). There are temporality concerns relevant to interpreting the findings of this study, particularly the limitation that serum TCEP levels were only assessed at a single timepoint, which didn't precede the development of the outcome. Due to limitations and limited details regarding methods, EPA assigned a medium overall quality determination to this study.

Canbaz et al. (2015) did not identify an association between TCEP levels from mattress dust in Swedish homes where 2-month-old children lived and the subsequent development of asthma when the children reached ages 4 or 8 years in a medium-quality study.

Laboratory Animals

NTP (1991b) reported no chemical-related changes in hematological parameters in rats or mice after 66 weeks of exposure and no histopathological changes in bone marrow, lymph nodes, spleen, or thymus; rats did show a statistically significant increased trend in mononuclear cell leukemia with increasing dose in a species (F344 rats) with a high rate of this cancer. No other *in vivo* animal toxicity studies were identified that studied specific immune system changes.

Mechanistic Information
Three *in vitro* studies examined immune effects. Zhang et al. (2017a) found that 12.5 to 200 mg/L TCEP was associated with a decrease of roughly 13 to 16 percent in the production of IL-6 at 24 or 48 hours in the supernatant of human hepatocytes (L02 cells). Another study (Zhang et al., 2017b) found that TCEP at 50 and 200 mg/L in liver cells for 24 or 48 hours also resulted in decreased IL-6 protein, up to roughly 33 percent. The adversity of these changes in IL-6 are unclear. For example, IL-6 can act as a pro-inflammatory cytokine but may also have anti-inflammatory properties (Scheller et al., 2011). Using the human hepatocellular carcinoma cell line HepG2, Krivoshiev et al. (2018) found that TCEP altered gene expression of effector and regulatory proteins in the inflammatory process and concluded that TCEP may influence inflammation and alter immune function. Zhang et al. (2017b) found that liver cells co-exposed to both TCEP and benzo[a]pyrene activated pathways associated with inflammation and increased expression of pro-inflammatory cytokines, whereas exposure to TCEP alone did not yield similar changes.

Evidence Integration Summary
Evidence from epidemiological studies in Canada, Sweden, or Japan did not identify an association between TCEP and childhood asthma, but a U.S. study found multiple associations with TCEP and respiratory outcomes. A study of SjS in China did not show an effect overall but when investigating different exposure quartiles, the authors identified a negative association at lower concentrations and a non-significant positive association at the highest concentration after adjusting for other factors; it is possible that there is a non-monotonic relationship. Mendy et al. (2024) note that short half-lives of chemicals such as TCEP (when measuring metabolites in urine) and lack of accounting for dietary and outdoor concentrations when using dust concentrations and loadings make interpretation of results somewhat difficult.

Overall, Mendy et al. (2024) is a high-quality study conducted in the United States that identified multiple positive associations with wheeze, allergy symptoms, and respiratory infections and was more robust in the numbers of measurements of exposure and outcomes than other studies of wheeze, allergy symptoms, and asthma. It is not clear why there were differences in outcomes among studies but Araki et al. (2020) had a very low participation rate and may not reflect a wider group. Also, Canbaz et al. (2015) focused only on children with diagnosed asthma (not symptoms). Differences in timing of when exposures were measured may have also led to differences in outcomes. EPA concluded that the human evidence is slight for immune effects, specifically for wheeze, allergies, and respiratory infections.

Animal studies did not identify histopathological changes in immune-related organs or in hematological parameters. A statistically significant increased trend in mononuclear cell leukemia with increasing dose was seen in rats. In mechanistic studies, TCEP was associated with decreases in an inflammatory cytokine and altered gene expression of inflammatory proteins in two studies, but a third study identified inflammatory changes only after co-exposure with benzo[a]pyrene. EPA has not determined whether the decreases of IL-6 in mechanistic studies are adverse.

Based on slight human evidence and indeterminate animal and mechanistic evidence, EPA concludes that the evidence suggests but is not sufficient to conclude that TCEP exposure causes immunological or hematological effects in humans under relevant exposure circumstances.

5.2.3.2.5 Thyroid

Humans
From the updated literature search (2019–2024), EPA identified a case-control study (Liu et al., 2022) in Shandong Province, eastern China that evaluated risk of thyroid cancer, but also measured the

association between TCEP in serum and thyroid hormone levels normalized by lipid weight within the control group. Among female controls, increases in TCEP exposure were associated with decreases in triiodothyronine (T3), free T3, and free thyroxine (T4) ($p < 0.05$). Thyroid stimulating hormone (TSH) was increased with increasing TCEP exposure ($p < 0.05$) among females. In male controls, the only statistically significant change was a decrease in free T4 ($p < 0.05$); males showed a decrease in TSH without statistical significance, and there was no obvious relationship with T3 for males. The model was adjusted for age, body mass index (BMI), smoking, alcohol consumption, and diabetes status. EPA gave this study a medium overall quality determination.

Liu et al. (2022) also evaluated TCEP's association with papillary thyroid cancer but found no statistically significant differences between case and control groups. Alternately, Hoffman et al. (2017) identified a statistically significant association between TCEP exposure and papillary thyroid cancer in a high-quality epidemiology study. Section 5.2.4.1 describes the cancer results in more detail.

Animals

Moser et al. (2015) found no changes in serum levels of total thyroxine (T4) and triiodothyronine (T3) in Long-Evans dams or offspring at PNDs 6 and 22 when dosed up to 90 mg/kg-day. NTP (1991b) evaluated histopathological changes in the thyroid and parathyroid in the 16-day, 16-week, and 2-year rat and mouse studies. In the 2-year study, 12 percent of male mice (6 of 50) exhibited follicular cell hyperplasia at 350 mg/kg-day vs. 6 percent of controls (3 of 60). NTP (1991b) identified increased incidences of thyroid neoplasms in rats in a 2-year cancer bioassay; the authors concluded that there is uncertainty regarding an association with TCEP exposure.

Evidence Integration Summary

Based on these data, both human and animal evidence for non-cancer thyroid effects is indeterminate. EPA did not identify any mechanistic information specific to the thyroid. Overall, the currently available evidence is inadequate to assess whether TCEP may cause non-cancer thyroid changes in humans under relevant exposure circumstances.

5.2.3.2.6 Endocrine (Other)

F0 male and female mice exhibited decreased adrenal weights after administration of 700 mg/kg-day TCEP for 18 weeks (NTP, 1991a).[25] Similar effects were not observed in other studies.

Based on indeterminate human and animal evidence and lack of mechanistic support, the currently available evidence is inadequate to assess whether TCEP may cause endocrine changes other than thyroid and reproductive hormones in humans.

Evidence related to reproductive hormones is assessed under discussed in Section 5.2.3.1.2 on reproductive and developmental toxicity endpoints.

5.2.3.2.7 Lung/Respiratory

Humans

Zhu et al. (2022) evaluated the association between natural log transformed BCEP urinary concentrations for 987 individuals aged 6 to 79 years who were part of the National Health and Nutrition Examination Survey (years 2011–2012) and five spirometry measures.

[25] Kawashima et al. (1983) measured changes in pituitary weights; this study is being translated and will be evaluated for the risk evaluation.

BCEP was related to a statistically significant decrease in forced vital capacity (FVC) in two models. In model 1 (adjusted for age, sex, and race), the beta for the natural log of BCEP concentrations associated with was –43.37 (with 95% CI: –79.79, –6.95; p = 0.02).

In the second model, FVC was decreased by a greater amount: by –79.34 (95% CI: –143.29, –15.39; p = 0.016). This model was adjusted for age, sex, race, body mass index (BMI), serum cotinine, smoking status, physical activity, family poverty/income ratio, educational level, and urinary creatinine.

FEV_1 divided by FVC was significantly related to increased BCEP in urine: β of 0.005 (95% CI: 0.001, 0.01; p = 0.007) for the first model. However, the second model was not statistically significant for this endpoint. The study received a medium overall quality determination.

In a high-quality study described in the above section on immunological and hematological outcomes, TCEP dust loadings were associated with lower FEV_1 at age 5 years (β: –6.17, 95% CI: –11.09, –1.25), but urinary BCEP at 26 weeks of gestation was associated with a higher FEV_1 (β: 4.88, 95% CI: 0.04–9.73), which would seemingly suggest a possible beneficial association (Mendy et al., 2024).

Laboratory Animals

Lung weight changes were identified after 16 weeks (an increase of 17.5 percent in absolute weight in 350 mg/kg-day female rats and decreases of 9 percent in absolute weight at 700 mg/kg-day in female mice with relative-to-body lung weight decreases of 11.7 and 8.4 percent at 350 and 700 mg/kg/day, respectively).[26] No changes were identified at the 66-week interim sacrifice in the 2-year bioassay, and no non-cancer changes in histopathology were seen in rats or mice after two years other than increased hemorrhage with dose in female rats presumed to be associated with cardiovascular collapse in dying animals (NTP, 1991b). All studies received high overall quality determinations.

Evidence Integration Summary

Based on two epidemiological studies with inconsistent results, human evidence is indeterminate. In addition, animal data are indeterminate (no relevant histopathological effects, lung weight changes in studies with high and uninformative overall quality determinations) based on high-quality studies. Therefore, the currently available evidence is inadequate to assess whether TCEP may cause lung or respiratory effects in humans under relevant exposure circumstances (see Appendix L.2).

5.2.3.2.8 Body Weight

Humans

Yang et al. (2022), a high-quality study, conducted a prospective cohort study of 340 mother-infant pairs in Cincinnati, Ohio and examined the association between (1) gestational exposure to the metabolite BCEP in urine of mothers at 16 and 26 weeks and gestational age; and (2) newborn weight, length, ponderal index (a relationship between weight and length), and head circumference in offspring.

Yang et al. (2022) found that higher maternal BCEP concentrations in urine at 16 weeks (but not 26 weeks) was associated with a measure of lower birth weight in female infants (β: –0.25 decrease in birth weight z-score; 95% CI: –0.46 to –0.04) for every 10-fold increase (µg/L) in BCEP urine concentration. This study is described in more detail in Section 5.2.3.2.9 below.

Laboratory Animals

[26] A decrease was also seen in female rats after 16 days, but the study is uninformative due to a viral infection in the lungs and salivary glands (NTP, 1991b).

Changes in body weight are of concern and can suggest an underlying toxicity. For TCEP, most studies ranging from 14 days at doses up to 1,000 mg/kg-day to two years at doses up to 88 and 350 mg/kg-day in rats and mice, respectively showed no body weight changes greater than 10 percent (Yang et al., 2018a; NTP, 1991a, b). Likewise, dams, fetuses, and pups exhibited no significant body weight changes when dams were dosed up to 940 mg/kg-day during gestation or gestation and lactation (Moser et al., 2015; Hazleton Laboratories, 1983). Changes were also not observed in adjusted pup weights, F0 or F1 dams at delivery, or in adult males in the continuous breeding study (NTP, 1991a).

Differences in body weights compared with controls were observed in only a few studies. Body weights of male ICR mice decreased as much as 14.8 percent at 300 mg/kg-day TCEP after 35 days (Chen et al., 2015a). Another study identified a 20 percent increase among female rats after 16 weeks exposure to 350 mg/kg-day TCEP (NTP, 1991b).

In the continuous breeding study, F0 dam weights were decreased at 350 and 700 mg/kg-day from PNDs 7 through 21 (statistically significant trend, with up to 30 percent decrease for the single dam evaluated at 700 mg/kg-day). In contrast, females in the 350 mg/kg-day group exhibited a 17 percent increase in body weight at weaning but not during weeks 28 through 30 (NTP, 1991a). Overall, TCEP effects on body weight were not consistent across studies and when observed, were not consistently increased, or decreased.

Evidence Integration Summary

EPA identified no human studies that had information on body weight changes and therefore, human evidence is indeterminate. In animal toxicity studies, TCEP effects on body weight were not consistent across multiple studies. When body weight changes were observed, they were not consistently increased or decreased. Therefore, the animal data are indeterminate. Overall, the currently available evidence is inadequate to assess whether TCEP may cause changes in body weight in humans under relevant exposure circumstances (see Appendix L.2).

5.2.3.2.9 Developmental Toxicity

U.S. EPA (1991) identifies death, structural abnormalities, altered growth, and functional deficits as the four major manifestations of developmental toxicity. This section describes relevant measurements related to these outcomes in epidemiolocal studies (as well as pre-term birth and differences in gestational age of offspring), in prenatal/postnatal studies in mice and rats, and in the continuous breeding study in mice. This section also describes effects in animals measured during adolescence, a relevant developmental lifestage (U.S. EPA, 1991). Mating and fertility outcomes resulting from the continuous breeding study are described in Section 5.2.3.1.2.

Humans

EPA identified four epidemiological studies evaluating growth and gestational age since publication of the draft risk evaluation. Crawford et al. (2020) was identified in the updated literature search (2019–2024) conducted by EPA. Peer reviewers identified three additional studies (Oh et al., 2024; Hernandez-Castro et al., 2023b; Yang et al., 2022). All studies examined the association between BCEP (a TCEP metabolite) in pregnant mothers' urine and growth/gestational age. The studies are described based on the size of the cohort, with the largest cohort discussed first. Additional studies related maternal exposures during gestation are described in Sections 5.2.3.1.1 and 5.2.3.2.4 for and neurotoxicity and immunological effects, respectively.

Oh et al. (2024), to which EPA assigned a medium overall quality determination, was the largest cohort study that investigated the association between urinary concentrations of BCEP (identified as BCETP in the study) and various measures of gestational age and birth weight. BCEP measurements were obtained

primarily from second and third trimesters from 6,646 pregnant women in 16 cohorts across the United States. BCEP concentrations were divided into non-detect, low exposure, and high exposure categories for analysis.

Oh et al. (2024) found that the association of the high vs. the non-detect BCEP group with gestational age and preterm birth differed by sex ($p < 0.01$). Female offspring had a shorter gestational age (β: −0:13 weeks; 95% CI: −0.24, −0.03) and a higher odds of pre-term birth for gestational age (odds ratio [OR]: 1.27; 95% CI: 1.03, 1.58). The associations for male offspring were not statistically significant but did show that they had longer gestational age (β = 0.06 weeks; 95% CI: −0.06, 0.18) and lower odds of pre-term birth (OR: 0.77; 95% CI: 0.55, 1.06). The regression models were adjusted for maternal measures of race/ethnicity, age at delivery, education, marital status, pre-pregnancy body mass index (BMI), smoking during pregnancy, and parity, as well as the year and season of sample collection.

The Spearman coefficients (–0.017 to 0.189) showed weak correlations among BCEP and other OPE metabolite biomarkers, and when regression models were run with all OPE biomarkers, results were similar to the primary results. The authors noted that removing one cohort at a time demonstrated that the model was robust; although some estimates were strengthened or weakened with removal of individual cohorts, the direction of the associations did not change (Oh et al., 2024).

Among measures of birth weight, high BCEP was associated with lower odds of being small for gestational age (SGA) for both sexes combined (OR: 0.83; 95% CI: 0.71, 0.96, $p < 0.01$). In addition to the model adjustments identified above, this full model (with both sexes combined) also adjusted for child's sex. In sex-specific models, the association for SGA was statistically significant for males for both the low BCEP group (OR: 0.73; 95% CI: 0.55, 0.97) and the high group (OR: 0.77; 95% CI: 0.62, 0.97) compared with the non-detects (Oh et al., 2024).[27]

Hernandez-Castro et al. (2023b) recruited pregnant women from health clinics, a private obstetrics and gynecological practice, community meetings, and advertisements in Los Angeles, California for a prospective cohort study that investigated the association between BCEP and both gestational age and birthweight among infants. The authors measured BCEP in third trimester urine samples from 421 mothers, who were primarily low income and Hispanic/Latina. The authors also investigated associations between exposure to mixtures of OPEs and these same birth outcomes. The authors did not identify any associations for BCEP measurements for the full model or when evaluating effects in female and male infants separately. The model was adjusted for recruitment site, maternal age, season of sample collection, gestational age, race/ethnicity, pre-pregnancy BMI, income, education, infant birth order, maternal hypertensive disorders of pregnancy; infant sex was also used as a covariate for the gestational age model that included both sexes. When evaluating the OPE mixture, Hernandez-Castro et al. (2023b) found an association with lower gestational age, primarily based on female infant data but the association was not statistically significant. Spearman correlations between BCEP and other OPE metabolite concentrations were generally weak and ranged from 0.02 to 0.22. EPA assigned a medium overall quality determination to this study.

Yang et al. (2022) conducted a prospective cohort study of 340 mother-infant pairs in Cincinnati, Ohio and examined the association between (1) gestational exposure to the metabolite BCEP in urine of mothers at 16 and 26 weeks and gestational age; and (2) newborn weight, length, ponderal index (a relationship between weight and length), and head circumference in offspring. BCEP at 26 weeks was

[27] Table S8 identifies 0.73 and 0.77 as β values. However, the text describes these male-specific effects as the odds of occurrence and Table 4 presents ORs for SGA (small for gestational age). Therefore, EPA assumes that the identification of these numbers in Table S8 should have been as ORs instead of β values.

positively associated with gestational age for both sexes combined (β: 0.33-week increase; 95% CI: 0.06–0.61) for every 10-fold unit increase in BCEP urine concentration (in µg/L), but not when stratified by sex. Increases of $\log_{10}$ BCEP concentrations at 26 weeks were also associated with lower pre-term birth (RR = 0.48; 95% CI: 0.27, 0.83), but the authors suggested that the results related to pre-term birth need to be interpreted with caution because there were only 30 pre-term births. The model was adjusted for maternal age at delivery, race, household income, education, marital status, infant sex, parity, and pre-pregnancy BMI, serum cotinine/blood lead levels at 16 weeks gestation.

Yang et al. (2022) ran two other models and identified similar associations with pre-term birth or gestational age. First, the Cox proportional hazard model showed that a $\log_{10}$ increase in BCEP concentrations at 26-weeks was inversely associated with pre-term birth (hazard ratio (HR): 0.40; 95% CI: 0.19, 0.83). A similar relationship using the Cox model was seen for the average of the 16 and 26-week BCEP concentrations and pre-term birth (HR: 0.41; 95% CI: 0.17, 0.98). When divided into tertiles with the lowest exposure tertile as reference, the 26-week BCEP concentration was positively associated with gestational age (p for trend = 0.02). These models were adjusted for the same covariates identified for the above model (Yang et al., 2022).

Yang et al. (2022) also found that higher maternal BCEP exposure at 16 weeks (but not 26 weeks) was associated with lower birth weight z-scores[28] in female infants (β: –0.25 decrease in birth weight z-score; 95% CI: –0.46 to –0.04) for every 10-fold increase (µg/L) in BCEP urine concentration. The 10-fold (µg/L) BCEP urine concentration increases were associated with decreased female infant length z-scores, with β of –0.31 (95% CI: –0.56, –0.07) at 16 weeks and β of –0.18 (95% CI: –0.35, –0.02) at 26 weeks. EPA assigned a high overall quality determination to this study.

A pilot prospective cohort study (Crawford et al., 2020) evaluated BCEP (identified as BCHP in the study) in urine among 56 pregnant women representing the general population of Rhode Island and examined the association between BCEP and various anthropometric measures in children born to these mothers at birth and 6 weeks of age. Seventy-one percent of the women provided samples for all three trimesters. BCEP was associated with an overall increased thigh skinfold thickness in males and females (β: 0.34 mm; 95% CI: 0.16, 0.52); and increased subscapular skinfold thickness in males (β: 0.14 mm; 95% CI: 0.002, 0.28) (p = 0.05 for the effect modification by infant sex). Results were adjusted for maternal age at delivery, income, pre-pregnancy BMI, parity, infant sex, and age at the time of measurement. As noted by Crawford et al. (2020), skinfold thickness measurements correlate well with subcutaneous fat distribution. Although researchers may measure skinfold thickness differently from each other, they received the same training, and any measurement errors were expected to be minimized. BCEP was not associated with infant weight, length, or head and abdominal circumferences. Also, there were no statistically significant effects of BCEP on infant feeding behavior, but BCEP did show a tendency for increased general appetite (β: 0.11; 95% CI: –0.04, 0.27). Crawford et al. (2020) note that based on the number of pregnant women included in this pilot study, it was underpowered to investigate the effects of OPE mixtures. The authors also did not analyze correlations among chemical exposures. EPA gave the study a high overall quality determination.

Percy et al. (2021), Percy et al. (2022), and Hernandez-Castro et al. (2023a) examined prenatal BCEP concentrations and neurobehavioral measures and found some negative association with IQ for individuals in certain groups associated with lower SES in one study (Percy et al., 2022) (see Section 5.2.3.1.1).

[28] Z-scores relate infant size measures to the population distribution. For example, an infant with an average body weight has a weight z-score of 0 and an infant with a weight one standard deviation higher than the mean population weight has a z-score of +1.

Mendy et al. (2024) identified some associations between mothers' exposure to BCEP during gestation and children's respiratory symptoms as described in Section 5.2.3.2.4.

Table 5-46. Associations between BCEP (TCEP Metabolite) in Urine of Pregnant Women and Growth and Gestational Age

#Mom: Infant Pairs	Geographic Region	BCEP During Pregnancy[a]	Birth Outcomes Measured[b]	Statistically Significant Associations	Model Adjustments	Co-Exposures Measured	Citation, OQD
6,646	16 individual cohorts across the United States	Second and third trimesters exposure groups: non-detect, low, high [50%ile: 0.52 ng/mL; 5%ile < LOD (0.02); 95%ile: 8.22]	GA, pre-term, early term, full term post/late term BW at birth (BW-GA z-scores, SGA, LGA, LBW)	*Both sexes:* ↓ SGA (high BCEP) *Females:* ↓ GA, ↑ pre-term[c] *Males:* ↓ SGA (low and high BCEP)	Maternal age at delivery, education, race/ethnicity, marital status, pre-pregnancy BMI, smoking during pregnancy, parity, as year and season of sample collection	Weak correlations between BCEP and other OPEs (-0.017 to 0.189); multiple OPE model supports BCEP results	Oh et al. (2024) OQD = Medium
421	Los Angeles, California; primarily low-income Hispanic/ Latina	Third trimester (31.5 +/- 2.0 wks) [GM = 0.31 ng/mL; 25, 50, 75 %ile = 0.03, 0.53, 1.62; min – ND; max - 168; LOD – 0.02]	GA, BW (as BW-GA z-scores)	*BCEP*: None for full model or by sex *OPE mixture, females (primarily):* ↓GA	Recruitment site, maternal age, season of sample collection, gestational age, race/ethnicity, pre-pregnancy BMI, income, education, infant birth order, maternal hypertensive disorders of pregnancy; infant sex (GA model with both sexes)	Weak correlations between BCEP and other OPEs (0.02 to 0.22); multiple OPEs *not* included in models	Hernandez-Castro et al. (2023b) OQD = Medium
340	Cincinnati, Ohio >50% non-Hispanic	16 and 26 wks [GM (GSD): 0.60 (3.16) µg/L at 16 wks 0.51 (4.33) µg/L at 26 wks]	GA, pre-term at birth, newborn BW, length, ponderal index (relationship between weight and height), and head circumference	*Both sexes*: ↑ GA (main model and tertiles/trend test, 26-wk BCEP); ↓ pre-term birth (main model and Cox, 26 or avg. 16/26-wk BCEP) *Females:* 16-wk BCEP: ↓ BW z-score; avg 16/26 wk BCEP: ↓ length z-score	Maternal age at delivery, race, household income, education, marital status, parity, and pre-pregnancy BMI, serum cotinine/blood lead levels at 16 weeks gestation; infant sex in models with both sexes	No	Yang et al. (2022) OQD = Medium
56	Rhode Island	12, 28, and/or 35 wks[d] [Median = 0.31 ug/L; 0.17 to 0.60 for interquartile range]	Skinfold thickness, BW, length, head, and abdominal circumferences, feeding behavior at birth and 6 wks	*Both sexes:* ↑ thigh skinfold thickness *Males:* ↑ subscapular skinfold thickness	Maternal age at delivery, income, pre-pregnancy BMI, parity, and age at the time of measurement; infant sex in models with both sexes	No	Crawford et al. (2020) OQD = High

[a] BCEP concentrations were normalized by specific gravity of the urine. [b] GA = Gestational age; BW = body weight; SGA and LGA (<10th and >90th percentiles of BW at gestation, respectively); LBW (birthweight <2,500 g at ≥37 weeks gestation); [c] Males had ↑GA and ↓ pre-term birth that was *not* statistically significant; [d] 71 percent of mothers had data for all three timepoints.

Laboratory Animals
EPA identified three prenatal/postnatal animal studies. Two studies received high overall quality determinations. Hazleton Laboratories (1983) administered 940 mg/kg-day TCEP via oral gavage to female CD-1 mice from GD 7 to 14. Dams exhibited clinical signs of neurotoxicity but no differences in measures of live or dead pups per litter. In addition, there were no changes in fetal or pup weights.

Similarly, Long-Evans rat dams were dosed from GD 10 to PND 22 via oral gavage at 0, 12, 40, and 90 mg/kg-day (decreased from 125 mg/kg-day after 5 days) in the developmental neurotoxicity study described in Section 5.2.3.1.1. There were no differences in litter size on PND 2 or changes in offspring weight (Moser et al., 2015).[29,30]

Kawashima et al. (1983) evaluated effects of TCEP exposure on developmental outcomes after dosing pregnant Wistar rats via oral gavage with 0, 50, 100, or 200 mg/kg-day from GD 7 through 15, and EPA assigned it a medium overall quality determination. Seven of 30 dams (23%) dosed with 200 mg/kg-day died during gestation, as noted in Section 5.2.3.1.1. Kawashima et al. (1983) is the only study that evaluated teratogenicity and skeletal variations after TCEP exposure. The authors identified no increased malformations or variations. Offspring exhibited a 27 percent decrease in absolute pituitary weight at the highest dose (with maternal toxicity) but no changes in multiple other organ weights (Kawashima et al., 1983). Offspring exposed to TCEP *in utero* showed no significant differences from controls in numbers of surviving pups, sex ratio, body weight, or mortality rate. Section 5.2.3.1.1 describes the functional and neurobehavioral effects in male offspring from this study.

In the RACB protocol NTP (1991a), the 350 and 700 mg/kg-day mice exhibited decreases in average number of litters per pair and live pups per litter ($p < 0.001$).

During crossbreeding of F0 mice, the 700 mg/kg-day male × control female group yielded decreased live F1 pups per litter (statistical analysis not possible because only one litter was delivered). Results of 700 mg/kg-day females crossed with control males also led to decreases in live F1 pups per litter ($p < 0.01$ males; $p < 0.05$ both sexes). Outcomes from treated males × control females were more pronounced, with production of just 1 litter with 3 live pups vs. 12 litters and 7.2 live pups per litter from treated females × untreated males. The control × control group resulted in 12 litters and 10.3 live pups per litter compared with either 700 mg/kg-day males or females crossbred with controls (NTP, 1991a).[31,32]

After F1 breeding, there were decreased numbers of live F2 pups per litter at the highest dose of 350 mg/kg-day ($p < 0.05$). Although live male F2 pups per litter were also reported as being significantly decreased at 175 mg/kg-day (NTP, 1991a), EPA identified a discrepancy in NTP's Table 4-4 in the proportion of males.

Effects were more pronounced across generations. The same dose (*e.g.*, 350 mg/kg-day) resulted in fewer live F2 pups per litter (7.6) than live F1 pups per litter (10.1) (NTP, 1991a).

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[29] Limited information from the unavailable Russia inhalation study in rats, Shepel'skaia and Dyshginevich (1981) identified decreased body weight and crown rump length in rat offspring at 0.5 mg/m^3.
[30] NTP (1991a) identified no effects on sex ratio in the first generation, and although significant differences in sex ratio from controls were observed in the second generation, there is uncertainty in the change due to a discrepancy in reporting of proportion of male offspring born alive at the highest dose (0.41 vs. 0.45). Moser et al. (2015) did not identify effects on sex ratio. Hazleton Laboratories (1983) did not describe whether sex ratio was measured.
[31] The number of breeding pairs examined ranged from 18 to 20 among dose groups.
[32] Shepel'skaia and Dyshginevich (1981) cited in (NTP, 1991a) (unobtainable Russian abstract) resulted in dams with significantly decreased litter size and increased pre- and post-implantation loss at 1.5 mg/m^3.

Mechanistic Information
Yonemoto et al. (1997) identified an IP50 (inhibitory concentration for cell proliferation) 3,600 µM of TCEP using rat embryo limb bud cells. The ID50 (inhibitory concentration for differentiation) was identified as 1,570 µM. The authors concluded that the high proliferation to differentiation ratio suggested that TCEP should be investigated more fully for developmental toxicity.

In vivo and *in vitro* studies found TCEP to affect male reproductive hormones as noted in Section 5.2.3.1.2 including decreases in both testosterone secretion and decreases in a gene associated with testosterone synthesis in mouse Leydig (TM3) cells (Chen et al., 2015a; 2015b). These reproductive studies may support observed developmental effects based on effects on offspring viability observed after crossbreeding treated males with control females.

In other *in vitro* studies, TCEP was not associated with estrogenic or anti-estrogenic effects or changes in AR-mediated gene expression or ERα and AhR target gene activation (Reers et al., 2016; Follmann and Wober, 2006). TCEP did not exhibit estrogenic activity in in MCF-7 cells but did yield anti-estrogenic activity when co-treated with E2 (Krivoshiev et al., 2016).

Evidence Integration Summary
EPA located four human epidemiological studies that found changes in gestational age and some growth measures, including some increased skinfold thickness in offspring associated with maternal urine concentrations of BCEP, a TCEP metabolite, but did not identify any associations with other body composition measurements. One study identified a decrease in IQ among lower SES children and another identified increased respiratory effects among children whose mothers' showed exposure to BCEP during gestation (or at delivery).

EPA has concluded that the human evidence is slight for developmental effects given that there are associations for several endpoints related to growth, gestational age, and other effects, but there are some inconsistencies among results (*e.g.*, increased gestational age for both sexes in one experiment; decreased gestational age and increased pre-term birth for females in another) and a lack of effect on gestational age and growth in one of the larger studies Hernandez-Castro et al. (2023b).

Animal studies show slight evidence for developmental effects. Developmental outcomes such as decreased live pups per litter were observed in the NTP RACB study (described in Section 5.2.3.1.2) with increased severity in the second generation. However, the prenatal and prenatal/postnatal studies did not result in developmental outcomes, except under severe maternal toxicity. Although differences in study protocols between the RACB and prenatal studies may explain differences in outcomes, there is only one acceptable study that investigated exposure prior to gestation. There is some support for the potential for developmental effects based on male reproductive toxicity observed in animal studies (see Section 5.2.3.1.2).

The limited mechanistic evidence of reproductive toxicity can be relevant as considerations for developmental toxicity. EPA considers the supporting mechanistic data to be slight.

Overall, EPA concluded that evidence suggests but is not sufficient to conclude that TCEP exposure causes developmental toxicity in humans under relevant exposure circumstances. This conclusion is based on effects in four epidemiological studies using the TCEP metabolite BCEP as a biomarker that

are inconsistent and fertility-related changes in the RACB study. The oral studies in mice and rats used as a basis for this decision evaluated doses of 12 to 700 mg/kg-day (Table_Apx L-3).

5.2.4 Cancer Hazard Identification, MOA Analysis, and Evidence Integration

The sections below outline human (Section 5.2.4.1) and animal evidence (Section 5.2.4.2) for carcinogenicity as well as and an MOA summary (Section 5.2.4.3) and a summary of evidence integration conclusions (Section 5.2.4.4).

5.2.4.1 Human Evidence

One high-quality case-control cancer study examined the association between TCEP/other flame-retardant exposure and papillary thyroid cancer in adults living near Duke University in North Carolina (Hoffman et al., 2017). TCEP concentrations in dust were measured in 70 age- and gender-matched cases and controls in 2014 to 2016; no biological measurements were collected for TCEP. The authors identified a median TCEP concentration of 400 ng/g in dust. Diagnosis of papillary thyroid cancer was positively associated with TCEP concentrations above the median. The odds ratio is 2.42 (CI: 1.10–5.33) ($p < 0.05$).

In contrast, another case-control study with a medium overall quality determination that was conducted in Shandong Province, eastern China (Liu et al., 2022) did not identify any statistically significant associations between TCEP and papillary thyroid cancer. The study compared upper quartiles of TCEP concentration in serum normalized by lipid weight with the lowest quartile concentration. The results were adjusted for age, BMI, smoking, alcohol consumption, and diabetes status in the regression model. ORs were greater than one only for males (1.24) for the 50 to 75th percentile (but not the highest (>75th) percentile) and only for females for the highest percentile comparison (1.14). None of the associations were statistically significant. See Section 5.2.3.2 for information on TCEP's association with thyroid hormones from this study.

Li et al. (2020) examined the association between TCEP and other OPEs in plasma and prevalence of gastrointestinal and colorectal cancers (all stages) in Wuhan, China. There were 34 cases of gastrointestinal cancer and 40 cases of colorectal cancer and 62 controls, who were health individuals without cancer. EPA gave this study a medium overall quality determination. TCEP was detected in gastrointestinal and in colorectal cancer patients more frequently than in the control group, for which TCEP was not detected ($p < 0.01$ for both comparisons); the concentrations were also higher in each of the cancer groups ($p < 0.01$). However, there was no statistically significant association between TCEP concentrations in plasma and the presence of either cancer when using binary logistic regression for an unadjusted model or for a model adjusted for sex and gender.

Liu et al. (2021) investigated TCEP in plasma and female-related cancers. A total of 258 women were recruited from a hospital in Wuhan, China in April 2019 with benign breast tumors (n = 45), breast cancer (n = 73), benign uterine tumors (n = 62), and cervical cancer (n = 78). The concentration of TCEP in the benign uterine tumor group was higher than the cervical cancer group ($p < 0.05$). No other associations were identified for TCEP between the benign and cancerous tumor groups. EPA gave this study a low overall quality determination partly based on lack of controls without tumors as well as small sample sizes per group, and use of cross-sectional design to evaluate a chronic risk such as cancer.

5.2.4.2 Animal Evidence

EPA identified one oral NTP cancer bioassay in which F344/N rats $B6C3F_1$ mice (50 per sex per dose of each species) were administered TCEP in corn oil via oral gavage for 5 days per week for 104 weeks.

Rats received 0, 44, or 88 mg/kg and mice received 0, 175, or 350 mg/kg (NTP, 1991b). The study received high overall quality determinations for the tumor incidence data.

NTP (1991b) identified multiple tumors and concluded that there is clear evidence of carcinogenic activity of renal tubule adenomas in male and female rats. The authors also concluded that thyroid follicular cell neoplasms and mononuclear cell leukemia in rats may have been related to TCEP administration but acknowledge uncertainty related to this association. There was equivocal carcinogenic evidence based on marginally increased incidence of renal tubule cell neoplasms in for male mice and marginally increased incidence of harderian gland adenomas in female mice.[33]

Kidney Tumors

Rats: At the 66-week sacrifice, one high-dose male had a renal tubule adenoma. At the end of the study, high-dose male rats exhibited increased incidences of renal tubule adenomas (48%) vs. control rats (2%) ($p < 0.001$) and a dose-response trend was evident ($p < 0.001$). Male rats also exhibited hyperplasia of the renal tubule epithelium, with 48 percent incidence at the high dose (vs. 0 percent in controls). One control and one high dose male developed a renal tubule carcinoma. High-dose females had a lower incidence of renal tubule adenomas (10%), but incidence was higher than controls (0%) ($p < 0.05$) with a statistically significant dose-response trend ($p < 0.001$). High dose females also exhibited a 32 percent incidence of focal hyperplasia of the renal tubule epithelium vs. 0 percent in controls.

Rats exhibited lower survival rates at 88 mg/kg-day after dosing with TCEP: 51 vs. 78 percent in controls in males and 37 vs. 66 percent in controls for females. Female survival started to decrease at week 70 and many rats exhibited brain lesions, whereas males' decreased survival was limited to the final month of the study.

Mice: Mice exhibited no decreases in survival. At the end of the study, eight percent of high-dose male mice had either renal tubule adenomas or adenocarcinomas compared with 2 percent in controls. Only one low dose female exhibited a renal tubule adenoma. Six percent of mice exhibited renal tubule cell hyperplasia. All treated mice had statistically significant increases in enlarged nuclei in renal tubule epithelial cells (NTP, 1991b). No kidney-related lesions were observed at the 66-week interim sacrifice.[34]

Other Tumors

Hematopoietic System: Mononuclear cell leukemia (MNCL) was increased in male rats at both doses (28 and 26 percent, respectively) vs. 10 percent in controls. Because these are fatal neoplasms, life table analyses are considered important and showed statistical significance for the low and high doses vs. controls ($p < 0.05$) and for a dose-response trend ($p = 0.01$). Female rats exhibited a slight increase at the high dose (40%) compared with controls (28%) and exhibited a dose-response trend ($p < 0.01$). Although MNCL may relate to TCEP exposure, the increase in male rats was not clearly dose-related and was partly due to incidence that was lower than expected in the controls. In addition, historical

[33] Takada et al. (1989) dosed ddY mice at 0, 0.012, 0.06, 0.3, or 1.5 percent TCEP to ddY mice in the diet for 18 months and identified increased incidence of tumors in multiple target organs; this study is not in English and was not translated or evaluated for data quality. Takada et al. (1989) was, however, described in the 2009 PPRTV for TCEP (U.S. EPA, 2009). U.S. EPA (2009) presented estimated doses for this study as 0, 9.3, 46.6, 232.8, and 1,687.5 for males and 0, 10.7, 53.3, 266.7, and 1,875 for females using measured data for body weight and food consumption from the bioassay in the following equation: % diet × 10,000 × estimated food consumption)/estimated body weight.

[34] Takada et al. (1989) identified an incidence of 82 percent renal cell adenomas and carcinomas in male mice at the highest concentration vs. 4 percent in controls ($p < 0.01$).

control values for these neoplasms are variable and all incidences in the current study were within historical controls (NTP, 1991b).[35]

Thyroid: Other notable tumors in rats identified in the NTP (1991b) bioassay included slightly increased incidences of thyroid combined follicular cell adenomas and carcinomas observed in high-dose males (10 vs. 2 percent control males) and in high-dose females (8 vs. 0 percent in controls). The incidence in females exhibited a statistically significant dose-response trend and pairwise comparison at the highest dose ($p < 0.05$). NTP concluded that these tumors may be related to TCEP exposure. However, the increases were considered marginal. In addition, female rats did not exhibit thyroid follicular hyperplasia, and NTP (1991b) states that most thyroid carcinogens also cause hyperplasia.

Harderian Gland: At the 66-week sacrifice in NTP (1991b), two high-dose female mice had adenomas of the harderian gland and a third had a harderian gland carcinoma. In female mice, combined incidence of harderian gland adenomas and carcinomas from both the 66-week and terminal sacrifices were increased (5, 13, and 17 percent for controls, low, and high doses). Both the high-dose incidence vs. controls and dose-response trend were statistically significant ($p < 0.05$).[36]

Liver: Male mice exhibited a significant positive trend for hepatocellular adenoma ($p < 0.05$) with 40, 36, and 56 percent incidence in controls, 175, and 350 mg/kg-day, respectively. However, the increase at the high dose compared with controls was not statistically significant and there was no increase in hepatocellular carcinomas compared with controls. Male mice also exhibited increased eosinophilic foci (16 vs. 0 percent at the high dose compared with controls) but no increase in basophilic or clear cell foci, which constitutes a morphological continuum with hepatocellular adenoma (NTP, 1991b).[37]

Uterine: Three female rats had uterine stromal sarcomas at the high dose but none in controls or the low-dose group. Although the trend test was significant ($p < 0.05$), the incidence in the high-dose group was not significantly greater than in concurrent or historical controls and thus, NTP (1991b) concluded that the uterine tumors were not related to TCEP administration.

Mammary Gland: Three high-dose female mice had adenocarcinomas of the mammary gland with a positive trend ($p < 0.05$). However, a fibroadenoma occurred in a female control; there was no significant trend for fibroadenoma, or adenocarcinoma combined; and the incidence of adenocarcinomas is within female historical vehicle controls. Therefore, NTP (1991b) concluded that the mammary gland adenocarcinomas were not related to TCEP treatment.

5.2.4.3 MOA Summary

The U.S. EPA (2005b) *Guidelines for Carcinogen Risk Assessment* defines mode of action as "a sequence of key events and processes, starting with the interaction of an agent with a cell, proceeding through operational and anatomical changes and resulting in cancer formation." Hard (2018) has identified modes of action for renal tubule carcinogens that include direct DNA reactivity, indirect DNA reactivity resulting from formation of free radicals, bioactivation involving glutathione conjugation, mitotic disruption, sustained cell proliferation resulting from direct cytotoxicity, sustained cell proliferation after disruption of a physiologic process (such as alpha 2u-globulin nephropathy), chemical exacerbation of chronic progressive nephropathy among others.

[35] Takada et al. (1989) found increased incidence of leukemia (type not specified) in female ddY mice (18 percent at ≈ 266.7 and 1,875 mg/kg-day) compared with two percent in controls ($p < 0.05$).
[36] There were no increases in harderian gland tumors in male or female ddY mice (Takada et al., 1989).
[37] Takada et al. (1989) identified increased hepatocellular adenomas or carcinomas in male ddY mice of 26 and 38 percent at 232.8 and 1,688 mg/kg-day in the diet compared with 8 percent in controls ($p < 0.01$).

The target organ with the most robust evidence of carcinogenicity for TCEP is the kidney. In addition to genotoxicity information on multiple cell types, EPA summarizes other biochemical and cellular effects primarily in renal cells and kidneys. EPA did not conduct a formal analysis using concordance tables to separately evaluate postulated MOAs according to the International Programme on Chemical Safety (IPCS) *Conceptual Framework for Evaluating a Mode of Action for Chemical Carcinogenesis* (Sonich-Mullin et al., 2001). Available data from *in vitro* studies identified effects associated with TCEP and that identify a variety of biochemical changes that might be relevant to induction of kidney tumors resulting from TCEP exposure. However, only sparse *in vivo* evidence was available to understand the temporality of precursor events associated with inducing kidney tumors.

Based on extensive data on tests of mutagenicity, EPA concludes that a mutagenic mode of action is not a likely MOA for TCEP, as noted in Section 5.2.3.2.9 and Appendix M.

TCEP was associated with effects in 28-day studies in kidneys (OSOM and cortex) at 350 mg/kg-day that included cell cycle deregulation, apoptosis, increases in regenerating tubules, and increased markers of cell proliferation (but no accompanying proliferative lesions) (2012b; Taniai et al., 2012a). The authors surmise that cell proliferation along with aberrant regulation of the cell cycle (*e.g.*, from the G2 phase during which macromolecules are produced to prepare for cell division and through the M phase of mitosis) may lead to chromosome instability linked to cancer. The accompanying apoptosis may reflect aberrant cell cycle regulation (Taniai et al., 2012b). It is also possible that DNA damage may have been a precipitating factor in the increase of one of the markers (topoisomerase IIα) (Taniai et al., 2012a).

In vitro studies showed that primary rabbit renal proximal tubule cells (PTCs) exposed to TCEP exhibited altered expression of cell cycle regulatory proteins, reduced DNA synthesis, inhibition of ion- and non-ion-transport functions (*e.g.*, decreased uptake of sodium, calcium, etc.), and induced cytotoxicity. Increased expression of pro-apoptotic regulatory proteins and decreased expression of proteins that inhibit apoptosis were also observed (Ren et al., 2012; Ren et al., 2009, 2008).

Studies of other tissues and cell types exposed to TCEP identified cell cycle changes, perturbation of cell signaling pathways, markers of oxidative stress, impaired mitochondrial function, inhibition of glutathione, and other effects (see Table_Apx L-6).

In NTP (1991b), the authors reported no hyperplasia in rats at the 66-week interim sacrifice in the narrative (data tables not included). Although focal hyperplasia was observed and can be expected to be a precursor to tumors, the only related finding regarding kidney tumors at the 66-week sacrifice was a single renal tubule adenoma seen in female rats. Therefore, evidence of temporal progression from hyperplasia to adenoma and then carcinoma is not available. At two years, hyperplasia was observed in male rats, but incidence was slightly lower (0, 2, and 24) than adenomas (1, 5, and 24) compared with hyperplasia at 0, 44, and 88 mg/kg-day. The lack of temporality and limited information on precursor lesions and their relationship with tumors leads to uncertainty regarding dose-response progression from hyperplasia to adenomas and carcinomas in males. Female rats did have higher rates of hyperplasia (0, 3, 16) than adenomas (0, 2, 5), at 0, 44, and 88 mg/kg-day, respectively.

Conclusion

Several studies have investigated biochemical and cellular changes in kidneys or renal cells that may be associated with steps in an MOA for kidney cancer. EPA has not performed a formal analysis on postulated MOAs (*e.g.*, as in Sonich-Mullin et al. (2001)). However, available *in vitro* studies and a few

in vivo studies that identify multiple biochemical changes that might be relevant to induction of kidney tumors. There is sparse information on temporality and dose-response of potential pre-cursor events within the *in vivo* studies and no clear NOAEL regarding tumor response to be able to confidently model tumor incidence with a non-linear/threshold dose response analysis as an alternate possible dose-response.

U.S. EPA's PPRTV (U.S. EPA, 2009) concluded that the overall weight of evidence for mutagenicity is negative and that no mechanistic data identify specific potential key events in an MOA for kidney or other tumors induced by TCEP exposure other than a general association with known proliferative and preneoplastic lesions.

5.2.4.4 Evidence Integration Summary

EPA concludes that TCEP is likely to be carcinogenic to humans using guidance from the Agency's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b). This conclusion is based on clear evidence of carcinogenic activity in rats based on renal tubule adenomas, equivocal evidence of kidney tumors in mice, the rarity of the kidney tumors in rodents, and equivocal evidence of several other tumors in rats or mice. Tumor incidence data are based on oral chronic bioassays in rats and mice that assessed dose levels between 44 and 350 mg/kg-day. Table_Apx L-6 provides details regarding EPA's evidence integration conclusion for cancer.[38]

There is indeterminate evidence in humans from four studies. One identified an association between TCEP and papillary thyroid cancer (Hoffman et al., 2017). A second one did not find an association for the same type of cancer (Liu et al., 2022). Although TCEP exposure was higher for cases, female-related cancers (Liu et al., 2021) and gastrointestinal/colorectal cancers (Li et al., 2020) were not clearly associated with TCEP exposure.

In laboratory animal studies, there is evidence of carcinogenicity in two species and both sexes in a single high-quality study. Evidence for kidney tumors is robust based on increased incidence of renal tubule adenomas in male and female F344/N rats and marginal increases in these tumors in male B6C3Fl mice (NTP, 1991b). The rarity of these tumors in F344/N rats and B6C3F1 mice strengthens the evidence.

Lesions observed in kidneys include focal hyperplasia, renal tubular cell enlargement (karyomegaly), and adenomas and carcinoma in rats and/or mice (NTP, 1991b). This continuum of has been observed with renal tubular cell cancer in humans (Beckwith, 1999). Two-year cancer bioassay for a similar chemical, tris (2,3-dibromopropyl) phosphate (CASRN 126-72-7), also resulted in kidney tumors in male and female rats and male mice and karyomegaly in mice (NTP, 1991b).

For MNCL, evidence is slight. NTP (1991b) observed significant pairwise increases and dose-response trends of MNCL in male and female F344/N rats. However, MNCL is common in F344 rats, its spontaneous incidence varies widely, and incidences in male rats exposed to TCEP were within historical controls. Occurrence of these tumors is rare in mice and other strains of rats (Thomas et al., 2007). Further, there is uncertainty regarding similarity to tumors in humans. MNCL may be similar to large granular lymphocytic leukemia (LGLL) in humans (Caldwell et al., 1999; Caldwell, 1999; Reynolds and Foon, 1984), particularly an aggressive form of CD3- LGL leukemia known as aggressive natural killer cell leukemia (ANKCL) (Thomas et al., 2007). However, Maronpot et al. (2016) note that

[38] Using the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), the equivalent conclusion is that TCEP likely causes cancer in humans under relevant exposure circumstances.

ANKCL is extremely rare with less than 98 cases reported worldwide, and the authors contend that ANKCL has an etiology related to infection with Epstein-Barr virus, not chemical exposure.

Animal evidence for thyroid follicular cell tumors was slight based on increases seen in significant pairwise increases of adenomas or carcinomas in female F344/N rats with a significant dose-response trend but only marginal increases in male rats and no increase in B6C3F1 mice (NTP, 1991b). Although U.S. EPA (1998a) notes that thyroid tumors in animal studies cannot be completely dismissed as a hazard for humans, it appears that that rodents are more sensitive than humans to thyroid follicular cell tumors induced by thyroid-pituitary disruption and thyroid stimulating hormone hyperstimulation (Dybing and Sanner, 1999; U.S. EPA, 1998a). There is also slight evidence in animals for harderian gland adenoma or carcinoma based on increased incidence in female B6C3F1 mice at the highest dose only, but no increased incidence in rats or male B6C3F1 mice (NTP, 1991b). Finally, slight evidence in animals exists for hepatocellular tumors based on a dose-related trend in tumor incidence in only in one sex of one species (male B6C3F1 mice) (NTP, 1991b).

The mechanistic evidence for carcinogenesis is slight. Available data indicates that TCEP has little of any genotoxic potential, but data are limited to assess in vivo genotoxicity. Limited additional data indicate that TCEP may influence cell signaling related to proliferation, apoptosis, and ion transport, induce oxidative stress, alter cellular energetics in kidney tissues and cells and in other cell types.

U.S. EPA's PPRTV (U.S. EPA, 2009) also concluded that TCEP is likely to be carcinogenic to humans based on information from oral animal bioassays that included clear evidence of renal tubule cell adenomas in F344/N rats in NTP (1991b), renal tubule adenomas and carcinomas in ddY mice in Takada et al. (1989) as well as the rarity of these tumors. The PPRTV also describes evidence for other tumors identified in these two bioassays as suggestive or equivocal.

The 2009 *European Union Risk Assessment Report* (ECB, 2009) concluded that TCEP has carcinogenicity potential and cites the EU classification category 3 and R40—limited evidence of carcinogenic effect. In contrast, the International Agency for Research on Cancer (IARC) designated TCEP as not classifiable as to its carcinogenicity to humans in 1990 and again in 1999 (IARC, 2019).

5.2.5 Dose-Response Assessment

According to U.S. EPA's 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), hazard endpoints that receive evidence integration judgments of *demonstrates* and *likely* would generally be considered for dose-response analysis. Endpoints with *suggestive* evidence can be considered on a case-by-case basis. Studies that received high or medium overall quality determinations (or low-quality studies if no other data are available) with adequate quantitative information and sufficient sensitivity can be compared.

There were no hazard outcome categories for which evidence *demonstrates* that TCEP causes the effect in humans. Therefore, hazard outcomes that received *likely* judgements are the most robust evidence integration decisions. The health effect with the most robust and sensitive POD among these *likely* outcomes was used for risk characterization for each exposure scenario to be protective of other adverse effects as described in the sections below.

Data for the dose-response assessment were selected from oral toxicity studies in animals. No acceptable toxicological data were available by the inhalation route, and no PBPK models are available to extrapolate between animal and human doses or between routes of exposure using TCEP-specific information.

The PODs estimated based on effects in animals were converted to HEDs or CSFs for the oral and dermal routes and HECs or IURs for the inhalation route. For this conversion, EPA used guidance from U.S. EPA (2011b) to allometrically scale oral data between animals and humans. Although the guidance is specific for the oral route, EPA used the same HEDs and CSFs for the dermal route of exposure as the oral route because the extrapolation from oral to dermal routes is done using the human oral doses, which do not need to be scaled across species. EPA accounts for dermal absorption in the dermal exposure estimates, which can then be directly compared to the dermal HEDs.

For the inhalation route, EPA extrapolated the daily oral HEDs and CSFs to HECs and IURs using human body weight and breathing rate relevant to a continuous exposure of an individual at rest. Based on existing data (Herr et al., 1991), absorption via the oral route may be greater than 95 percent. Therefore, EPA assumed that absorption for the oral routes is 100 percent; there is no information regarding absorption via the inhalation route, and therefore, EPA assumed 100 percent absorption via this route. Therefore, no adjustment specific to absorption is needed for the oral and inhalation routes. For consistency, all HEDs and the CSF are expressed as daily doses and all HECs are based on daily, continuous concentrations (24 hours per day) using a breathing rate for individuals at rest. Adjustments to exposure durations, exposure frequencies, and breathing rates are made in the exposure estimates used to calculate risks for individual exposure scenarios.

Appendix K.3 presents information on dose derivation, calculations for each of the PODs, and route-to-route extrapolations. Considerations regarding the BMD modeling process as well as modeling results for *likely* as well as *suggestive* TCEP outcomes are presented in the supplemental file *Benchmark Dose Modeling Results for TCEP* (U.S. EPA, 2024c). A comparison of the PODs for *likely* and *suggestive* health outcomes is presented visually in exposure response arrays within Appendix N, with calculations for these PODs in an Excel spreadsheet in the supplemental file *Human Health Hazard Points of Departure Comparison Tables* (U.S. EPA, 2024k).

5.2.5.1 Selection of Studies and Endpoints for Non-cancer Toxicity

EPA considered the suite of oral animal toxicity studies and *likely* individual adverse health effects outcomes when considering non-cancer PODs for estimating risks for acute and intermediate/chronic exposure scenarios, as described in Sections 5.2.5.1.1 and 5.2.5.1.2, respectively. Epidemiological studies were summarized for the weight of scientific evidence. EPA selected studies and relevant health effects based on the following considerations:

- Overall quality determinations;
- Exposure duration;
- Dose range;
- Relevance (*e.g.*, what species was the effect in, was the study directly assessing the effect, is the endpoint the best marker for the tox outcome?);
- Uncertainties not captured by the overall quality determination;
- Endpoint/POD sensitivity;
- Total UF; and
- Uncertainty and sensitivity of BMR selection from BMD modeling.

The following sections provide comparisons of the above attributes for studies and hazard outcomes for each of these exposure durations and details related to the studies considered for each exposure duration scenario.

5.2.5.1.1 Non-cancer Points of Departure for Acute Exposure

To calculate risks for the acute exposure duration in the risk evaluation, EPA used a daily HED of 9.46 mg/kg (NOAEL of 40 mg/kg) from a prenatal/postnatal neurodevelopmental toxicity study (Moser et al., 2015) based on very slight to moderate tremors within five days of dosing at 125 mg/kg-day in 13 dams. EPA gave this study a high overall quality determination, and a UF of 30 was used for the benchmark MOE during risk characterization.

Mice exhibited signs of neurotoxicity in other acute or short-term high-quality studies. In the NTP (1991b) 16-day study, mice exhibited ataxia and convulsive movements within three days at the two highest doses with a daily HED of 16.6 mg/kg; data were only qualitatively described. Pregnant mice administered 940 mg/kg-day TCEP via oral gavage were languid, prostrate, and exhibited jerking movements during GDs 7 through 14 with an HED of 125 mg/kg-day (Hazleton Laboratories, 1983). The HED from Moser et al. (2015) is more sensitive.

Tilson et al. (1990) found that in addition to convulsions, female Fischer 344 rats exhibited histopathological changes in the hippocampus and memory impairment in the Morris water maze after a single oral gavage administration of 275 mg/kg and an HED of 65.0 mg/kg. Although EPA gave Tilson et al. (1990) a high overall quality determination, the authors tested only a single dose level, which did not allow a full understanding of the dose-response for TCEP. The POD is associated with greater uncertainty because only a LOAEL was identified and a UF of 300 would be required for a benchmark MOE analysis.

The high-quality intraperitoneal injection study by Umezu et al. (1998) provides qualitative support for neurotoxicity; mice exhibited increased ambulatory activity at 100 and 200 mg/kg and 'light' convulsions at 200 mg/kg after single administration of these doses. EPA did not consider this study to be a candidate for the POD based on the exposure route.

Table 5-47 presents a comparison of the attributes of studies and hazard endpoints considered for the intermediate exposure scenario and Table 5-48 summarizes the study PODs and pertinent information, including HEDs and HECs. The bolded row represents the study and POD values used to calculate risks for acute scenarios in the risk evaluation.

Overall, the tremors observed in Moser et al. (2015) represent a sensitive endpoint that could occur in humans. The clinical signs of neurotoxicity (*e.g.*, convulsions) were consistently observed across acute/short-term studies.

Table 5-47. Comparison among Studies with Sensitive Neurotoxicity Endpoints Considered for Acute Exposure Scenarios

	Moser et al. (2015)	NTP (1991b)	Tilson et al. (1990)	Hazleton Laboratories (1983)
Overall Data Quality Determination	High	High	High	High
Exposure Duration	Within 5 days	Within 3 days	1 day	8 days
Dose Range	12, 40, 125 mg/kg-day (high dose changed to 90 mg/kg-day at 5 days)	0, 44, 88, 175, 350, 700 mg/kg-day	275 mg/kg	940 mg/kg-day
Relevance	Assumed to be relevant to humans; clearly adverse	Assumed to be relevant to humans (similar effect as chosen POD); clearly adverse	Assumed to be relevant to humans (similar effect as chosen POD); clearly adverse	Assumed to be relevant to humans (similar effect as chosen POD); clearly adverse
Uncertainties Not Captured Elsewhere	Effects observed only at the highest dose	BMD modeling not possible; only qualitative outcome information available	Precision of POD is limited because no NOAEL was identified	Precision of POD is limited because no NOAEL was identified
Sensitivity of POD for exposure scenario	Sensitive endpoint with an identified NOAEL	Less sensitive	Most sensitive when considering comparison with 300 benchmark MOE	Least sensitive
Total UF	30	30	300	300

Table 5-48. Dose-Response Analysis of Selected Studies Considered for Acute Exposure Scenarios

Target Organ/ System	Species	Duration	Study POD/ Type (mg/kg)[a]	Effect	HEC (mg/m^3) [ppm]	HED (mg/kg)	UFs	Reference	Overall Quality Determination
Neurotoxicity	**Long Evans rats (dams)**	**5 days**	**NOAEL = 40**	**Tremors**	**51.5 [4.41]**	**9.46**	**UF_A= 3 UF_H=10 *Total UF=30***	Moser et al. (2015)	**High**
Neurotoxicity	$B6C3F_1$ mice	16 days	NOAEL = 125	Convulsions, ataxia within 3 days	90.4 [7.75]	16.6	UF_A= 3 UF_H=10 *Total UF=30*	NTP (1991b)	High
Neurotoxicity	Fischer 344 rats (females)	1 day	LOAEL = 275	Convulsions brain lesions, behavior changes	354 [30.3]	65.0	UF_A= 3 UF_H=10 UF_L = 10 *Total UF=300*	Tilson et al. (1990)	High
Neurotoxicity	CD-1 mice (dams)	GD 7–14	LOAEL = 940	Jerking movements, languidity, prostration	680 [58.3]	125	UF_A= 3 UF_H=10 UF_L = 10 *Total UF=300*	Hazleton Laboratories (1983)	High

[a] The PODs are duration adjusted to 7 days per week; therefore, any PODs from studies that dosed for 5 days per week were multiplied by 5/7.

5.2.5.1.2 Non-cancer Points of Departure for Intermediate and Chronic Exposures

Figure 5-17 presents exposure response arrays of the HEDs for the *likely* hazard outcomes from the studies considered for the intermediate and chronic HEDs. The HEDs are presented within the hazard outcomes of reproductive toxicity (with developmental as comparison), kidney toxicity, and neurotoxicity and ordered from lowest to highest to view relative sensitivities more easily.

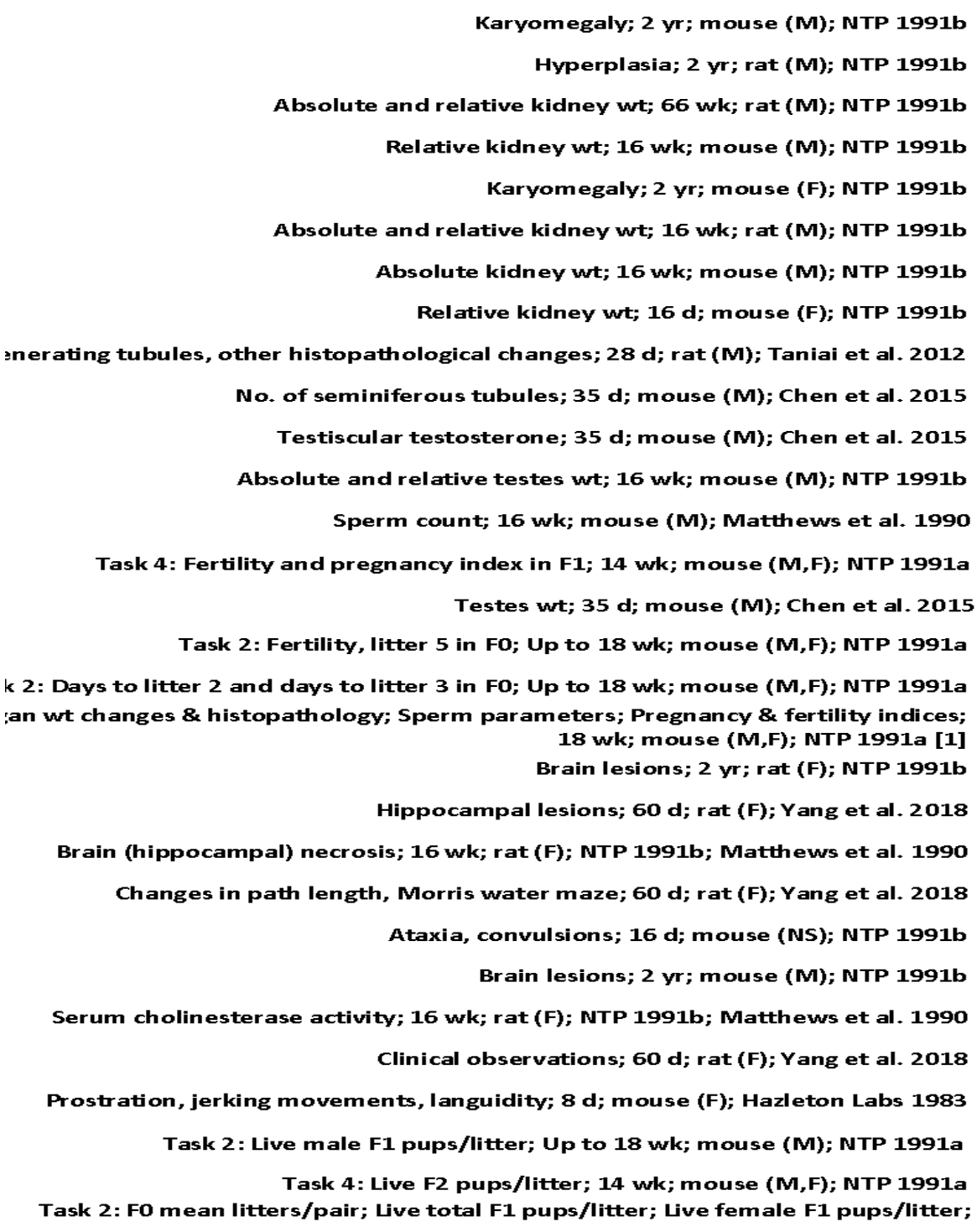

Figure 5-17. Exposure Response Array for Intermediate and Chronic Exposure Durations by Likely Hazard Outcomes (and Developmental Toxicity)

EPA is using Chen et al. (2015a), the 35-day study in adolescent mice, to estimate non-cancer risks for both the intermediate and chronic exposure scenarios. The study received a high overall quality determination, and the sensitive effect is a decrease in the numbers of seminiferous tubules (by 22 and 41 percent at 100 and 300 mg/kg-day, respectively) that is accompanied by absolute disintegration of tubules and decreased testosterone levels and testes weights at 300 mg/kg-day.

EPA conducted BMD modeling, and several continuous BMD models adequately fit the seminiferous tubule numbers, resulting in similar BMDL5s. The exponential 2 model fit resulted in the lowest Akaike information criterion (AIC) and a good fit upon visual inspection. U.S. EPA (2024c) presents additional details, including the fits for all seven continuous models that were run and BMDL values for BMRs of five percent RD and one SD.

For continuous data, EPA's BMD Technical Guidance recommends modeling the data using a BMR of one standard deviation (SD) (U.S. EPA, 2012b), but lower response rates should be used when effects are severe (*e.g.*, frank). Thus, EPA used a BMR of 5 percent based on biological severity and identified a BMDL5 of 21 mg/kg-day. The BMDLs for 1 SD and 10 percent were 61 and 43 mg/kg-day, respectively.

As stated in EPA's *Guidelines for Reproductive Toxicity Risk Assessment* (U.S. EPA, 1996), human males are particularly susceptible to chemicals that reduce numbers or quality of sperm. Chen et al. (2015a) did not directly evaluate sperm numbers or quality but due to potential for the endpoint to affect fertility, the magnitude of effects observed in the study, and the potential for human males to be more susceptible than rodents, EPA considers the significant effect on seminiferous tubules (which help produce, maintain, and store sperm) to be severe and thus, warrants use of a BMR of 5 percent.

BMRs of 5 percent were also used for other severe or frank effects in the TCEP risk evaluation, including decreased live pups per litter and brain necrosis. Furthermore, use of 5 percent for degeneration of seminiferous tubules has support from other authors, even at levels down to 5 percent degeneration of the tubules (Blessinger et al., 2020). When evaluating male phthalate syndrome, Blessinger et al. (2020) used a BMR of 5 percent for all endpoints associated with zero to moderate impacts on fertility. These endpoints included germ cell degeneration or depletion in seminiferous tubules ranging from 5 to 75 percent (Blessinger et al., 2020; Lanning et al., 2002).

EPA calculated a daily HED of 2.79 mg/kg-day for Chen et al. (2015a) that accounts for allometric scaling between mice and humans and is compared with a benchmark MOE of 30. HEDs for other reproductive effects ranged from 9.51 to 93.1 mg/kg-day. Many are within an order of magnitude of Chen et al. (2015a). The HEDs of 93.1 mg/kg-day are based on LOAELs that are 33 times greater (NTP, 1991a) and are used with a benchmark MOE of 300 instead of 30.

As noted in Section 5.2.3.1.2, hazard outcomes identified by Chen et al. (2015a) are supported by effects on sperm, reproductive organ weight changes, and testes hyperplasia (NTP, 1991a, b; Matthews et al., 1990). Other reproductive and developmental outcomes were observed, including decreases in fertility and live pups per litter in the continuous breeding toxicity study (NTP, 1991a).

There are uncertainties associated with using Chen et al. (2015a) for the POD. Other than minimal to mild hyperplasia, histopathological changes in the testes were not routinely identified in other studies (NTP, 1991a, b). However, Chen et al. (2015a) was conducted more than 20 years after the NTP studies and some methods differed from older studies (*e.g.*, preparation of tissues). Also, differences may reflect

use of different species or mouse strains, and in such cases, U.S. EPA (1996) recommends using the most sensitive species in the absence of information to suggest otherwise.

There are limitations of Chen et al. (2015a)'s study design and the BMD modeling analysis. Doses for this feeding study may be imprecise because information on body weight and food consumption were not reported. In addition, the sample size is small and as sample size decreases, uncertainty in the true response rate increases. Finally, although EPA considered BMD modeling as appropriate for this data set, in part because the lowest dose tested was a LOAEL, the BMR of 5 percent is lower than the biologically and statistically adverse responses observed in the study (22.2 and 40.7%). Overall, however, the significant and severe effect on seminiferous tubules is of concern for human males, a sensitive subpopulation.

Comparison of Studies Used for the Intermediate Exposure Scenario: In addition to Chen et al. (2015a), EPA considered sensitive effects from other studies ranging from a few days to 60 days for the intermediate POD that would be associated with a 30-day exposure scenario. Table 5-49 presents a comparison of the attributes of multiple studies and hazard endpoints considered for the intermediate exposure scenario. Table 5-50 provides details of the studies, including PODs from the study or from dose-response modeling, HECs, and HEDs. The bolded row represents the study and POD values used to calculate risks for intermediate and chronic scenarios in the risk evaluation.

HEDs for both Moser et al. (2015) and Yang et al. (2018a) are based on neurotoxicity, which are relevant hazard outcomes observed across multiple studies and are within an order of magnitude of the sensitive HED (2.79 mg/kg-day) from Chen et al. (2015a). In addition, they are oral gavage studies and thus, dose levels are expected to be more precise compared with Chen et al. (2015a), a dietary study. However, exposure durations (5 and 60 days) for these studies introduce some uncertainty regarding applicability to the target 30-day exposure scenario compared with Chen et al. (2015a), a 35-day study.

Even though the HED from Chen et al. (2015a) is based on using a BMR below the observed data, other intermediate study and endpoint candidates also have limitations related to dose-response relationships. Moser et al. (2015) observed effects only at the highest dose, and therefore, the HED is based on a NOAEL, not a BMDL that considers the full dose-response curve. Similarly, the lowest HED (11.8 mg/kg-day) from Yang et al. (2018a) is based on a NOAEL; a similar HED from Yang et al. (2018a) (13 mg/kg-day, based on a $BMDL_{20}$ of 55.0 mg/kg-day) also results in some uncertainty given typical variability in the modeled neurobehavioral endpoint.

Taniai et al. (2012a), a 28-day study resulting in kidney proximal tubule regeneration, has a relevant hazard outcome and an exposure duration closer to the intermediate scenario. However, even less is known about the dose-response relationship because the study used only a single dose level resulting in a LOAEL and a benchmark MOE of 300 rather than 30 used with Chen et al. (2015a).
Overall, using Chen et al. (2015a) for the intermediate exposure scenario in which adolescent male rats were evaluated during a potentially sensitive lifestage results in a sensitive POD for a relevant endpoint for the risk evaluation. EPA considers this POD to be protective of other adverse effects identified in TCEP toxicity studies.

Table 5-49. Comparison among Studies with Sensitive Endpoints Considered for Intermediate Exposure Scenarios

	Neurotoxicity (Moser et al., 2015)	Neurotoxicity (Yang et al., 2018a)	Reproductive Toxicity (Chen et al., 2015a)	Kidney Toxicity (Taniai et al., 2012a)
Overall Data Quality Determination	High	High	High	Medium
Exposure Duration	Within 5 days; less applicable to intermediate exposure	60 days; less applicable to intermediate exposure	35 days	28 days
Dose Range	12, 40, 125 mg/kg-day (high dose changed to 90 mg/kg-day at 5 days)	50, 100, 250 mg/kg-day	100, 300 mg/kg-day	350 mg/kg-day
Relevance	Endpoint assumed to be relevant to humans	Endpoint assumed to be relevant to humans	Endpoint assumed to be relevant to human male reproduction (U.S. EPA, 1996)	Endpoint assumed to be relevant to humans
Uncertainties Not Captured Elsewhere	Dose-response less precise: Use of NOAEL	Dose-response less precise: Use of NOAEL); Neurobehavioral outcomes (BMR of 20%) had a similar HED (13 mg/kg-day) but effect is typically variable	Dose precision unclear: dietary study and no information on food consumption or body weight	Lack of understanding of dose response and greater uncertainty due to use of single dose level resulting in a LOAEL
Sensitivity of Endpoint and POD	Within an order of magnitude of the most sensitive endpoint	Within an order of magnitude of the most sensitive endpoint	Most sensitive endpoint for the intermediate scenario	Less sensitive endpoint but is used with a larger benchmark MOE
Total UF/ Benchmark MOE	30	30	30	300
Uncertainty/ Sensitivity of BMR Selection	N/A	N/A	BMR of 5% is lower than responses in study	N/A

Table 5-50. Dose-Response Analysis of Selected Studies Considered for Intermediate Exposure Scenarios

Target Organ/ System	Species	Duration	Study POD/ Type (mg/kg-day)	Effect	HEC (mg/m^3) [ppm]	HED (mg/kg-day)	UFs	Reference	Overall Quality Determination
Reproductive Toxicity	**ICR mice (males)**	**35 days**	**$BMDL_5$ = 21[a]**	**Decreased numbers of seminiferous tubules**	**14.9 [1.27]**	**2.73**	**UF_A= 3 UF_H=10 *Total UF=30***	Chen et al. (2015a)	**High**
Neurotoxicity	Sprague-Dawley rats (females)	60 days	NOAEL = 50	Hippocampal lesions	64.3 [5.51]	11.8	UF_A= 3 UF_H=10 *Total UF=30*	Yang et al. (2018a)	High
Kidney Toxicity	F344 rats (males)	28 days	LOAEL = 350	Regenerating tubules in kidneys	450 [38.6]	82.8	UF_A= 3 UF_H=10 UF_L=10 *Total UF=300*	Taniai et al. (2012a)	Medium

[a] The BMDL based on 1SD is 61.2 mg/kg-day.

Comparison of Studies and Hazard Outcomes for the Chronic Exposure Scenario: EPA generally considers chronic studies to be those with exposure durations of ≥ 10 percent of a lifetime. For TCEP, these studies include the 16- and 18-week and 2-year NTP studies in rats and mice (NTP, 1991b). Also, many of the endpoints in the RACB study (NTP, 1991a) (especially the crossbreeding and second-generation effects) were measured after chronic exposure. Table 5-51 presents a comparison of the attributes of sensitive endpoints from studies considered for the chronic exposure scenario, and Table 5-52 provides study details including PODs from the study or BMD modeling results, HECs, and HEDs.

Although it is a study with a shorter exposure duration, EPA chose Chen et al. (2015a) for the chronic exposure scenarios because it resulted in an HED that is more sensitive (2.79 mg/kg-day) than most longer-term results and covers a potentially sensitive lifestage (adolescence).

Use of the shorter duration study by Chen et al. (2015a), however, does lend uncertainty to the risk evaluation because other longer-term studies are not as sensitive and because it is uncertain whether the POD would be lower if Chen et al. (2015a) extended the exposure duration.

For the endpoints that resulted in *likely* evidence integration conclusions, most chronic studies received high overall quality determinations. There were a few exceptions. EPA gave medium overall quality determinations to the sperm morphology and vaginal cytology results reported in the 16- and 18-week NTP studies (Matthews et al., 1990), primarily based on limited information regarding methods and results. Clinical observations described by NTP (1991b) for the 16- and 18-week studies in mice and rats received uninformative overall quality determinations due to the lack of reasonably available quantitative information for these effects.

The single chronic endpoint more sensitive than Chen et al. (2015a) was increased relative kidney weights for female rats from the 16-week NTP study, with an HED of 1.75 mg/kg-day (NTP, 1991b). However, EPA considered the changes in kidney weights for TCEP less relevant for predicting kidney toxicity than other endpoints (*i.e.*, kidney histopathology) because they were not consistently observed; female rats had increased relative kidney weights after 16 weeks but not after 66 weeks, and female mice had increased weights at 16 days but not at 16 weeks or the 66-week sacrifice. In addition, kidney weight changes did not correspond to histopathology changes (NTP, 1991b).

Histopathology is a more reliable endpoint for kidney effects and was observed in the 2-year studies (NTP, 1991b); daily HEDs associated with hyperplasia and karyomegaly ranged from 5.49 to 14.2 mg/kg-day; most are within a factor of three of Chen et al. (2015a) and 14.2 mg/kg-day is roughly five times higher.

Neurotoxicity was consistently observed across chronic studies with HEDs ranging from 7.43 to 22.8 mg/kg-day. These HEDs are all within an order of magnitude of Chen et al. (2015a).

The comparison of HEDs with reproductive endpoints described earlier and the comparisons with kidney and neurotoxicity endpoints observed in the chronic studies demonstrates some consistency across endpoints with respect to potency. These co-critical endpoints lend strength to using the sensitive endpoint from Chen et al. (2015a) for the chronic duration.

Similar to Chen et al. (2015a), only two dose groups (44 and 88 mg/kg-day) were used in the NTP (1991b) 2-year studies associated with the most sensitive of the kidney and neurotoxic effects, which somewhat limits the understanding of the dose response relationship for these endpoints.

Overall, the HED from Chen et al. (2015a) associated with a relevant hazard outcome is protective of other observed adverse effects from chronic exposure to TCEP that include neurotoxicity and kidney histopathological effects.

Table 5-51. Comparison among Studies with Sensitive Endpoints Considered for Chronic Exposure Scenarios

	Neurotoxicity (NTP, 1991b)	Reproductive Toxicity (Chen et al., 2015a)	Kidney (NTP, 1991b)
Overall Data Quality Determination	High	High	High
Exposure Duration	2-year; chronic	35-day; intermediate (< chronic)	2-year; chronic
Dose Range	44, 88 mg/kg-day	100, 300 mg/kg-day	44, 88 mg/kg-day
Relevance	Endpoint assumed to be relevant to humans	Endpoint assumed relevance to human male reproduction (U.S. EPA, 1996); severity identified	Endpoint assumed to be relevant to humans
Uncertainties Not Captured Elsewhere	Dose-response less precise (use of NOAEL)	Dose precision unclear based on dietary study with no information on food consumption or body weight changes	Some inconsistencies between kidney weight changes and histopathology
Sensitivity of Endpoint and POD	Most sensitive among chronic neurotoxic effects	Most sensitive across hazard outcomes (except increased kidney weight in 16-week study)	Most sensitive among chronic histopathological kidney effects; 16-week kidney weight change more sensitive
Total UF	30	30	30
Uncertainty/Sensitivity of BMR Selection	N/A	BMR of 5 percent, predicted BMD and BMDL values are lower than doses associated with responses observed in the study	BMR of 10 percent

Table 5-52. Dose-Response Analysis of Selected Studies Considered for Chronic Exposure Scenarios

Target Organ System	Species/Sex Exposed	Duration	Study POD/Type (mg/kg-day)	Effect	HEC (mg/m^3) [ppm]	HED (mg/ kg-day)	UFs	Reference	Overall Quality Determination
Reproductive Toxicity	**ICR mice (male)**	**35 days**	**$BMDL_5$ = 21[a]**	**Decreased numbers of seminiferous tubules**	**14.9 [1.27]**	**2.73**	**UF_A= 3 UF_H=10 *Total UF=30***	Johnson et al. (2003) Chen et al. (2015a)	**High**
Neurotoxicity	F344 rats (female)	2 years	NOAEL = 31.4	Brain lesions	40.4 [3.46]	7.43	UF_A= 3 UF_H=10 *Total UF=30*	NTP (1991b)	High
Kidney Toxicity	F344 rats (female)	2 years	$BMDL_{10}$ = 23.2	Renal tubule hyperplasia	30 [2.6]	5.49	UF_A= 3 UF_H=10 *Total UF=30*	NTP (1991b)	High

[a] The BMDL based on 1SD is 61.2 mg/kg-day.

5.2.5.1.3 Uncertainty Factors Used for Non-cancer Endpoints

For the non-cancer health effects, EPA used a total uncertainty factor (UF) of 30 for the benchmark MOEs for acute, intermediate, and chronic exposure durations for all exposure routes among studies that are used to estimate risks. Other endpoints that used LOAELs for which EPA used a LOEAL-to-NOAEL UF of 10 and a total benchmark MOE of 300.

1. **Interspecies Uncertainty Factor (UF_A) of 3**

   EPA uses data from oral toxicity studies in animals to derive relevant HEDs, and U.S. EPA (2011a) recommends allometric scaling (using the ¾ power of body weight) to account for interspecies toxicokinetics differences for oral data. When applying allometric scaling, EPA guidance recommends reducing the UF_A from 10 to 3. The remaining uncertainty is associated with interspecies differences in toxicodynamics. EPA also uses a UF_A of 3 for the inhalation HEC and dermal HED values because these values are derived from the oral HED.

2. **Intraspecies Uncertainty Factor (UF_H) of 10**

   EPA uses a default UF_H of 10 to account for variation in sensitivity within human populations due to limited reasonably available information regarding the degree to which human variability may impact the disposition of or response to, TCEP.

3. **LOAEL-to-NOAEL Uncertainty Factor (UF_L) of 1 or 10**

   The PODs chosen to calculate risks were either NOAELs or BMDL values and therefore, EPA used a UF_L of 1. EPA compared these values with other endpoints based on LOAELs, which used a UF_L of 10 to account for the uncertainty inherent in extrapolating from the LOAEL to the NOAEL.

U.S. EPA (1993a) and U.S. EPA (2002b) further discuss use of UFs in human health hazard dose-response assessment.

5.2.5.2 Selection of Studies and Endpoint Derivation for Carcinogenic Dose-Response Assessment

EPA considered the kidney tumors for derivation of toxicity values for the risk calculations based on the evidence integration conclusion that the tumors are sensitive and robust, and that cancer is *likely* to be caused by TCEP. The selection of representative cancer studies and tumors for dose-response analysis is described below based on the following considerations:

- Overall quality determination;
- Sufficiency of dose-response information;
- Strength of the evidence supporting the associated tumor type;
- MOA conclusions;
- Relevance (*e.g.*, what species was the effect in, was the study directly assessing the effect, is the endpoint the best marker for the tox outcome?);
- Uncertainties not captured by the overall quality determination; and
- Endpoint sensitivity.

Rodent bioassays identify increased incidences of kidney tumors in male F344/N rats, with a lower increase in female rats (NTP, 1991b). Treatment-related kidney tumors were also observed after two years in male $B6C3F_1$ mice (NTP, 1991b). EPA gave NTP (1991b) a high overall quality determination.

Based on a lack of adequate information on mechanisms or temporality and lack of dose-response data for precursor lesions to consider an alternate dose-response using a threshold analysis, EPA used linear

low-dose extrapolation to estimate risks. U.S. EPA's PPRTV also used linear low-dose extrapolation in the absence of specific mechanistic information.

EPA used the multistage models available in the BMD software and adjusted the data for mortality by using animals still alive on the first day of cancer incidence. Therefore, animals dying from other causes were not included in the analysis. For both male and female rats, kidney tumor incidence data adequately fit one or both multistage models and tumors in males (adenomas and carcinomas) resulted in the more sensitive CSF (0.0058 per mg/kg-day). The IUR is based on daily, continuous concentrations (24 hours per day) using a breathing rate for individuals at rest. Adjustments to exposure durations, exposure frequencies, and breathing rates are made in the exposure estimates used to calculate risks for individual exposure scenarios.

Table 5-53 presents the cancer PODs for modeled renal tumors. Because EPA has not concluded that TCEP acts via a mutagenic mode of action, an age-dependent adjustment factor (ADAF) (U.S. EPA, 2005c) was not applied when estimating cancer risk for kidney tumors from TCEP exposure. EPA did not use CSFs for combined tumors (across multiple target organs) for the risk evaluation but focused on the tumors with the most robust evidence from the animal data.

See Appendix K.3 for dose-response derivation, including details on route-to-route extrapolation. Considerations regarding the BMD modeling process for cancer and results are presented in *Benchmark Dose Modeling Results for TCEP* (U.S. EPA, 2024c).

EPA did not use CSFs for combined tumors (across multiple target organs) for the risk evaluation but focused on the tumors with the most robust evidence from the animal data.

Table 5-53. Dose-Response Analysis of Kidney Tumors[a] for Lifetime Exposure Scenarios

Tumors	Species (Sex)	Oral/Dermal CSF[a b]	IUR[a]	Extra Cancer Risk Benchmark
Renal tubule adenomas or carcinomas	F344 rats (male)	0.0245 per mg/kg-day	0.00451 per mg/m^3 (0.0526 per ppm)	1E–04 (occupational) 1E–04 to 1E–06 (consumer, general population)
Renal tubule adenomas	F344 rats (female)	0.0220 per mg/kg-day	0.00404 per mg/m^3 (0.0472 per ppm)	

[a] CSFs and IURs were derived based on continuous exposure scenarios; CSFs from BMD modeling prior to allometric scaling were 0.0058 and 0.0052 per mg/kg-day for male and female rats, respectively.
[b] U.S. EPA's PPRTV (U.S. EPA, 2009) calculated an oral CSF of 0.02 per mg/kg-day, also based on increased renal tubule adenomas or carcinomas in male rats from NTP (1991b).

5.2.6 Weight of Scientific Evidence Conclusions for Human Health Hazard

EPA used information described in previous sections and additional factors to arrive at confidence levels for key human health hazard outcome and exposure duration combinations. Evidence integration was conducted by considering Bradford Hill criteria, as described in Section 5.2.1, with conclusions for TCEP described in Sections 5.2.3 and 5.2.4.4. Only *likely* evidence integration conclusions were considered for dose-response and given weight of evidence confidence levels. As outlined in Section 5.2.5, factors in addition to the Bradford Hill criteria were considered when choosing studies for dose-response modeling and for each exposure scenario (acute, intermediate, and chronic).

The current section combines the evidence integration conclusions, factors already considered during dose-response as well as additional considerations (see Section 5.2.6.1) to choose the overall hazard confidence levels:

- Evidence integration conclusion (from Appendix L)
  - *Demonstrates* is rated as +++
  - *Likely* is rated as ++
  - *Suggests* is rated as +
- Selection of key endpoint and study
- Relevance to exposure scenario
- Dose-response considerations
- PESS sensitivity

Section 5.2.6.2 presents a summary table of confidence for each hazard endpoint and exposure duration.

5.2.6.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the Hazard Identification and Selection of PODs for Human Health Hazard Assessment

5.2.6.1.1 Acute Non-cancer

Evidence Integration Conclusions

Clinical signs of neurotoxicity, histopathological changes in the brain, and neurobehavioral changes measured in multiple studies were considered for the acute exposure scenario. EPA concluded that TCEP likely causes neurotoxicity in humans under relevant exposure circumstances and assigned high overall quality determinations to all acute studies considered.

Selection of the Key Endpoint and Study

Several human epidemiological studies evaluated neurotoxicity. There appears to be possible support for TCEP's association of prenatal exposure and impairment in cognitive abilities based on effects on IQ in lower SES children (Percy et al., 2022). The tremors observed in Moser et al. (2015) and similar neurotoxic effects in other studies are key because they are adverse, and neurotoxicity is consistently observed among acute and longer-term studies.

Offspring do not appear to be more sensitive for developmental neurotoxicity up to 90 mg/kg-day[39] after exposure of pregnant rats during gestation and the early postnatal period based on results from Moser et al. (2015). Viability and growth of offspring were also not affected after pregnant mice were dosed with 940 mg/kg-day (Hazleton Laboratories, 1983).[40]

Relevance to Exposure Scenario

The candidate studies and endpoints for acute exposure identified neurotoxicity after one to eight days, and EPA considered these durations relevant for the acute exposure scenario. The study, Moser et al. (2015), chose to calculate risks and identified tremors within five days of exposure. There is some uncertainty for this human exposure scenario given the lack of TCEP-specific information or models (*e.g.,* PBPK models) to extrapolate from animals to humans. EPA also extrapolated from oral HEDs to inhalation HECs and dermal HEDs, which lends uncertainty for these routes. It is not known whether

[39] The study began with a dose of 125 mg/kg-day, which was lower to 90 mg/kg-day after 5 days due to toxicity in dams at the highest dose.

[40] A prenatal study in Wistar rats (Kawashima et al., 1983) in a foreign language will be translated it into English and evaluated for the risk evaluation.

these assumptions for the chosen POD would lead to over- or underprediction of risk from acute exposure.

Dose-Response Considerations
None of the studies considered for acute exposure could be modeled using BMD models due to limited reasonably available dose-response information. EPA identified a NOAEL from Moser et al. (2015), and effects were seen only at the highest dose. The other acute studies also identified only a NOAEL or LOAEL with effects observed only at the highest dose or the only dose in the study.

Susceptible Subpopulations
Moser et al. (2015) evaluated effects in pregnant female rats. Given the lower HED for this study compared with other acute studies, pregnant dams may be a susceptible subpopulation. However, uncertainties exist because of limited dose response information for other studies. Non-pregnant female rats are also shown to be a sensitive species and sex for neurotoxicity in longer-term studies as identified in NTP (1991b). Offspring, as noted earlier, were not identified as more sensitive to neurotoxicity or other effects from gestational and postnatal exposure of the dams.

5.2.6.1.2 Intermediate and Chronic Non-cancer

Evidence Integration Conclusions
EPA considered multiple animal toxicity studies and multiple hazard outcomes – reproductive toxicity, neurotoxicity, and kidney toxicity – for the intermediate and chronic exposure scenarios. Of the animal toxicity studies/endpoints, EPA assigned high quality determinations except Taniai et al. (2012a), which EPA gave a medium quality determination.

Selection of the Key Endpoint and Study
The nature of the effect chosen for calculating risks—differences in numbers and degeneration of seminiferous tubules identified by Chen et al. (2015a)—is considered adverse, and the fertility of human males is known to be sensitive to changes in sperm numbers and quality (U.S. EPA, 1996). Neurotoxicity and kidney toxicity were also observed consistently among studies and HEDs were often within an order of magnitude of each other.

The effects of Chen et al. (2015a) were the most sensitive after intermediate exposure. Increased relative kidney weight was most sensitive after chronic exposure, but EPA considered these weight changes less predictive of kidney toxicity due to inconsistencies between intermediate and longer-term studies and lack of correlation with histopathology and clinical chemistry results in many cases.

Using Chen et al. (2015a) does lead to uncertainty because other studies did not report decreased numbers or disintegration of seminiferous tubules; furthermore, related male reproductive effects were only seen at higher doses in other studies. However, male reproduction was consistently affected in several studies along with fertility and offspring viability. Thus, EPA considers the sensitive effects in Chen et al. (2015a) to be relevant and differences might be due to species, test methods, or lifestage.

There are several considerations that lend uncertainty as to whether risks could be underpredicted using this POD. These include lack of human data specific for male reproductive effects; the known sensitivity of human males to reproductive insults; and uncertainty about certain sensitive effects that could not be considered for a POD due to an error in the results presented in the continuous breeding study (NTP, 1991a) or lack of full reports (see Section 5.2.3.1.2).

There is some uncertainty as to whether this POD is protective of a full range of effects. For example, chronic studies did not evaluate neurobehavioral batteries. In addition, EPA did not locate any studies that investigated TCEP's association with acoustic startle responses or social behaviors.

Relevance to Exposure Scenarios

The 35-day exposure used by Chen et al. (2015a) is more relevant than the shorter and longer studies of 5 or 60 days (*e.g.*, Moser et al. (2015) and Yang et al. (2018a)) for the intermediate exposure scenario, which EPA defines as a 30-day exposure for this risk evaluation. Although the 28-day Taniai et al. (2012a) study is well-suited for intermediate exposures, other study aspects limit its suitability, including testing at only 350 mg/kg-day.

There is inherent uncertainty in assuming that a 35-day toxicity study in rodents during male adolescence is applicable to a similar exposure duration in human adolescent males for the endpoint of decreased numbers of seminiferous tubules.

Using Chen et al. (2015a) to represent chronic exposure durations adds uncertainty to the risk evaluation. If the specific effect identified by Chen et al. (2015a) were measured in a chronic study in the same species starting in adolescence, the POD could be more sensitive. Therefore, it is possible that risks might be under-predicted. Yet, the available chronic HEDs were less sensitive than Chen et al. (2015a).

For all studies and endpoints, no TCEP-specific information was available for extrapolation to humans and EPA relied on allometric scaling based on $BW^{3/4}$. Route-to-route extrapolation to inhalation HECs and dermal HEDs results in additional uncertainty. EPA cannot predict whether the assumptions regarding route extrapolation for the chosen POD would lead to over- or underprediction of risk from intermediate exposure for the dermal route.[41]

Dose-Response Considerations

Chen et al. (2015a) fed TCEP to rats in a dietary study and did not report information on food consumption. Thus, EPA does not know the precise doses received by the rats. However, the data adequately fit several BMD models based on statistics and visual inspection and resulted in similar BMDLs among the fit models. Also, use of the BMDL allowed EPA to use a relatively low total UF of 30. Given the severity of the effect (large percent decrease in numbers of tubules and significant degeneration), EPA chose a BMR of 5 percent.

Although other intermediate studies with relevant sensitive effects used three treatment levels (vs. two for Chen et al. (2015a)), EPA identified limitations for these other studies that included the inability to conduct BMD modeling, use of only one dose (with LOAEL only) or an effect seen only at the highest dose. Sensitive chronic neurotoxic and kidney effects are from studies with two treatment levels; neurotoxicity could not be modeled (and only a NOAEL is available) but kidney hyperplasia could be modeled and yielded an appropriate BMDL.

[41] Data from Shepel'skaia and Dyshginevich (1981) (as cited in (NTP, 1991a)) suggests that testes effects occurred by inhalation. If they are adverse and occurred at either low or high dose (0.5 or 1.5 mg/m^3), the effect is more sensitive than the extrapolated HEC value from Chen et al. (2015a). Shepel'skaia and Dyshginevich (1981) was not readily available to EPA and appears to be only an abstract. Thus, EPA cannot consider Shepel'skaia and Dyshginevich (1981) for use in this risk evaluation.

Susceptible Subpopulations
Chen et al. (2015a) evaluated a sensitive sex lifestage (male adolescent mice) and identified a sensitive POD among critical endpoints. Other studies and endpoints considered for intermediate and chronic exposure identified sexes that might be more sensitive to certain effects. For example, female rats were more sensitive for neurotoxicity.

5.2.6.1.3 Cancer

Evidence Integration Conclusions
EPA concludes that TCEP is likely to be carcinogenic to humans using guidance from U.S. EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b) based on information from a high-quality study (NTP, 1991b).

Selection of the Key Endpoint and Study
Of the organs that exhibited tumors in NTP (1991b), EPA used the tumor type with the most robust evidence – kidney adenomas and carcinomas – and used a CSF that was the most sensitive among modeled kidney tumor incidence.

EPA considers increased incidence of renal tubule adenomas and carcinomas to be adverse, relevant to humans, and representative of a continuum of benign to malignant tumors and was the only target organ with robust evidence of increased tumors. There is possible support for TCEP's association with thyroid tumors in humans based on a case control study (Hoffman et al., 2017), but the evidence is mixed based on a lack of association in a second study (Liu et al., 2022).

Of the kidney tumors, NTP (1991b) identified primarily adenomas and only one carcinoma. Thus, the risk of malignant tumors is less certain; if humans are like rodents, use of the CSF from NTP (1991b) could result in an over prediction of malignant cancer. However, if humans are more sensitive and develop malignancies sooner, risks may be underpredicted.

Relevance to Exposure Scenarios
NTP (1991b) is a 2-year bioassay and is relevant for chronic exposures in humans. However, like non-cancer endpoints, use of allometric scaling among species and route-to-route extrapolation to inhalation HECs and dermal HEDs leads to some uncertainties and the impacts on risks are unknown.

Dose-Response Considerations
There is no complete understanding regarding mechanism(s) of cancer and there is also a lack of appropriate precursors to cancer in the available *in vivo* studies with respect to temporality and dose response (*e.g.*, the single dose used by Taniai et al. (2012a) is higher than doses associated with tumors). Therefore, EPA used linear low dose extrapolation a $BMDL_{10}$. Because direct mutagenicity is not likely to be the predominant MOA, using linear low dose extrapolation may be a health conservative analysis.

Use of tumor data for only one target organ (*i.e.*, not combining incidence with other target organ tumors) may result in some underestimation of risk, however. Therefore, the net effect of the dose-response modeling, considering the benchmark risk levels used in the risk evaluation (1 in 10,000 to 1 in 1,000,000) is not known.

Susceptible Subpopulations

The single human study identified regarding TCEP exposure and thyroid cancer did not identify a specific susceptible subpopulation (Hoffman et al., 2017). Availability of a high-quality animal study using two species and both sexes suggests possible sensitivities by sex (*e.g.*, higher incidence of kidney tumors in male rats).

The dose-response model applied to animal tumor data employed low-dose linear extrapolation, and this assumes *any* TCEP exposure is associated with some positive risk of getting cancer. However, EPA did not identify specific human groups that are expected to be more susceptible to cancer following TCEP exposure even though there is likely to be variability in susceptibility across the human population. Other than relying on animal tumor data for the more sensitive sex, the available evidence does not allow EPA to evaluate or quantify the potential for increased cancer risk in specific subpopulations. Given that a mutagenic mode of action is unlikely, EPA does not anticipate greater cancer risks from early life exposure to TCEP.

5.2.6.2 Human Health Hazard Confidence Summary

Table 5-54 summarizes the confidence ratings for each factor for key human health hazards considered for acute, intermediate, chronic, and lifetime exposure scenarios. The bolded rows are the health endpoints for each exposure scenario used to calculate risks. Alternate PODs for health outcomes are not bolded in the table.

Table 5-54. Confidence Summary for Human Health Hazard Assessment

Hazard Domain	Evidence Integration Conclusion	Selection of Most Critical Endpoint and Study	Relevance to Exposure Scenario	Dose-Response Considerations	PESS Sensitivity	Overall Hazard Confidence
Acute non-cancer						
Neurotoxicity	+ +	+ + +	+ +	+ +	+ +	**Moderate**
Intermediate non-cancer						
Reproductive	+ +	+ +	+ + +	+	+ +	**Moderate**
Neurotoxicity	+ +	+	+ +	+ +	+ +	Moderate
Kidney	+ +	+	+ + +	+	+	Moderate
Chronic non-cancer						
Reproductive	+ +	+ +	+	+	+ +	**Moderate**
Neurotoxicity	+ +	+	+ + +	+ +	+ +	Moderate
Kidney	+ +	+	+ + +	+ +	+	Moderate
Cancer						
Kidney Cancer	+ +	+ +	+ + +	+ +	+ +	**Moderate**

+ + + Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the hazard estimate.
+ + Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize hazard estimates.
+ Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.

5.2.7 Toxicity Values Used to Estimate Risks from TCEP Exposure

After considering hazard identification and evidence integration, dose-response evaluation, and weight of scientific evidence of POD candidates, EPA chose two non-cancer endpoints for the risk evaluation—

one for acute exposure scenarios and a second one for intermediate and chronic scenarios (Table 5-55). Cancer risks were estimated using increased kidney tumors in male rats (Table 5-56). HECs and IURs are based on daily continuous (24-hour) exposure and HEDs and CSFs are daily values. All studies received high overall quality determinations.

Table 5-55. Non-cancer HECs and HEDs Used to Estimate Risks

Exposure Scenario	Target Organ System	Species (Sex)	Duration	POD (mg/kg-day)	Effect	HEC (mg/m^3) [ppm]	HED (mg/ kg-day)	Benchmark MOE	Reference(s)
Acute	Neurotoxicity	Long Evans rats (dams)	5 days	NOAEL = 40	Tremors	51.5 [4.41]	9.46	UF_A = 3 UF_H = 10 *Total UF = 30*	Moser et al. (2015)
Intermediate and Chronic	Reproductive Toxicity	ICR mice (male)	35 days	$BMDL_5$ = 21	Decreased seminiferous tubules	14.9 [1.27]	2.73	UF_A= 3 UF_H=10 *Total UF = 30*	Johnson et al. (2003) Chen et al. (2015a)

Table 5-56. Cancer IUR and CSF Used to Estimate Risks

Exposure Scenario	Target Organ System	Species (Sex)	Duration	POD (mg/kg-day)	Effect	IUR (per mg/m^3) [per ppm]	CSF (per mg/ kg-day)	Benchmark Risk Levels	Reference
Chronic/ Lifetime	Kidney tumors	Fischer 344/N rats (male)	2 years	CSF from BMD model = 0.0058 per mg/kg-day	Increased renal tubule adenomas or carcinomas	0.00451 [0.0526]	0.0245	1E−04 (occupational) 1E−04 to 1E−06 (consumer, general population)	NTP (1991b)

Table 5-57. General Population Acute Drinking Water (Oral Ingestion) Non-cancer Risk Summary

COU		OES	Acute Oral Non-cancer MOEs *UFs = 30*											
			Drinking Water						Drinking Water (Diluted)					
Life Cycle Stage/Category	**Subcategory**		**Adult (≥21 yr)**	**Infant (<1 yr)**	**Youth (16–20 yr)**	**Youth (11–15 yr)**	**Child (6–10 yr)**	**Toddler (1–5 yr)**	**Adult (≥21 yr)**	**Infant (<1 yr)**	**Youth (16–20 yr)**	**Youth (11–15 yr)**	**Child (6–10 yr)**	**Toddler (1–5 yr)**
Manufacturing/ Import	Import	Repackaging	172	49	224	223	175	138	2.12E05	6.05E04	2.76E05	2.76E05	2.16E05	1.70E05
Processing/ Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	40	11	52	52	40	32	6.38E04	1.82E04	8.30E04	8.28E04	6.49E04	5.11E04
		Incorporation into paints and coatings – 2-part reactive coatings	44	13	57	57	45	35	7.03E04	2.00E04	9.15E04	9.13E04	7.15E04	5.64E04
	Polymers used in aerospace equipment and products	Formulation of TCEP containing 2-part reactive resin	38	11	49	49	38	30	1.63E04	4.64E03	2.12E04	2.11E04	1.66E04	1.30E04
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	4,292	1,223	5,586	5,571	4,366	3,440	5.30E06	1.51E06	6.89E06	6.87E06	5.39E06	4.24E06
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	73	21	95	95	74	59	9.02E04	2.57E04	1.17E05	1.17E05	9.17E04	7.23E04

Table 5-58. Acute Fish Ingestion Non-cancer Risk Summary Based on 50th Percentile Flow of Harmonic Mean

COU		OES	Acute Oral Non-cancer MOEs *UFs = 30*							
			General Population		Subsistence Fishers		Tribes (Current IR)[a]		Tribes (Heritage IR)[b]	
Life Cycle Stage/ Category	Subcategory		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
Manufacturing/ Import	Import	Repackaging	18	363	3	57	2	37	<1	5
Processing/ Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	4	82	1	13	<1	8	<1	1
		Incorporation into paints and coatings – 2-part reactive coatings	4	90	1	14	<1	9	<1	1
	Polymers used in aerospace equipment and products	Formulation of TCEP-containing reactive resin	3	65	<1	10	<1	7	<1	1
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	450	9,067	70	1,411	46	930	6	122
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	8	154	1	24	1	16	<1	2

[a] Current fish consumption rate at 216 g/day based on survey of Suquamish Indian Tribe in Washington (see Section 5.1.3.4.4)
[b] Heritage fish consumption rate at 1,646 g/day based on study of Kootenai Tribe in Idaho (see Section 5.1.3.4.4)

Table 5-59. General Population Chronic Water and Soil Ingestion Non-cancer Risk Summary

COU		OES	Chronic Non-cancer Oral MOEs *UFs = 30*							
Life Cycle Stage/Category	**Subcategory**		**Drinking Water (Diluted)**	**Drinking Water**	**Drinking Water (via Leaching to Groundwater)**	**Ambient Water (Incidental Ingestion)**	**Soil Intake (50th) at 100 m**	**Soil Intake (95th) at 100 m**	**Soil Intake (50th) at 1,000 m**	**Soil Intake (95th) at 1,000 m**
Manufacturing/ Import	Import	Repackaging	1.64E08	1.05E05	N/A	2.11E05	2.20E10	5.15E09	1.73E12	4.03E11
Processing/ Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	4.40E07	23,728	2.12E06	4.89E04	7.02E08	1.64E08	7.95E10	1.86E10
		Incorporation into paints and coatings – 2-part reactive coatings	4.85E07	26,171	N/A	5.39E04	4.85E09	1.13E09	3.68E11	8.59E10
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	9.89E06	18,706	N/A	4.62E04	4.41E09	1.03E09	3.46E11	8.07E10
Processing/ Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	N/A	N/A	2.12E06	N/A	5.15E08	1.20E08	5.05E10	1.18E10
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	4.10E09	2.60E06	N/A	5.30E06	4.60E08	1.07E08	4.20E10	9.81E09
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	6.96E07	4.47E04	N/A	8.98E04	2.98E05	6.96E04	5.72E07	1.34E07

Table 5-60. Chronic Fish Ingestion Non-cancer Risk Summary

COU		OES	General Population				Subsistence Fishers[b]		Tribes (Current)[c]		Tribes (Heritage)[d]	
Life Cycle Stage/Category	Subcategory		BAF 2,198[a]		BAF 109[a]		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
			CT[e]	HE	CT[e]	HE						
Manufacturing/ Import	Import	Repackaging	23	5	461	105	1	16	1	11	<1	1
Processing/ Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	5	1	104	24	<1	4	<1	2	<1	<1
		Incorporation into paints and coatings – 2-part reactive coatings	6	1	115	26	<1	4	<1	3	<1	<1
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	4	1	82	19	<1	3	<1	2	<1	<1
Processing/ Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	571	130	11,505	2,617	20	407	13	268	2	35
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	10	2	196	45	<1	7	<1	5	<1	1

[a] General population exposure estimates based on general population fish ingestion rate of 22.2 g/day.
[b] Subsistence fishers exposure estimates based on subsistence fisher ingestion rate of 142.2 g/day.
[c] Current fish consumption rate at 216 g/day based on survey of Suquamish Indian Tribe in Washington (see Section 5.1.3.4.4).
[d] Heritage fish consumption rate at 1,646 g/day based on study of Kootenai Tribe in Idaho (see Section 5.1.3.4.4).
[e] Exposure estimates based on a general population mean fish ingestion rate of 5.04 g/day.

Table 5-61. General Population Lifetime Cancer Oral Ingestion Risk Summary Table

COU		OES	Lifetime Cancer Oral Risk Estimates			
			Drinking Water		Drinking Water (Diluted)	
Life Cycle Stage/Category	Subcategory		Lifetime from Birth	Adult Lifetime	Lifetime from Birth	Adult Lifetime
Manufacturing/Import	Import	Repackaging	6.09E–07	6.37E–07	3.91E–10	4.09E–10
Processing/Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	2.70E–06	2.82E–06	1.45E–09	1.52E–09
		Incorporation into paints and coatings – 2–part reactive coatings	2.44E–06	2.56E–06	1.32E–09	1.38E–09
Processing/Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1.43E–06	1.50E–06	9.20E–10	9.63E–10
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	5.44E–08	5.69E–08	3.49E–11	3.65E–11
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	3.42E–06	3.58E–06	6.47E–09	6.77E–09

Table 5-62. Lifetime Cancer Risk Summary for Fish Consumption

COU		OES	Lifetime Cancer Oral Risk Estimates									
Life Cycle Stage/ Category	Subcategory		Adult Fish Ingestion General Population[a]				Adult Subsistence Fisher		Tribes (Current IR)		Tribes (Heritage IR)	
			BAF 2,198		BAF 109		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
			CT[b]	HE	CT[b]	HE						
Manufacturing/ Import	Import	Repackaging	2.32E–03	1.02E–02	1.15E–04	5.07E−04	6.57E−02	3.26E–03	9.96E–02	4.94E–03	7.59E–01	3.77E–02
Processing/ Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	1.03E–02	4.53E−02	5.11E–04	2.25E−03	2.91E–01	1.44E–02	4.41E–01	2.19E–02	3.36	1.67E–01
		Incorporation into paints and coatings – 2-part reactive coatings	9.34E–03	4.11E−02	4.63E–04	2.04E−03	2.64E–01	1.31E–02	4.00E–01	1.98E–02	3.05	1.51E–01
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1.31E–02	5.74E−02	6.48E−04	2.85E−03	3.69E–01	1.83E−02	5.60E–01	2.78E−02	4.27	2.12E–01
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	9.32E−05	4.10E−04	4.62E−06	2.03E−05	2.63E−03	1.31E−04	3.99E−03	1.98E−04	3.04E−02	1.51E−03
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	5.47E−03	2.41E−02	2.71E−04	1.19E−03	1.55E−01	7.67E−03	2.35E−01	1.16E−02	1.79	8.87E−02

[a] Cancer risk estimates for the adult general population are based on the high-end fish ingestion rate of 22.2 g/day.
[b] Exposure estimates are based on a general population mean fish ingestion rate of 5.04 g/day.

Table 5-63. General Population Dermal Acute and Chronic Non-cancer Risk Summary

COU		OES	Acute MOEs *UFs = 30*	Chronic Non-cancer MOE[a] *UFs = 30*				
Life Cycle Stage/Category	**Subcategory**		**Surface Water (Adult Swimming)**	**Surface Water (Adult Swimming)**	**Child Playing in Mud at 100 m[a]**	**Child Activities with Soil at 100 m[a]**	**Child Playing in Mud at 1,000 m[a]**	**Child Activities with Soil at 1,000 m[a]**
Manufacturing/Import	Import	Repackaging	6.82E03	4.55E05	6.95E06	1.43E09	5.44E08	1.12E11
Processing/Processing – Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	1.54E03	1.05E05	2.21E05	4.55E07	2.51E07	5.15E09
		Incorporation into paints and coatings – 2-part reactive coatings	1.70E03	1.14E05	1.53E06	3.14E08	1.16E08	2.39E10
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1.21E03	9.75E04	1.39E06	2.86E08	1.09E08	2.24E10
Processing/Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	N/A	N/A	1.62E05	3.34E07	1.59E07	3.27E09
Commercial use	Laboratory chemicals	Use of laboratory chemicals	1.70E05	1.13E07	1.45E05	2.98E07	1.33E07	2.72E09
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	2.90E03	1.95E05	9.4E01	1.93E04	1.80E04	3.71E06

[a] A soil concentration based of annual air deposition fluxes is used to estimate the acute exposures scenario of a child playing with mud and conducting activities in soil.

Table 5-64. Inhalation Chronic Risk Summary for General Population[a]

COU		OES	Chronic Inhalation MOEs *UFs = 30*	
Life Cycle Stage/Category	**Subcategory**		**Ambient Air 50th**	**Ambient Air 95th**
Manufacturing/Import	Import	Repackaging	9.34E07	5.10E07
Processing/Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	3.66E06	1.49E06
		Incorporation into paints and coatings – 2-part reactive coatings	2.22E07	7.18E06
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1.98E07	6.41E06
Processing/Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	2.41E06	1.82E06
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	2.10E06	1.48E06
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	1.23E03	4.98E02

[a] 2,500 lb Production Volume – High-End Release Estimate, Suburban Forest Scenario at 10 m, high meteorology conditions

Table 5-65. General Population Lifetime Cancer Inhalation Risk Summary Table[a]

COU		OES	Distances (m)	Lifetime Cancer Inhalation Risk			
				Central Tendency Meteorological Data		High-End Meteorological Data	
Life Cycle Stage/Category	Subcategory			Cancer Risk Estimate for 50th Percentile Air Concentration	Cancer Risk Estimate for 95th Percentile Air Concentration	Cancer Risk Estimate for 50th Percentile Air Concentration	Cancer Risk Estimate for 95th Percentile Air Concentration
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use in paints and coatings at job sites	10	4.55E–05	5.53E–05	5.42E–05	1.33E–04
			30	1.49E–05	2.50E–05	1.50E–05	3.74E–05
			30–60	6.74E–06	1.70E–05	7.16E–06	2.29E–05
			60	4.63E–06	1.07E–05	4.76E–06	1.20E–05
			100	1.66E–06	4.66E–06	1.69E–06	4.46E–06

[a] 2,500 lb Production Volume – High-End Release Estimate, Suburban Forest Scenario

5.2.8 Hazard Considerations for Aggregate Exposure

For use in this risk evaluation and assessing risks from other exposure routes, EPA conducted route-to-route extrapolation of the toxicity values from the oral studies for use in the dermal and inhalation exposure routes and scenarios. Because the health outcomes are systemic and EPA did not identify evidence of differences in toxicokinetics across exposure routes), EPA considers it is possible to aggregate risks across exposure routes for all exposure durations and endpoints for the selected PODs identified in Sections 5.2.5.1 and 5.2.5.2 but only if exposure scenarios indicate that aggregation is reasonable.

5.2.9 Genotoxicity Hazard Identification and Evidence Integration

For TCEP, several studies evaluated tests of clastogenicity (three *in vivo* micronucleus assays and one *in vitro* chromosomal aberrations assay in mammalian cells), gene mutations (one forward mutation assay in mammalian cells and six bacterial reverse mutation assays), and other genotoxicity and related endpoints (two sister chromatid exchange assays, three comet assays, two cell transformation assays, and one DNA binding assay) specific to TCEP. Although EPA did not evaluate these studies using formal data quality criteria, selected studies were reviewed by comparing against current OECD test guidelines and important deviations are noted below. EPA did not review the multiple studies that were negative for gene mutations. When interpreting the results of these studies, EPA also consulted OECD (2017).

Tests of clastogenicity and gene mutations can identify the potential for a chemical to induce permanent, transmissible changes in the amount, chemical properties, or structure of DNA. One of three *in vivo* micronucleus assays was readily available. Sala et al. (1982) administered TCEP via i.p. injection to Chinese hamsters up to 250 mg/kg-day. Study methods deviated from OECD TG 474 (OECD, 2016b) in several ways. Fewer erythrocytes (2,000 vs. 4,000) were scored than recommended, and the authors did not verify that TCEP reached the bone marrow, although statistically significant results suggest this was likely. Sala et al. (1982) used two hamsters per sex vs. five per sex recommended by OECD TG 474 and used an exposure route that was not recommended. A firm conclusion is not possible given several deviations from OECD TG 474. Also, the authors state that differences in the response between sexes with variations among doses make interpretation difficult, resulting in an equivocal conclusion. However, EPA combined results across sexes, based on a comparison of means test that indicated similar results across sex and dose. This allowed greater statistical power (OECD, 2017). These combined results showed statistically significant increases in micronuclei that showed a dose-response trend. No information was provided to allow comparison with historical controls.

Two negative *in vivo* micronucleus studies using mice cited in the 2009 *European Union Risk Assessment Report* (ECB, 2009) and a review article (Beth-Hubner, 1999) were not available for review.[42]

TCEP also did not induce chromosomal aberrations in an *in vitro* assay using Chinese hamster ovary cells (Galloway et al., 1987) that was mostly consistent with OECD TG 473 (OECD, 2016a), except that the authors scored only 100 cells per concentration compared with the recommended 300 per concentration needed to conclude that a test is clearly negative.

A forward gene mutation assay using Chinese hamster lung fibroblasts (Sala et al., 1982) and multiple bacterial reverse gene mutation assays (Follmann and Wober, 2006; Haworth et al., 1983; BIBRA, 1977;

[42] According to ECB (2009), the mouse i.p. study used doses from 175 to 700 mg/kg-day, and the oral study used a dose of 1,000 mg/kg. The original reports were not readily available for review.

Prival et al., 1977; Simmon et al., 1977) were all negative for the induction of gene mutations. Most *in vitro* gene mutation assays were conducted both with and without metabolic activation. In a study by Nakamura et al. (1979), TCEP induced gene mutations in two *Salmonella typhimurium* strains. In strain TA1535, increases of four to seven times the control response were observed only with metabolic activation and in TA100, increases were observed both with and without metabolic activation. The reason for the inconsistency in results between Nakamura et al. (1979) and the other studies is unclear because concentrations were comparable. One difference, however, is that Nakamura et al. (1979) used a mixture of PCBs (Kanechlor 500) for metabolic activation, whereas other studies used Aroclor 1254 or did not appear to induce enzymes in the S9 fractions.

In addition to clastogenicity and gene mutation tests, other genotoxicity tests that measured DNA damage or DNA binding been conducted using TCEP. Two sister chromatid exchange (SCE) assays identified (1) equivocal results in Chinese hamster ovary cells (Galloway et al., 1987), and (2) statistically significant differences from controls in Chinese hamster lung fibroblasts but no clear dose response (Sala et al., 1982). *In vitro* comet assays in peripheral mononuclear blood cells (PMBCs) identified DNA damage at the highest concentration, although it is not known whether this result was in the presence of cytotoxicity (Bukowski et al., 2019). Another comet assay did not identify DNA damage in Chinese hamster fibroblasts either with or without metabolic activation (Follmann and Wober, 2006). TCEP was also negative in a DNA binding assay (Lown et al., 1980).

Sala et al. (1982) identified a high level of cell transformation in Syrian hamster embryo (SHE) cells but a lower level using C3H10T1/2 cells with metabolic activation. These cell transformation results may reflect direct or indirect genetic interactions or non-genotoxic mechanisms (OECD, 2007).

Overall, direct mutagenicity is not expected to be a predominant mode of action. Appendix M provides additional details regarding TCEP genotoxicity studies as well as considerations regarding the quality of the studies.

U.S. EPA's PPRTV (U.S. EPA, 2009) concluded that the overall weight of scientific evidence for the mutagenicity of TCEP is negative. The PPRTV also acknowledged the weak positive result in the Ames assay by Nakamura et al. (1979) and characterized the *in vivo* micronucleus assay in Chinese hamsters (Sala et al., 1982) as equivocal.

5.3 Human Health Risk Characterization

Human Health Risk Characterization (Section 5.3):
Key Points

EPA evaluated all reasonably available information to support human health risk characterization. The key points of the human health risk characterization are summarized below:

- Dermal exposures drive risks to workers in occupational settings and both cancer risks and non-cancer MOEs that met benchmarks were observed for most COUs, whereas risks and MOEs from inhalation exposure met benchmarks for multiple commercial paints and coatings use scenarios within a single COU.
- Fish ingestion is the primary exposure route driving risks to the general population. People who are subsistence fishers may be at high risk if they eat TCEP-contaminated fish; tribal people for whom fish is important dietarily and culturally may have higher risks than the general population and subsistence fishers.
- Mouthing by infants and children is the primary exposure risk for consumer articles to which infants and children are exposed.
- Infants exposed through human milk ingestion are not more sensitive than the mothers. The COUs that present infant risks also result in maternal risks. There are no COUs that show infant risks but not maternal risks. Therefore, protecting the mother will also protect the infant from exposure via human milk.

5.3.1 Risk Characterization Approach

The exposure scenarios, populations of interest, and toxicological endpoints used for evaluating risks from acute, intermediate, and chronic/lifetime exposures are summarized in Table 5-66.

Table 5-66. Exposure Scenarios, Populations of Interest, and Hazard Values

Populations of Interest and Exposure Scenarios	**Workers** Male and female adolescents and adults (≥ 16 years old) directly working with TCEP under light activity (breathing rate of 1.25 m^3/hr) Exposure durations • *Acute* – 8 hours for a single workday (most OESs) • *Intermediate* – 8 hours per workday for 22 working days • *Chronic* – 8 hours per workday for 250 days per year for 31 or 40 working years Exposure routes – Inhalation and dermal
	Occupational Non-users Male and female adolescents and adults (≥ 16 years old) indirectly exposed to TCEP within the same work area as workers (breathing rate of 1.25 m^3/hr) Exposure durations • *Acute, Intermediate, and Chronic* – same as workers Exposure route – Inhalation
	Consumers Male and female infants, children and adults using articles that contains TCEP Exposure durations • *Acute* – 1 day exposure

Populations of Interest and Exposure Scenarios	• *Chronic* – 365 days per year Exposure routes • *Adults* – Inhalation and dermal • *Infants and Children* – Inhalation, dermal, and oral
	General Population Male and female infants, children and adults exposed to TCEP through drinking water, ambient water, ambient air, soil, and diet Exposure durations • *Acute* – Exposed to TCEP continuously for a 24-hour period • *Chronic* – Exposed to TCEP continuously up to 33 years Exposure routes – Inhalation, dermal, and oral (depending on exposure scenario)
	Infants (Human Milk Pathway) Infants exposed to TCEP through human milk ingestion Exposure durations • *Intermediate* – Exposed to TCEP continuously for 30 days • *Chronic* – Exposed to TCEP continuously for one year Exposure routes – Oral
Health Effects, Hazard Values, and Benchmarks	**Non-cancer Acute Hazard Values** [b] Sensitive health effect: Neurotoxicity $HEC_{Daily,\ continuous}$ = 51.5 mg/m^3 (4.41 ppm) HED_{Daily} = 9.46 mg/kg; dermal and oral Total acute UF (benchmark MOE) = 30 (UF_A = 3; UF_H = 10)[c]
	Non-cancer Intermediate/Chronic Values[b] Sensitive health effect: Male reproductive effects $HEC_{Daily,\ continuous}$ = 14.9 mg/m^3 (1.27 ppm) HED_{Daily} = 2.73 mg/kg; dermal and oral Total intermediate/chronic UFs (benchmark MOE) = 30 (UF_A = 3; UF_H = 10)[c]
	Cancer Hazard Values[b] Both values based on renal tumors $IUR_{Daily,\ continuous}$ = 0.00451 per mg/m^3 (0.0526 per ppm) CSF_{Daily} = 0.0245 per mg/kg-day

[a] The chronic duration is the most relevant exposure scenario for the consumer COUs and is used to assess chronic non-cancer and lifetime cancer risks. Acute exposure duration non-cancer risks are presented to help characterize risk.
[b] The inhalation HEC and IUR are extrapolated from the oral HED or CSF, which are estimated using allometric scaling ($BW^{3/4}$) and are associated with continuous or daily exposures. The HEC and IUR values assume a resting breathing rate (0.6125 m^3/hr). The dermal HED is assumed to equal the oral HED. See Appendix K.3 and Benchmark Dose Modeling Results for TCEP in U.S. EPA (2024c) for dose derivation.
[c] Total UFs in the benchmark MOE.
UF_A = interspecies (animal to human); UF_H = intraspecies (human variability)

5.3.1.1 Estimation of Non-cancer Risks

EPA used a margin of exposure (MOE) approach to identify potential non-cancer risks. The MOE is the ratio of the non-cancer POD divided by a human exposure dose. Acute, intermediate, and chronic MOEs for non-cancer inhalation and dermal risks were calculated using the following equation:

Equation 5-26.

$$MOE = \frac{Non - Cancer\ Hazard\ Value\ (POD)}{Human\ Exposure}$$

Where:

MOE	=	Margin of exposure for acute, intermediate, or chronic risk comparison (unitless)
Non-cancer Hazard Value (POD)	=	HEC (mg/m^3) or HED (mg/kg-day)
Human Exposure	=	Exposure estimate (mg/m^3 or mg/kg-day)

MOE risk estimates may be interpreted in relation to benchmark MOEs. Benchmark MOEs are typically the total UF for each non-cancer POD. The MOE estimate is interpreted as a human health risk of concern if the MOE estimate is less than the benchmark MOE (*i.e.,* the total UF). On the other hand, if the MOE estimate is equal to or exceeds the benchmark MOE, the risk is not considered to be of concern and mitigation is not needed. Typically, the larger the MOE, the more unlikely it is that a non-cancer adverse effect occurs relative to the benchmark. When determining whether a chemical substance presents unreasonable risk to human health or the environment, calculated risk estimates are not "bright-line" indicators of unreasonable risk, and EPA has the discretion to consider other risk-related factors in addition to risks identified in the risk characterization.

5.3.1.2 Estimation of Cancer Risks

Extra cancer risks for repeated exposures to a chemical were estimated using the following equations:

Equation 5-27.

$$Inhalation\ Cancer\ Risk = Human\ Exposure \times IUR$$

or

$$Dermal\ or\ Oral\ Cancer\ Risk = Human\ Exposure \times CSF$$

Where:

Risk	=	Extra cancer risk (unitless)
Human Exposure	=	Exposure estimate (LADC in ppm)
IUR	=	Inhalation unit risk (risk per mg/m^3)
CSF	=	Cancer slope factor (risk per mg/kg-day)

Estimates of extra cancer risks are interpreted as the incremental probability of an individual developing cancer over a lifetime following exposure (*i.e.,* incremental or extra individual lifetime cancer risk).

EPA considers a range of extra cancer risk from 1×10^{-4} to 1×10^{-6} to be relevant benchmarks for risk assessment (U.S. EPA, 2017b). Consistent with NIOSH guidance (Whittaker et al., 2016), under TSCA, EPA typically applies a 1×10^{-4} benchmark for occupational scenarios in industrial and commercial work environments subject to OSHA requirements. The Agency typically considers the general population and consumer benchmark for cancer risk to be within the range of 1×10^{-6} and 1×10^{-4}. Again, it is important to note that these benchmarks are not bright lines and EPA has discretion to find unreasonable risks based on other risk-related considerations based on analysis. Exposure-related considerations (*e.g.,* duration, magnitude, population exposed) can affect EPA's estimates of the excess lifetime cancer risk.

5.3.2 Summary of Human Health Risk Characterization

5.3.2.1 Summary of Risk Estimates for Workers

EPA estimated cancer risks and non-cancer MOEs for workers exposed to TCEP for multiple COUs based on the occupational exposure estimates described in Section 5.3.2.1.1. Complete risk calculations and results for the occupational OES/COUs are available in *Risk Evaluation for Tris(2-chloroethyl)*

Phosphate (TCEP) – Supplemental Information File: Risk Calculator for Occupational Exposures (U.S. EPA, 2024m).

5.3.2.1.1 COUs/OESs with Quantitative Risk Estimates

Table 5-67. summarizes cancer and non-cancer risk estimates for the inhalation and dermal exposures for all OESs assessed. These risk estimates are based on exposures estimated for workers who do not use PPE such as gloves or respirators. When both monitoring and modeling data were available for inhalation exposures, EPA only presented the risk estimates for the most reliable data source in the summary table. Estimates for inhalation and dermal exposures that have PPE factored in are contained in the *Risk Evaluation for TCEP – Supplemental Information File: Risk Calculator for Occupational Exposures* (U.S. EPA, 2024m).

Exposure data for ONUs were not available for most COUs except for recycling (with recycling e-waste as the relevant OES). For the COUs and OESs without ONU-specific exposure data, EPA assumed risks would be equal to or less than risks to workers who handle materials containing TCEP as part of their job. The inhalation risk values used for workers are also presented for ONUs in Table 5-67.. EPA assumed that ONUs are not exposed dermally.

Within the commercial use of paints and coatings COU, EPA did not calculate short-term or chronic, non-cancer risks or lifetime cancer risks for the 1-day spray application for commercial paint and coating scenarios (OES #7 and #10) because risks were most appropriately assessed using only the inhalation HEC and dermal HED values for acute exposures. Likewise, EPA did not calculate chronic non-cancer or lifetime cancer risks for the 2-day commercial paint and coating spray application (OES #8 and #11) given the very limited number of days per year of exposure. However, for OESs exposures longer than 1 day per year, EPA also compared exposure with the acute hazard PODs.

EPA was informed by the Auto Alliance that it is possible that TCEP containing articles, replacement parts, and paints could be in use in the automotive industry, however, no data regarding product(s), operating site(s), etc was provided. As reflected below, in Table 5-58, in the absence of reasonably available data to refine the modelling, EPA expects modelled environmental releases and occupational exposures to be similar to the OES's already previously assessed for other industry sectors on a per generic site basis. Specifically, this means that that the recently added industrial use of paints and coatings would be similar to the commercial use of paints and coatings and that the recently modified COUs of Incorporation into article and Installation of article would be similar regardless of whether or not it is being done for aircraft or automobiles.

Risks from Inhalation Exposure

Cancer inhalation risk estimates were above 1 in 10,000 for the commercial use of paints and coatings COU for both central tendency and high-end exposures. These risks were associated with two OESs: 250-day applications of either 1- or 2-part sprays. Risk estimates were less than 1 in 10,000 for the remaining six occupational COU subcategories.

In addition, inhalation non-cancer MOEs were less than benchmark MOEs for the commercial use of paints and coatings COU for high-end exposures. Within this COU, high-end acute exposure for all three OESs associated with 2-part spray applications resulted in MOEs less than the benchmark MOE of 30. For high-end short-term/chronic exposures, MOEs were less than the benchmark MOE of 30 for the 250-day applications of either 1- or 2-part sprays. No other COU/OES combinations resulted in MOEs less than the non-cancer benchmark MOEs; this includes the commercial and industrial uses for the

installation of articles, which used surrogate monitoring data to estimate inhalation exposures that could occur during these activities.

Risks from Dermal Exposure

More COU categories were associated with worker dermal risks above 1 in 10,000. Cancer dermal risk estimates were above 1 in 10,000 for both central tendency and high-end exposures for certain subcategories and OESs within the following five COU categories: (1) Import; (2) Incorporation into formulation, mixture, or reaction products; (3) Processing – Incorporation into an article; (4) Commercial use of paints and coatings; and (5) Other commercial use – Laboratory chemicals.

Additional dermal cancer risks above 1 in 10,000 were observed for only high-end exposures within a single COU category (Processing – Incorporation into formulations, mixtures, or reaction products) and two associated OESs (Incorporation into 2-part paints and coatings and Formulation of 2-part reactive resins).

Three COU categories had chronic non-cancer dermal MOEs less than the benchmark value of 30 for both high-end and central tendency exposures. These were, (1) Processing – Incorporation into articles; (2) Commercial use of paints and coatings; and (3) Other commercial use – Laboratory chemicals. Two additional COUs were associated with MOEs lower than 30 for only high-end exposures, these were (1) Import; and (2) Processing – Incorporation into formulation, mixture, or reaction products.

For the short-term exposure scenario, MOEs were less than 30 for five COUs for at least some OESs. Within two of these COUs, certain OESs had MOEs less than 30 for only high-end exposures—Flame retardant in paints and coatings manufacture (2-part coatings and polymers in aerospace equipment) and Commercial use of paints and coatings (2-day application for 1-part coatings).

For the acute exposure scenario, five COUs had dermal MOEs of less than 30 for both central tendency and high-end exposures. One of these five COUs (commercial use of paints and coatings) also had some OESs (1-part sprays) for which MOEs were less than 30 for only high-end exposures.

Processing/Recycling was the single COU with cancer dermal risks less than 1 in 10,000 and all non-cancer MOEs greater than benchmark values. Dermal risk estimates were not calculated for industrial and commercial use of aerospace equipment and automotive products because EPA does not expect dermal exposure for this COU because TCEP will be entrained in the polymer matrix.

Table 5-67. Occupational Risk Summary for 2,500 lb Production Volume

COU		OES	Population	Exposure Route and Duration	Exposure Level	Estimates for No PPE				Overall Confidence in Risk Estimates
Life Cycle Stage/ Category	Subcategory					Acute Non-cancer MOE *UFs = 30*	Intermediate Non-cancer MOE *UFs = 30*	Chronic Non-cancer MOE *UFs = 30*	Lifetime Cancer Risk	
Manufacturing / Import	Import	Repackaging	Worker	Inhalation 8-hr TWA	CT	6.8E03	1.4E04	1.7E05	1.5E−07	Moderate
					High-End	1.9E03	4.0E03	4.9E04	5.5E−07	
			ONU[a]	Inhalation 8-hr TWA	CT	6.8E03	1.4E04	1.7E05	1.5E−07	Slight
					High-End	1.9E03	4.0E03	4.9E04	5.5E−07	
			Worker	Dermal	CT	4.3	9.4	1.14E02	2.3E−04	Moderate
					High-End	1.4	1.8	2.2E01	1.6E−03	
Processing/ Processing – Incorporation into formulation, mixture, or reaction product	Flame retardant in: Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	Worker	Inhalation 8-hr TWA	CT	4.6E03	6.7E03	7.7E04	3.3E−07	Moderate
					High-End	7.3E02	1.6E03	1.9E04	1.4E−06	
			ONU[a]	Inhalation 8-hr TWA	CT	4.6E03	6.7E03	7.7E04	3.3E−07	Slight
					High-End	7.3E02	1.6E03	1.9E04	1.4E−06	
			Worker	Dermal	CT	4.3	6.3	7.6E01	3.5E−04	Moderate
					High-End	1.4	5.7E−01	4.0	8.6E−03	
		Incorporation into paints and coatings – 2-part coatings	Worker	Inhalation 8-hr TWA	CT	7.9E02	6.5E03	7.9E04	3.2E−07	Moderate
					High-End	1.9E02	1.6E03	1.9E04	1.4E−06	
			ONU[a]	Inhalation 8-hr TWA	CT	7.9E02	6.5E03	7.9E04	3.2E−07	Slight
					High-End	1.9E02	1.6E03	1.9E04	1.4E−06	
			Worker	Dermal	CT	4.3	3.8E01	4.6E02	5.8E−05	Moderate
					High-End	1.4	6.3	7.6E01	4.5E−04	
	Polymers used in aerospace equipment and products	Formulation of TCEP into 2-part reactive resin	Worker	Inhalation 8-hr TWA	CT	1.0E04	6.7E03	8.1E04	3.1E−07	Moderate
					High-End	1.9E02	1.5E03	1.8E04	1.5E−06	
			ONU[a]	Inhalation 8-hr TWA	CT	1.0E04	6.7E03	8.1E04	3.1E−07	Slight
					High-End	1.9E02	1.5E03	1.8E04	1.5E−06	
			Worker	Dermal	CT	4.3	3.8E01	4.6E02	5.8E−05	Moderate
					High-End	1.4	2.1	2.5E01	1.4E−03	
Processing/ Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	Worker	Inhalation 8-hr TWA	CT	2.2E04	9.0E03	3.8E04	6.6E−07	Moderate
					High-End	4.2E03	1.8E03	6.3E03	4.1E−06	
			ONU[a]	Inhalation 8-hr TWA	CT	2.2E04	9.0E03	3.8E04	6.6E−07	Slight
					High-End	4.2E03	1.8E03	6.3E03	4.1E−06	
			Worker	Dermal	CT	1.1E01	4.3	1.6E01	1.7E−03	Moderate
					High-End	3.6	1.4	1.5	2.3E−02	

COU		OES	Population	Exposure Route and Duration	Exposure Level	Estimates for No PPE				Overall Confidence in Risk Estimates
Life Cycle Stage/ Category	Subcategory					Acute Non-cancer MOE *UFs = 30*	Intermediate Non-cancer MOE *UFs = 30*	Chronic Non-cancer MOE *UFs = 30*	Lifetime Cancer Risk	
Processing/ Recycling	Recycling	Processing – Recycling e-waste	Worker	Inhalation 8-hr TWA	CT	7.6E08	3.0E08	3.2E08	8.4E−11	Moderate
					High-End	7.8E04	3.1E04	3.3E04	1.0E−06	
			ONU	Inhalation 8-hr TWA	CT	7.6E08	3.0E08	3.2E08	8.4E−11	Moderate
					High-End	4.0E05	1.6E05	1.7E05	2.0E−07	
			Worker	Dermal	CT	5.2E05	2.0E05	2.2E05	1.2E−07	Moderate
					High-End	2.2E05	8.5E4	9.1E04	3.8E−07	
Commercial and Industrial use/Paints and coatings	Paints and coatings	Paints and coatings – Spray (1-part coatings, 1-day application)	Worker	Inhalation 8-hr TWA	CT	4.5E02	N/A	N/A	N/A	Moderate
					High-End	6.9E01	N/A	N/A	N/A	
			ONU[a]	Inhalation 8-hr TWA	CT	4.5E02	N/A	N/A	N/A	Slight
					High-End	6.9E01	N/A	N/A	N/A	
			Worker	Dermal	CT	3.2E01	N/A	N/A	N/A	Moderate
					High-End	5.9	N/A	N/A	N/A	
		Paints and coatings – Spray (1-part coatings, 2-day application)	Worker	Inhalation 8-hr TWA	CT	4.5E02	1.9E03	N/A	N/A	Moderate
					High-End	6.9E01	3.0E02	N/A	N/A	
			ONU[a]	Inhalation 8-hr TWA	CT	4.5E02	1.9E03	N/A	N/A	Slight
					High-End	6.9E01	3.0E02	N/A	N/A	
			Worker	Dermal	CT	3.2E01	1.4E02	N/A	N/A	Moderate
					High-End	5.9	2.6E01	N/A	N/A	
		Paints and coatings – Spray (1-part coatings, 250-day application)	Worker	Inhalation 8-hr TWA	CT	4.5E02	1.8E02	1.9E02	1.4E−04	Moderate
					High-End	6.9E01	2.7E01	2.9E01	1.2E−03	
			ONU[a]	Inhalation 8-hr TWA	CT	4.5E02	1.8E02	1.9E02	1.4E−04	Slight
					High-End	6.9E01	2.7E01	2.9E01	1.2E−03	
			Worker	Dermal	CT	3.2E01	1.3E01	1.3E01	2.0E−03	Moderate
					High-End	5.9	2.3	2.5	1.4E−02	
		Paints and coatings – Spray (2-part coatings, 1-day application)	Worker	Inhalation 8-hr TWA	CT	9.0E01	N/A	N/A	N/A	Moderate
					High-End	1.4E01	N/A	N/A	N/A	
			ONU[a]	Inhalation 8-hr TWA	CT	9.0E01	N/A	N/A	N/A	Slight
					High-End	1.4E01	N/A	N/A	N/A	
			Worker	Dermal	CT	6.4	N/A	N/A	N/A	Moderate
					High-End	1.2	N/A	N/A	N/A	
		Paints and coatings – Spray (2-part	Worker	Inhalation 8-hr TWA	CT	9.0E01	3.9E02	N/A	N/A	Moderate
					High-End	1.4E01	5.9E01	N/A	N/A	
			ONU[a]	Inhalation 8-hr TWA	CT	9.0E01	3.9E02	N/A	N/A	Slight
					High-End	1.4E01	5.9E01	N/A	N/A	

COU		OES	Population	Exposure Route and Duration	Exposure Level	Estimates for No PPE				Overall Confidence in Risk Estimates
Life Cycle Stage/ Category	**Subcategory**					**Acute Non-cancer MOE *UFs = 30***	**Intermediate Non-cancer MOE *UFs = 30***	**Chronic Non-cancer MOE *UFs = 30***	**Lifetime Cancer Risk**	
Commercial and Industrial use/Paints and coatings	Paints and coatings	coatings, 2-day application)	Worker	Dermal	CT	6.4	2.8E01	N/A	N/A	Moderate
					High-End	1.2	5.1	N/A	N/A	
		Paints and coatings – Spray (2-part coatings, 250-day application)	Worker	Inhalation 8-hr TWA	CT	9.0E01	3.8E01	3.8E01	7.1E−04	Moderate
					High-End	1.4E01	5.4	5.8	6.0E−03	
			ONU[a]	Inhalation 8-hr TWA	CT	9.0E01	3.8E01	3.8E01	7.1E−04	Slight
					High-End	1.4E01	5.4	5.8	6.0E−03	
			Worker	Dermal	CT	6.4	2.5	2.7	9.9E−03	Moderate
					High-End	1.2	4.6E−01	5.0E−01	6.9E−02	
Industrial Use/Other Use	Aerospace equipment products and automotive articles and replacement parts containing TCEP	Installation of articles	Worker	Inhalation 8-hr TWA	CT	5.8E06	2.3E06	2.5E06	1.1E−08	Slight
					High-End	5.8E06	2.3E06	2.5E06	1.1E−08	
			ONU[a]	Inhalation 8-hr TWA	CT	5.8E06	2.3E06	2.5E06	1.1E−08	Slight
					High-End	5.8E06	2.3E06	2.5E06	1.1E−08	
			Worker	Dermal	CT	N/A	N/A	N/A	N/A	N/A
					High-End	N/A	N/A	N/A	N/A	
Commercial Use/Other Use	Aerospace equipment products and automotive articles and replacement parts containing TCEP	Use and/or maintenance of articles	Worker	Inhalation 8-hr TWA	CT	5.8E06	2.3E06	2.5E06	1.1E−08	Slight
					High-End	5.8E06	2.3E06	2.5E06	1.1E−08	
			ONU[a]	Inhalation 8-hr TWA	CT	5.8E06	2.3E06	2.5E06	1.1E−08	Slight
					High-End	5.8E06	2.3E06	2.5E06	1.1E−08	
			Worker	Dermal	CT	N/A	N/A	N/A	N/A	N/A
					High-End	N/A	N/A	N/A	N/A	
Commercial Use/Other Use	Laboratory chemicals	Laboratory chemicals	Worker	Inhalation 8-hr TWA	CT	1.0E05	5.1E04	5.5E04	4.0E−07	Moderate
					High-End	6.5E04	3.2E04	3.5E04	6.8E−07	
			ONU[a]	Inhalation 8-hr TWA	CT	1.0E05	5.1E04	5.5E04	4.0E−07	Slight
					High-End	6.5E04	3.2E04	3.5E04	6.8E−07	
			Worker	Dermal	CT	4.3	1.7	2.7	9.7E−03	Moderate
					High-End	1.4	5.7E−01	7.6E−01	4.5E−02	
Disposal/ Disposal	Disposal	Disposal	Evaluated as part of each OES as opposed to a standalone OES							
CT = Central Tendency										

5.3.2.1.2 COUs/OESs Without Quantitative Risk Estimates

Distribution in Commerce

For purposes of assessment in this risk evaluation, distribution in commerce of TCEP consists of the transportation associated with the moving of sealed containers of TCEP. EPA expects TCEP to be transported in sealed containers from import sites to downstream processing and use sites, or for final disposal of TCEP. The steps of loading and unloading that are assessed during other COUs/OESs consists of unloading TCEP into the formulation process and loading refers to packaging the finished product prior to shipment. Loading and unloading activities that occur during a distribution in commerce scenario would only refer to loading or unloading sealed containers from a transport vehicle. More broadly under TSCA, "distribution in commerce" and "distribute in commerce" are defined under TSCA section 3(5). EPA expects under standard operating procedures that exposures and releases that could occur during distribution in commerce are expected to be negligible because containers remain sealed during transport.

Commercial Uses that TCEP is No Longer Actively Incorporated Into

EPA determined that the following commercial use COUs for TCEP consists of use of existing products and end of service life disposal:

- Commercial use – Furnishing, cleaning, treatment/care products – Fabric and textile products;
- Commercial use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products;
- Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation; and
- Commercial use – construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites.

TCEP was used for these purposes in the past, but the COUs were phased out beginning in the late 1980s or early 1990s and replaced by other flame retardants or flame-retardant formulations. EPA did not locate data to estimate (1) the amount of TCEP used in these products, (2) the amounts of these products that have already reached the end of their service life, or (3) the amounts that have already been disposed. Based on the years that the phase-out occurred, many of these products are likely to no longer be in use because the end of their service life was already reached (*e.g.*, commercial roofing has an estimated lifespan of 17–20 years). EPA assumes that any of these products still used commercially represent a fraction of the overall amount of TCEP previously used for these purposes.

EPA acknowledges that workers may handle fabric and textile products, foam seating and bedding products, building/construction materials – wood and engineered wood products – wood resin composites, building/construction materials insulation that were previously manufactured when TCEP was more often present as a flame retardant in these articles. There exists some potential for exposures to workers in commercial settings that handle or repair older articles that used TCEP.

In addition, office workers may be exposed to fabric and textile products, foam seating and bedding products, building/construction materials – wood and engineered wood products – wood resin composites, building/construction materials insulation, in the commercial environments and offices. Office workers exposed to such articles would mirror the consumer assessment described in Section 5.1.2 due to the similarity of exposure scenarios between office workers and consumers. The consumer assessment for these articles resulted in no consumer risk for inhalation, ingestion, or dermal risk for adults for the COUs with moderate confidence. Therefore, EPA has moderate confidence that under similar exposure durations and exposure frequencies, the Agency expects these articles to pose no

commercial risk for inhalation, ingestion, and dermal risk to COUs to commercial workers who use articles in a similar fashion to consumers.

Commercial uses may be higher or lower than consumer exposures for a multitude of reasons. Commercial uses for PESS such as truck drivers (auto-foam), gym teachers (foam mats, and wood flooring), and DIY hobbyists (wood resin products) may have elevated exposures to these finished articles due to their increased activity and use patterns with such articles. Whereas commercial environments may frequently replace older furniture articles with newer articles which no longer include TCEP.

Due to the lack of reasonably available information on (1) the amount of TCEP used in these products; (2) the amounts of these products that have already reached the end of their service life; or (3) the amount of articles that remain in commercial environments; EPA is unable to quantify the exposure to commercial COUs listed above.

Disposal

Waste handling, disposal, and/or treatment includes waste disposal (landfilling or incineration) as well as water (*e.g.,* releases to wastewater treatment and POTWs) and air releases (*e.g.,* fugitive and stack air emissions). Workers engaged in these activities at the facilities where TCEP is processed and used, as well as workers at off-site waste treatment and disposal facilities (*e.g.*, landfills, incinerators, POTWs) could be exposed to TCEP.

EPA estimated releases to landfills for the following two COU/OES combinations:

- Processing – Incorporation into formulation, mixture, or reaction product – Paint/coating manufacture – 1-part coating OES; and
- Processing – Incorporation into articles – Aerospace equipment and products and automotive articles and replacement parts containing TCEP – Processing in two-part resin article OES.

EPA estimated releases to incinerators for the following two COU/OES combinations:

- Processing – Incorporation into formulation, mixture, or reaction product – Paint/coating manufacture – 2-part coating OES; and
- Processing – Incorporation into formulation, mixture, or reaction product – Polymers in aerospace equipment and products – Formulation of reactive resins OES.

Both releases to landfills and incinerators rely on inputs provided by ESDs or GSs. However, the ESDs and GSs do not specify the proportion of the throughput that goes to either of these two disposal practices. Therefore, EPA was unable to further quantify environmental releases related to these two disposal processes.

For three of the COUs/OESs listed above, EPA was able to perform quantitative risk characterization that included releases to on-site wastewater treatment or discharge to POTWs, where applicable (see Table 3-2). Any worker exposures associated with on-site waste treatment were combined with other exposures as relevant for the above COUs.

Waste treatment or disposal is expected to be negligible for industrial and commercial uses related to installing articles for aerospace applications and automotive articles and replacement parts containing TCEP. For the COUs of manufacturing/repackaging, commercial use of paints and coatings, commercial use of laboratory chemicals, and disposal to landfills or incinerators are not expected but EPA estimated surface water releases that could include release to wastewater treatment or POTWs.

For the commercial uses that have been phased out, any currently used products that contain TCEP are expected to be disposed in landfills but will represent just a fraction of previous amounts from when TCEP was used more widely. Data are lacking with which to estimate exposure and risk from disposal or waste treatment activities for these COUs and EPA has not quantified such risks. For e-waste recycling, there is also too little information to estimate exposure from disposal and only a small portion of e-waste is expected to contain TCEP. Therefore, EPA's confidence in these exposures is indeterminate and cannot quantify risk for the disposal or waste treatment activities for these COUs. However, EPA acknowledges that while some releases and exposures could occur during the disposal of the wide variety of items that TCEP has found its way into, these exposures are expected to be negligible.

5.3.2.2 Summary of Risk Estimates for Consumers

5.3.2.2.1 COUs with Quantitative Risk Estimates

Table 5-68 summarizes the dermal, inhalation, and ingestion MOEs used to characterize non-cancer risk for acute, intermediate, and chronic exposure and presents these values for all life stages for each COU. Table 5-69 summarizes the dermal, inhalation, and oral lifetime cancer risk estimates for each consumer COU. Risk estimates in Table 5-68 and Table 5-69 are only presented for COUs, routes, and age groups that are below the non-cancer risk benchmarks or above the lifetime cancer benchmarks. For cancer, EPA uses a cancer benchmark range from 1 in 10,000 to 1 in 1,000,000 to consider and characterize lifetime cancer risks from consumer exposure. Table 5-69 presents the risk estimates that were above the lifetime cancer benchmark of 1 in 1,000,000.

Although CEM 3.2 provides inhalation exposure doses for each age group, inhalation exposure risk estimates were calculated for the adult exposure scenario. Inhalation risk estimates for other life stages are presented in Appendix J. These adjusted inhalation exposure doses are estimated using breathing rate and body weight considerations for each age group. Body weight- and inhalation rate-adjusted inhalation risk estimates for younger life stages should be interpreted with caution. Despite accounting for breathing rate and body weight, adjusted inhalation exposures for younger age groups may be inaccurate because there are other considerations (*e.g.*, elimination kinetics) that may differ among age groups (U.S. EPA, 2012a). Information on the inputs used for consumer modeling using CEM 3.2 are presented in Section 5.1.2 and Appendix J.

Acute and Chronic Risks

Children and infants have acute oral MOEs less than the benchmark of 30 for foam toy blocks, roofing insulation, and wood flooring. Infants have acute oral MOEs less than the benchmark of 30 for all of the COUs except acoustic ceilings. Chronic oral MOEs for children and infants are below the benchmark of 30 for fabric and textiles, foam seating and bedding products, wood articles (*e.g.*, wood flooring and wooden TV stands). Infants and children have a greater susceptibility to TCEP exposure due to mouthing behaviors associated with toys (*e.g.*, outdoor play structures, foam blocks). As discussed in Section 5.1.2.2.4, EPA selected a high mouthing parameter (50 cm^2) for the COUs that were designed for children. For other products that had the potential for mouthing, EPA selected medium mouthing parameters (10 cm^2). Mouthing duration had a pronounced impact on the oral exposures for children and infants (see Appendix J).

Section 5.1.2.2.3 describes the parameters selection and assumptions considered for the dermal exposure assessment. Acute and chronic dermal MOEs for all life stages are below the benchmark of 30 for wood flooring. Chronic dermal MOEs for children and infants are below the benchmark of 30 for wood articles (*e.g.*, wood flooring and wooden TV stands). Sensitivity analyses indicated that the initial SVOC

concentration in the article (a product of the article density and the weight fraction) is a driver of dermal exposures. The consumer modeling suggests direct contact with wooden articles (*e.g.,* wood flooring, wooden TV stands) results in greater exposure than dermal doses mediated from dust generated from consumer articles.

Chronic inhalation MOEs for acoustic ceilings, wood flooring, and insulation are below the benchmark of 30. Acute inhalation MOEs for textiles in outdoor play structures, acoustic ceilings, wood flooring, wooden TV stands, and insulation are below the benchmark of 30. Furthermore, figuresFigure 5-20,Figure 5-21, andFigure 5-22 demonstrate the chronic exposure estimates may be high due to an initial off-gassing period where there are high gas phase air concentrations. Depending on the COU, after a few weeks to a few months, there is a precipitous drop in the gas phase air concentrations. These findings suggest that inhalation exposure estimates are higher for newer consumer articles where the off-gassing period dominates the exposure versus older consumer articles that have already undergone the off-gassing period. Sensitivity analyses indicated that the initial SVOC concentration in the article (a product of the article density and the weight fraction) is a driver of inhalation exposures for insulation. For more information on the inhalation exposure estimates, see Section 5.1.2.2.2.

Lifetime Cancer Risks

Inhalation from wood resin composites (*e.g.,* wood flooring) presents the highest lifetime cancer risk (3.19×10^{-2}), followed by inhalation exposure from insulation (2.55×10^{-2}) (Table 5-69). In comparing inhalation risks among wood resin composites, wood flooring has a larger cancer inhalation risk estimate by two orders of magnitude than wood TV stands. This suggests that the space (surface area) a wood article occupies in the home environment has a relationship to the magnitude of inhalation risk. Lifetime cancers risks for wood resin composites (*e.g.,* wood flooring) is dominated by the inhalation route whereas lifetime cancer risks for wood resin composites (*e.g.,* wooden TV stands) is dominated by the ingestion of dust. This may be explained by the relatively large surface area for wood flooring vs. wooden TV stands. Wood articles (*e.g.,* wood flooring, wooden TV stands) have a higher lifetime cancer risk for dermal exposures (6.35×10^{-3} and 2.03×10^{-3}) compared to oral exposure (4.85×10^{-4} and 3.99×10^{-4}). Carpet and foam products (*e.g.,* mattresses, foam furniture, automobile foams) are dominated by oral cancer risks relative to other routes. The contribution of mouthing exposure from these articles at younger life stages may be contributing to the overall cancer risk.

Table 5-68. Acute and Chronic Non-cancer Consumer Risk Summary

COU		Consumer Use Scenario	Exposure Route	Age Group (years)	Non-cancer MOEs[a]		Overall Confidence Non-cancer MOEs
Life Cycle Stage/Category	Subcategory				Acute MOE *UFs* = 30	Chronic MOE *UFs* = 30	
Consumer use/ Furnishing, cleaning, treatment, and care products	Fabric and textile products	Carpet back coating	Oral	Infant: 1–2	35	10	Moderate
			Oral	Infant: <1	18	5	
		Textile for children's outdoor play structures	Inhalation	Adult: >21	9	45	Moderate
			Oral	Infant: 1–2	30	10	
			Oral	Infant: <1	17	5	
	Foam seating and bedding products	Foam auto	Oral	Infant: 1–2	35	10	Moderate
			Oral	Infant: <1	18	5	
		Foam living room	Oral	Infant: 1–2	35	10	Slight
			Oral	Infant: <1	18	5	
		Mattress	Oral	Infant: 1–2	35	10	Slight
			Oral	Infant: <1	18	5	
		Foam-other (toy block)	Oral	Child: 6–10	88	26	Slight
			Oral	Child: 3–5	52	15	
			Oral	Infant: 1–2	7	2	
			Oral	Infant: <1	4	1	
Consumer use/ Construction, paints, electrical, and metal products	Building/ construction materials – Insulation	Roofing insulation	Inhalation	Adult: >21	<1	2	Slight
			Oral	Child: 6–10	21	75	
			Oral	Child: 3–5	7	27	
			Oral	Infant: 1–2	8	30	
			Oral	Infant: <1	10	37	
		Acoustic ceiling	Inhalation	Adult: >21	2	24	

COU		Consumer Use Scenario	Exposure Route	Age Group (years)	Non-cancer MOEs[a]		Overall Confidence Non-cancer MOEs
Life Cycle Stage/Category	Subcategory				Acute MOE *UFs* = 30	Chronic MOE *UFs* = 30	
Consumer use/ Construction, paints, electrical, and metal products	Building/ construction materials – Wood and engineered wood products – Wood resin composites	Wood flooring	Dermal	Adult: >1	18	11	Slight
			Dermal	Youth: 16–20	19	12	
			Dermal	Youth: 11–15	17	11	
			Dermal	Child: 6–10	14	9	
			Dermal	Child: 3–5	11	7	
			Dermal	Infant: 1–2	9	6	
			Dermal	Infant: <1	8	5	
			Inhalation	Adult: >21	<1	3	
			Oral	Child: 6–10	17	95	
			Oral	Child: 3–5	6	48	
			Oral	Infant: 1–2	6	9	
			Oral	Infant: <1	6	5	
		Wooden TV stand	Dermal	Child: 6–10	93	27	Moderate
			Dermal	Child: 3–5	75	22	
			Dermal	Infant: 1–2	65	19	
			Dermal	Infant: <1	55	16	
			Inhalation	Adult: >21	8	416	
			Oral	Infant: 1–2	34	10	
			Oral	Infant: <1	18	5	

[a] Risk estimates are only presented for COUs, routes, and age groups that are below the non-cancer risk benchmarks or above the lifetime cancer benchmarks.

Table 5-69. Lifetime Cancer Consumer Risk Summary

COU		Consumer Use Scenario	Exposure Route	Lifetime Cancer Risk Estimates[a]	Overall Confidence in Cancer Risk Estimate
Life Cycle Stage/Category	Subcategory				
Consumer use/ Furnishing, cleaning, treatment, and care products	Fabric and textile products	Carpet back coating	Dermal	9.24E–06	Moderate
			Inhalation	1.48E–04	
			Oral	3.99E–04	
	Foam seating and bedding products	Foam automobile	Dermal	1.52E–04	Moderate
			Inhalation	2.52E–08	
			Oral	3.99E–04	
		Foam living room	Dermal	3.38E–04	Moderate
			Inhalation	4.51E–08	
			Oral	3.99E–04	
		Mattress	Dermal	4.12E–05	Slight
			Inhalation	2.15E–06	
			Oral	3.99E–04	
Consumer use/ Construction, paints, electrical, and metal products	Building/construction materials – Insulation	Roofing insulation	Dermal	1.85E–05	Slight
			Inhalation	2.55E–02	
			Oral	1.29E–04	
		Acoustic ceiling	Dermal	1.94E–06	Slight
			Inhalation	3.63E–03	
			Oral	1.43E–05	
	Building/construction materials – Wood and engineered wood products – Wood resin composites	Wood flooring	Dermal	6.35E–03	Slight
			Inhalation	3.19E–02	
			Oral	4.85E–04	
		Wooden TV stand	Dermal	2.03E–03	Moderate
			Inhalation	2.08E–04	
			Oral	3.99E–04	

[a] Risk estimates are only presented for COUs, routes, and age groups that are below the non-cancer risk benchmarks or above the lifetime cancer benchmarks.

5.3.2.2.2 COUs Without Quantitative Risk Estimates

Paints and Coatings Including Those Found on Automotive Articles and Replacement Parts

Domestic retail production and manufacturing of paints and coatings, including those found on automotive articles and replacement parts containing TCEP, has ceased, and consumers can no longer purchase these products from store shelves in the United States. However, TCEP containing paints and coatings are still used in commercial applications, and consumers could potentially buy commercial paints and coatings containing TCEP on-line. In addition, consumers may have old cannisters of paints and coatings that were purchased prior to the phase out of TCEP in the consumer market. Consumers are unlikely to obtain TCEP containing paints and coatings because the domestic retail production and manufacturing of TCEP containing paints and coatings has ceased and TCEP containing paints and coatings represent a small fraction of the paints and coatings on the market.

In the early 2000s, Ingerowski et al. (2001) detected TCEP in 85 percent of 983 household products in Germany and reported TCEP in wood preservation coatings at 1.0 percent. Also, Haumann and Thumulla (2002) detected TCEP in paints at a maximum of 840 mg/kg (0.084%) in Germany prior to 2002 (TERA, 2013). Commercial paints and coatings containing TCEP have weight fractions up to 25 percent (Section 3.3.1 of the Engineering Supplemental File Appendix C.4.13), whereas communications with the Alliance for Automotive Innovation indicated that TCEP is being used in smaller weight fractions ranging from 0.1 percent to maximum 1 percent.

If a consumer obtains a TCEP-containing paint or coating, exposure could occur through inhalation and dermal exposure from the application (*e.g.*, spray applied or using a paint brush) of the paint or coating; or inhalation, oral and dermal exposure from the dried paint or coating through dust generated from abrasion of the painted or coated article.

Application Scenario

There is a dearth of information regarding the consumer application of paints and coatings exposure scenario. No consumer SDS are available for TCEP in consumer paints and coatings. EPA has determined that it is not likely for consumers to obtain TCEP containing paints and coatings products that are available for commercial applications. Therefore, EPA does not expect exposure to consumers from the application of TCEP containing paints and coatings.

Dried Paint or Coating Scenario

The dried paint or coating scenario is akin to the article scenarios (*e.g.*, wood resin articles) already modeled in the TCEP risk evaluation (Section 5.3.2.1.1). EPA is uncertain whether the TCEP used in paints and coatings is bonded to the polymer matrix (covalently bonded), or whether the TCEP is used as an additive (non-covalently bonded). For the 2-part resin OES, EPA expects TCEP releases and dermal exposures to be limited by TCEP being entrained into the hardened polymer matrix (covalently bonded). If TCEP is entrained in a hardened polymer matrix, a chronic exposure is more relevant than an acute scenario as the article would either need to be abraded, degrade or the TCEP would have to migrate from the entrained polymer to the surface of the article and subsequently volatilize. An acute exposure scenario is more applicable when TCEP is used as an additive, where the TCEP could quickly off-gas. How TCEP is bonded in the paint or coating, or in its subsequent application may influence the likelihood of TCEP being released from the consumer product or article.

CEM 3.2 does not include input parameters to specify whether TCEP is bonded to the polymer matrix or used as an additive. Rather, CEM 3.2 is a set of deterministic models that use physical chemical properties to estimate inhalation, dermal and ingestion exposures to consumer products and articles. The

CEM results in Figures Figure 5-20, Figure 5-21 and Figure 5-22, describe how inhalation exposures of articles containing TCEP are expected to be initially high and then subsequently off gas. Older articles in the home may have already undergone off-gassing of TCEP; thus, there is uncertainty as to the relevance of continued inhalation exposure from older consumer articles containing TCEP as much of the exposure may have already occurred in the first few weeks to months.

Consumers use automotive replacement parts that have been painted with TCEP containing paints and coatings well after the physical painting and coating has occurred in industrial and commercial applications. However, the chronic exposure scenario (abrasion or delayed release from entrained polymer matrix) may be more relevant, as TCEP is likely used for its intumescent properties in automotive applications. Consumers with long commutes, truck drivers, and others who remain in the indoor vehicle cabin for longer durations may have greater potential for exposure than consumers with shorter exposure durations in vehicle cabins.

Due to limited reasonably available information regarding the use of articles containing dried paints and coatings, including those found on automotive articles and replacement parts, and the uncertainties surrounding the weight fraction, activity, and use patterns, and duration of use for consumers, EPA did not quantitatively assess the consumer use of articles containing dried paints and coatings including those used in automotive articles and replacement parts.

To qualitatively evaluate articles containing dried paints and coatings for consumers, EPA looked at the consumer analysis for other articles (*e.g.,* wood resin articles) containing TCEP. EPA's consumer analysis for articles containing TCEP, resulted in no chronic inhalation, ingestion, or dermal risk for adults for the COUs with moderate confidence. However, the consumer analysis did reveal chronic dermal risk for wood resin composites, and ingestion risk for multiple articles for infants and children.

Based on the similarity of the exposure scenarios, EPA expects articles containing dried paints and coatings, including those found on automotive articles and replacement parts containing TCEP, to mirror the other consumer use articles (*e.g.,* wood resin articles) scenarios assessed in Section 5.3.2.2.1. EPA expects dermal and ingestion risk to infants and children from the use of articles containing dried paints and coatings, including those found on automotive articles and replacement parts containing TCEP, due to its similarity to the other consumer article scenarios (*e.g.,* wood resin articles) with an overall confidence of slight.

Disposal of Wastewater, Liquid Wastes, and Solid Wastes

Consumers may be exposed to articles containing TCEP during disposal and the handling of waste. The removal of articles in DIY scenarios may lead to direct contact with articles and the dust generated from the articles. EPA believes that the monitoring data found for the commercial COU of e-waste recycling would represent similar exposures that could occur during the removal and/or disposal of other articles containing TCEP. Risk to workers was not found during these activities and therefore it is not expected that risk would be found in a DIY scenario involving the removal and/or disposal of TCEP containing articles.

5.3.2.3 Summary of Risk Estimates for the General Population

5.3.2.3.1 COUs with Quantitative Risk Estimates

EPA quantitatively assessed human exposures to TCEP concentrations via oral ingestion of drinking water, soil, and fish, dermal exposures to soil and surface water, and inhalation of ambient air. EPA assessed risk associated with each of these exposure scenarios by comparing doses to acute, short-term,

and chronic human equivalent concentrations and doses. Furthermore, EPA assessed the lifetime cancer risk from TCEP exposure via these routes. As noted previously, EPA uses a range of cancer benchmarks from 1 in 10,000 to 1 in 1,000,000 to characterize lifetime cancer risks for the general population.

Table 5-70 and Table 5-71 summarize the MOEs used to characterize acute non-cancer risks for oral exposures for the applicable COUs. Table 5-72 and Table 5-73 summarizes the chronic non-cancer MOE estimates for the applicable COUs. Table 5-74 summarizes the lifetime cancer oral risk for the applicable COUs. Oral ingestion non-cancer MOEs and cancer risks are presented for drinking water, diluted drinking water, landfill leachate to groundwater and subsequent migration to drinking water, incidental ingestion during swimming, fish ingestion, and soil ingestion for children playing with soil. Table 5-75 summarizes the acute and chronic non-cancer dermal MOEs for incidental dermal exposures during swimming and dermal ingestion of soils for children playing with soil associated with applicable COUs.

Table 5-76 presents the general population chronic inhalation MOEs used to characterize risk for the applicable COUs. Table 5-77 presents the general population lifetime cancer inhalation risk estimates for the applicable COUs. Inhalation MOEs and risk estimates are provided for various distances from a hypothetical facility for two meteorology conditions (Sioux Falls, South Dakota, for central tendency meteorology; and Lake Charles, Louisiana, for higher-end meteorology).

Ingestion

Drinking Water and Incidental Surface Water Ingestion: Table 5-70 summarizes the acute drinking water risk estimates for all COUs and lifestages. The non-cancer MOE values for the acute drinking water ingestion exposure by infants for four scenarios—Incorporation into paints and coatings (1-part coatings), Incorporation into paints and coatings (2-part coatings), Use in paints and coatings at job sites, and Formulation of TCEP containing reactive resin—are less than the benchmark MOE of 30. When factoring in dilution, none of the lifestages have acute drinking water MOE of less than the benchmark for any scenario.

Because TCEP is recalcitrant to drinking water treatment removal processes, a 0 percent drinking water treatment removal efficiency was used to calculate the oral drinking water exposure doses. The non-diluted acute risk estimates assume the general population was drinking water at the site of the facility outfall. To approximate a more typical drinking water concentration, distances between drinking water intake locations and facilities based on SIC codes were used to calculate a dilution factor to estimate a diluted drinking water concentration (see Section 5.1.3.4.1). All non-cancer MOEs from acute incidental ingestion via swimming were larger than the benchmark MOE of 30 for adults, youth, and children (see Appendix I).

None of the chronic MOEs from drinking water, diluted drinking water, incidental ingestion via swimming, and drinking water contamination from landfill leachate were lower than the benchmark MOE of 30. Drinking water MOEs are presented for both diluted and non-diluted surface water concentrations. The diluted drinking water MOEs represent typical case scenarios, whereas MOEs based on the non-diluted concentrations represent worst-case scenarios.

The DRAS model described in Section 3.3.3.8 estimated TCEP groundwater concentrations from landfill leachate. Only two industrial and commercial release scenarios have anticipated releases to landfill (Incorporation into paints and coatings – 1-part coatings and Processing into 2-part resin article). The DRAS model estimated groundwater concentrations by a range of loading rates and a range of leachate concentrations rather than the release estimate generated by the two industrial uses (21.5

kg/site-year for 1-part coatings, and 42.9 kg/site-year for 2-part resin article). Nevertheless, estimates via the full production volume did not result in chronic oral MOEs below 30 for drinking water.

Lifetime (from birth) oral ingestion cancer risk greater than 1 in 1,000,000 is associated with releases from four OESs: (1) Incorporation into paints and coatings – 1-part coatings; (2) Incorporation into paints and coatings – Resins/solvent-borne; (3) Use in paints and coatings at job sites; and (4) Processing into 2-part resin article. There was also oral ingestion cancer risk greater than 1 in 1,000,000 for the adult lifetime for the same scenarios, except for the use in paints and coatings at job sites. Under diluted drinking water conditions, no lifetime risks from birth or for the adult lifetimes exceeded 1 in 1,000,000.

Fish Ingestion: For the adult general population, acute exposure estimates via fish ingestion using a BAF of 2,198 L/kg showed MOEs less than 30 for all OESs except laboratory use of chemicals (Table 5-32). No OESs had an acute risk estimate less than 30 based on a BAF of 109 L/kg. For the adult subsistence fisher, EPA only had one fish IR that resulted in the same doses for both acute and chronic exposure. EPA estimated non-cancer MOEs by comparing that same dose with both the acute and chronic HEDs. Exposure estimates based on a BAF of 2,198 L/kg showed MOEs less than the acute benchmark for all OESs except laboratory use of chemicals. Using a BAF of 109, Laboratory use of chemicals and import and repackaging showed MOEs above the acute benchmark. For Tribes, the same approach was to estimate acute and chronic risks as the subsistence fisher. A BAF of 2,198 showed MOEs less than the acute benchmark for all OESs except Laboratory use of chemicals based on the current IR and for all OESs based on the heritage IR. A BAF of 109 showed MOEs less than the acute benchmark for all COUs except Import and repackaging and Laboratory use of chemicals based on the current mean IR (for the Suquamish Tribe). The BAF of 109 also had MOEs less than the acute benchmark for all COUs except Laboratory use of chemicals based on the heritage IR (for the Kootenai Tribe).

Chronic exposure for the general population resulted in MOEs less than the chronic benchmark of 30 for all OESs except Laboratory use of chemicals for both fish IRs and a BAF of 2,198/kg (Table 5-72). The table presents adult general population risk estimates based on only the 90th percentile IR even though two values were used, as discussed in Section 5.1.3.4.2. The MOEs based on the central tendency IR will be 4.4 times higher. When estimating exposure and risks based on a BAF of 109 L/kg, there are some differences in risks between the two IRs. The 90th percentile IR results in risks below their chronic benchmark for three OESs: (1) Incorporation into paints and coatings – 1-part coating; (2) Incorporation into paints and coatings – 2-part reactive coatings; and (3) Formulation of TCEP containing reactive resin. The central tendency IR did not result in any OESs with risk estimates below their chronic benchmark.

Chronic exposure for the subsistence fisher and Tribes resulted in MOEs less than 30 for all OESs based on a BAF of 2,198 L/kg and all IRs. A BAF of 109 L/kg showed risk estimates less than the chronic benchmark for all OESs except Laboratory use of chemicals.

Exposure estimates were not calculated for younger age groups. For younger age groups, acute and chronic MOEs less than benchmark values are reasonably expected because these age groups generally have higher fish ingestion rates per kilogram body weight (Table_Apx I-2). For Tribes, adults were reported to have the highest IR per kilogram of body weight (see Section 2265.1.3.4.4).

For the adult general population, subsistence fisher, and Tribe, cancer risk estimates are above 1 in 1,000,000 for all OESs and for both BAF values, as well as current and heritage IRs for Tribes. Table

5-75 shows the lifetime cancer risk estimates for fish ingestion. Cancer risk estimates were not calculated for fish ingestion among younger age groups. Similar to non-cancer risk, cancer risks for younger age groups are reasonably expected to be higher than older groups because of the higher fish ingestion rate per kilogram of body weight or because adults have the highest IR by body weight. (Table_Apx I-2).

The above risk estimates are based on the harmonic mean surface water flows representing the 50th percentile stream flows of all facilities in each industry sector. EPA also estimated exposure and risks using the 90th percentile stream flow of the harmonic mean. Acute and chronic non-cancer risk estimates are above their corresponding benchmarks for most of the COUs for all population groups using the 90th percentile stream flow. However, cancer risks estimates exceed 1 in 1,000,000 for almost all the COUs, population groups, and two BAFs. Results are presented in Table 5-75. These results indicate the critical role of receiving water flow as an input in determining TCEP concentrations in surface water and thus exposure via fish ingestion.

Soil Ingestion: Chronic oral non-cancer MOEs from soil were estimated for children 3 to 6 years of age based on soil concentrations that were calculated from air deposition for various distances from a hypothetical facility releasing TCEP (see Section 3.3.3.2). Oral doses were calculated for two exposure scenarios: (1) a child conducting activities with soil, and (2) a child playing in mud (see Section 5.1.3.4.4). No MOEs were less than the benchmark of 30 for the children's soil ingestion scenario for any of the COUs. In addition, there was no lifetime cancer risk for soil ingestion for any of the COUs.

Dermal

Incidental Dermal from Swimming: Non-cancer MOEs were not lower than benchmark values for the acute and chronic incidental dermal exposures swimming scenario for any of the COUs.

Children's Dermal Exposure from Playing in Soil: Dermal exposure estimates from soil were estimated for children 3 to 6 years of age because these ages are expected to play in mud and perform activities with soil. Soil concentrations were calculated via annual air deposition fluxes for various distances from a hypothetical facility releasing TCEP (see Section 3.3.3.2). Dermal exposure doses were also calculated for a child conducting activities with soil and a child playing in mud (see Section 5.1.3.3.2). No non-cancer MOEs for chronic exposures were less than the benchmark MOE of 30 at 100 or 1,000 m for either scenario of children playing in mud or children conducting activities with soil.

Many uncertainties are associated with the dermal exposure estimate used for the chronic dermal MOE that was less than the benchmark, including the lack of release information, site information, and reasonableness of the exposure scenario. The source of the exposure is a hypothetical facility that releases TCEP to the air for 2 days. Because no site information was available, EPA's release assessment estimated a 50th percentile of 27 sites to a 95th percentile of 203 sites per the OES for the commercial use of paints and coatings. To observe an MOE less than the benchmark, a child would have to be playing in mud at 100 m from the hypothetical facility. TCEP would deposit to the soil after deposition from air releases. Section 3.3.3.2 describes how EPA calculates soil concentrations from annual modeled air deposition. No U.S. studies recorded TCEP in soil. Modeled soil concentrations at 100 m (4.15×10^3 ng/g) were two orders of magnitude higher than the TCEP concentrations found in Germany (23.5 ng/g) (Mihajlovic and Fries, 2012). The study from Germany also indicated increased soil concentration of TCEP due to snow melt (see Section 3.3.3.1).

Inhalation

Table 5-78 shows the COUs where EPA found lifetime inhalation cancer risk estimates greater than 1 in 1,000,000 for the 2,500 lb production volume, high-end release estimate, suburban forest scenario and when using both central-tendency and high-end meteorological data. EPA found inhalation cancer risks greater than the benchmark for the 50th percentile air concentrations for the use of paints and coatings at job sites at distances as far as 100 m from the site. EPA also found cancer risk above this benchmark for the 95th percentile air concentrations for the use of paints and coatings out to 100 m from the job site.

Table 5-77 displays the chronic inhalation non-cancer risk estimates for the 2,500 lb production volume, high-end release estimate, suburban forest scenario, high-end meteorological data at 10 m from the facility. No non-cancer inhalation MOEs were less than the acute (total UF = 30) or chronic (total UF = 30) benchmark MOEs for any COUs. The lowest MOE for the chronic exposure scenario was 498 (the use of paints and coatings scenario, high meteorological station data, at 10 m, 95th percentile). The lowest MOE for the acute exposure scenario was 295,000 for the Processing into 2-part resin article, high meteorological station data, at 10 m, 95th percentile scenario (not shown). Ambient air is a minor environmental compartment as described in Section 2.2.

It is unlikely that individual residences will be within 10 m of the stack or fugitive air release from these facilities. However, these estimates suggest that fence line communities living within 100 m downwind of facilities that use TCEP in paints and coatings at job sites may be at an increased risk of developing cancer over their lifetimes.

When compared to the monitoring literature, the maximum modeled ambient air concentrations 2.55 ng/m^3, are within an order of magnitude of the ambient concentrations described in Bradman et al. (2014) that recorded a maximum concentration of 1.60 ng/m^3, mean of 0.72 ng/m^3, at 14 early childhood education facilities in California between May 2010 and May 2011. For deposition, Moran et al. (2023) reports 1.36×10^{-6} g/m^2/day which equates to 4.74×10^{-4} g/m^2 a year. These modeled deposition values are one to two orders of magnitude higher than the high-end deposition values observed 1,000 m from the hypothetical releasing facility for the Use of Paints and Coatings – Spray Application OES, 2,500 lb Production Volume, 95th Percentile Release Estimate, Suburban Forest Land Category Scenario (9.51×10^{-3} to 1.47×10^{-2} g/m^2 a year).

Table 5-70. General Population Acute Drinking Water (Oral Ingestion) Non-cancer Risk Summary

COU		OES	Acute Oral Non-cancer MOEs *UFs = 30*											
			Drinking Water						Drinking Water (Diluted)					
Life Cycle Stage/Category	**Subcategory**		**Adult (≥21 yr)**	**Infant (<1 yr)**	**Youth (16–20 yr)**	**Youth (11–15 yr)**	**Child (6–10 yr)**	**Toddler (1–5 yr)**	**Adult (≥21 yr)**	**Infant (<1 yr)**	**Youth (16–20 yr)**	**Youth (11–15 yr)**	**Child (6–10 yr)**	**Toddler (1–5 yr)**
Manufacturing/ Import	Import	Repackaging	172	49	224	223	175	138	2.12E05	6.05E04	2.76E05	2.76E05	2.16E05	1.70E05
Processing/ Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	40	11	52	52	40	32	6.38E04	1.82E04	8.30E04	8.28E04	6.49E04	5.11E04
		Incorporation into paints and coatings – 2-part reactive coatings	44	13	57	57	45	35	7.03E04	2.00E04	9.15E04	9.13E04	7.15E04	5.64E04
	Polymers used in aerospace equipment and products	Formulation of TCEP containing 2-part reactive resin	38	11	49	49	38	30	1.63E04	4.64E03	2.12E04	2.11E04	1.66E04	1.30E04
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	4,292	1,223	5,586	5,571	4,366	3,440	5.30E06	1.51E06	6.89E06	6.87E06	5.39E06	4.24E06
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	73	21	95	95	74	59	9.02E04	2.57E04	1.17E05	1.17E05	9.17E04	7.23E04

Table 5-71. Acute Fish Ingestion Non-cancer Risk Summary Based on 50th Percentile Flow of Harmonic Mean

COU		OES	Acute Oral Non-cancer MOEs *UFs = 30*							
			General Population		Subsistence Fishers		Tribes (Current IR)[a]		Tribes (Heritage IR)[b]	
Life Cycle Stage/ Category	Subcategory		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
Manufacturing/ Import	Import	Repackaging	18	363	3	57	2	37	<1	5
Processing/Processing –Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	4	82	1	13	<1	8	<1	1
		Incorporation into paints and coatings – 2-part reactive coatings	4	90	1	14	<1	9	<1	1
	Polymers used in aerospace equipment and products	Formulation of TCEP-containing reactive resin	3	65	<1	10	<1	7	<1	1
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	450	9,067	70	1,411	46	930	6	122
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	8	154	1	24	1	16	<1	2

[a] Current fish consumption rate at 216 g/day based on survey of Suquamish Indian Tribe in Washington (see Section 5.1.3.4.4).
[b] Heritage fish consumption rate at 1,646 g/day based on study of Kootenai Tribe in Idaho (see Section 5.1.3.4.4).

Table 5-72. General Population Chronic Water and Soil Ingestion Non-cancer Risk Summary

COU		OES	Chronic Non-cancer Oral MOEs *UFs = 30*							
Life Cycle Stage/Category	**Subcategory**		**Drinking Water (Diluted)**	**Drinking Water**	**Drinking Water (via Leaching to Groundwater)**	**Ambient Water (Incidental Ingestion)**	**Soil Intake (50th) at 100 m**	**Soil Intake (95th) at 100 m**	**Soil Intake (50th) at 1,000 m**	**Soil Intake (95th) at 1,000 m**
Manufacturing/ Import	Import	Repackaging	1.64E08	1.05E05	N/A	2.11E05	2.20E10	5.15E09	1.73E12	4.03E11
Processing/ Processing – Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	4.40E07	23,728	2.12E06	4.89E04	7.02E08	1.64E08	7.95E10	1.86E10
		Incorporation into paints and coatings – 2-part reactive coatings	4.85E07	26,171	N/A	5.39E04	4.85E09	1.13E09	3.68E11	8.59E10
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	9.89E06	18,706	N/A	4.62E04	4.41E09	1.03E09	3.46E11	8.07E10
Processing/ Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	N/A	N/A	2.12E06	N/A	5.15E08	1.20E08	5.05E10	1.18E10
Commercial use//Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	4.10E09	2.60E06	N/A	5.30E06	4.60E08	1.07E08	4.20E10	9.81E09
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	6.96E07	4.47E04	N/A	8.98E04	2.98E05	6.96E04	5.72E07	1.34E07

Table 5-73. Chronic Fish Ingestion Non-cancer Risk Summary

COU		OES	General Population				Subsistence Fishers[b]		Tribes (Current)[c]		Tribes (Heritage)[d]	
Life Cycle Stage/Category	Subcategory		BAF 2,198[a]		BAF 109[a]		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
			CT[e]	HE	CT[e]	HE						
Manufacturing/ Import	Import	Repackaging	23	5	461	105	1	16	1	11	<1	1
Processing/Processing – Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	5	1	104	24	<1	4	<1	2	<1	<1
		Incorporation into paints and coatings – 2-part reactive coatings	6	1	115	26	<1	4	<1	3	<1	<1
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	4	1	82	19	<1	3	<1	2	<1	<1
Processing/Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	571	130	11,505	2,617	20	407	13	268	2	35
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	10	2	196	45	<1	7	<1	5	<1	1

[a] General population exposure estimates based on general population fish ingestion rate of 22.2 g/day.
[b] Subsistence fishers exposure estimates based on subsistence fisher ingestion rate of 142.2 g/day.
[c] Current fish consumption rate at 216 g/day based on survey of Suquamish Indian Tribe in Washington (see Section 5.1.3.4.4).
[d] Heritage fish consumption rate at 1,646 g/day based on study of Kootenai Tribe in Idaho (see Section 5.1.3.4.4).
[e] Exposure estimates based on a general population mean fish ingestion rate of 5.04 g/day.

Table 5-74. General Population Lifetime Cancer Oral Ingestion Risk Summary Table

COU		OES	Lifetime Cancer Oral Risk Estimates			
			Drinking Water		Drinking Water (Diluted)	
Life Cycle Stage/Category	Subcategory		Lifetime from Birth	Adult Lifetime	Lifetime from Birth	Adult Lifetime
Manufacturing/Import	Import	Repackaging	6.09E–07	6.37E–07	3.91E–10	4.09E–10
Processing/Processing – Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	2.70E–06	2.82E–06	1.45E–09	1.52E–09
		Incorporation into paints and coatings – 2–part reactive coatings	2.44E–06	2.56E–06	1.32E–09	1.38E–09
Processing/Processing –Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1.43E–06	1.50E–06	9.20E–10	9.63E–10
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	5.44E–08	5.69E–08	3.49E–11	3.65E–11
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	3.42E–06	3.58E–06	6.47E–09	6.77E–09

Table 5-75. Lifetime Cancer Risk Summary for Fish Consumption

COU		OES	Lifetime Cancer Oral Risk Estimates									
Life Cycle Stage/ Category	Subcategory		Adult Fish Ingestion General Population[a]				Adult Subsistence Fisher		Tribes (Current IR)		Tribes (Heritage IR)	
			BAF 2,198		BAF 109		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
			CT[b]	HE	CT[b]	HE						
Manufacturing/ Import	Import	Repackaging	2.32E–03	1.02E–02	1.15E–04	5.07E−04	6.57E−02	3.26E–03	9.96E–02	4.94E–03	7.59E–01	3.77E–02
Processing/ Processing – Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	1.03E–02	4.53E−02	5.11E–04	2.25E−03	2.91E–01	1.44E–02	4.41E–01	2.19E–02	3.36	1.67E–01
		Incorporation into paints and coatings – 2-part reactive coatings	9.34E–03	4.11E−02	4.63E–04	2.04E−03	2.64E–01	1.31E–02	4.00E–01	1.98E–02	3.05	1.51E–01
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1.31E–02	5.74E−02	6.48E−04	2.85E−03	3.69E–01	1.83E−02	5.60E–01	2.78E−02	4.27	2.12E–01
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	9.32E−05	4.10E−04	4.62E−06	2.03E−05	2.63E−03	1.31E−04	3.99E−03	1.98E−04	3.04E−02	1.51E−03
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	5.47E−03	2.41E−02	2.71E−04	1.19E−03	1.55E−01	7.67E−03	2.35E−01	1.16E−02	1.79	8.87E−02

[a] Cancer risk estimates for the adult general population are based on the high-end fish ingestion rate of 22.2 g/day.
[b] Exposure estimates are based on a general population mean fish ingestion rate of 5.04 g/day.

Table 5-76. General Population Dermal Acute and Chronic Non-cancer Risk Summary

COU		OES	Acute MOEs *UFs = 30*	Chronic Non-cancer MOE[a] *UFs = 30*				
Life Cycle Stage/Category	**Subcategory**		**Surface Water (Adult Swimming)**	**Surface Water (Adult Swimming)**	**Child Playing in Mud at 100 m[a]**	**Child Activities with Soil at 100 m[a]**	**Child Playing in Mud at 1,000 m[a]**	**Child Activities with Soil at 1,000 m[a]**
Manufacturing/Import	Import	Repackaging	6.82E03	4.55E05	6.95E06	1.43E09	5.44E08	1.12E11
Processing/Processing –Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	1.54E03	1.05E05	2.21E05	4.55E07	2.51E07	5.15E09
		Incorporation into paints and coatings – 2-part reactive coatings	1.70E03	1.14E05	1.53E06	3.14E08	1.16E08	2.39E10
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1.21E03	9.75E04	1.39E06	2.86E08	1.09E08	2.24E10
Processing/Processing –Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	N/A	N/A	1.62E05	3.34E07	1.59E07	3.27E09
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	1.70E05	1.13E07	1.45E05	2.98E07	1.33E07	2.72E09
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	2.90E03	1.95E05	9.4E01	1.93E04	1.80E04	3.71E06

[a] Soil concentration based of annual air deposition fluxes is used to estimate the acute exposures scenario of a child playing with mud and conducting activities in soil.

Table 5-77. Inhalation Chronic Risk Summary for General Population[a]

COU		OES	Chronic Inhalation MOEs *UFs = 30*	
Life Cycle Stage/Category	**Subcategory**		**Ambient Air 50th**	**Ambient Air 95th**
Manufacturing/Import	Import	Repackaging	9.34E07	5.10E07
Processing/Processing – Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	3.66E06	1.49E06
		Incorporation into paints and coatings – 2-part reactive coatings	2.22E07	7.18E06
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1.98E07	6.41E06
Processing/Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	2.41E06	1.82E06
Commercial use/Laboratory chemicals	Laboratory chemical	Use of laboratory chemicals	2.10E06	1.48E06
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use of paints and coatings at job sites	1.23E03	4.98E02

[a] 2,500 lb Production Volume – High-End Release Estimate, Suburban Forest Scenario at 10 m, high meteorology conditions

Table 5-78. General Population Lifetime Cancer Inhalation Risk Summary Table[a]

COU		OES	Distances (m)	Lifetime Cancer Inhalation Risk			
				Central Tendency Meteorological Data		High-End Meteorological Data	
Life Cycle Stage/Category	Subcategory			Cancer Risk Estimate for 50th Percentile Air Concentration	Cancer Risk Estimate for 95th Percentile Air Concentration	Cancer Risk Estimate for 50th Percentile Air Concentration	Cancer Risk Estimate for 95th Percentile Air Concentration
Commercial and Industrial use/Paints and coatings	Paints and coatings	Use in paints and coatings at job sites	10	4.55E–05	5.53E–05	5.42E–05	1.33E–04
			30	1.49E–05	2.50E–05	1.50E–05	3.74E–05
			30–60	6.74E–06	1.70E–05	7.16E–06	2.29E–05
			60	4.63E–06	1.07E–05	4.76E–06	1.20E–05
			100	1.66E–06	4.66E–06	1.69E–06	4.46E–06

[a] 2,500 lb Production Volume – High-End Release Estimate, Suburban Forest Scenario

5.3.2.3.2 COUs Without Quantitative Risk Estimates

Distribution in Commerce

Distribution in commerce includes transporting TCEP or TCEP-containing products between work sites or to final use sites, as well as loading and unloading from transport vehicles. The general population may be in the proximity of vehicles that transport TCEP or TCEP-containing products.

TCEP production volumes have declined and recent reports (*e.g.*, the 2020 CDR) indicate that production volumes may be below reporting levels; therefore, the precise volume is unknown. The general decline in production volume would logically lead to decreased distribution into commerce. Therefore, exposure and risk would also likely have declined with time. Exposure is possible from ongoing manufacturing, processing, industrial, and commercial uses. However, EPA lacks the data to assess the full set of risks to the general population from this COU. Due to this limited data, EPA's confidence in these exposures is indeterminate. Nevertheless, given that TCEP and/or TCEP containing products or articles are expected to be transported in sealed containers or packages; EPA anticipates that exposure and releases during Distribution in Commerce will be negligible.

Processing – Recycling

EPA did not quantify risks to the general population from releases during recycling of either e-waste or recycled foam products due to limited reasonably available information and limited use of TCEP in electronics.

EPA did not find data to quantify releases of TCEP from e-waste recycling facilities. The total releases are expected to be low for several reasons: The volume of TCEP in e-waste products is low; only a fraction of the products is recycled; and recycling will likely be dispersed over many e-waste sites. Although EPA located information on the presence of TCEP at e-waste recycling facilities during systematic review, the data sources did not provide the volume of TCEP-contained electronics processed at any of the facilities identified. Therefore, EPA's confidence in these exposures is indeterminant and cannot quantify risk from e-waste recycling. However, EPA acknowledges that while some releases and exposures could occur during the disposal of the wide variety of items that TCEP has found its way into, these are expected to be minimal and dispersed, and exposures are expected to be negligible.

TCEP may be present within flexible foam, fabric, textile, and other applications that have been made from recycled foam scraps generated during trimming of original TCEP-containing manufactured foam products. EPA was not able to determine, with reasonable accuracy, the exact flame retardants that are used in these products and did not locate information on releases during recycling of such foam.

Industrial and Commercial Use (Other) – Aerospace Equipment and Products and Automotive Articles and Replacement Parts Containing TCEP

EPA does not expect significant releases to the environment for the following COUs:

- Industrial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP; OES: installing article (containing 2-part resin); and
- Commercial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP; OES: installing article (containing 2-part resin).

After TCEP-containing resins have cured within products that are installed, EPA expects TCEP releases and dermal exposures will be limited by TCEP being entrained into the hardened polymer matrix. During installation it is possible that very small levels of dust could be generated, these were quantified in Table 5-67. and do not indicate risk to workers from inhalation nor do they indicate the generation of

significant dust releases occurring. Releases may occur via the mechanism of blooming (volatilization from the cured resin surface) during the service life of the article, but EPA expects that such releases during installation will be negligible (OECD, 2009; NICNAS, 2001). Installation of aerospace equipment and products would be installed without any type of further processing of the article that would lead to potential releases (sanding, drilling, etc.). Therefore, the potential risk to workers and the general population from releases during installation of TCEP-containing articles is low.

Commercial Uses that TCEP Is No Longer Actively Incorporated Into

EPA determined that the following commercial use COUs for TCEP consists of use of existing products and end of service life disposal:

- Commercial use – Furnishing, cleaning, treatment/care products – Fabric and textile products;
- Commercial use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products;
- Commercial use – Construction, paint, electrical, and metal products – building/construction materials – Insulation; and
- Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites.

These COUs were being phased out beginning in the late 1980s or early 1990s and replaced by other flame retardants or flame-retardant formulations. EPA did not locate data to estimate (1) the amount of TCEP that was historically used in these products, (2) the amounts of these products that have already reached the end of their service life, or (3) the amounts of these products that have already been disposed. Based on the years that the phase-out occurred, many of these products not likely to be in use because the end of their service life was already reached (*e.g.*, commercial roofing has an estimated lifespan of 17–20 years). EPA assumes that any of these products still used commercially represent a fraction of the overall amount of TCEP previously used for these purposes. Therefore, releases to the environment from these commercial uses would also represent only a fraction of previous release amounts.

The consumer assessment for these articles resulted in no consumer risk for inhalation, ingestion dermal risk for adults for the COUs with moderate confidence. Therefore, under similar exposure durations and exposure frequencies, EPA expects these articles to pose no commercial risk for inhalation, ingestion, and dermal risk to COUs to the general population who use articles in a similar fashion to consumers. Furthermore, exposure to the releases from the disposals of these articles are anticipated to be less than the consumer use of these articles in the home.

Disposal

Disposal is possible throughout the life cycle of TCEP and TCEP-containing products, including waste treatment and disposal resulting from manufacturing, processing, commercial and consumer uses.

For processing COUs, EPA estimated releases to landfills or incinerators (see Section 5.3.2.1) for

- Incorporation into formulation, mixture, or reaction product – Paint/coating manufacture – 1-part coating OES (landfill);
- Incorporation into articles – Aerospace equipment and products and automotive articles and replacement parts containing TCEP – Processing in two-part resin article OES (landfill);
- Incorporation into formulation, mixture, or reaction product – Paint/coating manufacture – 2-part coating OES (incineration); and

- Incorporation into formulation, mixture, or reaction product – Polymers in aerospace equipment and products – Formulation of reactive resins OES (incineration).

Both releases to landfills and incinerators rely on inputs provided by ESDs or GSs, but the ESDs and GSs do not specify the proportion of the throughput that goes to either of these two disposal practices. Therefore, EPA was unable to further quantify environmental releases related to these two disposal processes. For three of these processing COUs, EPA was able to perform quantitative risk characterization for releases to surface water (which includes on-site wastewater treatment or discharge to POTWs, where applicable) (see Table 3-2); any releases to on-site waste treatment or POTWs were combined with other exposures and this combined risk to the general population was quantified for these processing COUs.

Waste treatment (POTW or on-site) or disposal (landfill or incineration) is expected to be negligible for industrial and commercial uses related to installing articles. For the COUs of Manufacturing/repackaging, Commercial use of paints and coatings, and Commercial use of laboratory chemicals, Disposal to landfills or incinerators is not expected but EPA estimated surface water releases that could include release to wastewater treatment or POTWs and any resulting risks to the general population were assessed for the individual COUs.

For the commercial uses that have been phased out, any currently used products that contain TCEP are expected to be disposed in landfills but will represent just a fraction of previous amounts when TCEP was used more widely. Landfills would likely contain TCEP in commercial articles from these COUs, but data are lacking with which to estimate exposure and risk from disposal or waste treatment activities for these COUs, and EPA has not quantified such risks. For e-waste recycling, there is also too little information to estimate exposure from disposal and only a small portion of e-waste is expected to contain TCEP.

The DRAS model described in Section 3.3.3.8 estimated TCEP groundwater concentrations from landfill leachate. Only two industrial and commercial release scenarios have anticipated releases to landfill (Incorporation into paints and coatings – 1-part coatings and Processing into 2-part resin article). The DRAS model estimated groundwater concentrations by a range of loading rates and a range of leachate concentrations rather than the release estimate generated by the two industrial uses (21.5 kg/site-year for 1-part coatings, and 42.9 kg/site-year for 2-part resin article). Nevertheless, estimates via the full production volume did not result in chronic oral MOEs below 30 for drinking water.

While this analysis estimates potential exposures to the general population via disposal to landfills, this approach does not capture all the types of disposals that may occur throughout the manufacturing, processing, distribution, import, industrial use, commercial use and consumer use life cycle of TCEP containing products and articles. There may be releases to the environment from consumer articles containing TCEP via end-of-life disposal and demolition of consumer articles in the built environment, and the associated down-the-drain release of TCEP from domestic laundry that removes TCEP containing dust from clothing to wastewater. It is difficult for EPA to quantify these end-of-life and down-the-drain laundry exposures due to limited reasonably available information on source attribution of the consumer COUs.

EPA's confidence in these general population exposures from the disposal scenario is *indeterminant*. The Agency acknowledges that while some releases and exposures could occur during the disposal of the wide variety of items that TCEP has been incorporated into, these exposures are expected to be negligible.

5.3.2.4 Summary of Risk Estimates for Infants from Human Milk

EPA estimated infant risks from milk ingestion based on TCEP concentrations in milk modeled for maternal exposures associated with consumer, occupational, and general population groups. Infant exposures through milk were estimated for both mean (105 mL/kg-day) and upper (153 mL/kg-day) milk intake rates. Risk estimates for intermediate and chronic infant exposures through milk were calculated for both cancer and non-cancer endpoints for each COU within each maternal group. Short-term risks, which have an averaging time of 30 days or less, were estimated based on the infant's first month of life. The first month of life generally had the highest doses because of the highest milk ingestion rate per kilogram of body weight; thus, it is most protective for estimating intermediate risks. For chronic non-cancer risks, exposure typically occurs over at least 10 percent of lifetime in adults. However, it cannot be ruled out that continuous exposure during the first year of life will result in permanent health effects through adulthood. Chronic risks were thus considered for infant doses in the first year of life. Similarly, cancer risks were also estimated using the linear low-dose extrapolation even though exposure did not occur over the lifetime.

Acute infant doses were not estimated because the Verner Model is designed to estimate TCEP concentrations in milk and doses from continuous exposure rather than an acute, 1-day dose. However, if short-term or chronic doses result in risk estimates below their corresponding benchmark MOEs, EPA estimated acute risks by comparing short-term and chronic doses with an acute POD. Appendix I.5.1 through Appendix I.5.5 presents risk estimates for all iterations that EPA considered.

For the consumer exposure pathways, intermediate and chronic infant risk estimates were above the corresponding benchmark MOEs for all COUs. Infant cancer risk estimates are above 1 in 1,000,000 for two consumer exposure scenarios regardless of milk intake rate: (1) Building/construction materials not covered elsewhere (roofing insulation), and (2) Building/construction materials – Wood and engineered wood products (wood flooring). The infant cancer risk estimates for these two COUs range from 7.04×10^{-6} to 1.03×10^{-5}. The maternal cancer risk estimates for the same COUs range from 8.11×10^{-6} to 4.5×10^{-2} (Table 5-69). Although the lower bound of the cancer risk estimates for the mother and infant are similar, it is important to note that maternal risks are calculated by separate exposure routes (*i.e.,* oral, dermal, and inhalation). Dermal exposure to roofing insulation resulted in the lowest maternal cancer risk estimates, and all other routes resulted in risk estimates that were two to four magnitudes higher. Other COUs with cancer risk estimates above 1 in 1,000,000 for the mother were below this level for the infant ingesting human milk. Therefore, infant risks are not proportionally higher than maternal risks. Furthermore, the maternal risk estimates in Table 5-69 are based on doses for an adult weighing 80 kg. If they were adjusted for women of reproductive age, the risk estimates for this population will increase given the higher dose. This underscores the conclusion that minimizing maternal exposure to TCEP is most important for protecting an infant, as the mother is more sensitive.

For the occupational exposure pathways, 1- and 2-day application of spray paints and coatings were not evaluated because the Verner model is intended to estimate only continuous maternal exposure. Among the evaluated OESs, short-term and chronic infant risk estimates were below their benchmark MOEs for Commercial use – Paints and coatings – Spray (2-part coatings, 250-day application) regardless of the maternal dose type (chronic or subchronic) and milk intake rate (mean or upper). For Laboratory chemicals, a mean milk intake rate resulted in short-term risk estimates below their benchmark MOEs based on a subchronic maternal dose. An upper milk intake rate for the same OES resulted in short-term and chronic infant risk estimates below their benchmark MOEs regardless of the maternal dose type. Lastly, for Incorporation into paints and coatings – 1-part coatings, a mean milk intake rate resulted in short-term risk estimates below their benchmark MOEs based on a subchronic maternal dose. An upper milk intake rate and subchronic maternal dose for the same OES resulted in short-term and chronic

infant risk estimates below the benchmark MOE. However, acute infant risk estimates were above the MOE for all of the above OESs.

Cancer risk estimates vary depending on the maternal worker dose type and the milk intake rate. For subchronic maternal doses, infant cancer risk estimates exceeded 1 in 1,000,000 for 8 out of the 10 OESs regardless of milk intake rate:

- Import and repackaging;
- Incorporation into paints and coatings – 1-part coatings;
- Incorporation into paints and coatings – 2-part reactive coatings;
- Processing – Formulation of TCEP into 2-part reactive resins;
- Processing – Processing into 2-part resin article;
- Commercial use – Paints and coatings – Spray (1-part, 250-day application);
- Commercial use – Paints and coatings – Spray (2-part reactive coatings, 250-day application); and
- Laboratory chemicals.

For the above OESs, infant cancer risk estimates ranged from 2.67×10^{-6} to 6.06×10^{-5}. The OES that showed short-term and chronic infant risks also showed the highest infant cancer risk estimates: commercial use – paints and coatings – spray (2-part coatings, 250-day application). For this OES, infant cancer risk estimates based on a mean and upper milk intake rate were 3.61×10^{-5} and 6.06×10^{-5}, respectively.

For chronic maternal doses, infant cancer risk estimates exceeded 1 in 1,000,000 for five or seven OESs, depending on the milk intake rate:

- Import and repackaging *(only for upper milk intake rate)*;
- Incorporation into paints and coatings – 1-part coatings;
- Processing – Formulation of TCEP into 2-part reactive resins *(only for upper milk intake rate)*;
- Processing – Processing into 2-part resin article;
- Commercial use – Paints and coatings – Spray (1-part coatings, 250-day application);
- Commercial use – Paints and coatings – Spray (2-part reactive coatings, 250-day application); and
- Laboratory chemicals.

For the above OESs, infant cancer risk estimates ranged from 1.06×10^{-6} to 4.91×10^{-5}. Again, Commercial use – Paints and coatings – Spray (2-part coatings, 250-day application) had the highest infant cancer risk estimate at 3.37×10^{-5} and 4.91×10^{-5} for a mean and upper milk intake rate, respectively. Overall, for occupational exposure pathways, the risk estimates for short-term, chronic, and cancer effects are lower in the infants compared to the mothers.

EPA estimated risks to infants in Tribal communities where the mothers were exposed to TCEP through fish ingestion. As discussed in Section 5.1.3.4.4, a current mean ingestion rate (IR) and heritage IR was used. The milk intake rate (mean vs. upper) did not significantly change risk estimates. For the high BAF, both milk intake rates and both fish IRs resulted in MOEs below the short-term and chronic benchmarks for all OESs except Laboratory use of chemicals. All OESs had cancer risk estimates above 1 in 1,000,000. The low BAF and current IR did not show any MOEs below the short-term and chronic benchmarks for all OESs and both milk intake rates. However, cancer risks exceeded 1 in 1,000,000 for all OESs except Laboratory use of chemicals. The low BAF, heritage IR, and mean milk intake rate resulted in risk estimates below the short-term and chronic benchmarks for the same three OESs, plus

one OES with only short-term risk estimates below the benchmark. Cancer risks exceeded 1 in 1,000,000 for the low BAF, heritage IR, and mean milk intake rate for all OESs except Laboratory use of chemicals. Lastly, the OESs that had MOEs below the short-term and chronic benchmarks were also compared against the acute benchmark to determine if there are acute risks at that exposure level. A high BAF had MOEs below the acute benchmark (3–5 OESs depending on the fish and milk IR type). A low BAF had no risk estimates below the acute benchmark except for one scenario.

For the general population, EPA focused on maternal oral exposures because they resulted in significantly higher doses than dermal or inhalation. Within the oral routes, ingestion of fish (at the general population's 90th percentile IR of 22.2 g/day) and undiluted drinking water were among the sentinel pathways for mothers. EPA estimated infant risks using these pathways and did not combine across other routes. Using a low BAF, no OESs had short-term or chronic risk estimates below the MOE based on the mean and upper milk uptake rate. Cancer risk estimates did not exceed 1 in 1,000,000 for any of the OESs based on the mean intake rate. However, based on the upper milk intake rate, the cancer risk estimate for Formulation of TCEP containing reactive resin did exceed 1 in 1,000,000 (1.21×10^{-6}).

For the general population adult fish ingestion based on the high BAF, no OESs had risk estimates below their short-term and chronic MOEs for both milk intake rates. Cancer risk estimates exceeded 1 in 1,000,000 for all OESs except Laboratory use of chemicals. Under the mean milk intake rate, cancer risk estimates ranged from 2.96×10^{-6} to 1.66×10^{-5}. Under the upper milk intake rate, cancer risk estimates ranged from 4.32×10^{-6} to 2.43×10^{-5}. The OES with the highest cancer risk estimate is Formulation of TCEP containing reactive resin. Risk estimates for infants of subsistence fisher were not calculated but are expected to fall in between those for the adult general population and Tribal population.

Due to the uncertainties in estimating fish ingestion exposure as discussed in Section 5.3.2.3, EPA also considered ingestion of undiluted drinking water. This pathway did not result in any non-cancer risk estimates below the benchmark MOE or cancer risk estimates above 1 in 1,000,000. No maternal risks were observed either. While it is possible that combining other exposure routes, such as dermal absorption from swimming, can result in additional scenarios showing infant risk estimates below their benchmark MOEs, results from consumer, occupational, and general population fish ingestion demonstrated that the mothers are more sensitive than the infants. There are no COUs or OESs across all maternal groups that showed higher risk estimates in the infants compared to the mothers. In fact, some COUs resulted in maternal doses and risk estimates that are several magnitudes higher for the mothers than the infants. Therefore, protecting the mother will also protect the infant from exposure via human milk.

5.3.3 Risk Characterization for PESS

EPA considered PESS throughout the exposure assessment and throughout the hazard identification and dose-response analysis. The Agency has identified several PESS factors that may contribute to a group having increased exposure or biological susceptibility. Examples of these factors include lifestage, occupational and certain consumer exposures, nutrition, and lifestyle activities.

For the TCEP risk evaluation, EPA accounted for the following PESS groups: infants exposed through human milk from exposed individuals, children and male adolescents who use consumer articles or are among the exposed general population, subsistence fishers, Tribal populations, pregnant women, workers and consumers who experience aggregated or sentinel exposures, people who live in fenceline communities near facilities that emit TCEP, and firefighters.

The non-cancer hazard values used in the risk evaluation are based on pregnant rats that may be more susceptible to neurotoxic effects and male reproductive endpoints.

Table 5-79 summarizes how PESS were incorporated into the risk evaluation and also summarizes the remaining sources of uncertainty related to consideration of PESS. Appendix D provides additional details on PESS considerations for the TCEP risk evaluation.

Table 5-79. Summary of PESS Considerations Incorporated into the Risk Evaluation

PESS Categories	Potentially Exposed Individuals		Susceptible Subpopulations	
	Potential Increased Exposures Incorporated into Exposure Assessment	**Sources of Uncertainty for Exposure Assessment**	**Potential Sources of Biological Susceptibility Incorporated into Hazard Assessment**	**Sources of Uncertainty for Hazard Assessment**
Lifestage	• Lifestage-specific exposure scenarios included infants exposed through human milk. • Exposure factors by age group were applied to calculate consumer oral and dermal exposures. • Children scenarios of playing in mud and activities with soil considered for dermal and oral soil ingestion. • Mouthing of consumer articles considered for infants and children.	• The level of exposure via milk is uncertain as described in Section 5.1.3.7.2. • Uncertainties regarding the appropriateness for adjusting inhalation values to younger lifestages for the consumer analysis.	• There is potential susceptibility related to different lifestages using adolescent male mice as the POD for intermediate and chronic exposure. Potential differences in other lifestages, such as older individuals, which might relate toxicokinetic or toxicodynamic differences was addressed through a 10× UF for human variability (see Section 5.2.7 for POD and UFs). • The intermediate/chronic POD is expected to be protective of adolescent, developmental, and adult outcomes (including pregnant females) based on comparison with existing developmental and reproductive studies and a 2-year bioassay for TCEP. Pregnant females are the basis of the acute POD. • Human data lend slight evidence of possible developmental effects in infants and children.	• The magnitude of differences in toxicokinetics and toxicodynamics for some individuals may be greater than accounted for by the UF_H of 10. • Inability to use some reproductive/developmental data due to errors in one study results in uncertainty regarding the magnitude of some effects in offspring. • Some uncertainty exists based on limited number of studies and differences in specific outcomes among studies.
Pre-existing Disease	• EPA did not identify pre-existing disease factors influencing exposure.		• Pre-existing diseases and conditions, especially those that lead to neurological and behavioral effects, reproductive effects, and cancer may increase susceptibility to the effects of TCEP. • This greater susceptibility is addressed through the 10× UF for human variability.	• The increase in susceptibility is not known and is a source of uncertainty; differences may be greater than the UF_H of 10.

PESS Categories	Potentially Exposed Individuals		Susceptible Subpopulations	
	Potential Increased Exposures Incorporated into Exposure Assessment	**Sources of Uncertainty for Exposure Assessment**	**Potential Sources of Biological Susceptibility Incorporated into Hazard Assessment**	**Sources of Uncertainty for Hazard Assessment**
Lifestyle Activities	• EPA evaluated exposures resulting from subsistence fishing and considered increased intake of fish in these populations, as well as Tribal populations.	• There is a high level of uncertainty in the BAF values because of limited monitoring data. There is also uncertainty in the modeled surface water concentrations.	• EPA did not identify lifestyle factors that specifically influence susceptibility to TCEP and that could be quantified. Generally, certain factors (*e.g.*, smoking, alcohol consumption, diet) can affect health outcomes.	• This is a remaining source of uncertainty.
Occupational and Consumer Exposures	• Monitoring data suggest that firefighters have elevated TCEP exposures because of firefighting activities (indicated by elevated urine concentrations of BCEP, a metabolite of TCEP (Mayer et al., 2021; Jayatilaka et al., 2017)). • Consumer articles intended for use by children (children's play structures, toy foam blocks) considered in the assessment of COUs.	• Uncertainties in duration of use of consumer articles in the home.	• EPA did not identify occupational and consumer exposures that influence susceptibility.	• This is a remaining source of uncertainty.
Socio-demographic	• EPA did not evaluate exposure differences between racial groups.	• Monitoring literature indicates TCEP levels in dust are significantly associated with the presence of extremely worn carpets. This may be relevant for lower socioeconomic status families (Castorina et al., 2017).	• Slight evidence that sociodemographic factors may influence susceptibility to TCEP (decreased IQ).	• There is still some uncertainty given the limited, slight evidence.
Nutrition	• EPA did not identify nutritional factors influencing exposure.		• Nutrition can affect susceptibility to disease generally. EPA did not identify specific evidence that nutritional factors influence susceptibility to TCEP.	• This is a remaining source of uncertainty.

PESS Categories	Potentially Exposed Individuals		Susceptible Subpopulations	
	Potential Increased Exposures Incorporated into Exposure Assessment	**Sources of Uncertainty for Exposure Assessment**	**Potential Sources of Biological Susceptibility Incorporated into Hazard Assessment**	**Sources of Uncertainty for Hazard Assessment**
Genetics/ Epigenetics	• EPA did not identify genetic or epigenetic factors influencing exposure.		• Genetic disorders may increase susceptibility to male reproductive effects; this was addressed through a 10× UF for human variability (see Section 5.2.5.1.2).	• The magnitude of the impact of genetic disorders is unknown and is a source of uncertainty; differences may be greater than the UF_H of 10.
Unique Activities	• EPA did not evaluate activities that are unique to Tribal populations (*e.g.*, sweat lodges, powwows). The evaluation of high fish consumption among Tribal populations is included in the category Lifestyle Activities.	• There is uncertainty in how exposure factors (*e.g.*, water consumption rate) change for specific Tribal lifeways.	• EPA did not identify unique activities that influence susceptibility.	• This is a remaining source of uncertainty.

PESS Categories	Potentially Exposed Individuals		Susceptible Subpopulations	
	Potential Increased Exposures Incorporated into Exposure Assessment	**Sources of Uncertainty for Exposure Assessment**	**Potential Sources of Biological Susceptibility Incorporated into Hazard Assessment**	**Sources of Uncertainty for Hazard Assessment**
Aggregate Exposures	• Occupational dermal and inhalation exposures aggregated. • Consumer inhalation, dermal, and oral ingestion exposures are presented by individual but are aggregated in Appendix J.	• Uncertainty is associated with several exposures that EPA did not aggregate (see Section 5.1.4): ○ Inhalation and drinking water for the general population from co-located facilities due to the lack of reasonably available site-specific data for TCEP. ○ Across consumer, commercial, or industrial COUs due to a lack of data indicating such co-exposures exist for TCEP. Across exposure scenarios based on release estimates for the general population because such assumptions could result in double-counting. Across other exposure scenarios (*e.g.*, mouthing consumer articles, drinking water) due to a lack of data indicating the co-exposure of TCEP.	• Not relevant to susceptibility	
Other Chemical and Non-chemical Stressors	• EPA did not identify factors influencing exposure.		• *In vitro* data on co-exposure with benzo[a]pyrene showed increased impacts on inflammation and proliferation pathways. • TCEP showed anti-estrogenic activity *in vitro* after co-exposure with 17β-estradiol.	• There is insufficient data to quantitatively address potential increased susceptibility due to these factors; this is a remaining source of uncertainty.

EPA considered susceptibility when conducting hazard identification and dose-response analysis for TCEP. Some observations may be made regarding factors that may increase susceptibility to the effects of TCEP. Human data suggest there may be susceptible subpopulations, although as identified in Section 5.2.3, human evidence is only slight or indeterminate. Percy et al. (2022) found increased BCEP in urine was associated with lower IQ in children with SES using more than one measure related to SES. Developmental effects related to growth and gestational age show male infants alone and both sexes of offspring were less likely to be small for their gestational age in one study (Oh et al., 2024) and have increased skinfold thickness (two measures of thickness for males; one measure for both sexes) in another study (Crawford et al., 2020). Female children had a greater incidence of being pre-term (Oh et al., 2024) and lower birthweight and length (Yang et al., 2022). Effects may differ by gender, as identified by some epidemiological studies, including the developmental effects on growth and gestational age and possible greater susceptibility by female children to hay fever/allergies (Mendy et al., 2024).

Animal studies showed slight evidence for developmental effects (NTP, 1991a). EPA identified some sensitive sexes for certain health outcomes (higher incidence of neurotoxicity in female rats (NTP, 1991b), greater sensitivity of male (vs. female) mice in reproductive effects (Chen et al., 2015a)), and EPA quantified risks based on male reproductive effects in the risk evaluation. An acute POD based on neurotoxicity was identified for pregnant rats (Moser et al., 2015).

As identified in Table_Apx E-2, other susceptibility factors are generally considered to increase susceptibility of individuals to chemical hazards. These factors include pre-existing diseases, alcohol use, diet, stress, among others. The effect of these factors on susceptibility to health effects of TCEP is not known; therefore, EPA is uncertain about the magnitude of any possible increased risk from effects associated with TCEP exposure.

For non-cancer endpoints, EPA used a default value of 10 for human variability (UF_H) to account for increased susceptibility when quantifying risks from exposure to TCEP. The Risk Assessment Forum, in *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002b), discusses some of the evidence for choosing the default factor of 10 when data are lacking and describe the types of populations that may be more susceptible, including different lifestages (*e.g.*, of children and elderly). U.S. EPA (2002b), however, did not discuss all the factors presented in Table_Apx E-2. Thus, uncertainty remains regarding whether these additional susceptibility factors would be covered by the default UF_H value of 10 chosen for use in the TCEP risk evaluation.

For cancer, the dose-response model applied to animal tumor data employed low-dose linear extrapolation, and this assumes *any* TCEP exposure is associated with some positive risk of getting cancer. EPA made this assumption in the absence of an established MOA for TCEP and according to guidance from U.S. EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b). Assuming all TCEP exposure is associated with some risk is likely to be health conservative because EPA does not believe that a mutagenic MOA is likely for TCEP. However, information is lacking with which to determine whether there is an MOA that acts via a non-linear dose-response. Even though the cancer dose-response modeling assumes any exposure is associated with a certain risk, EPA presents risk estimates in comparison with benchmark risk levels (1 in 1,000,000 to 1 in 10,000).

Although there is likely to be variability in susceptibility across the human population, EPA did not identify specific human groups that are expected to be more susceptible to cancer following TCEP exposure. Other than relying on animal tumor data for the more sensitive sex, the available evidence does not allow EPA to evaluate or quantify the potential for increased cancer risk in specific

subpopulations, such as for individuals with pre-existing diseases or those who smoke cigarettes. Given that a mutagenic mode of action is unlikely, EPA does not anticipate greater cancer risks from early life exposure to TCEP. Therefore, EPA is not applying an age-dependent adjustment factor.

EPA also considered PESS throughout the exposure assessment. EPA estimated infant risks from milk ingestion based on TCEP concentrations in milk modeled for maternal exposures associated with consumer, occupational, and general population groups. Infant exposures through milk were estimated for both mean (105 mL/kg-day) and upper (153 mL/kg-day) milk intake rates. Risk estimates for intermediate and chronic infant exposures through milk were calculated for both cancer and non-cancer endpoints for each COU within each maternal group. Although EPA only had slight confidence in the exposure estimates for infants for this pathway, EPA did determine that infants exposed through human milk ingestion are not more sensitive than the mothers. Protecting the mother will also protect the infant from exposure via human milk. Results of that analysis are included in Section 5.3.2.4.

For the general population, EPA also identified subsistence fishers, children, infants, and people who live in fenceline communities as PESS groups. In its evaluation, EPA considered the increased intake of fish in subsistence fishers. Although there was not enough reasonably available information to assess exposures for Tribal populations specifically, EPA quantitatively evaluated the Tribal fish ingestion pathway for TCEP. Children, infants, and people who live in fenceline communities were also identified as a PESS group for the general population through the drinking water pathway and soil ingestion pathways. The fish ingestion analysis and the analysis of children's exposure through drinking water and soil can be found in Section 5.3.2.3.1.

For occupational exposures, EPA also conducted a qualitative assessment for firefighters. Monitoring data suggests that firefighters have elevated TCEP exposures as a result of firefighting activities. Elevated levels of flame retardants have been found in dust collected from fire stations and in firefighter personal equipment (Shen et al., 2018). A study on firefighters reported increased urine concentrations of BCEP, a metabolite of TCEP, from pre-fire to 3- and 6-hour post fire collections. Although the results were not statistically significant, pre-fire vs. post fire concentrations indicate that firefighters may be at increased risk of TCEP exposures during structure fires (Mayer et al., 2021). Researchers from the CDC measured urine samples for BCEP in 76 members of the general population and 146 firefighters who performed structure firefighting while wearing full protective clothing and SCBA respirators. BCEP was detected in 10 percent of the general population at a median level that was below the detection limit and in 90 percent of firefighters at a median of 0.86 ng/mL (Jayatilaka et al., 2017). TCEP was measured at five fire stations across the United States (California, Minnesota, New Hampshire, New York, and Texas) at median concentrations of 1,040 ng/g. In comparing chemical concentrations by vacuum use, this study did not observe any differences in TCEP concentrations due to cleaning practices (vacuuming) (Shen et al., 2018). These levels are less than the median (2,700 ng/g) concentrations measured in 2011 in California house dust (Dodson et al., 2012). The U.S. Fire Profile study states that the total number of firefighters in 2020, 364,300 (35%) were career, while 676,900 (65%) were volunteers. The U.S. Fire Profile study also states that the number of fire departments for career firefighters is up to a total of 5,244 establishments and a total of 24,208 establishments for volunteer firefighters (NFPA, 2022).

For consumer exposures, EPA identified and evaluated the exposure for PESS groups including children and infants through exposure to consumer products. Risk estimates for these PESS groups can be found in Section 5.3.2.2. EPA has moderate confidence in the fabric and textile products COU, and slight to moderate confidence in the foam seating and bedding products and building/construction materials-wood resin COUs. Confidence ratings are derived from consideration of variety of factors including

confidence in the model used, the default values, and the input parameters (*e.g.*, density, use duration, weight fraction, dermal parameters), and the corroborating monitoring data (see Table 5-18).

Limited information was available on the TCEP COUs. However, the Ecology Washington database (WSDE, 2023) sampled consumer articles that children under 3 years of age are expected to contact and/or mouthed. Of the 268 products related to TSCA COUs, 24 articles were detected to have TCEP. Eleven out of twenty-four (4 percent of total) articles were related to fabric and textiles uses, whereas 13 out of 24 (5 percent of total) were in foam articles. Products were sampled in the summer of 2012.

Ionas et al. (2014) sampled children's toys in Antwerp, Belgium, and reported an overall detection frequency of 28 percent (32 out of 114) of TCEP detected in children toys produced around the year 2007. Two out of eight articles were for wooden toys. Fang et al. (2013) reported a detection frequency of 95 percent (19 out of 20) of V6/TCEP in vehicles with an average model year of 2004. Stapleton et al. (2012) detected only one instance of V6/TCEP in 102 foam couches across the United States during 2011-2012.

Table 5-80. Summary of Detection Frequencies and Sampling Dates for Relevant Consumer Products Containing TCEP

COU		Detection Frequency	n	Source	Sampling Date
Life Cycle Stage/ Category	**Subcategory**				
Consumer Use/ Furnishing, cleaning, treatment/care products	Fabric and textile products	4%	268	Ecology Washington database (WSDE, 2023)	2012
	Foam seating and bedding products (foam couches)	1%	102	Stapleton et al. (2012)	2011–2012
		5%	268	Ecology Washington database (WSDE, 2023)	2012
		70%	20	Fang et al. (2013)	2009–2011
	Foam seating and bedding products (auto foam)	95%	20	Fang et al. (2013)	2009–2011 vehicle average model year 2004
Construction, paint, electrical, and metal products	Building/ construction materials – Wood and engineered wood products – Wood resin composites	100%	1	SCHER (2012)	1997
		25%	8	Ionas et al. (2014)	2007

Table 5-80 provides a summary of the detection frequencies of the monitoring literature. It is significant that all these frequency estimates were estimated before the implementation of California TB 117-2013, and it is anticipated that manufacturers have phased out TCEP from their product due to the introduction of the less stringent flammability standards for upholstered furniture (TB 117-2013).

Table 5-81. Suggested Consumer Population Sizes Based on Characterization of Consumer Article Detection Frequencies

COU		Detection Frequency	Adjusted Detection Frequency: Current Use	Total U.S. Population (of 331,449,281)[a]	Total U.S. Children under 5 years (of 18,400,235)[a]	Total U.S. Females of Reproductive Age (of 118,273,566)[a]
Life Cycle Stage/Category	Subcategory					
Furnishing, cleaning, treatment/Care products	Fabric and textile products	4%	0.4%	1,325,797	73,601	473,094
	Foam seating and bedding products	5%	0.5%	1,657,246	92,001	591,368
Construction, paint, electrical, and metal products	Building/ construction materials – Wood and engineered wood products – Wood resin composites	1%[b]	1%	3,314,493	184,002	1,182,736

[a] Values from the 2020 U.S. Census.
[b] It was the assessor's judgement to overwrite literature detection frequency value. Only 9 samples presented TCEP use in wooden products.

Table 5-81 assigns a detection frequency value for each COU above slight-moderate confidence. Four percent is chosen for Fabric and Textile Products, and five percent is selected for foam seating and bedding products. Although Fang et al. (2013) indicates higher detection frequencies in vehicles (95%), the vehicles selected in this study were from an average model year of 2003.5, and it is understood that auto manufacturers have moved away from using V6/TCEP formulations in their vehicles. A detection frequency value of one percent is selected for wood resin products, due to the scarce number of examples indicating TCEP use in wood articles.

An order of magnitude correction to adjust the detection frequencies to current uses was applied for fabric and textile products and foam seating and bedding products to adjust for TB 117-2013. The adjustment did not apply to wood resin composites because TB 117-2013 applies to upholstered furniture.

To characterize the population utilizing these consumer articles, the adjusted detection frequencies are multiplied by the total U.S. population, total U.S. population of children under 5 years of age, and total U.S. population of females of reproductive age from the 2020 US census. This calculation provides a ballpark figure of the expected number of individuals who are exposed to current consumer articles.

Major assumptions in the characterization of this population include the idea that the use of these consumer articles scale linearly with the detection frequency of detection among consumer articles, the detection frequencies in the monitoring literature is representative of the use of TCEP compared to other flame retardants in the marketplace, and that the order of magnitude adjustment is sufficient to reflect the phase away from TCEP to other OPFRs.

5.3.4 Risk Characterization for Aggregate and Sentinel Exposures

Section 2605(b)(4)(F)(ii) of TSCA requires EPA, as a part of the risk evaluation process, to describe whether aggregate or sentinel exposures under the COUs were considered and the basis for their consideration.

The term aggregate is defined as "the combined exposures to an individual from a single chemical substance across multiple routes and across multiple pathways" in the Agency's final rule, *Procedures for Chemical Risk Evaluation Under the Toxic Substances Control Act* (82 FR 33726, July 20, 2017) (see Appendix A.2).

In the procedural rule, EPA defines sentinel exposure as "the exposure from a chemical substance that represents the plausible upper bound of exposure relative to all other exposures within a broad category of similar or related exposures" (40 CFR 702.33). In this evaluation, EPA considered sentinel exposures by considering risks to populations who may have upper bound exposures, including workers and ONUs who perform activities with higher exposure potential and people who live in fenceline communities. EPA characterized high-end exposures using modeling approaches and if available, using monitoring data. Where information on the distribution of exposures is available, EPA typically uses the 95th percentile value of the available dataset to characterize high-end exposure for a given COU.

Across Routes

Figure 5-15 aggregates the consumer exposure estimates by route (inhalation, dermal, ingestion) for each COU and lifestage combination. In addition, this supplemental file includes risk tables that indicate whether aggregation across routes results in risk. Figure 5-18 and Figure 5-19 provide two examples where an aggregation across routes could result in chronic and acute risk, whereas consideration from a single route would not result in risk. For example, for Figure 5-18, if dermal, ingestion, and inhalation routes were considered individually, the exposure estimates are lower than the intermediate/chronic HED of 2.73 mg/kg-day or HEC 14.9 mg/m^3 divided by the benchmark MOE of 30. However, when aggregating dermal and inhalation exposures, the toxicity value of 2.73 mg/kg-day divided by the benchmark MOE is exceeded.

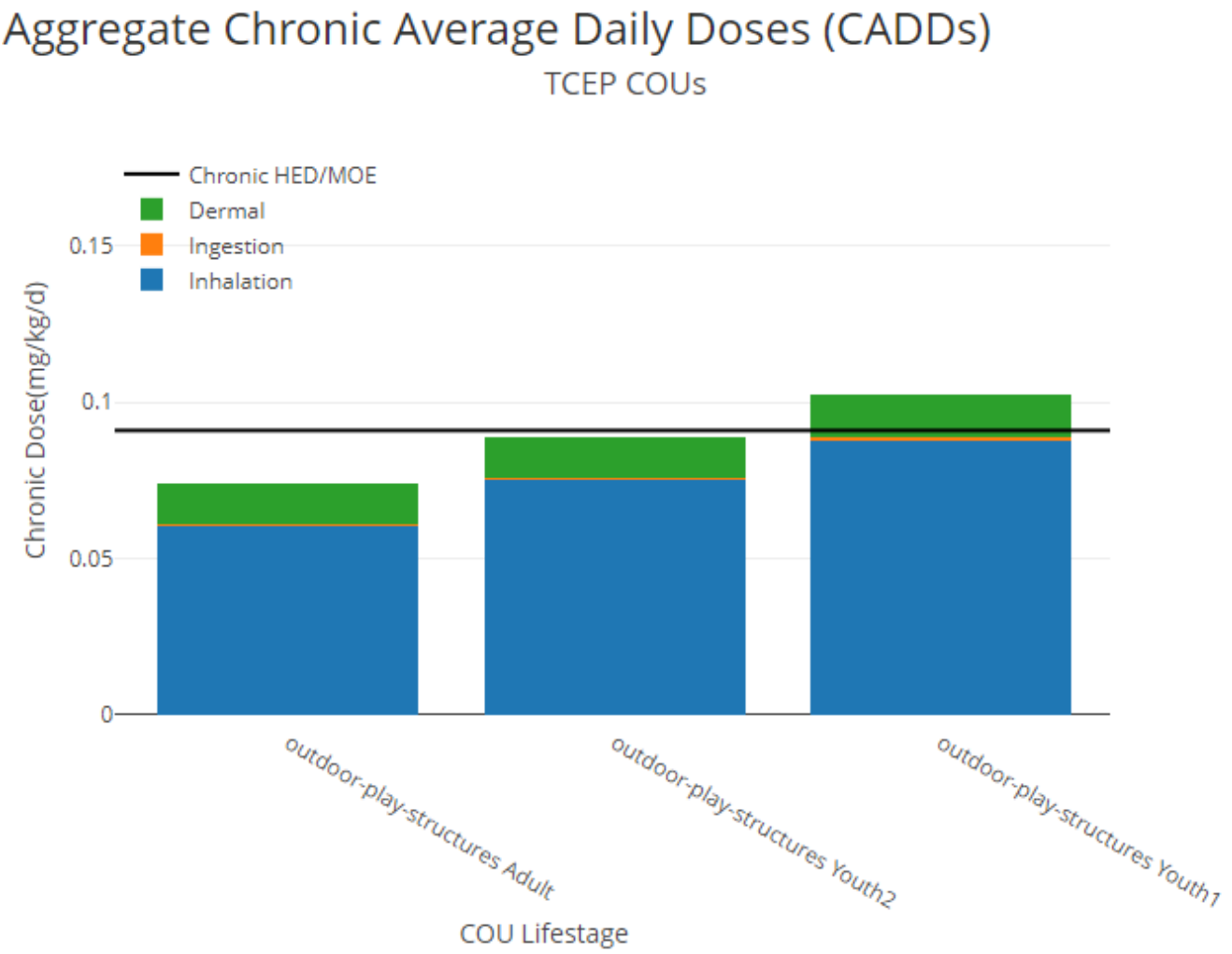

Figure 5-18. Aggregate CADDs for Consumer Use of Textiles in Outdoor Play Structures at Adult, Youth2, and Youth1 Lifestages

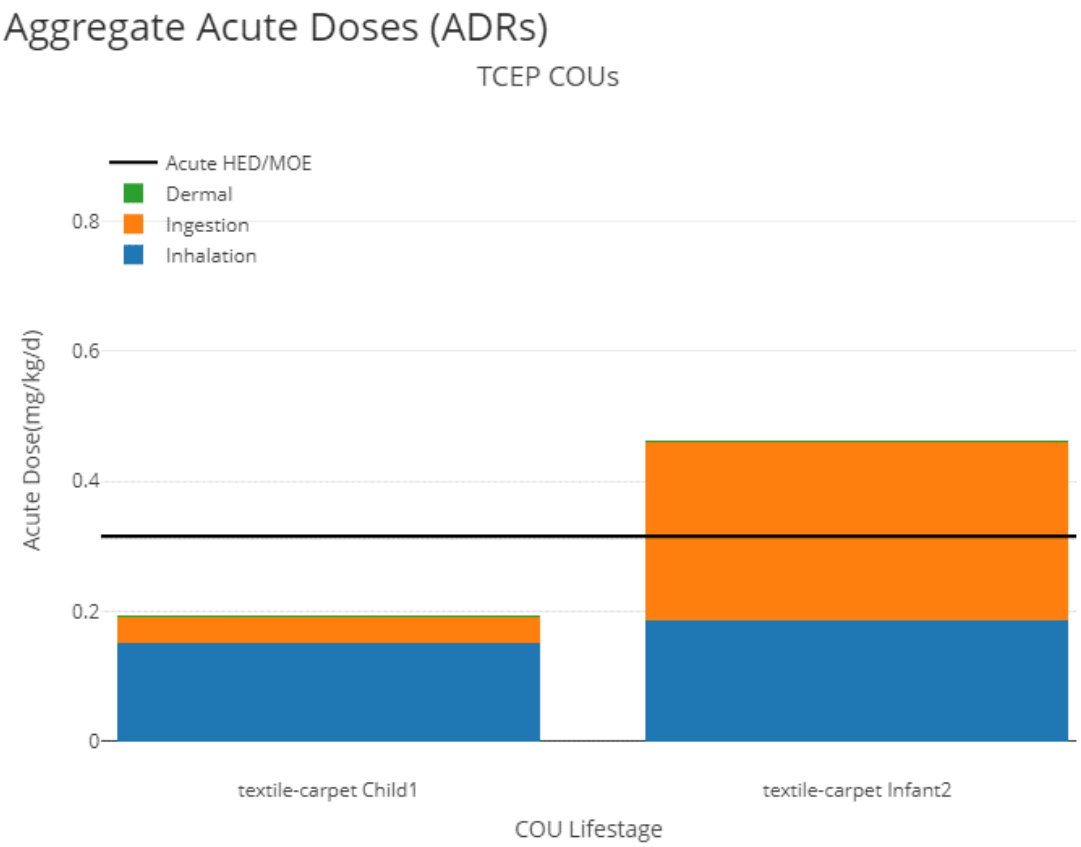

Figure 5-19. Aggregate ADRs for Carpet Back Coating, Child1, and Infant2 Lifestages

There were no instances of aggregate lifetime risk for any COU where there was not already risk to the COU from an individual route. The supplemental file includes risk tables that can further be toggled to explore aggregate risks.

EPA combined exposures for the milk pathway across all routes for each COUs/OESs within workers and consumers. However, for the general population, EPA only assessed the oral route when assessing the milk pathway because exposure estimates showed that oral doses were several magnitudes higher than dermal or inhalation doses. As a result, oral exposures will be the primary driver for infant risks via the milk pathway. Furthermore, within the adult oral pathways that include fish ingestion, drinking water ingestion, and incidental water ingestion from swimming, EPA only considered fish and drinking water ingestion. These two pathways constitute the highest oral doses, thus having the greatest potential to result in infant risks from human milk ingestion. Indeed, infant cancer risk estimates exceeded 1 in 1,000,000 for all COUs/OESs based on maternal fish ingestion (high BAF). Aggregating other exposure scenarios will not further inform risk characterization.

Across Exposure Scenario

The confidence in the general population exposure scenarios for drinking water ingestion, fish ingestion (lowBAF), and inhalation (100 m) is moderate. For the formulation of TCEP containing reactive resin OES, chronic non-diluted drinking water exposure estimates are 1.46×10^{-4} mg/kg/d. For the same OES, chronic fish ingestion concentrations are two to three orders of magnitude higher for the general population and subsistence fishers at 0.033 and 0.94 mg/kg/d, respectively. Chronic inhalation exposure estimates are given in mg/m^3 and do not exhibit risk—even at 10 m from a hypothetical facility. Therefore, aggregate exposure across general population exposure scenarios does not result in an appreciable difference as the exposure is dominated by the sentinel exposure (fish ingestion).

Furthermore, since the general population and subsistence fisher estimates result in chronic risk for all COUs, aggregating additional exposure scenarios (*e.g.*, consumer, occupational) with the general exposure scenarios (fish ingestion) is uninformative in characterizing risks.

The confidence in the consumer COUs is moderate for the subcategories of carpet back coating, textile in outdoor play structures, living room foam, automobile foam, and wooden TV stands. Chronic ingestion estimates are above the chronic benchmark (0.091 mg/kg/d) for each of these subcategories (carpet back coating, textile in outdoor play structures, living room foam, automobile foam, and wooden TV stands), and chronic dermal estimates are above the benchmark for wooden TV stands. Because the consumer exposure estimates result in chronic risk, aggregating additional exposure scenarios (*e.g.*, general population, occupational) with the consumer exposure scenarios is uninformative in characterizing risk.

The other consumer exposure scenario subcategories (*e.g.*, insulation, mattress, wood resin) have slight confidence. Aggregating these subcategories with additional exposure scenarios (*e.g.*, general population, occupational) would be uninformative in characterizing risk due to the slight confidence in these scenarios.

5.3.5 Overall Confidence and Remaining Uncertainties in Human Health Risk Characterization

EPA took fate, exposure (occupational, consumer, and general population), and human health hazard considerations into account when characterizing the human health risks of TCEP. Human health risk characterization evaluated confidence from occupational, consumer, and general population exposures and human health hazards. Hazard confidence and uncertainty is represented by health outcome and exposure duration as reported in Section 5.2.6, which presents the confidence, uncertainties, and limitations of the human health hazards for TCEP. Confidence in the exposure assessment has been synthesized in the respective weight of scientific evidence conclusion sections for occupational exposures (see Section 5.1.1.4), consumer exposures (see Section 5.1.2.4), and general population exposures (see Section 5.1.3.7). Table 5-82 provides a summary of confidence for exposures and hazards for non-cancer endpoints for the COUs that resulted in any non-cancer risks, and Table 5-83 provides a confidence summary for cancer for the COUs that resulted in cancer risks.

Uncertainties associated with the occupational exposure assessment include a lack of reported data from databases such as TRI, NEI, DMR, and more recently, CDR. Site-specific data were only available for a small number of current processors, and it is not clear if this data are representative of these industries and workplace practices.

Uncertainties associated with the general population exposures assessment included the lack of site-specific information, the incongruence between the modeled concentrations and doses with the monitoring data, and the complexity of the assessed exposure scenarios. Section 5.1.3.7 illustrates the confidence in the assessment of the general population exposure scenarios.

5.3.5.1 Occupational Risk Estimates

Exposure Monitoring Data and Use of Models

EPA only identified monitoring data for dust occurring within an e-waste recycling facility. Monitoring data for the remaining COUs/OESs was not found. Surrogate monitoring data were found to assess potential exposure to TCEP during installation of articles and this estimated inhalation exposure used TCEP monitoring data for furniture manufacturing (Mäkinen et al., 2009). Surrogate monitoring data are

also used for the assessment of paints and coatings use during spray application. It is unclear if these COUs have similar worker activities and if they are fully representative of worker exposure for the OESs of installation of articles and use of paints and coatings. The remaining COUs/OESs used modelling approaches to estimate worker exposures.

Where sufficient data were reasonably available, the 95th and 50th percentile exposure concentrations were calculated using these data. The underlying distribution of the data, and the representativeness of the reasonably available data, are not known. Where discrete data were not reasonably available, EPA used reported statistics from the Monte Carlo simulations (*i.e.*, 50th and 95th percentile). Because EPA could not verify these values, there is an added level of uncertainty.

For OESs that do not have monitoring data, EPA used relevant GSs and/or ESDs to identify worker activities and exposure routes that are reasonably expected to occur. Exposure distributions were then created using Monte Carlo simulation with 100,000 iterations and the Latin hypercube sampling method.

EPA calculated ADC and LADC values assuming workers and ONUs are regularly exposed during their entire working lifetime, which likely results in an overestimate. Individuals may change jobs during their career such that they are no longer exposed to TCEP; therefore, actual ADC and LADC values would be lower than the estimates presented.

Although EPA has confidence in the models used, it is possible that they may not account for variability of exact processes and practices at an individual site. Furthermore, there are no 2020 CDR reports for TCEP and only one from 2016. Therefore, EPA made assumptions about pounds per site-year (2,500 lb presented in risk tables) that leads to uncertainty in these estimates.

Assumptions Regarding Occupational Non-users

Exposures for ONUs can vary substantially and most data sources do not sufficiently describe the proximity of these employees to the TCEP exposure source. As such, exposure levels for the "occupational non-user" category will have high variability depending on the work activity; therefore, all ONU exposure estimates except for recycling of e-waste are considered to have only slight confidence. For the OES of recycling of e-waste, monitoring data were available for workers conducting activities consistent with the activities of ONUs, this results in a confidence rating of moderate to robust.

Modeled Dermal Exposures

The Fractional Absorption Model is used to estimate dermal exposure to TCEP in occupational settings. The model also assumes a single exposure event per day based on existing framework of the EPA/OPPT 2-Hand Dermal Exposure to Liquids Model and does not address variability in exposure duration and frequency. Additionally, the studies used to obtain the underlying values of the quantity remaining on the skin (Q_u) did not take into consideration the fact that liquid retention on the skin may vary with individuals and techniques of application on and removal from the hands. Also, the data used were developed from three kinds of oils; therefore, the data may not be applicable to other liquids. Based on these uncertainties, EPA has a moderate level of confidence in the assessed baseline exposure.

Number of Workers

There are several uncertainties surrounding the estimated number of workers potentially exposed to TCEP. Most are unlikely to result in a systematic underestimate or overestimate but could result in an inaccurate estimate. CDR data were not available to estimate the number of workers associated with manufacturing, processing, or use of TCEP. There are also uncertainties with BLS data, which are used to estimate the number of workers for the remaining COUs. EPA had to use higher-level NAICS codes (at 3- to 5-digit level) combined with assumptions from the U.S. Census' SUSB, which could result in inaccuracies if the distribution of workers in occupations with TCEP exposure differs from the overall distribution of workers in each NAICS. Also, EPA needed to designate which industries and occupations have potential exposures, and this may result in over- or underestimation. However, any inaccuracies would not be likely to systematically either overestimate or underestimate the number of exposed workers.

Weight Fraction Considerations

The COU that had the lowest TCEP concentration weight fraction in the products assessed was the Use of Paints and Coatings. The weight fraction, from the product SDS's, ranged from 0.1 to 25 percent (Table 5-14). A Monte Carlo simulation using this weight fraction range was used to assess risk. For human cancer inhalation risk estimates were above 1 in 10,000 for both central tendency and high-end exposures and dermal presented some risks as well. For environmental, this OES presents risk to the aquatic environment at the high end but not at the central tendency by 1 day of exceedance. Although risk does seem to decrease based on lower concentrations of TCEP being used in certain OESs, EPA has not estimated a weight fraction value alone that would eliminate risk.

5.3.5.2 Consumer Risk Estimates

Lack of Weight Fraction Data

No safety data sheets (SDSs) were available for consumer products containing TCEP. Monitoring literature and databases suggest that TCEP is used in consumer articles (*e.g.*, fabric and textiles, home furnishings, automobile foams, children's toys, and building materials such as insulation). Section 5.1.2.2 highlights the available information on the consumer COUs and relevant exposure scenarios. EPA only had a few U.S. studies and databases (Castorina et al., 2017; Fang et al., 2013), including the Ecology Washington Database (WSDE, 2023), which provides information on article weight fractions for the consumer COUs. Where there were gaps, EPA utilized foreign data (Ionas et al., 2014; Marklund et al., 2003; Ingerowski et al., 2001) to help select values for product weight fraction data. EPA is unclear on how relevant the foreign weight fraction data are for consumer articles used in the United States. Moreover, one of these European studies (Ingerowski et al., 2001) had a low-quality data evaluation rating and was from the early 2000s. In addition, there are limitations in the data integrity in the Washington State Database (WSDE, 2023). There is a possibility that a chemical could be a contaminant rather than a component of the formulation of the consumer article. In addition, there are some quality assurance and quality control issues with the database suggesting that it might be unreliable.

Nevertheless, due to the paucity of information, EPA used low-quality information where higher quality information was unavailable. In general, EPA has slight confidence in the building and construction materials COUs (*e.g.*, insulation and acoustic ceiling); slight-moderate confidence in the wood resin products and foam seating and bedding products exposure scenarios; and moderate confidence in the fabric and textile COUs (*e.g.*, carpet back coating).

Complexity of Exposure Scenarios
The indoor air and indoor dust literature indicate that TCEP is present at higher values in indoor vs. outdoor environments suggesting amplified exposures in the home. Uncertainties in the particle and gas distribution (see Section 3.3.1.2.1) of TCEP builds further uncertainty on the reliability of direct inhalation estimates vs. dust-mediated exposure via dermal absorption and oral ingestion.

SVOCs such as TCEP exhibit complex behaviors in the indoor environment. Shin et al. (2014) indicates that TCEP has a relatively high emission rate compared to other semivolatile organic compounds. Shin et al. (2014) observed that dust parameters such as removal rate from vacuuming, and dust loading onto carpets and indoor furnishings are important variables that influence emission rates. CEM 3.2 does incorporate defaults for cleaning frequency and cleaning efficiency from settled floor dust; however, EPA was not able to obtain data on dust loading onto carpets when assessing the consumer COUs. The uncertainties related to the behavior of TCEP in the indoor dust matrix further builds uncertainty into the consumer risk estimates.

Model and Parameter Uncertainties
CEM 3.2 is a deterministic (rather than a population-based) model that provides point estimates of TCEP exposure to population of interest. CEM is not equipped to model complex emission profiles or activity patterns of residents other than those pre-populated within CEM. EPA used the CEM 3.2's sensitivity mode to vary certain parameters to help understand which parameters influence the exposure estimates. The initial concentration of SVOC in the article (a product of weight fraction and product density) was the most important parameter for consumer modeling. Best judgments were used to approximate product density of consumer articles where defaults were unavailable. The uncertainties in the weight fraction and density information are reflected in EPA's overall confidence in consumer modeling.

Dermal absorption parameter of fraction absorbed (F_{abs}) was estimated at 35.1 percent for all consumer article scenarios from Abdallah et al. (2016). This value overrode the embedded CEM calculation for dermal absorption. Estimates derived from the literature were of higher confidence then the CEM 3.2 calculated dermal absorption parameters. Nevertheless, there are uncertainties as to the applicability of this one fraction absorbed value for all scenarios. Fraction absorbed can be a function of duration of article or dust contact; however, because EPA was uncertain as to how often consumers, infants, and children would wash their hands, EPA retained a conservative fraction absorbed value for the purposes of consumer modeling.

Monitoring vs. Modeled Concentrations and Doses
The incongruence between modeled and measured concentrations and doses helps illustrate further uncertainties in the consumer exposure assessment. Modeled indoor air concentrations for the building/construction materials, insulation scenario (12.07 mg/m^3) are six orders of magnitude higher than the highest indoor air TCEP concentration observed in the United States (95th percentile of 35 ng/m^3) (Dodson et al., 2017). This discrepancy suggests major uncertainties in the insulation exposure scenario.

The highest observed modeled dust intake in the reported modeled literature was 1.38 µg/kg-day reported for children at a kindergarten in Hong Kong (Deng et al., 2018b). This value is within one to two orders of magnitude of EPA's highest oral and dermal modeled intakes for children. EPA's highest modeled oral intakes was 6.92×10^{-2} mg/kg-day (69.2 µg/kg-day) for the foam toy block scenario. EPA's highest observed dermal intakes via dermal absorption was 3.07×10^{-1} mg/kg-day (307 µg/kg-day) for

the wood flooring scenario. These comparisons suggest that the oral and dermal intakes are more like values reported in the literature than the modeled inhalation estimates.

Timeseries of Inhalation Exposure Estimates

CEM 3.0 estimates a chronic inhalation exposure by averaging the exposure over 365 days. Chronic consumer inhalation exposures from TCEP containing articles are initially dominated by the gas phase concentrations (due to off-gassing of TCEP). Figure 5-20 and Figure 5-21 display the time series air concentrations for acoustic ceilings and wood flooring scenarios. After 4 weeks for the acoustic ceiling scenario and 2 weeks for the wood flooring scenario, chronic consumer inhalation exposures are dominated by the dust air concentrations. Chronic inhalation concentrations from insulation were dominated by the gas phase concentrations; however, Figure 5-22 displays a precipitous drop in concentration from the insulation article after the first few months.

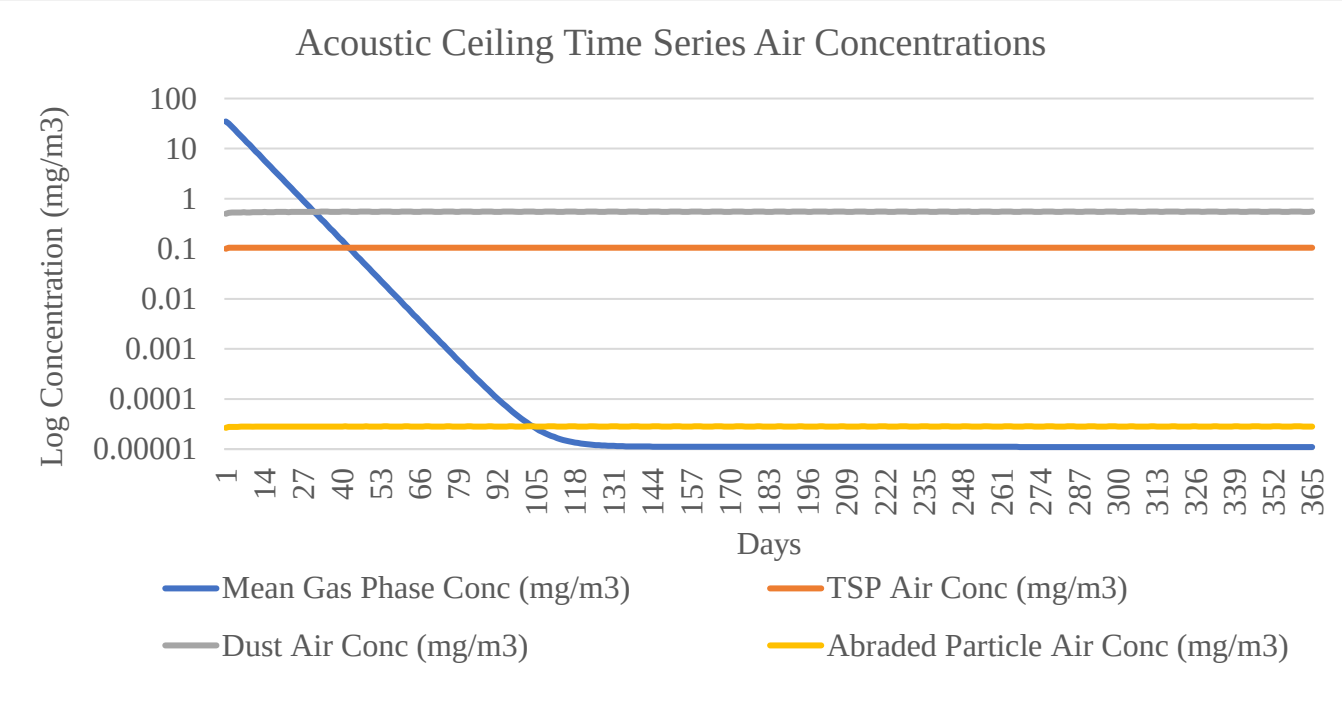

Figure 5-20. Consumer Modeling Time Series Results for Acoustic Ceilings

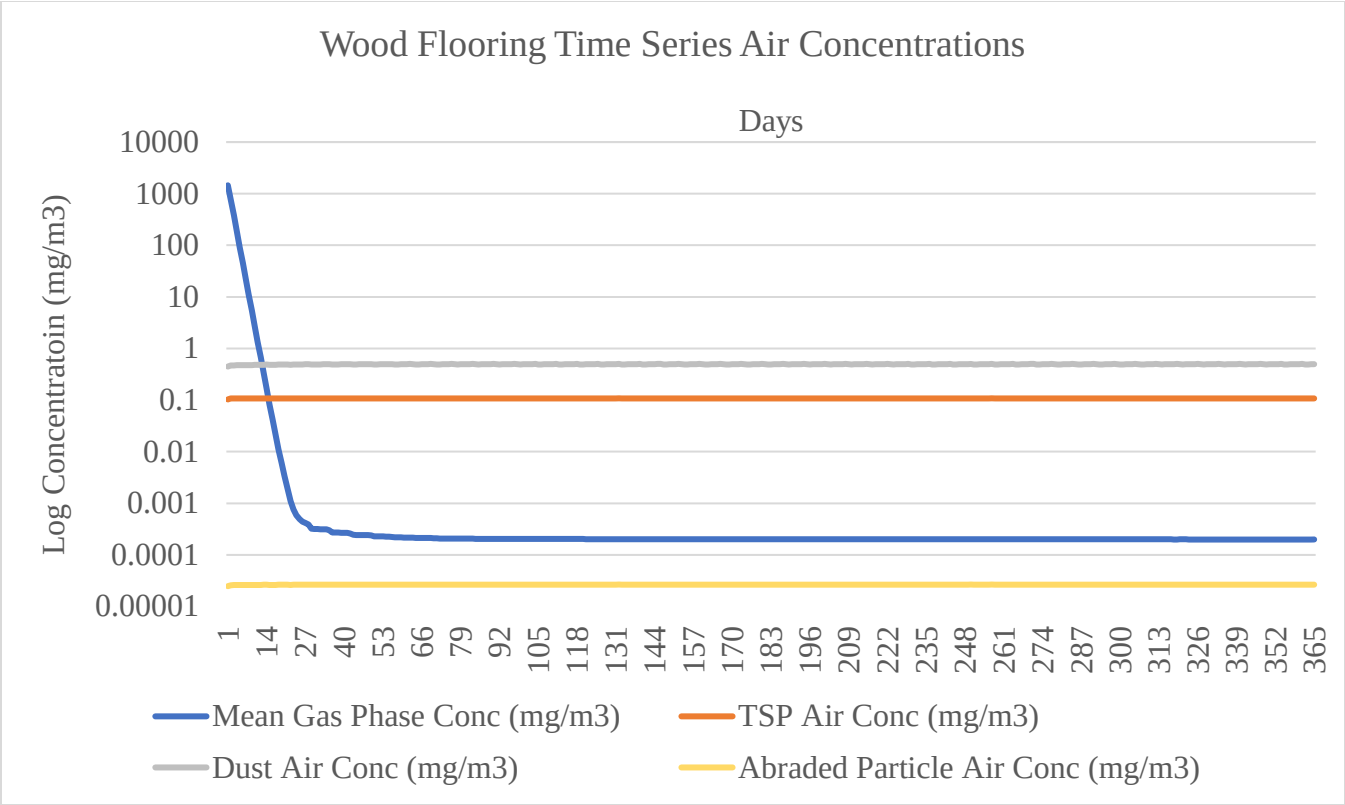

Figure 5-21. Consumer Modeling Time Series Results for Wood Flooring

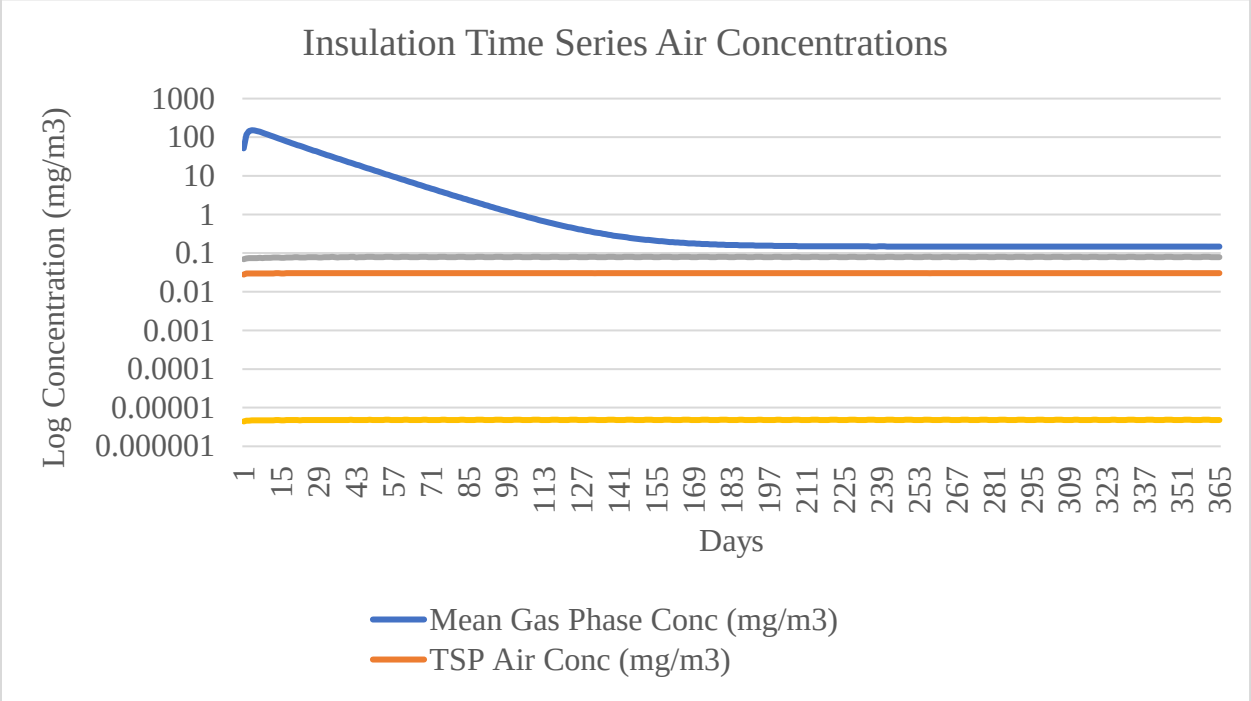

Figure 5-22. Consumer Modeling Time Series Results for Insulation

Consumer articles containing TCEP are no longer manufactured in the United States. Consumers may obtain new products containing TCEP only via import. Older articles in the home may have already

undergone off-gassing of TCEP; thus, there is uncertainty as to the relevance of continued inhalation exposure from older consumer articles containing TCEP as much of the exposure may have already occurred in the first few weeks.

Risk Estimates for Conservative Scenarios
EPA did not utilize a range of estimates to model a central tendency and high-end for consumer exposures. Detection frequencies of TCEP were low for various consumer products in the Washington State Database and accompanying monitoring data, and rather than utilize a central tendency (that potentially was below realistic detection limits), EPA selected plausible worst-case values for weight fractions. Due to this approach, EPA has more confidence in scenarios that did not exhibit risk than scenarios that exhibited risk.

Sensitivity for Minimum Weight Fraction Values
Table 5-68 indicates that EPA has moderate confidence in the non-cancer MOEs for the Fabric and textile products scenario – textile for children's outdoor play structures, Foam seating and bedding products – foam auto, and building/construction materials – wood and engineered wood products – wood resin composites – wooden TV stand scenarios. EPA used CEM 3.2's batch mode, to vary weight fractions two to three orders of magnitude.

The wooden TV stand scenario had the highest weight fraction (3%) of the scenarios with moderate confidence. When keeping all other parameters constant, varying the weight fraction to 1 percent no longer resulted in dermal risk for infants and children, and no longer resulted in ingestion risk for children. However, even after varying the weight fraction to 1 percent, 0.1 percent, and 0.01 percent, EPA still saw inhalation and ingestion risk for Infants.

EPA also varied weight fractions for the textile for children's outdoor play structures scenario. This scenario had an initial weight fraction of 1.3 percent and moderate confidence. When keeping all other parameters constant, and varying the weight fraction to 1 percent, EPA still saw inhalation risk to children and infants, and ingestion risk to infants. When varying the weight fraction to 0.1 percent, EPA no longer saw inhalation risk for any lifestages, but still saw ingestion risk to infants. Even after varying the weight fraction to 0.01 percent EPA still saw ingestion risk for Infants.

EPA varied weight factions for the automobile foam scenario (0.74 percent weight fraction and moderate confidence). Even after varying the weight fraction to 0.1 percent and 0.01 percent, EPA still saw ingestion risk for Infants.

These results indicate that TCEP weight fraction does not appear to be a good predictor of risk from TCEP ingestion and inhalation exposure to Infants. More information on this sensitivity analysis can be found in the (U.S. EPA, 2024b)

5.3.5.3 General Population Risk Estimates

Location Information
Due to the lack of reasonably available site-specific information, the exposures assessment relied on assumptions for location specific model inputs. This lack of data results in uncertainties surrounding these location specific parameters (*e.g.*, flow parameters and meteorological data). The AERMOD Model included two meteorological conditions (Sioux Falls, South Dakota, for central tendency meteorology and Lake Charles, Louisiana, for higher-end meteorology), in addition to different land coverage scenarios (Suburban Forests and Oceans) to characterize potential amounts of annual TCEP

deposition to soil from air. It is unclear how relevant these meteorological conditions and land cover scenarios are to TCEP facilities as there are no available site-specific information.

EPA modeled air concentrations and deposition fluxes at various distances from the hypothetical facility releasing TCEP. The Agency selected various distances to calculate exposure doses and inhalation concentrations for the general population (*e.g.*, ambient air exposure to the general population, soil dermal and oral intakes for children). In general, EPA has more confidence in risk estimates at further distances from the hypothetical facility than risk estimates at closer distances. For example, EPA has less confidence soil dermal exposure at 100 m of the facility than it does with soil dermal exposure at 1,000 m of the facility.

Due to the lack of reasonably available site-specific information for industrial and commercial releases of TCEP, EPA could not estimate the proximity of general population residents to drinking water intake locations. Drinking water estimates were calculated for non-diluted (*i.e.*, drinking water intake locations are at the site of the surface water release) conditions as a worst-case scenario. Drinking water estimates were also calculate for diluted conditions by estimating the distance between intake location and the site of release via drinking water intake information available for various SIC codes. EPA has more confidence in these estimates as they represent a more plausible distance from which the general population would receive their drinking water.

EPA Region 9 (2023) conducted a biological evaluation and habitat assessment for discharges to the Pacific Ocean with the city of Los Angeles Hyperion Water Reclamation Plant and wastewater reclamation systems. This study indicated average mass loadings of TCEP from 0.19 to 0.28 lb/day. In addition, this study reported effluent concentrations of TCEP between 0.10 to 0.15 µg/L after wet and dry events in Fall of 2019. Furthermore, this study estimated annual average discharge rates of 230 to 236 MGD (millions of gallons per day) which equates to 871 to 893 MLD (millions of liters per day).

Due to the absence of information on specific sites and water bodies receiving TCEP water releases, EPA utilized SIC codes to estimate flow rates. EPA used 50th percentile harmonic mean flow rates ranging from 6.51 to 11.46 MLD. In the *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: E-FAST Modeling Results* (U.S. EPA, 2024g), EPA has included surface water modeling estimates for the 90th percentile flow rates which range from 1,780 to 14,942 MLD).

The effluent flows presented in the analysis conducted by EPA Region 9 (2023) (871–893 MLD) are in between the 50th percentile and 90th surface water flow rates utilized in EPA's surface water modeling. Surface water flow rates are generally larger than wastewater flow rates. Although, EPA utilized the more conservative 50th percentile harmonic mean flow (3.51–11.46) in its risk calculations, the flows described in the EPA Region 9 (2023) assessment should be interpreted with caution as they are only one data point compared to thousands of facilities that are sampled from the SIC codes. The approach for calculating 50th percentile and 90th percentile flows are described in Appendix I.2.2.

Results of the 90th percentile harmonic mean stream flows and central tendency estimates are presented in *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: E-FAST Modeling Results* (U.S. EPA, 2024g). These results indicate the critical role of receiving water flow as an input in determining TCEP concentrations in surface water.

Monitoring vs. Modeled Concentrations and Doses

The incongruence between modeled and measured concentrations and doses helps illustrate further uncertainties in the general population. WQP data on surface water TCEP concentrations is three to five

orders of magnitude lower than modeled surface water concentrations (see Sections 3.3.2.4 and 3.3.2.5). TCEP fish tissue concentrations within the Great Lakes (Guo et al., 2017b) are two to three orders of magnitude lower than the TCEP tissue concentrations calculated using a whole organism BCF value from another high-quality study (Arukwe et al., 2018). Modeled soil concentrations were within one order of magnitude of a single study from published literature (Mihajlovic and Fries, 2012); however, it is important to note that similarity with a single study is not enough to build confidence in the relevance or accuracy of modeled results.

Complexity of Exposures Scenarios

The dermal absorption and ingestion from soil exposures scenarios require a complex understanding of fate and transport of TCEP. Soil concentrations were calculated by modeling deposition fluxes of TCEP at various distances from a hypothetical facility. Soil intakes were estimated for two exposures scenarios—a child playing in mud and a child performing activities with soil. Parameters to calculate these exposures, such as surface areas, absorption factors, and intake rates, were available in EPA's *Exposure Factors Handbook* (U.S. EPA, 2017d); however, there is high uncertainty in the scenario due to the multiple unknowns (*e.g.,* hypothetical facility, hypothetical release estimate, unknown distance between homes and facility).

Model and Parameter Uncertainties

An additional uncertainty for the general population and consumer assessment are model uncertainties. VVWM-PSC allowed for the application of a standard, conservative, set of parameters and adjust for physical-chemical properties of TCEP. For example, stream reach was set to represent a shallow waterway with a width of 5 m and depth of 1 m. There are uncertainties on the applicability of this shallow water body volume.

Ambient and drinking water estimates via VVWM-PSC and EFAST utilized a 0 percent drinking water treatment removal efficiency (see Section F.2.5.3). Although TCEP has been shown to be recalcitrant to removal treatment processes, EPA is uncertain whether advanced treatment methods can remove TCEP from water.

For AERMOD, EPA specified deposition parameters for such as the fraction of gas vs. particle phase, diffusivity in air, diffusivity in water, and the MMAD. Further sensitivity analysis can illustrate the effects these parameters have on the deposition fluxes. Conflicting information in the peer-reviewed literature creates uncertainties on the appropriate values of these parameters. Okeme (2018) has described the complexities associated with the gas and particle partitioning of TCEP and has suggested reported high concentrations of TCEP in particulates may be a result of sampling artifact (see Section 3.3.1.2.1).

A major uncertainty in fish ingestion exposure estimates was the selection of BAF values. Appendix F.2.6 provides a review of BAFs found in the literature. The BAF of 2,198 for walleye (*Sander vitreus)* from Guo et al. (2017a) was initially selected as a representative study of the U.S. population as it sampled surface water and fish tissue concentrations in the Great Lakes. Walleye also represent a cool-water top predator that serves as an important food fish. This species potentially preys on secondary and tertiary consumers; however, it is uncertain what localized conditions affect BAF values within Guo et al. (2017a). Furthermore, the surface water concentration and fish tissue concentrations were collected in different years, thus it is difficult to hypothesize if TCEP surface water concentrations at the time of sample collection influenced BAF values. A possible explanation for the resulting high oral risk estimates could be an issue specific to BAFs for walleye (*Sander vitreus)* within the selected study Guo et al. (2017a).

Uncertainties in Production Volume and Leachate Concentrations for Disposals Analysis
As demonstrated in Figure 1-3, the production volume of TCEP has decreased over the past decade. Current production volume levels are difficult to predict. EPA conducted a bounding analysis to estimate groundwater concentrations resulting from the disposal of TCEP to landfills utilizing the DRAS software (see Section 3.3.3.8). The bounding analysis varied the production volume four orders of magnitude to account for past disposal practices. In addition, EPA varied the leachate concentrations, bounding the top of the range by TCEP's solubility (see Section 3.3.3.8).

Although efforts were made to estimate the potential migration of TCEP to groundwater from the disposal of TCEP containing wastes, there are considerable uncertainties in this assessment approach. Uncertainties in the loading rate, the leachate concentration and the absence of site-specific information make it difficult to characterize this exposure scenario.

When estimating drinking water risk using the estimated groundwater concentrations from the DRAS analysis, EPA only found lifetime cancer risk when using a leachate concentration of 1,000 mg/L and production volume above 250,000 lb. No lifetime cancer risk was observed for a leachate concentration of 100 mg/L and a production volume of 2,500,000 lb. TCEP's solubility is 7,820 mg/L (see Table 2-1). EPA believes that 2,500 lb is the most suitable production volume for current uses of TCEP, and the highest reported literature value of TCEP in leachate concentrations was 0.177 mg/L (Masoner et al., 2014a).

Therefore, this analysis suggests that even with the uncertainties of no site-specific information, assuming groundwater concentrations in place of drinking water concentrations, and assuming the general population living in proximity to poorly managed landfills, production volumes would have to be three orders of magnitude higher than current levels, and leachate concentrations would have to be four orders of magnitude higher than the maximum observed in the monitoring literature for TCEP to display risk via drinking water ingestion.

Risk Estimates for Conservative Scenarios
To help characterize risk EPA uses a range of central tendency and high-end estimates, as well as varying scenarios. EPA has more confidence in a risk estimate when risk is observed using conservative assumptions. In addition, EPA has more confidence in risk estimates when risk is not observed using fewer conservative assumptions. No risk observed with conservative parameters can build confidence that the OES/COU is not a risk to consumers or the general population. For example, drinking water risks were estimated for drinking water, diluted drinking water, incidental ingestion via swimming and drinking water contamination from landfill leachate. None of these scenarios resulted in chronic oral risk. Lifetime cancer risks were found for a few OESs (Incorporation into 1-part and 2-part reactive paints and coatings, Commercial use of paints and coatings, and Processing of 2-part resin articles); however, when adjusting for dilution to drinking water intake locations, these OESs no longer show lifetime cancer risk.

Due to the uncertainties in the BAF for walleye, EPA considered BAF values from all reviewed studies to capture a range conditions (see Section 2.12.2). Liu et al. (2019a) measured BAFs for multiple aquatic species in China and reported the lowest value of 109 to 202 L/kg for mud carp (*Cirrhinus molitorella*). Samples were collected from an e-waste polluted pond in South China. Risk estimates using this lowest BAF value (109 L/kg) still resulted in risks for fish consumption (see Table 5-70). Lastly, EPA's modeled surface water concentrations are generally several magnitudes higher than measured concentrations, thus resultant fish tissue concentrations and doses are high regardless of BAF. However, the Agency still relied on modeled data because of the paucity of measured data.

5.3.5.4 Hazard Values

EPA has moderate confidence in all hazard values used to modeled risks from TCEP. Although additional toxicity data can always help to refine the risk evaluation, EPA believes the available human studies and oral animal toxicity studies address the relevant endpoints for TCEP (*e.g.*, neurotoxicity, reproductive toxicity, developmental toxicity, other repeated-dose endpoints, carcinogenicity). There are uncertainties that are common to all values. EPA identified several epidemiology studies that added to the weight of scientific evidence, but the studies were not considered useful for dose-response analysis. Therefore, the Agency used TCEP values from oral toxicity studies in animals, and these values required extrapolation to inhalation and dermal hazard values. The impact of these uncertainties on the direction of risk (under- or overprediction) is unknown. Additional uncertainties specific to individual hazard values are described below, with details presented in Section 5.2.6.

Acute HED and HEC

Based on the weight of scientific evidence analysis of the reasonably available toxicity studies from animals, the key acute exposure effect is neurotoxicity. EPA identified a POD from high-quality acute animal toxicity study to calculate risks for acute exposure scenarios for TCEP. Tilson et al. (1990) identified neurotoxicity in female rats, and EPA concluded that these types of effects are likely to be caused by TCEP. EPA did not identify human data or other animal toxicity data using acute exposure durations, and there is uncertainty because the POD does not account for all the effects associated with acute exposure.

Intermediate/Chronic HED and HEC

EPA concluded that reproductive toxicity in humans is likely to be caused by TCEP and identified a high-quality 35-day study in adolescent male mice that identified decreases in seminiferous tubule numbers as the non-cancer POD for both intermediate and chronic exposure scenarios (Chen et al., 2015a). The observed effect is adverse and fertility due to male reproductive effects is known to be sensitive in humans. Using Chen et al. (2015a) for the POD is expected to be protective of other hazards (*e.g.*, neurotoxicity) for these exposure durations. There is uncertainty about the precision of the doses because Chen et al. (2015a) is a dietary study and the authors did not state the amount of food consumed. Using a 35-day toxicity study for chronic exposure durations adds some uncertainty (*e.g.*, the POD for the same effect may be lower after chronic exposure) but based on the weight of scientific evidence for other studies with male reproductive toxicity at higher doses and limited data from an unobtainable inhalation study that identified effects related to male reproductive toxicity and fertility, EPA believes the use of this study is relevant for the chronic duration.

Cancer CSF and IUR

Integrating evidence from humans, animals, and mechanistic studies resulted in a conclusion that TCEP is likely to cause cancer in humans under relevant exposure circumstances. EPA used a sensitive endpoint, kidney tumors in male rats, from a high-quality study (NTP, 1991b) to estimate cancer risks from exposure to TCEP. The increased incidence of renal tubule adenomas and carcinomas is considered adverse, relevant to humans, and representative of a continuum of benign to malignant tumors. Increased incidence of tumors was identified in one epidemiological study that identified an association between TCEP and thyroid tumors (Hoffman et al., 2017). Because NTP (1991b) identified primarily benign kidney tumors (adenomas), the incidence of malignant tumors is less certain. However, humans may be more sensitive and develop malignancies sooner than rats. Use of linear low dose extrapolation is also uncertain because direct mutagenicity is not likely to be the predominant MOA; thus, risks may be overpredicted using linear low dose extrapolation. Use of only kidney tumors could result in some underestimation of risk.

Table 5-82. Overall Confidence for Acute, Intermediate, and Chronic Human Health Non-cancer Risk Characterization for COUs Resulting in Risks[a] [b]

COU			Route/Exposed Group	Exposure Confidence	Hazard Confidence	Risk Characterization Confidence
Life Cycle Stage	**Category**	**Subcategory**				
Occupational						
Manufacturing	Import	Import	Dermal/Worker	++	++	Moderate
Processing	Processing – Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Dermal/Worker	++	++	Moderate
	Processing – Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Dermal/Worker	++	++	Moderate
	Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Dermal/Worker	++	++	Moderate
Commercial Use	Paints and coatings	Paints and coatings	Inhalation/Worker	++	++	Moderate
			Inhalation/ONU	+	++	Slight
			Dermal/Worker	++	++	Moderate
	Laboratory chemicals	Laboratory chemical	Dermal/Worker	++	++	Moderate
Consumer						
Consumer Use	Paints and coatings	Paints and coatings	N/A	N/A	++	N/A
	Furnishing, cleaning, treatment/care products	Fabric and textile products	Oral	++	++	Moderate
			Inhalation	++	++	Moderate
	Furnishing, cleaning, treatment/care products	Foam seating and bedding products	Oral	++	++	Moderate
	Construction, paint, electrical, and metal products	Building/construction materials	Inhalation	+	++	Slight
			Oral	++	++	Moderate

COU			Route/Exposed Group	Exposure Confidence	Hazard Confidence	Risk Characterization Confidence
Life Cycle Stage	**Category**	**Subcategory**				
	Construction, paint, electrical, and metal products	Building/construction materials – Wood and engineered wood products – Wood resin composites	Inhalation	++	++	Moderate
			Dermal	++	++	Moderate
Disposal	Disposal	Disposal	N/A	N/A	++	N/A
General population exposures						
Manufacturing	Import	Import	Oral	+	++	Slight
Processing	Processing – Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Oral	++	++	Moderate
	Processing – Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Oral	++	++	Moderate
	Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Oral	+	++	Slight
Commercial Use	Paints and coatings	Paints and coatings	Oral	++	++	Moderate
			Dermal	++	++	Moderate
	Laboratory chemicals	Laboratory chemical	Oral	+	++	Slight

[a] This table identifies COUs that have any non-cancer risk (acute, intermediate, or chronic) and the route associated with the risk.
[b] Intermediate risks were evaluated for workers only, not consumers or the general population.

Table 5-83. TCEP Evidence Table Summarizing Overall Confidence for Lifetime Human Health Cancer Risk Characterization for COUs Resulting in Risks

COUs			Route/Exposed Group	Exposure Confidence	Hazard Confidence	Risk Characterization Confidence
Life Cycle Stage	**Category**	**Subcategory**				
Occupational						
Manufacturing	Import	Import	Dermal/Worker	++	++	Moderate
Processing	Processing – Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Dermal/Worker	++	++	Moderate
	Processing – Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Dermal/Worker	++	++	Moderate
	Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Dermal/Worker	++	++	Moderate
Commercial Use	Paints and coatings	Paints and coatings	Inhalation/Worker	++	++	Moderate
			Inhalation/ONU	+	++	Slight
			Dermal/Worker	++	++	Moderate
	Laboratory chemicals	Laboratory chemical	Dermal/Worker	++	++	Moderate
Consumer						
Consumer Use	Paints and coatings	Paints and coatings	N/A	N/A	++	N/A
	Furnishing, cleaning, treatment/care products	Fabric and textile products	Oral	++	++	Moderate
			Inhalation	++	++	Moderate
	Furnishing, cleaning, treatment/care products	Foam seating and bedding products	Oral	++	++	Moderate
	Construction, paint, electrical, and metal products	Building/construction materials	Inhalation	+	++	Slight
			Oral	++	++	Moderate

COUs			Route/Exposed Group	Exposure Confidence	Hazard Confidence	Risk Characterization Confidence
Life Cycle Stage	**Category**	**Subcategory**				
	Construction, paint, electrical, and metal products	Building/construction materials - Wood and engineered wood products – wood resin composites	Inhalation	++	++	Moderate
			Dermal	++	++	Moderate
Consumer Use	Paints and coatings	Paints and coatings	N/A	N/A	++	N/A
	Furnishing, cleaning, treatment/care products	Fabric and textile products	Oral	++	++	Moderate
			Inhalation	++	++	Moderate
			Dermal	++	++	Moderate
	Furnishing, cleaning, treatment/care products	Foam seating and bedding products	Oral	++	++	Moderate
			Inhalation	++	++	Moderate
			Dermal	++	++	Moderate
	Construction, paint, electrical, and metal products	Building/construction materials	Oral	+	++	Slight
			Inhalation	+	++	Slight
			Dermal	+	++	Slight
	Construction, paint, electrical, and metal products	Building/construction materials – Wood and engineered wood products – Wood resin composites	Oral	++	++	Moderate
			Dermal	++	++	Moderate
Disposal	Disposal	Disposal	N/A	N/A	++	N/A
General population exposures						
Manufacturing	Import	Import	Oral	+	++	Slight
Processing	Processing – Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Oral	++	++	Moderate
	Processing – Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Oral	++	++	Moderate

COUs			Route/Exposed Group	Exposure Confidence	Hazard Confidence	Risk Characterization Confidence
Life Cycle Stage	**Category**	**Subcategory**				
	Processing – Incorporation into article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Oral	+	++	Slight
Commercial Use	Paints and coatings	Paints and coatings	Oral	++	++	Moderate
			Inhalation	++	++	Moderate
	Laboratory chemicals	Laboratory chemical	Oral	+	++	Slight

6 UNREASONABLE RISK DETERMINATION

TSCA section 6(b)(4) requires EPA to conduct a risk evaluation to determine whether a chemical substance presents an unreasonable risk of injury to health or the environment, without consideration of costs or other nonrisk factors, including an unreasonable risk to a PESS identified by EPA as relevant to this risk evaluation, under the COUs.

EPA has determined that TCEP presents an unreasonable risk of injury to health and the environment under the COUs. This unreasonable risk determination is based on the information in previous sections of this risk evaluation and the appendices and supporting documents in accordance with TSCA section 6(b). It is also based on TSCA's best available science (TSCA section 26(h)), weight of scientific evidence standards (TSCA section 26(i)), and relevant implementing regulations in 40 CFR part 702, including the amendments to the procedures for chemical risk evaluation under TSCA finalized in May 2024 (89 FR 37028; May 3, 2024).

EPA will initiate risk management for TCEP by applying one or more of the requirements under TSCA section 6(a) to the extent necessary so that TCEP no longer presents an unreasonable risk. The risk management requirements will likely focus on the COUs significantly contributing to the unreasonable risk. However, under TSCA section 6(a), EPA is not limited to regulating the specific COUs found to significantly contribute to unreasonable risk and may select from among a suite of risk management options related to manufacture, processing, distribution in commerce, commercial use, and disposal to address the unreasonable risk. For instance, EPA may regulate upstream COUs (*e.g.*, processing, distribution in commerce) to address downstream COUs that significantly contribute to unreasonable risk (*e.g.*, consumer use)—even if the upstream COUs are not significant contributors to the unreasonable risk. The Agency would also consider whether such risk may be prevented or reduced to a sufficient extent by action taken under another federal law, such that referral to another agency under TSCA section 9(a) or use of another EPA-administered authority to protect against such risk pursuant to TSCA section 9(b), as appropriate.

The COUs evaluated for TCEP are listed in Table 1-1. The following COUs significantly contribute to the unreasonable risk:

- Manufacturing (Import);
- Processing – Incorporation into formulation, mixture, or reaction product – Paint and coating manufacturing;
- Processing – Incorporation into formulation, mixture, or reaction product – Polymers used in aerospace equipment and products;
- Processing – Incorporation into article – Aerospace equipment and products and automotive articles and replacement parts containing TCEP;
- Industrial use – Paints and coatings;
- Commercial use – Paints and coatings;
- Commercial use – Laboratory chemicals;
- Consumer use – Furnishing, cleaning, treatment/care products – Fabric and textile products;
- Consumer use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products; and
- Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites.

The following COUs do *not* significantly contribute to the unreasonable risk:

- Processing – Recycling;
- Distribution in commerce;
- Industrial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP;
- Commercial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP;
- Commercial use – Furnishing, cleaning, treatment/care products – Fabric and textile products;
- Commercial use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products;
- Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation;
- Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites;
- Consumer use – Paints and coatings, including those found on automotive articles and replacement parts;
- Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation; and
- Disposal.

Because TCEP production volumes and uses have declined, and no companies reported manufacture or import of TCEP in the 2020 CDR, EPA had limited data available to evaluate certain COUs. In determining whether COUs that the Agency had limited information significantly contributed to the unreasonable risk of TCEP, EPA integrated reasonably available information in a qualitative risk characterization using professional judgement of read-across evidence. The qualitative analyses are a best estimate of what EPA expects given the weight of scientific evidence without overstating the science. Environmental and human health risk characterizations for those COUs with limited data are in Sections 4.3.6.2 and 5.3.2. of this risk evaluation. Additional explanation regarding the qualitative risk characterizations and EPA's conclusion about whether the COU significantly contributes to unreasonable risk are included in Sections 6.1.4, 6.1.5, and 6.1.6. The COUs that significantly contribute to unreasonable risk from TCEP are based on risk estimates that assume a production volume of 2,500 lb, which EPA has estimated, based on the reasonably available data, is reflective of current domestic TCEP use.

Whether EPA makes a determination of unreasonable risk for a particular chemical substance under TSCA depends upon risk-related factors beyond exceedance of benchmarks, such as the endpoint under consideration, the reversibility of effect, exposure-related considerations (*e.g.*, duration, magnitude, frequency of exposure, population exposed), and the confidence in the information used to inform the hazard and exposure values. For COUs evaluated quantitatively, to determine if a COU contributed significantly to unreasonable risk, EPA compared the risk estimates of the scenario used to evaluate the COUs and considered whether the risk from the COU was best represented by the central tendency or high-end risk estimates. Additionally, in the risk evaluation, the Agency describes the strength of the scientific evidence supporting the human health and environmental assessments as robust, moderate, or slight. Robust confidence suggests thorough understanding of the scientific evidence and uncertainties, and the supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the exposure estimate. Moderate confidence suggests some understanding of the scientific evidence and uncertainties, and the supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize exposure estimates. Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the Agency is making the best scientific assessment possible in the

absence of complete information. This risk evaluation discusses important assumptions and key sources of uncertainty in the risk characterization, and these are described in more detail in the respective weight of scientific evidence conclusions sections for fate and transport, environmental release, environmental exposures, environmental hazards, and human health hazards. It also includes overall confidence and remaining uncertainties sections for human health and environmental risk characterizations.

In the TCEP unreasonable risk determination, EPA has considered risk estimates with an overall confidence rating of slight, moderate, or robust, and the Agency considered COUs with indeterminate exposures and COUs with limited reasonably available information. In general, EPA makes an unreasonable risk determination based on risk estimates that have an overall confidence rating of moderate or robust—because those confidence ratings indicate the scientific evidence is adequate to characterize risk estimates despite uncertainties or is such that it is unlikely the uncertainties could have a significant effect on the risk estimates (see Appendix F.2.3.1).

6.1 Unreasonable Risk to Human Health

Calculated risk estimates (MOEs or cancer risk estimates) can provide a risk profile of TCEP by presenting a range of estimates for different health effects for different COUs. When characterizing the risk to human health from occupational exposures during risk evaluation under TSCA, EPA conducts baseline assessments of risk and makes its determination of unreasonable risk from a baseline scenario that does not assume use of respiratory protection or other PPE.[43] A calculated MOE that is less than the benchmark MOE supports a determination of unreasonable risk of injury to health, based on non-cancer effects. Similarly, a calculated cancer risk estimate that is greater than the cancer benchmark supports a determination of unreasonable risk of injury to health from cancer. It is important to emphasize that these calculated risk estimates alone are not bright-line indicators of unreasonable risk.

6.1.1 Populations and Exposures EPA Assessed to Determine Unreasonable Risk to Human Health

EPA has evaluated risk to workers, including ONUs and male and female adolescents and adults (≥16 years old); consumer users; the general population; and infants via human milk from exposed individuals, using reasonably available monitoring and modeling data for inhalation, dermal, and ingestion exposures, as applicable. EPA has evaluated risk from inhalation and dermal exposure of TCEP to workers as well as inhalation exposures to ONUs. The Agency also has evaluated risk from oral, dermal, and inhalation exposures to consumers. For the general population, EPA has evaluated risk from (1) ingestion exposures via drinking water, incidental surface water ingestion, fish ingestion (including subsistence fishers), and soil ingestion by children; (2) dermal exposures to swimmers and children playing in the mud and other activities with soil; and (3) chronic inhalation exposure. For infants consuming the human milk of exposed individuals, EPA has evaluated risk from milk ingestion based on milk concentrations modeled for maternal exposures associated with occupational, consumer, and general population COUs. Descriptions of the data used for human health exposure and human health hazards are provided in Sections 5.1 and 5.2 of this risk evaluation. Uncertainties for overall exposures and hazards are presented in Section 5.3.5 and are summarized in Table 5-82 and Table 5-83 and are considered in the unreasonable risk determination.

6.1.2 Summary of Human Health Effects

EPA has determined that the unreasonable risk presented by TCEP is due to

[43] It should be noted that, in some cases, baseline conditions may reflect certain mitigation measures, such as engineering controls, in instances where exposure estimates are based on monitoring data at facilities that have engineering controls in place.

- Non-cancer effects and cancer in workers from dermal and inhalation exposures;
- Non-cancer effects and cancer in consumers from ingestion, dermal, and inhalation exposures; and
- Non-cancer effects and cancer in the general population, including subsistence and Tribal fishers from fish consumption (ingestion).

With respect to health endpoints upon which EPA has based this unreasonable risk determination, the Agency has moderate overall confidence in the following PODs: (1) acute neurotoxicity, (2) intermediate and chronic male reproductive effects, (3) acute and chronic kidney effects, and (4) kidney cancer. Only *likely* evidence integration conclusions, or those PODs with moderate overall confidence, were considered for dose-response. The confidence on the PODs is explained in Section 5.2.6 and Appendix L. EPA's exposure and overall risk characterization confidence levels varied and are summarized in Table 5-82 and Table 5-83.

The health risk estimates for workers, ONUs, consumers, the general population, and infants through the milk pathway are presented in Section 5.3.2. For consumer and general population exposures, risk estimates are provided in Section 5.3 of this risk evaluation only when MOEs were less than the benchmark MOEs for non-cancer effects or when cancer risks exceeded benchmark risk levels of 1 in 1,000,000 (1×10^{-6}). A complete list of health risk estimates for consumers and the general population is in the following supplemental files of the risk evaluation (see also Appendix C): *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Exposure E-FAST 2014 Surface Water Modeling Inputs, Flow Data, and General Population Exposure Estimates and Risk Calculations* (U.S. EPA, 2024g), *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Exposure Air Concentration Risk Calculations* (U.S. EPA, 2024j), and *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Consumer Exposure Modeling Results for TCEP* (U.S. EPA, 2024f).

6.1.3 Basis for EPA's Determination of Unreasonable Risk to Human Health

In developing the exposure and hazard assessments for TCEP, EPA has analyzed reasonably available information to ascertain whether some human populations may have greater exposure and/or susceptibility than the general population to the hazard posed by TCEP. For the TCEP risk evaluation, EPA has accounted for the following PESS: infants exposed through human milk from exposed individuals, children and male adolescents who use consumer articles that contain TCEP or are among the exposed general population, subsistence fishers, Tribal populations, pregnant women, workers and consumers who experience aggregate or sentinel exposures, people who live in fenceline communities that are near facilities that emit TCEP, and firefighters (see Section 5.3.3, Table 5-79, and Appendix E).

EPA has slight confidence in the inputs to the calculation of the risk estimates for infants ingesting human milk from exposed individuals and cannot determine that the human milk pathway significantly contributes to the unreasonable risk of TCEP for any COU (see Section 5.1.3.4.7 and Appendix I.5.5). There are no COUs that showed higher risk estimates in the infants compared to the mothers; in fact, some COUs resulted in maternal doses and risk estimates that are one to two orders of magnitudes higher for the mothers than the infants. Therefore, EPA has moderate confidence that protecting the mother will also protect the infant from exposure via human milk.

Risk estimates based on high-end exposure levels (*e.g.*, 95th percentile) are generally intended to cover individuals with sentinel exposure levels, whereas risk estimates at the central tendency exposure are generally estimates of average or typical exposures. EPA has aggregated exposures across certain routes for consumers and identified at least two COUs where aggregating exposures across routes resulted in

risk where there was not risk when considering a single route. EPA has not aggregated exposures across consumer COUs, because each COU already presented chronic risk to consumers. Because risk to the general population was driven by sentinel exposures via fish ingestion, EPA has not aggregated risk across routes or exposure scenarios for this population. EPA has not characterized aggregate risk to workers. There were no instances of aggregate lifetime risk for any COU where there was not already risk from the COU from an individual route (see Section 5.3.4). The UF of 10 for human variability that EPA has applied to MOEs accounts for increased susceptibility of populations, such as children and elderly populations. EPA also generally relies on high-end exposure levels to make an unreasonable risk determination to capture vulnerable populations that are expected to have higher exposures. The non-cancer PODs are based on susceptible populations. The acute POD is based on effects observed during pregnancy, and the intermediate and chronic PODs are based on reproductive effects observed in adolescent males. For cancer, although there is likely to be variability in susceptibility across the human population, EPA has not identified specific human groups that are expected to be more susceptible to cancer following TCEP exposure. More information on how EPA characterized sentinel and aggregate risks is provided in Section 5.3.4.

6.1.4 Workers

Based on the occupational risk estimates and related risk factors, EPA has determined that cancer and non-cancer effects from worker dermal exposure to TCEP in occupational settings for all COUs with quantified risk estimates and worker inhalation exposure to TCEP from commercial and industrial use of paints and coatings significantly contribute to the unreasonable risk presented by TCEP. More information on occupational risk estimates is in Section 5.3.2.1 of this risk evaluation. One COU, Processing – Recycling, has a cancer risk estimate for the high end at the benchmark for workers due to inhalation; however, the central tendency is below the benchmark. In general, TCEP manufacturing and processing was phased out starting in the late 1980s or early 1990s in favor of other flame retardants or flame-retardant formulations, and the overall TCEP releases from the Processing – Recycling COU are expected to continue to be lower over time. Therefore, EPA considered the central tendency as the best way to represent the contribution to unreasonable risk from this COU and determined that Processing – Recycling does not significantly contribute to the unreasonable risk of cancer to workers from inhalation exposure to TCEP.

EPA has determined that firefighters may be at increased risk of TCEP exposures during structure fires; therefore, exposures to firefighters contribute to the unreasonable risk presented by TCEP. However, EPA was not able to identify which specific COUs or pathway of exposures lead to the elevated TCEP exposures during firefighting activities.

EPA used a Fractional Absorption Model to estimate dermal exposure to TCEP in occupational settings. The model assumes a single exposure event per day and does not address variability in exposure duration and frequency. However, even with these uncertainties and limitations, EPA has considered the weight of scientific evidence for dermal risk estimates generated by the model to be sufficient for determining whether a COU significantly contributes to unreasonable risk. More information on EPA's confidence in these risk estimates and the uncertainties associated with them can be found in Section 5.1.1.4 of this risk evaluation.

The following occupational COUs do not have quantitative risk estimates for workers. However, EPA has qualitatively evaluated the COUs by integrating limited amounts of reasonably available information using professional judgement of read-across evidence. The qualitative analyses are a best estimate of what EPA expects given the weight of scientific evidence without overstating the science (see Section 5.3.2.1.2):

- Distribution in commerce: EPA expects TCEP to be transported in sealed containers from import sites to downstream processing and use sites, or for final disposal of TCEP. EPA expects under standard operating procedures that exposures and releases that could occur during distribution in commerce would not lead to a significant contribution to the unreasonable risk presented by TCEP.
- Commercial use – Furnishing, cleaning, treatment/care products – Fabric and textile products; Commercial use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products; Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation; Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites: TCEP was used for these purposes in the past, but these COUs were phased out beginning in the late 1980s or early 1990s and replaced by other flame retardants or flame-retardant formulations. EPA assumes that any of these products still used commercially represent a fraction of the overall amount of TCEP previously used for these purposes. Although there might be some workers that are exposed to those older products, the consumer assessment for these articles resulted in no indication of consumer risk for inhalation, ingestion, or dermal exposures for adults for these COUs. Therefore, EPA has moderate confidence that under similar exposure durations and exposure frequencies, the Agency expects these articles to pose no commercial risk for inhalation, ingestion, or dermal exposures to commercial workers who use articles in a similar fashion to consumers for these COUs. Thus, these COUs do not significantly contribute to the unreasonable risk presented by TCEP.
- Disposal: For the commercial uses that have been phased out, any currently used products that contain TCEP are expected to be disposed in landfills but will represent just a fraction of previous amounts from when TCEP was used more widely. And similarly, only a small portion of e-waste is expected to contain TCEP. However, although some releases and exposures could occur during the disposal of the wide variety of items into which TCEP has been incorporated, these are expected to be minimal and dispersed, and not expected to significantly contribute to the unreasonable risk presented by TCEP.

6.1.5 Consumers

Based on the consumer risk estimates and related risk factors, EPA has found that the Consumer use – Furnishing, cleaning, treatment/care products – Fabric and textile products COU significantly contributes to lifetime cancer risks from inhalation and oral exposures. In addition, the Consumer use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products COU significantly contributes to lifetime cancer risk due to dermal and oral exposures. The Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites COU significantly contributes to the lifetime cancer risk from inhalation, dermal, and oral exposures. The cancer risk estimates represent exposures at younger life stages that may significantly contribute to the lifetime cancer risk.

In addition, EPA has found that non-cancer effects to infants through age 2 from ingestion of dust and mouthing of articles covered by the Consumer use – Furnishing, cleaning, treatment/care products – Fabric and textile products and foam seating and bedding products COUs, as well as from ingestion of dust contaminated with TCEP from other articles in the home covered by the Consumer use – construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites consumer COU, significantly contribute to the unreasonable risk presented by TCEP.

Additionally, dermal contact with TCEP from the Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites COU significantly contributes to chronic risk for infants and children.
For adults, risk estimates from acute inhalation exposures to TCEP are below the MOE for two COUs: Consumer use – Furnishing, cleaning, treatment/care products – Fabric and textile products and Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites. Inhalation risks from these COUs primarily occur within the first few weeks after the article is produced or painted due to off-gassing of TCEP. Because of this, EPA has not anticipated a significant contribution to the unreasonable risk via inhalation of TCEP from TCEP-containing products that have already been in commerce longer than the off-gassing period, but the Agency has considered acute inhalation exposures from newer or imported products containing TCEP. In the case of the Consumer use – Furnishing, cleaning, treatment/care products – Fabric and textile products COU, TCEP incorporation into those consumer articles was mostly phased out beginning in the late 1980s or early 1990s and replaced by other flame retardants or flame-retardant formulations. EPA has concluded that this COU does not significantly contribute to the unreasonable risk of TCEP due to acute inhalation by adults from older articles in the home that may have already undergone off-gassing of TCEP; thus, the Agency does not expect acute inhalation exposure to adults from consumer fabric and textile articles containing TCEP unless they are newer articles (*i.e.*, imported into the United States for use by infants and young children. EPA has reasonably available information that imported fabric and textile articles expected to be used by individuals in this age group could contain TCEP (Section 5.1.1.2). EPA is concluding that the fabric and textile products COU significantly contributes to the unreasonable risk of TCEP due to acute inhalation by adults who are exposed to newer, imported articles containing TCEP. Additionally, in the case of the Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites, consumers may obtain new wood resin composite products containing TCEP by importing them, therefore, EPA has concluded that this COU significantly contributes to the unreasonable risk of TCEP due to acute inhalation by adults.

Two consumer COUs, Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation and Consumer use – Paints and coatings, including those found on automotive articles and replacement parts, were found to not significantly contribute to the unreasonable risk of TCEP because of the Agency's slight confidence in their risk estimates. Further discussion can be found below and in Sections 5.3.2.2.1 and 5.3.2.2.2.

EPA's overall confidence in the acute, intermediate, and chronic consumer inhalation, ingestion, and dermal exposure risk estimates ranges from slight to moderate—although only those risk estimates with overall confidence of moderate or robust were considered in the unreasonable risk determination. More information on the consumer analysis can be found in Sections 3.2.2, 3.4, 5.1.2, and 5.3.2.2 of the risk evaluation.

The following consumer COU and associated disposal do not have quantitative risk estimates. However, EPA has qualitatively evaluated the COUs by integrating limited amounts of reasonably available information using professional judgement of read-across evidence. The qualitative analyses are a best estimate of what EPA expects given the weight of scientific evidence without overstating the science. The qualitative analyses are a best estimate of what the Agency expects given the weight of scientific evidence without overstating the science (see Section 5.3.2.2.2).

- Consumer use – Paints and coatings, including those found on automotive articles and replacement parts: Consumers are unlikely to obtain TCEP containing paints and coatings, including those found on automotive articles and replacement parts, because the domestic retail

production and manufacturing of TCEP containing paints and coatings has ceased and TCEP containing paints and coatings represent a small fraction of the paints and coatings on the market. For consumer application of paints and coatings, EPA has determined that it is not reasonably foreseen for consumers to obtain TCEP containing paints and coatings products that are available for commercial applications and therefore does not expect exposure to consumers from the application of TCEP containing paints and coatings. For consumer use of articles containing dried paints and coatings, including those found on automotive articles and replacement parts, EPA expects the exposure scenario to mirror the other consumer use articles (*e.g.,* wood resin articles) scenarios assessed in Section 5.3.2.2.1. EPA's consumer analysis for articles containing TCEP resulted in no chronic inhalation, ingestion, or dermal risk for adults for the COUs with moderate confidence. However, the consumer analysis did reveal chronic dermal risk for wood resin composites and ingestion risk for multiple articles for infants and children. Although the analysis revealed dermal and ingestion risk to infants and children from the use of articles containing dried paints and coatings due to its similarity to the other consumer article scenarios (*e.g.,* wood resin articles), EPA's confidence in the analysis of the consumer risk from articles with TCEP containing paint is slight. Therefore, EPA has determined this COU does not significantly contribute to the unreasonable risk to consumers presented by TCEP.

- Disposal : Consumers may be exposed to articles containing TCEP during disposal and the handling of waste. The removal of articles in DIY scenarios may lead to direct contact with articles and the dust generated from the articles. EPA believes that the monitoring data found for the commercial COU of e-waste recycling would represent similar exposures that could occur during the removal and/or disposal of other articles containing TCEP. Risk to workers was not found during these activities and therefore it is not expected that risk would be found in a DIY scenario involving the removal and/or disposal of TCEP containing articles. Therefore, this COU does not significantly contribute to the unreasonable risk to consumers presented by TCEP.

6.1.6 General Population

EPA has identified the following exposure routes as significantly contributing to the unreasonable risk of TCEP for the following sub-populations:

Fish Ingestion

Based on the risk estimates for the general population, including subsistence fishers[44], Tribal fishers, and other related risk factors, EPA has determined that fish ingestion by the general population, including subsistence fishers and Tribal fishers, for all COUs evaluated quantitatively contribute to the unreasonable risk due to cancer. One COU significantly contributes to the unreasonable risk due to chronic non-cancer effects due to general population fish ingestion. Additionally, EPA has determined that three COUs significantly contribute to the unreasonable risk due to acute non-cancer effects for subsistence fishers in the general population, and four COUs significantly contribute to the unreasonable risk due to chronic non-cancer effects for subsistence fishers in the general population.

To make a determination of unreasonable risk based on fish consumption, EPA used a BAF of 109 L/kg and an ingestion rate of 22.2 g/day (142.4 g/day for subsistence fishers) for adults aged 16 to less than 70 years to calculate risk estimates (see Section 5.1.3.4.3 and Table_Apx I-2). EPA's confidence in the risk estimates using the BAF of 109 L/kg is moderate. Acute and chronic non-cancer risk estimates to the general population for oral fish ingestion are provided in Table 5-71 and Table 5-73 of this risk evaluation. Cancer risk estimates for oral fish ingestion are in Table 5-75 of this risk evaluation.

[44] Subsistence fishers represent a PESS group for TCEP due to their increased exposure via fish ingestion (142.4 g/day compared to a high-end of 22.2 g/day for the general population).

EPA has not determined whether Tribal populations were exposed to fish containing TCEP; however, the Agency estimated Tribal ingestion rates based on available information, resulting in all COUs quantitatively evaluated significantly contributing to the unreasonable risk due to acute and chronic non-cancer effects and cancer. EPA has also estimated risk based on fish consumption at heritage rates. Heritage rates refer to those that existed prior to non-indigenous settlement on Tribal fishery resources, as well as changes in culture and lifeways (see Section 5.1.3.4.4). Additionally, based on the risk estimates for adults, EPA has estimated that TCEP presents unreasonable risk of acute and chronic non-cancer effects and cancer for children aged 15 years or less who consume fish tissue contaminated with TCEP due to their higher rate of ingestion per kg of body weight.

Additionally, EPA has evaluated the following sub-populations and routes of exposure but did not identify any significant contribution to the unreasonable risk of TCEP from these routes:

Drinking Water and Incidental Surface Water Ingestion
EPA has estimated that ingestion of drinking water (diluted), drinking water from groundwater contaminated with TCEP leaching from landfills, and incidental surface water ingestion during swimming do not significantly contribute to the unreasonable risk of TCEP for any COU. Acute oral non-cancer risk estimates for drinking water and drinking water (diluted) ingestion for any age group (*i.e.*, adults ≥21, youths 16–20, youths 11–15, children 6–10, and toddlers 1–5 years) are presented in Table 5-70 of this risk evaluation. Chronic non-cancer risk estimates for drinking water and incidental surface water ingestion are provided in Table 5-72; cancer risk estimates from drinking water are presented in Table 5-74.

Soil Ingestion
EPA has estimated that chronic soil ingestion does not significantly contributes to the unreasonable risk of TCEP for any COU. EPA's confidence in the risk estimates at 1,000 m is moderate. Chronic non-cancer risk estimates for soil ingestion are presented in Table 5-72 of this risk evaluation.

Incidental Dermal from Swimming
EPA has estimated that incidental dermal exposure to an adult swimming does not significantly contribute to the unreasonable risk of TCEP for any COU. Dermal acute and chronic non-cancer risk estimates for swimming are provided in Table 5-76 of this risk evaluation. EPA's confidence in the risk estimates is moderate.

Children's Dermal Exposure from Playing in Mud and Soil Activities
EPA has estimated that chronic dermal exposure to children 3 to 6 years old playing in mud and conducting soil activities does not significantly contribute to the unreasonable risk of TCEP for any COU. EPA's confidence in the risk estimates at 1,000 m is moderate. Dermal, chronic non-cancer risk estimates for children playing in mud and soil activities are included in Table 5-76 of this risk evaluation.

Inhalation
EPA has calculated risk estimates for one COU, Commercial use – Paints and coatings. Chronic inhalation non-cancer risk estimates indicating no risk for even the very conservative distance of 10 m are in Table 5-77. Cancer risk estimates do not indicate risk for the distance between 100 to 1,000 m. Cancer inhalation risk estimates are presented in Table 5-78 of this risk evaluation.

The following COUs do not have quantitative risk estimates for the general population. However, EPA has qualitatively evaluated the COU by integrating limited amounts of reasonably available information

using professional judgement of read-across evidence. The qualitative analyses are a best estimate of what EPA expects given the weight of scientific evidence without overstating the science (see Section 5.3.2.3.2):

- Processing – Recycling: EPA has not found reasonably available data to quantify environmental releases of TCEP from e-waste facilities. The total releases are expected to be low because TCEP is not typically used in electronics. For commercial and consumer COUs evaluated qualitatively, according to literature sources, TCEP was used for these commercial and consumer COUs in the past, but manufacturing and processing was phased out starting in the late 1980s or early 1990s in favor of other flame retardants or flame-retardant formulations. The Agency assumes that commercial and consumer products with TCEP that are still in use, but are no longer manufactured or processed, represents a fraction of the overall amount of TCEP previously used. Therefore, TCEP releases from this COU (Processing – Recycling) is expected to be lower than those associated with COUs already quantified in this risk evaluation, and this COU does not significantly contribute to the unreasonable risk of TCEP.
- Distribution in commerce: TCEP production volumes have declined and recent reports (*e.g.,* the 2020 CDR cycle) indicate that production volumes may be below reporting levels; therefore, the precise volume is unknown. The general decline in production volume would logically lead to decreased distribution into commerce. Therefore, exposure and risk would also likely have declined with time. Exposure is possible from ongoing manufacturing, processing, industrial, and commercial uses. Nevertheless, given that TCEP and/or TCEP containing products or articles are expected to be transported in sealed containers or packages; EPA anticipates that exposure and releases to the general population during distribution in commerce will be negligible and not lead to a significant contribution to the unreasonable risk presented by TCEP.
- Commercial use – Furnishing, cleaning, treatment/care products – Fabric and textile products; Commercial use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products; Commercial use – Construction, paint, electrical, and metal products – building/construction materials – Insulation; and Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites: These COUs were being phased out beginning in the late 1980s or early 1990s and replaced by other flame retardants or flame-retardant formulations. EPA did not locate data to estimate (1) the amount of TCEP that was historically used in these products, (2) the amounts of these products that have already reached the end of their service life, or (3) the amounts of these products that have already been disposed of. Based on the years that the phase-out occurred, many of these products are not likely to be in use because the end of their service life was already reached (*e.g.,* commercial roofing has an estimated lifespan of 17 to 20 years). EPA assumes that any of these products still used commercially represent a fraction of the overall amount of TCEP previously used for these purposes. Therefore, releases to the environment from these commercial uses would also represent only a fraction of previous release amounts and these COUs do not significantly contribute to the unreasonable risk of TCEP.
- Industrial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP; and Commercial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP: After TCEP-containing resins have cured within products that are installed, EPA expects TCEP releases and dermal exposures will be limited by TCEP being entrained into the hardened polymer matrix. During installation it is possible that very small levels of dust could be generated, these were quantified in Table 5-67. and do not indicate risk to workers from inhalation nor do they indicate the generation of significant dust releases occurring. Releases may occur via the mechanism of

blooming (volatilization from the cured resin surface) during the service life of the article, but EPA expects that such releases during installation will be negligible (OECD, 2009; NICNAS, 2001). Installation of aerospace equipment and products would be installed without any type of further processing of the article that would lead to potential releases (sanding, drilling, etc.). Therefore, the potential risk to the general population from releases during installation of TCEP-containing articles is low. EPA has concluded these COUs do not significantly contribute to the unreasonable risk presented by TCEP.

- Disposal: Disposal is possible throughout the life cycle of TCEP and TCEP-containing products, including waste treatment and disposal resulting from manufacturing, processing, commercial and consumer uses. A robust discussion of the qualitative analyses done to estimate risk to the general population from the disposal of TCEP and TCEP-containing products can be found in Section 5.3.2.3.2. EPA qualitatively discussed releases to landfills or incinerators, surface water, groundwater, and considered releases due to e-waste recycling, end-of-life disposal, demolition, and down-the-drain releases. The Agency acknowledges that although some releases and exposures could occur during the disposal of the wide variety of items that TCEP has been incorporated into, these exposures are expected to be negligible, and not expected to significantly contribute to the unreasonable risk presented by TCEP.

6.2 Unreasonable Risk to the Environment

Calculated RQs can provide a risk profile by presenting a range of estimates for different environmental hazard effects for different COUs. An RQ equal to 1 indicates that the exposures are the same as the concentration that causes effects. An RQ less than 1, when the exposure is less than the effect concentration, generally indicates that there is not risk of injury to the environment that would support a determination of unreasonable risk for the chemical substance. An RQ greater than 1, when the exposure is greater than the effect concentration, generally indicates that there is risk of injury to the environment that would support a determination of unreasonable risk for the chemical substance. Additionally, if a chronic RQ is 1 or greater, the Agency evaluates whether the chronic RQ is 1 or greater for 30 days or more based on the exposure period of the hazard toxicity tests before making a determination of unreasonable risk.

6.2.1 Populations and Exposures EPA Assessed to Determine Unreasonable Risk to the Environment

For aquatic organisms, EPA has evaluated exposures via surface water and sediment (including pore water). For terrestrial organisms, the Agency has evaluated exposures via soil, air, and surface water. EPA has assessed terrestrial organism exposures from air deposition of TCEP to soil. Additionally, the Agency has estimated terrestrial organism exposures from trophic transfer of TCEP from soil and surface water.

6.2.2 Summary of Environmental Effects

EPA has determined that all five COUs assessed quantitatively, and one COU assessed qualitatively, significantly contribute to the unreasonable risk presented by TCEP due to chronic effects (mortality of yellow catfish) using empirical fish data.

Risks to terrestrial organisms from air deposition to soil and from trophic transfer from the five COUs quantitatively assessed do not significantly contribute to the unreasonable risk to the environment presented by TCEP.

6.2.3 Basis for EPA's Determination of Unreasonable Risk of Injury to the Environment

Consistent with EPA's determination of unreasonable risk to human health, the RQ is not treated as a bright-line and other risk-based factors may be considered (*e.g.,* confidence in the hazard and exposure characterization, duration, magnitude, uncertainty) for purposes of making an unreasonable risk determination. TCEP is described as a "ubiquitous" contaminant because it is commonly found in various environmental compartments such as outdoor air, surface water, drinking water, groundwater, soil, sediment, biota, and precipitation all over the world (see Section 3). Additionally, TCEP is persistent in water, soil, and sediment, and EPA has robust confidence that TCEP can undergo long-range transport.

EPA has moderate confidence in the acute and chronic aquatic hazards and aquatic exposures significantly contributing to unreasonable risk. Additionally, the Agency has robust to slight confidence in the terrestrial exposures and hazards, which do not significantly contribute to unreasonable risk. EPA has determined the terrestrial food web to be the driver of exposure and does not expect exposure to TCEP via air or surface water to significantly contribute to unreasonable risk to terrestrial organisms. Similarly, EPA does not expect exposure to TCEP via biosolids to significantly contribute to unreasonable risk to the environment. The Agency's overall environmental risk characterization confidence levels were varied and are summarized in Table 4-23 of this risk evaluation.

In making a determination of unreasonable risk, EPA has considered aggregating environmental exposures for aquatic and terrestrial organisms but did not because the surface water and sediment pathways for aquatic organisms and the soil pathway for terrestrial organisms were such significant contributors to unreasonable risk (see Section 4.3.6.1).

The following COUs do not have quantitative risk estimates for the environment. However, EPA has qualitatively evaluated the COUs by integrating limited amounts of reasonably available information using professional judgement of read-across evidence. The qualitative analyses are a best estimate of what EPA expects given the weight of scientific evidence without overstating the science (see Section 4.3.6.2):

- Processing – Recycling: As noted before, the total releases are expected to be low because TCEP is not typically used in electronics. Therefore, TCEP releases for this COU are expected to be lower than those associated with COUs with quantified environmental risk estimates for terrestrial receptors, with the Recycling COU expected to have lower risk than the quantified COUs. Thus, EPA does not expect this COU to significantly contribute to the unreasonable risk presented by TCEP to the environment.
- Distribution in commerce: EPA expects TCEP to be transported in sealed containers from import sites to downstream processing and use sites. Under standard operating procedures, the Agency expects that environmental releases from sealed containers are not expected to occur. Transportation of TCEP and TCEP containing articles for final disposal would be less than the releases expected from disposal, which are expected to be minimal and dispersed; therefore, distribution in commerce is not expected to significantly contribute to the unreasonable risk presented by TCEP to the environment.
- Industrial use – Paints and coatings: Similar to the occupational exposures, EPA expects modeled environmental releases to be similar to the OESs previously assessed for other industry sectors. This means that that the risk estimates for this COU would be similar to the commercial use of paints and coatings; therefore, this COU also significantly contributes to the unreasonable risk presented by TCEP to the environment.

- Industrial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP and Commercial use – Other use – Aerospace equipment and products and automotive articles and replacement parts containing TCEP: EPA does not expect significant releases to the environment to occur during the installation of TCEP-containing aircraft, aerospace, or automotive articles into or onto the relevant transportation equipment. After TCEP-containing resins have cured, EPA expects TCEP release will be limited by the hardened polymer matrix. Therefore, the Agency does not expect these COUs to significantly contribute to the unreasonable risk presented by TCEP to the environment.
- Commercial use – Furnishing, cleaning, treatment/care products – Fabric and textile products; Commercial use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products; Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation; and Commercial use – Construction, paint, electrical, and metal products – Building/construction materials – Wood and engineered wood products – Wood resin composites: EPA has confirmed from literature sources that TCEP was used for these purposes in past decades. However, these commercial uses began phasing out beginning in the late 1980s or early 1990s and were replaced by other flame retardants or flame-retardant formulations. However, because TCEP releases are expected to be lower relative to other quantified scenarios, these commercial COUs would be expected to have lower risk than the quantified COUs. Therefore, EPA does not expect these COUs to significantly contribute to the unreasonable risk presented by TCEP to the environment.
- Consumer use – Furnishing, cleaning, treatment/care products – Fabric and textile products; Consumer use – Furnishing, cleaning, treatment/care products – Foam seating and bedding products; Consumer use – Construction, paint, electrical, and metal products – Building/construction materials – Insulation; Consumer use – Construction, paint, electrical, and metal products – Building/construction materials wood and engineered wood products – Wood resin composites; and Consumer use – Paints and coatings, including those found on automotive articles and replacement parts: Consumer releases to the environment are expected to be less than occupational releases; wastewater concentrations from manufacturing, commercial and processing COUs were shown to be significantly lower than the acute and chronic COCs identified in Section 4.2. Therefore, EPA does not expect these COUs to significantly contribute to the unreasonable risk presented by TCEP to the environment.
- Disposal: As noted before, although some environmental releases could occur during the disposal of the wide variety of items that TCEP has been incorporated in to, these releases are expected to be minimal and dispersed and are not expected to significantly contribute to the unreasonable risk presented by TCEP to the environment.

6.3 Additional Information Regarding the Basis for the Unreasonable Risk Determination

Table 6-1, Table 6-2, and Table 6-3 summarize the basis for this unreasonable risk determination of injury to human health and the environment presented in this TCEP risk evaluation for those COUs with a qualitative evaluation. In these tables, a checkmark (✓) indicates how the COU significantly contributes to the unreasonable risk by identifying the type of effect (*e.g.*, non-cancer and cancer for human health; acute or chronic environmental effects) and the exposure route to the population or receptor that results in such significant contribution. Not all COUs, exposure routes, or populations or receptors evaluated are included in the tables. The tables only includes the relevant exposure route, or the population or receptor that supports the conclusion that the COU significantly contributes to the TCEP unreasonable risk determination. As explained in Section 6.2, for this unreasonable risk

determination, EPA has considered the effects of TCEP to human health at the central tendency and high-end, as well as effects of TCEP to human health and the environment from the exposures associated from the condition of use, risk estimates, and uncertainties in the analysis. See Section 5.3.2 for a summary of risk estimates.

Table 6-1. Supporting Basis for the Unreasonable Risk Determination for Human Health (Occupational COUs)

COU			Population	Exposure Route	Human Health Effects			
Life Cycle Stage	Category	Subcategory			Acute Non-cancer	Intermediate Non-cancer	Chronic Non-cancer	Lifetime Cancer
Manufacturing	Import	Import	Worker	Dermal	✓	✓	✓	✓
			General Population	Fish Ingestion		N/A		✓
			General Population – Subsistence Fishers	Fish Ingestion		N/A	✓	✓
			General Population – Tribal Fishers – Current IR	Fish Ingestion		N/A	✓	✓
			General Population – Tribal Fishers – Heritage IR	Fish Ingestion	✓	N/A	✓	✓
Processing	Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Worker	Dermal [a]	✓	✓	✓	✓
			General Population	Fish Ingestion		N/A		✓
			General Population – Subsistence Fishers	Fish Ingestion	✓	N/A	✓	✓
			General Population – Tribal Fishers – Current IR	Fish Ingestion	✓	N/A	✓	✓
			General Population – Tribal Fishers – Heritage IR	Fish Ingestion	✓	N/A	✓	✓
	Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Worker	Dermal	✓	✓	✓	✓
			General Population	Fish Ingestion		N/A	✓	✓
			General Population – Subsistence Fishers	Fish Ingestion	✓	N/A	✓	✓
			General Population – Tribal Fishers – Current IR	Fish Ingestion	✓	N/A	✓	✓
			General Population – Tribal Fishers – Heritage IR	Fish Ingestion	✓	N/A	✓	✓
	Incorporation into article	Aerospace equipment products and automotive articles and replacement parts containing TCEP	Worker	Dermal [b]	✓	✓	✓	✓
Commercial and Industrial Use	Paints and coatings	Paints and coatings	Worker	Inhalation [b]	✓	✓	✓	✓
				Dermal [b]	✓	✓	✓	✓
			General Population	Fish Ingestion		N/A		✓
			General Population – Subsistence Fishers	Fish Ingestion	✓	N/A	✓	✓
			General Population – Tribal Fishers – Current IR	Fish Ingestion	✓	N/A	✓	✓
			General Population – Tribal Fishers – Heritage IR	Fish Ingestion	✓	N/A	✓	✓

COU			Population	Exposure Route	Human Health Effects			
Life Cycle Stage	**Category**	**Subcategory**			**Acute Non-cancer**	**Intermediate Non-cancer**	**Chronic Non-cancer**	**Lifetime Cancer**
Commercial Use	Laboratory chemicals	Laboratory chemical	Worker	Dermal	✓	✓	✓	✓
			General Population	Fish Ingestion		N/A		✓
			General Population – Subsistence Fishers	Fish Ingestion		N/A		✓
			General Population – Tribal Fishers – Current IR	Fish Ingestion		N/A		✓
			General Population – Tribal Fishers – Heritage IR	Fish Ingestion		N/A		✓

[a] The risk estimate exceeded is based on the most conservative OES (1-part coatings).
[b] The risk estimate exceeded is based on the most conservative OES (2-part coatings, 250-day).

Table 6-2. Supporting Basis for the Unreasonable Risk Determination for Human Health (Consumer COUs)

COU			Population[a]	Exposure Route for Non-cancer	Acute Non-cancer	Chronic Non-cancer	Lifetime Cancer[b]
Life Cycle Stage	**Category**	**Subcategory**					
Consumer Use	Furnishing, cleaning, treatment/care products	Fabric and textile products	Adult	Inhalation	✓		✓ (inhalation and oral)
			Infant	Ingestion – Dust and Mouthing	✓	✓	
	Furnishing, cleaning, treatment/care products	Foam seating and bedding products	Adult	No Non-cancer Risk			✓ (dermal and oral)
			Infant	Ingestion – Dust and Mouthing	✓	✓	
	Construction, paint, electrical, and metal products	Building/ construction materials – Wood and engineered wood products – Wood resin composites	Adult	Inhalation	✓		✓ (dermal, inhalation, and oral)
			Child	Dermal		✓	
			Infant	Ingestion – Dust	✓	✓	
				Dermal		✓	

[a] "Child" represents ages 3–10 years, and "Infant" represents ages 0–2 years.
[b] Risk estimates considered represent lifetime cancer risk and includes exposures at younger life stages that may significantly contribute to the overall cancer risk.
[c] Consumer use of paints and coatings does not have quantitative risk estimates; EPA has conducted a qualitative assessment.
[d] Inhalation of vapors generated from application of paints and coatings containing TCEP.
[e] Ingestion of dust generated by dried paints and coatings containing TCEP.

Table 6-3. Supporting Basis for the Unreasonable Risk Determination for the Environment

COU			Population/ Receptor	Compartment	Environmental Effects	
Life Cycle Stage	Category	Subcategory			Acute	Chronic
Manufacturing	Import	Import	Aquatic	Surface water		✓
				Sediment		✓
Processing	Incorporation into formulation, mixture, or reaction product	Paint and coating manufacturing	Aquatic	Surface water		✓
				Sediment		✓
	Incorporation into formulation, mixture, or reaction product	Polymers used in aerospace equipment and products	Aquatic	Surface water		✓
				Sediment		✓
Commercial and Industrial Use	Paints and coatings	Paints and coatings	Aquatic	Surface water		✓
				Sediment		✓
Commercial Use	Laboratory chemical	Laboratory chemicals	Aquatic	Surface water		✓
				Sediment		✓

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APPENDICES

Appendix A ABBREVIATIONS, ACRONYMS, AND GLOSSARY OF SELECT TERMS

A.1 Key Abbreviations and Acronyms

AC	Acute exposure concentrations
AChE	Acetylcholinesterase
ADC	Average daily concentrations
ADME	Absorption, distribution, metabolism, and elimination
AERMOD	American Meteorological Society (AMS)/EPA Regulatory Model
AF	Assessment factor
ALP	Alkaline phosphatase
ALT	Alanine transferase
AST	Aspartate transaminase
ATSDR	Agency for Toxic Substances and Disease Registry
BAF	Bioaccumulation factor
BCCP	Bis(2-chloroethyl) carboxymethyl phosphate
BCEP	Bis(2-chloroethyl) phosphate
BCF	Bioconcentration factor
BCGP	The glucuronide of bis(2-chloroethyl) 2-hydroxyethyl phosphate
BCHP	Bis(2-chloroethyl) hydrogen phosphate
BLS	Bureau of Labor Statistics
BMD	Benchmark dose
BMDL	Benchmark dose lower confidence limit
BMF	Biomagnification factor
BMR	Benchmark response
BSAF	Biota-sediment accumulation factor
CASRN	Chemical Abstracts Service Registry Number
CBI	Confidential business information
CDR	Chemical Data Reporting (Rule)
CEPA	Canadian List of Toxic Substances
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CFR	Code of Federal Regulations
ChV	Chronic health value
CI	Confidence interval
COC	Concentration(s) of concern
COU	Condition of use
CoCAP	Cooperative Chemicals Assessment Program
CPSA	Consumer Product Safety Act
CPSC	Consumer Product Safety Commission
CSCL	Chemical Substances Control Law
CSF	Cancer slope factor
CSHO	Certified Safety and Health Official
CTD	Characteristic travel distance
DIY	Do-it-yourself
DMR	Discharge Monitoring Report

DOT	Department of Transportation
DRAS	(Hazardous Waste) Delisting Risk Assessment Software (EPA model)
DT50	Time needed to eliminate 50 percent of the substance
DT90	Time needed to eliminate 90 percent of the substance
DWTP	Drinking water treatment plant
EC50	Effect concentration at which 50 percent of test organisms exhibit an effect
ECHA	European Chemicals Agency
ECOSAR	Ecological Structure Activity Relationships (model)
EPA	Environmental Protection Agency
EPCRA	Emergency Planning and Community Right-to-Know Act
ESD	Emission Scenario Document
EU	European Union
FIR	Food intake rate
GS	Generic Scenario
HC05	Hazard concentration that is protective of 95 percent of the species in the sensitivity distribution
HEC	Human equivalent concentration
HED	Human equivalent dose
HERO	Health and Environmental Research Online (Database)
HHE	Health hazard evaluation
IARC	International Agency for Research on Cancer
IMAP	Inventory Multi-Tiered Assessment and Prioritization
IR	Ingestion rate
IRIS	Integrated Risk Information System
IUR	Inhalation unit risk
K_{AW}	Air:water partition coefficient
K_{OC}	Soil organic carbon:water partitioning coefficient
K_{OW}	Octanol:water partition coefficient
Kp	Permeability coefficient
LADC	Lifetime average daily concentrations
LADD	Lifetime average daily dose
LCD	Life cycle diagram
LC50	Lethal concentration at which 50 percent of test organisms die
LD50	Lethal dose at which 50 percent of test organisms die
LOAEL	Lowest-observable-adverse-effect level
LOD	Limit of detection
LOEC	Lowest-observed-effect concentration
LOQ	Limit of quantification
Log K_{AW}	Logarithmic air:water partition coefficient
Log K_{OA}	Logarithmic octanol:air partition coefficient
Log K_{OC}	Logarithmic organic carbon:water partition coefficient
Log K_{OW}	Logarithmic octanol:water partition coefficient
LRAT	Long-range transport via long-range atmospheric transport
MOA	Mode of action
MOE	Margin of exposure
MSW	Municipal solid waste
MSWLF	Municipal solid waste landfills
NAICS	North American Industry Classification System
NATA	National Scale Air-Toxics Assessment

ND	Non-detect
NEI	National Emissions Inventory
NHANES	National Health and Nutrition Examination Survey
NICNAS	National Industrial Chemicals Notification and Assessment Scheme
NIH	National Institutes of Health
NIOSH	National Institute for Occupational Safety and Health
NITE	National Institute of Technology and Evaluation
NMAM	NIOSH Manual of Analytical Methods
NOAA	National Oceanic and Atmospheric Administration
NOEL	No-observed-effect level
NOAEL	No-observed-adverse-effect level
NPDES	National Pollutant Discharge Elimination System
NTP	National Toxicology Program
NWIS	National Water Information System
OCSPP	Office of Chemical Safety and Pollution Prevention
OECD	Organisation for Economic Co-operation and Development
OES	Occupational exposure scenario
ONU	Occupational non-user
OPP	Office of Pesticide Programs
OPPT	Office of Pollution Prevention and Toxics
OSHA	Occupational Safety and Health Administration
PBPK	Physiologically based pharmacokinetic
PBZ	Personal breathing zone
PECO	Population, exposure, comparator, and outcome
PEL	Permissible exposure limit (OSHA)
PESS	Potentially exposed or susceptible subpopulations
PMOC	Persistent mobile organic compound
POD	Point of departure
POTW	Publicly owned treatment works
PPE	Personal protective equipment
PV	Production volume
QSAR	Quantitative structure-activity relationship (model)
RCRA	Resource Conservation and Recovery Act
REACH	Registration, Evaluation, Authorization and Restriction of Chemicals (European Union)
RP	Respirable particle
RQ	Risk quotient
SACC	Scientific Advisory Committee on Chemicals
SCADC	Subchronic average daily concentration
SCE	Sister chromatid exchange
SDS	Safety data sheet
SIDS	Screening Information Dataset
SOC	Standard Occupational Classification (BLS codes)
SSD	Species sensitivity distribution
STEL	Short-term exposure limit
STEV	Short-term occupational exposure value
STORET	STOrage and RETrieval and Water Quality exchange
SVOC	Semi-volatile compound
TE	Transfer efficiency
TESIE	Toddler's Exposure to SVOCs in the Indoor Environment (study)

TGD	Technical Guidance Document (European Commission)
TCEP	Tris(2-chloroethyl) phosphate
TMF	Trophic magnification factor
TRI	Toxics Release Inventory
TRV	Toxicity reference value
TSCA	Toxic Substances Control Act
TWA	Time-weighted average
UF	Uncertainty factor
U.S.	United States
USGS	United States Geological Survey
V6	2,2-Bis(chloromethyl)-propane-1,3-diyltetrakis(2-chloroethyl) bisphosphate
VOC	Volatile organic compound
VP	Vapor pressure
Web-ICE	Web-based Interspecies Correlation Estimation
WHO	World Health Organization
WQP	Water Quality Portal
WWTP	Wastewater treatment plant
7Q10	The lowest 7-day average flow that occurs (on average) once every 10 years
30Q5	The lowest 30-day average flow that occurs (on average) once every 5 years

A.2 Glossary of Select Terms

Condition of use (COU) (15 U.S.C. § 2602(4)): "means the circumstances, as determined by the Administrator, under which a chemical substance is intended, known, or reasonably foreseen to be manufactured, processed, distributed in commerce, used, or disposed of."

Margin of exposure (MOE) (U.S. EPA, 2002a): "a numerical value that characterizes the amount of safety to a toxic chemical–a ratio of a toxicological endpoint (usually a NOAEL [no observed adverse effect level]) to exposure. The MOE is a measure of how closely the exposure comes to the NOAEL."

Mode of action (MOA) (U.S. EPA, 2000c): "a series of key events and processes starting with interaction of an agent with a cell and proceeding through operational and anatomical changes causing disease formation."

Point of departure (POD) (U.S. EPA, 2002a): "dose that can be considered to be in the range of observed responses, without significant extrapolation. A POD can be a data point or an estimated point that is derived from observed dose-response data. A POD is used to mark the beginning of extrapolation to determine risk associated with lower environmentally relevant human exposures."

Potentially exposed or susceptible subpopulations (PESS) (15 U.S.C. § 2602(12)): "means a group of individuals within the general population identified by the Agency who, due to either greater susceptibility or greater exposure, may be at greater risk than the general population of adverse health effects from exposure to a chemical substance or mixture, such as infants, children, pregnant women, workers, or the elderly."

Reasonably available information (40 CFR 702.33): "means information that EPA possesses or can reasonably generate, obtain, and synthesize for use in risk evaluations, considering the deadlines specified in TSCA section 6(b)(4)(G) for completing such evaluation. Information that meets the terms of the preceding sentence is reasonably available information whether or not the information is confidential business information, that is protected from public disclosure under TSCA section 14."

Routes (40 CFR 702.33): "means the ways a chemical substance enters an organism after contact, *e.g.*, by ingestion, inhalation, or dermal absorption."

Sentinel exposure (40 CFR 702.33): "means the exposure from a chemical substance that represents the plausible upper bound of exposure relative to all other exposures within a broad category of similar or related exposures."

Appendix B REGULATORY AND ASSESSMENT HISTORY

B.1 Federal Laws and Regulations

Table_Apx B-1. Federal Laws and Regulations

Statutes/Regulations	Description of Authority/Regulation	Description of Regulation
EPA statutes/regulations		
Toxic Substances Control Act (TSCA) – Section 5(a)	Once EPA finalizes a Significant New Use Rule (SNUR) determining that a use of a chemical substance is a significant new use under TSCA section 5(a), persons are required to submit a significant new use notice (SNUN) to EPA at least 90 days before they manufacture (including import) or process the chemical substance for that use.	In June, 2023, EPA proposed a significant new use rule (SNUR) to designate manufacture or processing of TCEP for any use as a significant new use, with the exception that the conditions of use the Agency expects to consider within the scope of the TSCA section 6 risk evaluations are not proposed as significant new uses. (88 FR 40741, June 22, 2023).
Toxic Substances Control Act (TSCA) – Section 6(b)	EPA is directed to identify high-priority chemical substances for risk evaluation; and conduct risk evaluations on at least 20 high priority substances no later than three and one-half years after the date of enactment of the Frank R. Lautenberg Chemical Safety for the 21st Century Act.	TCEP is one of the 20 chemicals EPA designated as a High-Priority Substance for risk evaluation under TSCA (84 FR 71924, December 30, 2019). Designation of TCEP as high-priority substance constitutes the initiation of the risk evaluation on the chemical.
Toxic Substances Control Act (TSCA) – Section 8(a)	The TSCA section 8(a) CDR Rule requires manufacturers (including importers) to give EPA basic exposure-related information on the types, quantities and uses of chemical substances produced domestically and imported into the United States.	TCEP manufacturing (including importing), processing and use information is reported under the CDR rule (40 CFR part 711).
Toxic Substances Control Act (TSCA) – Section 8(b)	EPA must compile, keep current and publish a list (the TSCA Inventory) of each chemical substance manufactured (including imported) or processed for commercial purposes in the United States.	TCEP was on the initial TSCA Inventory and therefore was not subject to EPA's new chemicals review process under TSCA section 5 (60 FR 16309, March 29, 1995). The chemical is on the active inventory.
Toxic Substances Control Act (TSCA) – section 8(d)	Provides EPA with authority to issue rules requiring manufacturers (including importers), processors, and distributors of a chemical substance or mixture to submit lists and/or copies of ongoing and completed, unpublished health and safety studies. EPA's Health and Safety Data Reporting Rule at 40 CFR part 716 generally requires such submissions for manufacturers (including importers) and (if specified) processors of substances covered by part 716.	Two submissions received in 2021 (U.S. EPA, Chemical Data Access Tool. accessed November 25, 2022).

Statutes/Regulations	Description of Authority/Regulation	Description of Regulation
Toxic Substances Control Act (TSCA) – Section 4	Provides EPA with authority to issue rules and orders requiring manufacturers (including importers) and processors to test chemical substances and mixtures.	Three chemical data submissions from test rules received for TCEP: all three were monitoring reports (1978, 1980, and 1981) (U.S. EPA, ChemView, accessed April 3, 2019).
Emergency Planning and Community Right-To-Know Act (EPCRA) – Section 313	EPCRA Section 313 – also known as the Toxic Release Inventory (TRI), requires annual reporting from facilities in specific industry sectors that employ 10 or more full-time equivalent employees and that manufacture, process or otherwise use a TRI-listed chemical in quantities above threshold levels. A facility that meets reporting requirements must submit a reporting form for each chemical for which it triggered reporting, providing data across a variety of categories, including activities and uses of the chemical, releases, and other waste management (*e.g.*, quantities recycled, treated, combusted) and pollution prevention activities (under section 6607 of the Pollution Prevention Act). These data include on- and off-site data as well as multimedia data (*i.e.*, air, land, and water).	TCEP is a listed substance subject to reporting requirements under 40 CFR 372.65 effective as of November 30, 2022.

B.2 State Laws and Regulations

Table_Apx B-2. State Laws and Regulations

State Actions	Description of Action
State Prohibitions	Three states have adopted prohibitions for the use of TCEP in children's products, including Maryland (MD Health Gen § 24-306), New York (TRIS-free Children and Babies Act (NY Envir Conser § 37-0701 *et seq.*)), Minnesota (Four flame Retardants in Furniture Foam and Children's Products (Minn. Stat. § 325F.071)). California adopted a prohibition, effective on January 1, 2020, on the selling and distribution in commerce of new, not previously owned juvenile products, mattresses, or upholstered furniture that contains, or a constituent component of which contains, covered flame retardant chemicals at levels above 1,000 parts per million (A.B. 2998, Legislative Council, Sess. 2017-2018, C.A. 2018).
State Drinking Water Standards and Guidelines	Minnesota developed a health-based guidance value for TCEP in drinking water (Minn R. Chap. 4720).
Chemicals of High Concern to Children	Several states have adopted reporting laws for chemicals in children's products containing TCEP, including Maine (38 MRSA Chapter 16-D), Minnesota (Toxic Free Kids Act Minn. Stat. 116.9401 to 116.9407), Oregon (Toxic-Free Kids Act, Senate Bill 478, 2015), Vermont (18 V.S.A § 1776) and Washington State (Wash. Admin. Code 173-334-130).
Other	California listed TCEP on Proposition 65 in 1992 due to cancer (Cal Code Regs. Title 27, § 27001).

State Actions	Description of Action
	California issued a Health Hazard Alert for TCEP (Hazard Evaluation System and Information Service, 2016). California lists TCEP as a designated priority chemical for biomonitoring (California SB 1379). TCEP is listed as a Candidate Chemical under California's Safer Consumer Products Program (Health and Safety Code § 25252 and 25253). The regulation for Children's Foam-Padded Sleeping Products containing TCEP as a Priority Product went into effect on July 1, 2017: Manufacturers of this product must notify the Department by September 1, 2017 (California Department of Toxic Substances Control, Accessed April 12, 2019).

B.3 International Laws and Regulations

Table_Apx B-3. International Laws and Regulations

Country/ Organization	Requirements and Restrictions
Canada	TCEP (Ethanol, 2-chloro-, phosphate (3:1)) is on the Canadian List of Toxic Substances (CEPA 1999 Schedule 1). TCEP was added to Schedule 2 of the *Canada Consumer Product Safety Act (CCPSA)*, based on concerns for carcinogenicity and impaired fertility. (Government Canada Chemical Safety portal. Accessed April 10, 2019). In January 2013, a Significant New Activity was adopted for TCEP (*Canada Gazette*, April 3, 2014; Vol. 148, No. 9).
European Union	In June 2017, TCEP was added to Annex XIV of REACH (Authorisation List) with a sunset date of August 21, 2015 (European Chemicals Agency (ECHA, 2019) database, Accessed April 10, 2019). In 2010, TCEP was listed on the Candidate list as a Substance of Very High Concern (SVHC) under regulation (EC) No 1907/2006 - REACH (Registration, Evaluation, Authorization and Restriction of Chemicals due to its reproductive toxicity (category 57C)).
Australia	Ethanol, 2-chloro-, phosphate (3:1) (TCEP) was assessed under Human Health Tier II and III of the Inventory Multi-Tiered Assessment and Prioritisation (IMAP). Uses reported include commercial: (NICNAS, 2016, *Ethanol, 2-chloro-, phosphate (3:1): Human health tier II assessment*, Accessed April 8, 2019) (NICNAS, 2017, *Ethanol, 2-chloro-, phosphate (3:1): Human health tier III assessment*, Accessed April 8, 2019).
Japan	TCEP is regulated in Japan under the following legislation: • Act on the Evaluation of Chemical Substances and Regulation of Their Manufacture, etc. (Chemical Substances Control Law; CSCL), • Act on Confirmation, etc. of Release Amounts of Specific Chemical Substances in the Environment and Promotion of Improvements to the Management Thereof,

Country/ Organization	Requirements and Restrictions
	• Air Pollution Control Law (National Institute of Technology and Evaluation [NITE] Chemical Risk Information Platform [CHRIP], April 8, 2019).
Basel Convention	Waste substances and articles containing or contaminated with polychlorinated biphenyls (PCBs) and/or polychlorinated terphenyls (PCTs) and/or polybrominated biphenyls (PBBs) are listed as a category of waste under the Basel Convention. Although the United States is not currently a party to the Basel Convention, this treaty still affects U.S. importers and exporters. http://www.basel.int/Portals/4/Basel%20Convention/docs/text/BaselConventionText-e.pdf.

B.4 Assessment History

Table_Apx B-4. Assessment History of TCEP

Authoring Organization	Publication
EPA publications	
U.S. EPA, Superfund Health Risk Technical Support Center, Office of Research and Development (ORD)	Provisional Peer-Reviewed Toxicity Values (PPRTV) for Tris(2-chloroethyl)phosphate (TCEP) (CASRN 115-96-8) U.S. EPA (2009)
U.S. EPA, Design for the Environment Program	Design for the Environment (DfE) Alternatives Assessments
Other U.S.-based organizations	
Agency for Toxic Substances and Disease Registry (ATSDR)	Toxicological Profile for Phosphate Ester Flame Retardants (2012)
National Toxicology Program (NTP), National Institutes of Health (NIH)	Technical Report on Toxicology and Carcinogenesis Studies of Tris(2-chloroethyl) Phosphate (CASRN 115-96-8) in F344/N Rats and B6C3F1 Mice (Gavage Studies) (1991)
International	
Organisation for Economic Co-operation and Development (OECD), Cooperative Chemicals Assessment Program (CoCAP)	SIDS initial assessment profile for SIAM 23: Tris(2-chloroethyl)phosphate (CAS no. 115-96-8) (2006)
International Agency for Research on Cancer (IARC)	Monographs on the Evaluation of Carcinogenic Risks to Humans Volume 71 (1999)
European Union, European Chemicals Agency (ECHA)	European Union Risk Assessment Report: CAS: 115-96-8: Tris (2-chloroethyl) phosphate, TCEP (2009)
Government of Canada, Environment Canada, Health Canada	Screening Assessment for the Challenge Ethanol, 2-chloro-, phosphate (3:1) (Tris(2-chlrorethyl) phosphate [TCEP]) (2009)

Authoring Organization	Publication
National Industrial Chemicals Notification and Assessment Scheme (NICNAS), Australian Government	Ethanol, 2-chloro-, phosphate (3:1): Human health tier II assessment (2016), and Ethanol, 2-chloro-, phosphate (3:1): Human health tier III assessment (2017)

Appendix C LIST OF SUPPLEMENTAL DOCUMENTS

Appendix C incudes a list and citations for all supplemental documents included in the Risk Evaluation for TCEP. See Docket EPA-HQ-OPPT-2018-0476 for all publicly released files associated with this risk evaluation package; see Docket EPA-HQ-OPPT-2023-0265 for all publicly released files associated with peer review and public comments.

Associated **Systematic Review Protocol and Data Quality Evaluation and Data Extraction** Documents – Provide additional detail and information on systematic review methodologies used as well as the data quality evaluations and extractions criteria and results.

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Protocol (U.S. EPA, 2024p) – In lieu of an update to the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances*, also referred to as the "2021 Draft Systematic Review Protocol" (U.S. EPA, 2021a), this systematic review protocol for the Risk Evaluation for TCEP describes some clarifications and different approaches that were implemented than those described in the 2021 Draft Systematic Review Protocol in response to (1) SACC comments, (2) public comments, or (3) to reflect chemical-specific risk evaluation needs. This supplemental file may also be referred to as the "TCEP Systematic Review Protocol."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties (U.S. EPA, 2024v) – Provides a compilation of tables for the data extraction and data quality evaluation information for TCEP. Each table shows the data point, set, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation of physical and chemical properties. This supplemental file may also be referred to as the "TCEP Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport (U.S. EPA, 2024t) – Provides a compilation of tables for the data extraction and data quality evaluation information for TCEP. Each table shows the data point, set, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation for Environmental Fate and Transport. This supplemental file may also be referred to as the "TCEP Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure (U.S. EPA, 2024u) – Provides a compilation of tables for the data extraction and data quality evaluation information for TCEP. Each table shows the data point, set, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation of environmental release and occupational exposure. This supplemental file may also be referred to as the "TCEP Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Dermal Absorption (U.S. EPA, 2024s) – Provides a compilation of tables for the data extraction and data quality evaluation information for TCEP. Each table shows the data point, set, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation for

Dermal Absorption. This supplemental file may also be referred to as the "TCEP Data Quality Evaluation and Data Extraction Information for Dermal Absorption."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure. (U.S. EPA, 2024x) – Provides a compilation of tables for the data quality evaluation information for TCEP. Each table shows the data point, set, or information element that was evaluated from a data source that has information relevant for the evaluation of general population, consumer, and environmental exposure. This supplemental file may also be referred to as the "TCEP Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Extraction Information for General Population, Consumer, and Environmental Exposure (U.S. EPA, 2024r) – Provides a compilation of tables for the data extraction for TCEP. Each table shows the data point, set, or information element that was extracted from a data source that has information relevant for the evaluation of general population, consumer, and environmental exposure. This supplemental file may also be referred to as the "TCEP Data Extraction Information for General Population, Consumer, and Environmental Exposure."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Quality Evaluation Information for Human Health Hazard Epidemiology (U.S. EPA, 2024z) – Provides a compilation of tables for the data quality evaluation information for TCEP. Each table shows the data point, set, or information element that was evaluated from a data source that has information relevant for the evaluation of epidemiological information. This supplemental file may also be referred to as the "TCEP Data Quality Evaluation Information for Human Health Hazard Epidemiology."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Quality Evaluation Information for Human Health Hazard Animal Toxicology (U.S. EPA, 2024y) – Provides a compilation of tables for the data quality evaluation information for TCEP. Each table shows the data point, set, or information element that was evaluated from a data source that has information relevant for the evaluation of human health hazard animal toxicity information. This supplemental file may also be referred to as the "TCEP Data Quality Evaluation Information for Human Health Hazard Animal Toxicology."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Quality Evaluation Information for Environmental Hazard (U.S. EPA, 2024w) – Provides a compilation of tables for the data quality evaluation information for TCEP. Each table shows the data point, set, or information element that was evaluated from a data source that has information relevant for the evaluation of environmental hazard toxicity information. This supplemental file may also be referred to as the "TCEP Data Quality Evaluation Information for Environmental Hazard."

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Systematic Review Supplemental File: Data Extraction Information for Environmental Hazard and Human Health Hazard Animal Toxicology and Epidemiology (U.S. EPA, 2024q) – Provides a compilation of tables for the data extraction for TCEP. Each table shows the data point, set, or information element that was extracted from a data source that has information relevant for the evaluation of environmental hazard and human health hazard animal toxicology and epidemiology information. This supplemental file may

also be referred to as the "TCEP Data Extraction Information for Environmental Hazard and Human Health Hazard Animal Toxicology and Epidemiology."

Associated **Supplemental Information Documents** – Provide additional details and information on exposure, hazard, and risk assessments.

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Environmental Release and Occupational Exposure Assessment (U.S. EPA, 2024n).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: E-FAST Modeling Results (U.S. EPA, 2024g).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: IIOAC Modeling Input and Results (U.S. EPA, 2024l).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Environmental Monitoring Concentrations Reported by Media Type (U.S. EPA, 2024i).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Environmental Monitoring and Biomonitoring Concentrations Summary Table (U.S. EPA, 2024h).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Consumer Exposure Modeling Inputs (U.S. EPA, 2024e).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental File Folder: Supplemental Information on Consumer Exposure Modeling Results (U.S. EPA, 2024f).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Human Health Hazard Points of Departure Comparison Tables (U.S. EPA, 2024k) – Provides an Excel spreadsheet of PODs for all studies and hazard outcomes resulting in *likely* or *suggestive* evidence integration conclusions. Basic study details as well as the PODs from each study and associated HEDs, HECs, and total UFs for non-cancer endpoints, as well as CSFs and IURs for cancer endpoints are presented.

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Benchmark Dose Modeling Results for TCEP (U.S. EPA, 2024c) – Provides inputs to BMD modeling as well as outputs for individual health effects associated with hazard outcomes that have *likely* evidence integration conclusions. Information includes goodness of fit details for all models that were run, as well as BMD and BMDL values for the selected BMR and any comparison BMRs. Graphs of the chosen models are also presented.

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Risk Calculator for Occupational Exposures (U.S. EPA, 2024m).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Exposure Air Concentration Risk Calculations (U.S. EPA, 2024j).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Water Quality Portal Processed Water Data (U.S. EPA, 2024o).

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) - Supplemental File Folder: Supplemental Information on Human Milk PBPK Verner Modeling Results (U.S. EPA, 2024a)

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) - Supplemental File Folder: Biosolids Screening Tool Modeling Results (U.S. EPA, 2024d)

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental File Folder: Consumer Modeling Risk Calculations, Sensitivity Analysis, and Visualizations of Results and Environmental Water Quality Portal Data (U.S. EPA, 2024b)

Appendix D CONDITIONS OF USE DESCRIPTIONS

The following descriptions are intended to include examples of uses so as not to exclude other activities that may also be included in the COUs of the chemical substance. To better describe the TSCA COU, EPA considered CDR submissions from the last two CDR cycles for DINP (CASRN 28553-12-0 and CASRN 68515-48-0), and the COU descriptions reflect what EPA identified as the best fit for that submission.

D.1 Manufacturing (Import)

Import refers to the import of TCEP into the customs territory of the United States. This condition of use includes loading/unloading and repackaging (but not transport) associated with the import of TCEP. In general, chemicals may be imported into the United States in bulk via water, air, land, and intermodal shipments. These shipments take the form of oceangoing chemical tankers, railcars, tank trucks, and intermodal tank containers (U.S. EPA, 2021c).

Examples of CDR Submissions. In 2016, one company, reported importation of TCEP (CASRN 115-96-8) in wet-solid form with a production volume of 158,728 lb. However, the company ceased operations in 2019 (Aceto US LLC, 2024). There were no reporters of TCEP in the 2020 CDR.

D.2 Processing – Incorporation into Formulation, Mixture, or Reaction Product – Paint and Coating Manufacturing

This COU refers to the preparation of a product; that is, the incorporation of TCEP into formulation, mixture, or a reaction product which occurs when a chemical substance is added to a product (or product mixture), after its manufacture, for distribution in commerce, in this case—processing TCEP into coating products for commercial (non-consumer) use, including waterborne coatings and resin-/solvent-based coatings. The general processes for the formulation of waterborne coatings and resin/solvent-based coatings is similar but with shorter blending/mixing times for the resin-/solvent-based products.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, EPA has met with stakeholders and received comment indicating continued use of TCEP in paints and coatings used in the aerospace and automotive industries (Alliance for Automotive Innovation, 2023a, b; FCC, 2021; AIA, 2019).

D.3 Processing – Incorporation into Formulation, Mixture, or Reaction Product – Polymers used in Aerospace Equipment and Products

This COU refers to the preparation of a product; that is, the incorporation of TCEP into formulation, mixture, or a reaction product which occurs when a chemical substance is added to a product (or product mixture), after its manufacture, for distribution in commerce, in this case—processing TCEP into formulations of aerospace products. In aerospace products, TCEP is used as a flame-retardant additive component of 2-part polymer and prepolymer resin systems used in potting and casting applications; as an additive plasticizer and viscosity regulatory with flame-retarding properties for polyurethane, polyesters, polyvinyl chloride, and other polymers; in the production of unsaturated polyester resins and in acrylic resins and coatings; and for production of polyurethane foam.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, stakeholders provided information regarding the use of TCEP in polymers used in the aerospace industry (AIA, 2021a, b, 2020, 2019).

D.4 Processing – Incorporation into Article – Aerospace Equipment and Products and Automotive Articles and Replacement Parts Containing TCEP

This COU refers to the preparation of an article; that is, the incorporation of TCEP into articles, meaning TCEP becomes a component of the article, after its manufacture, for distribution in commerce. In this case, TCEP is present as a flame-retardant and plasticizer additive in polymer resins used in potting and casting applications in the aerospace industry as well as for production of polyurethane foam in aircraft and aerospace products. EPA identified that plastic products with TCEP-containing cured paints and coatings are currently used (via incorporation) in articles for automotive applications and that the TCEP-containing foam products are currently used (via incorporation) in articles for aircraft and aerospace applications. Specific aerospace industrial uses include resins and elastomeric coatings, polyurethane casting for aircraft interiors and as a flame retardant for aircraft furniture. Specific automotive processing uses include installation of products and replacement parts with cured TCEP-containing paints and coatings into automobiles.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, EPA has received public comment that TCEP may be processed into automotive articles and replacement parts (Alliance for Automotive Innovation, 2023b).

D.5 Processing – Recycling

This COU refers to the process of treating generated waste streams (*i.e.*, which would otherwise be disposed of as waste), containing TCEP, which are collected, either on-site or transported to a third-party site, for commercial purpose. TCEP may be present as an additive in components of electronics and electrical equipment that is recycled. E-waste recycling activities include receiving e-waste at the facility, dismantling or shredding the e-waste, and sorting the recycled articles and generated scrap materials. TCEP has been identified in articles that are commonly recycled such as insulation, plastics, and foam. TCEP may be present within flexible foam, fabric, textile, and other applications that have been made from recycled foam scraps generated during trimming of original TCEP-containing manufactured foam products.

Examples of CDR Submissions. EPA notes that although TCEP was not reported for recycling in the 2016 or 2020 CDR reporting periods, the Agency is assuming that recycling waste streams likely contain TCEP.

D.6 Distribution in Commerce

For purposes of assessment in this risk evaluation, distribution in commerce consists of the transportation associated with the moving of TCEP or TCEP-containing products between sites manufacturing, processing, or recycling TCEP or TCEP-containing products, or to final use sites, or for final disposal of TCEP or TCEP-containing products. More broadly under TSCA, "distribution in commerce" and "distribute in commerce" are defined under TSCA section 3(5).

D.7 Industrial Use – Other Use – Aerospace Equipment and Products and Automotive Articles and Replacement Parts Containing TCEP

This COU is referring to the industrial use of products or articles containing TCEP in aerospace equipment and automotive articles and replacement parts. Meaning the use of TCEP after it has already been incorporated into a product or article, as opposed to when it is used upstream (*e.g.*, when TCEP is processed into the product or article). TCEP is present in the cured resin or foam components of articles

that are installed in aircrafts or aerospace vehicles and in the cured paints and coatings in automotive articles and replacement parts. Examples of possible TCEP uses in aircraft and aerospace products include its presence as a flame retardant in aircraft furniture foams, electronics, or structural components. TCEP-containing articles are used as received at the site, with minimal or no reshaping or processing of the article prior to manual installation into the aircraft or aerospace vehicle and manual and/or robotic installation into the automobile. Industrial use of TCEP for this COU involves maintenance of aerospace equipment and products and workers handling automotive articles and replacement parts at automotive manufacturers.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, EPA has received comment that TCEP is used in the aerospace industry as well as in automotive articles and replacement parts (Alliance for Automotive Innovation, 2023b; Boeing, 2023; AIA, 2021a, 2019).

D.8 Industrial Use – Paints and Coatings

This COU is referring to the use of TCEP in various industrial sectors as a component of industrial paints and coatings. Meaning the use of TCEP after it has already been incorporated into a paint or coating product or mixture, as opposed to when it is used upstream (*e.g.*, when TCEP is processed into the paint or coating formulation). TCEP is an additive component in paints and coatings for industrial and commercial use as a flame-retardant coating to achieve flame spread or fire protection standards for automobile articles and replacement parts. Products include TCEP-containing paints and coatings that are applied in an industrial setting to automobile bodies. Coating application methods for TCEP-containing paints and coatings in the automotive industry include robotic and manual applications. TCEP will remain in the coating as an additive in the dried/cured coating on the substrate.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, EPA has received comment indicating continued use of TCEP in paints and coatings used in the aerospace and automotive industries (Alliance for Automotive Innovation, 2023a, b; AIA, 2019).

D.9 Commercial Use – Other Use – Aerospace Equipment and Products and Automotive Articles and Replacement Parts Containing TCEP

This COU is referring to the commercial use of TCEP in aerospace equipment and products and automotive articles and replacement parts, which already have TCEP incorporated into them. Meaning the use of TCEP-containing aerospace equipment and products and automotive articles and replacement parts in a commercial setting, such as a worker operating in unoccupied parts of airplanes or driving a vehicle or an automotive parts business, as opposed to upstream use of TCEP (*e.g.*, when TCEP containing products are used in the manufacturing of the airplane or automotive) or use in an industrial setting. TCEP is present in the cured resin in aerospace equipment and products, the cured paints and coatings in automotive articles and replacement parts and in foam components of articles that are installed in aircrafts or aerospace vehicles. Examples of possible TCEP uses in aircraft and aerospace products include its presence as a flame retardant in aircraft furniture foams, electronics or structural components. TCEP-containing articles are used as received at the site, with minimal or no reshaping or processing of the article prior to manual installation into the aircraft or aerospace vehicle and manual installation into the automobile. Commercial use of TCEP for this COU involves use of aerospace equipment or products by airliners and workers (*e.g.* auto repair shops and commercial drivers) handling automotive articles and replacement parts.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, EPA has received comment that TCEP is used in the aerospace industry as well as in automotive articles and replacement parts (Alliance for Automotive Innovation, 2023b; Boeing, 2023; AIA, 2021a, 2019).

D.10 Commercial Use – Paints and Coatings

This COU is referring to the commercial use of TCEP already incorporated as a flame retardant in paints and coatings. TCEP is an additive component in paints and coatings for industrial and commercial use as a flame-retardant coating to achieve flame spread or fire protection standards for structural and electrical components and include waterborne coatings and resin-/solvent-based coatings. Products include 1-part coatings and 2-part epoxy resins that are typically used on electrical cables, exterior (masonry) surfaces, or unoccupied parts of a building such as mechanical rooms, attics, and crawl spaces that may contain foam that needs to be coated in flame retardants. Other applications include coating large industrial steel or aluminum structures. Coating application methods for TCEP-containing paints and coatings include spray gun, brush, and trowel coating for use on structures or equipment. TCEP will remain in the coating as an additive in the dried/cured coating on the substrate. These products would be purchased by commercial operations and applied by professional contractors in various commercial settings.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, EPA has received comment indicating use of TCEP in paints and coatings used in the aerospace and automotive industries (Alliance for Automotive Innovation, 2023a, b; AIA, 2019).

D.11 Commercial Use – Laboratory Chemicals

This COU is referring to the commercial use of TCEP in laboratory chemicals. TCEP is used as a laboratory chemical, such as in a chemical standard or reference material during analyses. The users of products under this category would be expected to apply these products through general laboratory use applications. Commercial use of laboratory chemicals may involve handling TCEP by hand-pouring or pipette and either adding to the appropriate labware in its pure form to be diluted later or added to dilute other chemicals already in the labware. Laboratory TCEP products are pure TCEP in neat liquid form. The Agency notes that the same applications and methods used for quality control can be applied in industrial and commercial settings.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, EPA has received public comment indicating laboratory use of TCEP (NASA, 2020).

D.12 Commercial Use – Furnishing, Cleaning, Treatment/Care Products – Fabric and Textile Products

This COU is referring to the commercial use of TCEP already incorporated as a flame-retardant plasticizer in fabric and textile products. TCEP was previously used as a flame-retardant plasticizer in unsaturated polyester resins, which was used in polyester yarn and fabric products including clothing, bed sheets, blankets, upholstered furniture, industrial polyester fibers, fabrics for conveyor and safety belts, and coating fabric. EPA has not found evidence of TCEP currently being used in fabric and textile products for both commercial and consumer uses. The Agency has confirmed from literature sources that TCEP was used for these purposes in the past but was phased out of these uses starting in the late 1980s or early 1990s in favor of other flame retardants or flame-retardant formulations. Because

manufacturing and processing of TCEP is not ongoing, this COU focuses on the end of service life disposal for human health and the environment.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

D.13 Commercial Use – Furnishing, Cleaning, Treatment/Care Products – Foam Seating and Bedding Products

This COU is referring to the commercial use of TCEP already incorporated in foam seating and bedding products and furnishings. EPA has confirmed that the manufacturing and processing of TCEP into foam seating and bedding products for commercial and industrial use, outside of aircrafts and aerospace products, has been phased out. TCEP was previously used in flexible urethane cushions that were used in institutional mattresses, furniture foam padding, automotive seat cushions and padding, carpet underlay, and pillow and mattress padding. EPA identified during scoping that foams for furniture was a major use of TCEP prior to the 1990s. The Agency has confirmed from literature sources that TCEP was used for these purposes in the past but was phased out of these uses starting in the late 1980s or early 1990s in favor of other flame retardants or flame-retardant formulations. Because manufacturing and processing of TCEP is not ongoing, this COU focuses on the end of service life disposal for human health and the environment.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

D.14 Commercial Use – Construction, Paint, Electrical, and Metal Products – Building/Construction Materials – Insulation

This COU is referring to the commercial use of TCEP in commercial sectors associated with construction products that contain TCEP as a plasticizer or flame retardant, such as at a business or at a job site as opposed to upstream use of TCEP (*e.g.*, when TCEP is processed into the construction material) or use in an industrial setting. Rigid polyurethane foams for insulation, specifically commercial roofing insulation, was identified as a potential application for TCEP and was identified as a major use of TCEP prior to the 1990s. EPA has not found any modern evidence that TCEP is still being manufactured or processed for incorporation into building and construction materials. The Agency has confirmed from literature sources that TCEP was used for these purposes in the past but was phased out of these uses starting in the late 1980s or early 1990s in favor of other flame retardants or flame-retardant formulations. Because manufacturing and processing of TCEP is not ongoing, this COU focuses on the end of service life disposal for human health and the environment.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

D.15 Commercial Use – Construction, Paint, Electrical, and Metal Products – Building/Construction Materials – Wood and Engineered Wood Products – Wood Resin Composites

This COU is referring to the commercial use of TCEP in commercial sectors associated with construction products that contain TCEP as a plasticizer or flame retardant, such as at a business or at a job site as opposed to upstream use of TCEP (*e.g.*, when TCEP is processed into the construction material) or use in an industrial setting. TCEP was previously incorporated into construction, paint,

electrical, and metal products, such as roofing insulation and rigid foam. It is possible that TCEP was used in the resins that bond wood products together. TCEP use in engineered wood products was a minor use in only niche products, such as furniture production as opposed to larger scale uses in building construction. EPA has not found any modern evidence that TCEP is still being manufactured or processed for incorporation into building and construction materials. The Agency has confirmed from literature sources that TCEP was used for these purposes in the past but was phased out of these uses starting in the late 1980s or early 1990s in favor of other flame retardants or flame-retardant formulations. Because manufacturing and processing of TCEP is not ongoing, this COU focuses on the end of service life disposal for human health and the environment.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

D.16 Consumer Use – Paints and Coatings, Including Those Found on Automotive Articles and Replacement Parts

This COU is referring to the consumer use of TCEP already incorporated as a flame retardant in paints and coatings. TCEP-containing paints and coatings are unlikely to be available for purchase by consumers, and EPA did not find any evidence indicating the continued use of TCEP-containing paints and coatings by consumers. However, the Agency has found that some automotive articles and replacement parts painted with TCEP-containing paint may be available for purchase and installation by DIY users. Consumers may be exposed to TCEP if entering unoccupied parts of buildings such as attics or crawl spaces that might contain electrical cables or foams treated paints and coatings that may contain TCEP. Consumers can also be exposed to TCEP if using automobiles containing articles or replacement parts painted with TCEP-containing paint.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR. However, EPA has met with stakeholders who report the use of TCEP-containing paints and coatings on automotive articles and replacement parts that may be accessible to consumers (Alliance for Automotive Innovation, 2023a, b).

D.17 Consumer Use – Furnishing, Cleaning, Treatment/Care Products – Fabric and Textile Products

This COU is referring to the consumer use of TCEP already incorporated as a flame-retardant plasticizer in fabric and textile products. TCEP was previously used as a flame-retardant plasticizer in unsaturated polyester resins, which was used in polyester yarn and fabric products including clothing, bed sheets, blankets, upholstered furniture, industrial polyester fibers, fabrics for conveyor and safety belts, and coating fabric for carpets and upholstery. EPA has not found modern evidence of TCEP currently being used in fabric and textile products for both commercial and consumer uses.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

D.18 Consumer Use – Furnishing, Cleaning, Treatment/Care Products – Foam Seating and Bedding Products

This COU is referring to the consumer use of TCEP already incorporated in foam seating and bedding products and furnishings. EPA has confirmed that the manufacturing and processing of TCEP into foam seating and bedding products for commercial and industrial use, outside of aircrafts and aerospace products, has been phased out. TCEP was previously used in flexible urethane cushions that were used

in institutional mattresses, furniture foam padding, automotive seat cushions and padding, carpet underlay, and pillow and mattress padding.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

D.19 Consumer Use – Construction, Paint, Electrical, and Metal Products – Building/Construction Materials – Insulation

This COU is referring to the consumer use of TCEP associated with construction products that contain TCEP as a plasticizer or flame retardant. TCEP was previously incorporated into construction, paint, electrical, and metal products, such as roofing insulation and rigid foam. EPA has not found any modern evidence that TCEP is still being manufactured or processed for incorporation into building and construction materials.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

D.20 Consumer Use – Construction, Paint, Electrical, and Metal Products – Building/Construction Materials – Wood and Engineered Wood Products – Wood Resin Composites

This COU is referring to the consumer use of TCEP associated with construction products that contain TCEP as a plasticizer or flame retardant. TCEP was previously incorporated into construction, paint, electrical, and metal products, such as roofing insulation and rigid foam. It is possible that TCEP was used in the resins that bond wood products together. TCEP use in engineered wood products was a minor use in only niche products, such as furniture production as opposed to larger scale uses in building construction. EPA has not found any modern evidence that TCEP is still being manufactured or processed for incorporation into building and construction materials.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

D.21 Disposal

Each of the COUs of TCEP may generate waste streams of the chemical. For purposes of the TCEP risk evaluation, this COU refers to the TCEP in a waste stream that is collected and transported to third-party sites for disposal or treatment. This COU also encompasses TCEP contained in wastewater discharged to POTW or other, non-public treatment works for treatment, and other wastes. TCEP is expected to be released to other environmental media, such as introductions of biosolids to soil or migration to water sources, through waste disposal (*e.g.*, disposal of formulations containing TCEP, plastic and rubber products, textiles, and transport containers). Disposal may also include destruction and removal by incineration (U.S. EPA, 2021b). Recycling of TCEP and TCEP containing products is considered a different COU. Environmental releases from industrial sites are assessed in each COU. Disposal of TCEP occurs during the disposal of TCEP-containing articles, such as furniture, clothing, and plastic waste, as well as during demolition of and disposal of building materials that may contain TCEP, such as roofing insulation.

Examples of CDR Submissions. There were no reporters for TCEP for this use in neither the 2016 nor the 2020 CDR.

Appendix E DETAILED EVALUATION OF POTENTIALLY EXPOSED OR SUSCEPTIBLE SUBPOPULATIONS

E.1 PESS Based on Greater Exposure

In this section, EPA addresses the following potentially exposed populations expected to have greater exposure to TCEP. Table_Apx E-1 presents the quantitative data sources that were used in the PESS exposure analysis for incorporating increased background and COU-specific exposures.

Table_Apx E-1. PESS Evidence Crosswalk for Increased Exposure

Category	Subcategory	Increased Background Exposure	Increased COU or Pathway Specific Exposures	Quantitative Data Sources
Lifestage	Embryo/fetus	• Transfer of exposure from the parent (placenta to fetus) • Ratio of placenta: maternal serum (R_{pm}) concentrations shown to range from 0.76 for TCEP		• (Wang et al., 2021)
	Children (infants, toddlers)	• EPA did not identify sources of increased background exposure anticipated for this lifestage	• Hand to mouth behavior leads to increased ingestion of household dust • Age-appropriate behavior patterns (elevated soil ingestion exposure (children's activities with soil, children playing mud) • Human milk exposure from maternal doses derived from TSCA sources • Different exposure factors • Drinking water exposure from TSCA sources	• EPA Age Grouping Guidance • *Exposure Factors Handbook* (U.S. EPA, 2017d) • See Section 5.1.3.4.7
	Geriatric	• Older populations that generally use supplements may be at higher exposure to TCEP due to use of Fish oil supplements	• EPA did not identify sources of increased COU or pathway specific exposure for this lifestage	• (Poma et al., 2018)
Sociodemo-graphic/ Lifestyle	Race/Ethnicity	• EPA did not identify sources of increased background exposure anticipated for this lifestage	• TCEP levels in dust are significantly associated with the presence of extremely worn carpets; lower socioeconomic status (SES) populations are more prone to having homes with older carpets due to their cost of replacement • Fenceline populations (typically lower SES) may live closer to emitting sources	• (Castorina et al., 2017)
	Subsistence Fishing	• EPA did not identify sources of increased background exposure anticipated for this lifestage	• Subsistence fishing populations that consumer more fish have elevated levels of TCEP exposure	• See Section 5.1.3.4.3
Occupational	Firefighters	• Firefighters may be at increased risk of TCEP exposures during structure fires (Mayer et al., 2021).	• EPA did not identify sources of increased COU or pathway specific exposure for firefighters	• See qualitative discussion Section 5.3.3 • (Jayatilaka et al., 2017).

Category	Subcategory	Increased Background Exposure	Increased COU or Pathway Specific Exposures	Quantitative Data Sources
Consumer	High frequency consumers High duration consumers	• Non-TSCA source such as dietary exposures through food, food packaging, drugs, and personal care products that contain TCEP	• Consumer products designed for children (*e.g.*, children's outdoor play structures, toy foam blocks) may lead to elevated exposures for children and infants.	• Use Report • EPA's *Exposure Factors Handbook* (Ch. 17) • See Sections 5.1.2.2 and 5.1.3.4.8

E.2 PESS Based on Greater Susceptibility

In this section, EPA addresses subpopulations expected to be more susceptible to TCEP exposure than other populations. Table_Apx E-2 presents the data sources that were used in the PESS analysis evaluating susceptible subpopulations and identifies whether and how the subpopulation was addressed quantitatively in the risk evaluation of TCEP.

Some observations may be made regarding factors that may increase susceptibility to the effects of TCEP. Human data are available on health effects of TCEP that may suggest there are susceptible subpopulations, although as identified in Section 5.2.3, human evidence is only slight (with evidence as indeterminate for thyroid effects). Percy et al. (2022) found increased BCEP in urine was associated with *lower* IQ in children with low socioeconomic status (SES) using more than one measure related to SES. In contrast, for the full sample of children (a wider range of economic status), the authors found a small positive (but not statistically significant) relationship between BCEP IQ (Percy et al., 2022). Similarly, in a group of children across SES levels, maternal BCEP exposure was marginally associated with higher IQ (Percy et al., 2021).

Human epidemiological studies show slight evidence for developmental effects related to growth and gestational age examining associations with a TCEP metabolite (BCEP) in mother's urine during gestation. Both sexes and male offspring alone who were less likely to be small for their gestational age in one study (Oh et al., 2024). Increased BCEP was also associated with increased skinfold thickness in male offspring (two measures of thickness) and both sexes (one measure) (Crawford et al., 2020). Female children had a greater incidence of being pre-term (Oh et al., 2024) and lower birthweight and length (Yang et al., 2022).

Effects may differ by gender, as identified by some epidemiological studies. As noted above, female developmental outcomes related to growth and gestational age may differ from males. Mendy et al. (2024) found that TCEP in dust had a more pronounced effect on hay fever and allergies in female children compared with males. Also, TCEP in serum was associated with more statistically significant changes in thyroid hormones in females than in males (Liu et al., 2022) but evidence is indeterminate.

Animal studies identified developmental effects (NTP, 1991a) as well as sensitive sexes for certain health outcomes—higher incidence of neurotoxicity in female rats (NTP, 1991b) and greater sensitivity of male (vs. female) mice in reproductive effects (Chen et al., 2015a)—and EPA quantified risks based on these endpoints in the risk evaluation. It is possible that these differences in rodents reflect differences in humans. However, if sex differences in susceptibility among rodents are due solely to differences in toxicokinetics, there is uncertainty for humans given a lack of metabolic differences among sexes in experiments using human liver tissues (Chapman et al., 1991).

As identified in Table_Apx E-2, many other susceptibility factors that are generally considered to increase susceptibility of individuals to chemical hazards. These factors include pre-existing diseases, alcohol use, diet, stress, among others. The effect of these factors on susceptibility to health effects of TCEP is not known; therefore, EPA is uncertain about the magnitude of any possible increased risk from effects associated with TCEP exposure.

For non-cancer endpoints, EPA used a default value of 10 for human variability (UF_H) to account for increased susceptibility when quantifying risks from exposure to TCEP. The Risk Assessment Forum, in *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002b), discusses some of the evidence for choosing the default factor of 10 when data are lacking and describe the types

of populations that may be more susceptible, including different lifestages (*e.g.*, of children and elderly). U.S. EPA (2002b), however, did not discuss all the factors presented in Table_Apx E-2. Thus, uncertainty remains regarding whether these additional susceptibility factors would be covered by the default UF_H value of 10 chosen for use in the TCEP risk evaluation. In addition, given that EPA is using a default UF_H in the absence of data regarding whether adverse effects from TCEP exposure differ for certain subpopulations (such as those with genetic polymorphisms or underlying diseases), it is also not known whether the chosen default UF_H would fully cover pre-existing diseases or disorders U.S. EPA (2002b).

For cancer, the dose-response model applied to animal tumor data employed low-dose linear extrapolation, and this assumes *any* TCEP exposure is associated with some positive risk of getting cancer. EPA made this assumption in the absence of an established MOA for TCEP and according to guidance from U.S. EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b). Assuming all TCEP exposure is associated with some risk is likely to be health conservative because EPA does not believe that a mutagenic MOA is likely for TCEP and a threshold below which cancer does not occur is expected to exist. However, information is lacking with which to determine an appropriate threshold. Even though the cancer dose-response modeling assumes any exposure is associated with a certain risk, EPA presents risk estimates in comparison with benchmark risk levels (1 in 1,000,000 to 1 in 10,000).

Although there is likely to be variability in susceptibility across the human population, EPA did not identify specific human groups that are expected to be more susceptible to cancer following TCEP exposure. Other than relying on animal tumor data for the more sensitive sex, the available evidence does not allow EPA to evaluate or quantify the potential for increased cancer risk in specific subpopulations, such as for individuals with pre-existing diseases or those who smoke cigarettes. Given that a mutagenic mode of action is unlikely, EPA does not anticipate greater cancer risks from early life exposure to TCEP. Therefore, EPA is not applying an age-dependent adjustment factor.

Table_Apx E-2. PESS Evidence Crosswalk for Biological Susceptibility Considerations

Susceptibility Category	Examples of Specific Factors	Direct Evidence this Factor Modifies Susceptibility to TCEP		Indirect Evidence of Interaction with Target Organs or Biological Pathways Relevant to TCEP		Susceptibility Addressed in Risk Evaluation?
		Description of Interaction	Key Citations	Description of Interaction	Key Citation(s)	
Lifestage	Embryos/ fetuses/infants	Slight evidence for growth/gestational age effects in humans. Slight animal evidence for developmental toxicity (*e.g.*, decreased fertility and live births with some increased severity in the second generation) Lack of uniquely sensitive effects on neurodevelopment in animals (doses up to 90 mg/kg-day)	Oh et al. (2024) Crawford et al. (2020) Yang et al. (2022) NTP (1991a) Moser et al. (2015)			POD for male reproductive endpoints protective of effects in offspring [a]
	Pregnancy/ lactating status	Rodent dams not particularly susceptible during pregnancy and lactation except in one prenatal study, in which 7 of 30 dams died at 200 mg/kg-day	NTP (1991a) Hazleton Laboratories (1983) Moser et al. (2015) Kawashima et al. (1983)			POD for male reproductive endpoints protective of effects in dams
	Males of reproductive age	Reproductive outcomes (effects on seminiferous tubules) in adolescent male mice	Chen et al. (2015a)	Possible contributors to male reproductive effects/infertility (see also factors in other rows): • Enlarged veins of testes • Trauma to testes • Anabolic steroid or illicit drug use • Cancer treatment	CDC (2023b)	POD for this endpoint and study used to calculate non-cancer risks
	Children	Slight evidence for lower IQ in children with low SES Reproductive outcomes (effects on seminiferous tubules) in adolescent male mice	Percy et al. (2022) Chen et al. (2015a)			Adolescent animal POD used to calculate non-cancer risks; other variability and uncertainty addressed using default UF_H
	Elderly	No direct evidence identified				Use of default UF_H

Susceptibility Category	Examples of Specific Factors	Direct Evidence this Factor Modifies Susceptibility to TCEP		Indirect Evidence of Interaction with Target Organs or Biological Pathways Relevant to TCEP		Susceptibility Addressed in Risk Evaluation?
		Description of Interaction	Key Citations	Description of Interaction	Key Citation(s)	
Pre-existing disease or disorder	Health outcome/ target organs	No direct evidence identified		Several conditions may contribute to male reproductive effects/infertility: • Hormone disorders (hypothalamus/ pituitary glands) • Diabetes, cystic fibrosis, autoimmune disorders, certain infections Viruses such as human papilloma virus can increase susceptibility to cancer	CDC (2023b) CDC (2023a)	Use of default UF_H
	Toxicokinetics	Sex differences in toxicokinetic parameters might have resulted in differences in susceptibility.	Herr et al. (1991) Burka et al. (1991) Chapman et al. (1991)			Use of PODs for the more sensitive sex; Use of default UF_H
Lifestyle activities	Smoking	No direct evidence identified		Heavy smoking may increase susceptibility for reproductive outcomes and cancer.	CDC (2023a) CDC (2023b)	Qualitative discussion in this section (D.2) and this table
	Alcohol consumption	No direct evidence identified		Heavy alcohol use may affect susceptibility to reproductive outcomes and cancer.	CDC (2023b)	Qualitative discussion in this section (D.2) and this table
	Physical Activity	No direct evidence identified		Insufficient activity may increase susceptibility to multiple health outcomes. Overly strenuous activity may also increase susceptibility.	CDC (2022)	Qualitative discussion in this section (D.2) and this table

Susceptibility Category	Examples of Specific Factors	Direct Evidence this Factor Modifies Susceptibility to TCEP		Indirect Evidence of Interaction with Target Organs or Biological Pathways Relevant to TCEP		Susceptibility Addressed in Risk Evaluation?
		Description of Interaction	Key Citations	Description of Interaction	Key Citation(s)	
Sociodemo-graphic status	Race/ethnicity	No direct evidence identified (*e.g.*, no information on polymorphisms in TCEP metabolic pathways or diseases associated race/ethnicity that would lead to increased susceptibility to effects of TCEP by any individual group)				Qualitative discussion in this section (D.2) and this table
	Socioeconomic status	Slight evidence for lower IQ in children with low SES	Percy et al. (2022)	Individuals with lower incomes may have worse health outcomes due to social needs that are not met, environmental concerns, and barriers to health care access.	ODPHP (2023b)	Qualitative discussion in this section (D.2) and this table
	Sex/gender	*Humans:* Slight evidence for some differences (developmental, immunotoxicity); indeterminate for thyroid *Males (mice):* Potentially more sensitive regarding reproductive effects *Females (rats):* More sensitive for neurotoxicity Metabolism experiments using liver slices and microsomes show differences in metabolism by sex for rats, but not for humans. Thus, there is uncertainty regarding whether human females and males are susceptible subpopulations.	Oh et al. (2024) Crawford et al. (2020) Yang et al. (2022) Mendy et al. (2024) NTP (1991a) NTP (1991b) Chen et al. (2015a) Chapman et al. (1991)			PODs are used in the risk evaluation for both endpoints.

Susceptibility Category	Examples of Specific Factors	Direct Evidence this Factor Modifies Susceptibility to TCEP		Indirect Evidence of Interaction with Target Organs or Biological Pathways Relevant to TCEP		Susceptibility Addressed in Risk Evaluation?
		Description of Interaction	Key Citations	Description of Interaction	Key Citation(s)	
Nutrition	Diet	No direct evidence identified		Poor diets can lead to chronic illnesses such as heart disease, type 2 diabetes, and obesity. Obesity can increase susceptibility to cancer.	CDC (2023a) CDC (2020) CDC (2023c)	Qualitative discussion in this section (D.2) and this table
	Malnutrition	No direct evidence identified		Micronutrient malnutrition can lead to multiple conditions that include birth defects, maternal and infant deaths, preterm birth, low birth weight, poor fetal growth, childhood blindness, undeveloped cognitive ability. Thus, malnutrition may increase susceptibility to some/all health outcomes associated with TCEP.	CDC (2021) CDC (2023c)	Qualitative discussion in this Section (D.2) and this table
Genetics/ epigenetics	Target organs	No direct evidence identified		Genetic disorders, such as Klinefelter's syndrome, Y-chromosome microdeletion, myotonic dystrophy can affect male reproduction/fertility	CDC (2023b)	Use of default UF_H to assess variability among humans
	Toxicokinetics	No direct evidence identified		Specific enzymes have not been identified for TCEP's metabolic pathways. Therefore, potential polymorphisms are not known.		Use of default UF_H to assess variability among humans

Susceptibility Category	Examples of Specific Factors	Direct Evidence this Factor Modifies Susceptibility to TCEP		Indirect Evidence of Interaction with Target Organs or Biological Pathways Relevant to TCEP		Susceptibility Addressed in Risk Evaluation?
		Description of Interaction	Key Citations	Description of Interaction	Key Citation(s)	
Other chemical and nonchemical stressors	Built environment	No direct evidence identified		Poor-quality housing is associated with a variety of negative health outcomes.	ODPHP (2023a)	Qualitative discussion in this Section (D.2) and this table
	Social environment	No direct evidence identified		Social isolation and other social determinants (*e.g.*, decreased social capital, stress) can lead to negative health outcomes.	CDC (2023d) ODPHP (2023c)	Qualitative discussion in this Section (D.2) and this table
	Chemical co-exposures	An *in vitro* study of liver cells co-exposed to TCEP and benzo[a]pyrene activated pathways associated with cell proliferation and inflammation and increased expression of pro-inflammatory cytokines, whereas exposure to TCEP alone did not. TCEP showed anti-estrogenic activity (32 percent inhibition) *in vitro* using the breast adenocarcinoma cell line, MCF-7 after co-exposure with 17B-estradiol.	Zhang et al. (2017b) Krivoshiev et al. (2016)			Qualitative discussion in this Section (D.2) and this table

[a] An error in reporting the results in NTP (1991a) precluded using sex ratio; use of this endpoint would have resulted in using a LOAEL of 175 mg/kg-day with an HED of 23.3 mg/kg-day and a benchmark MOE of 300. This would have resulted in similar but slightly greater risk.

Appendix F PHYSICAL AND CHEMICAL PROPERTIES AND FATE AND TRANSPORT DETAILS

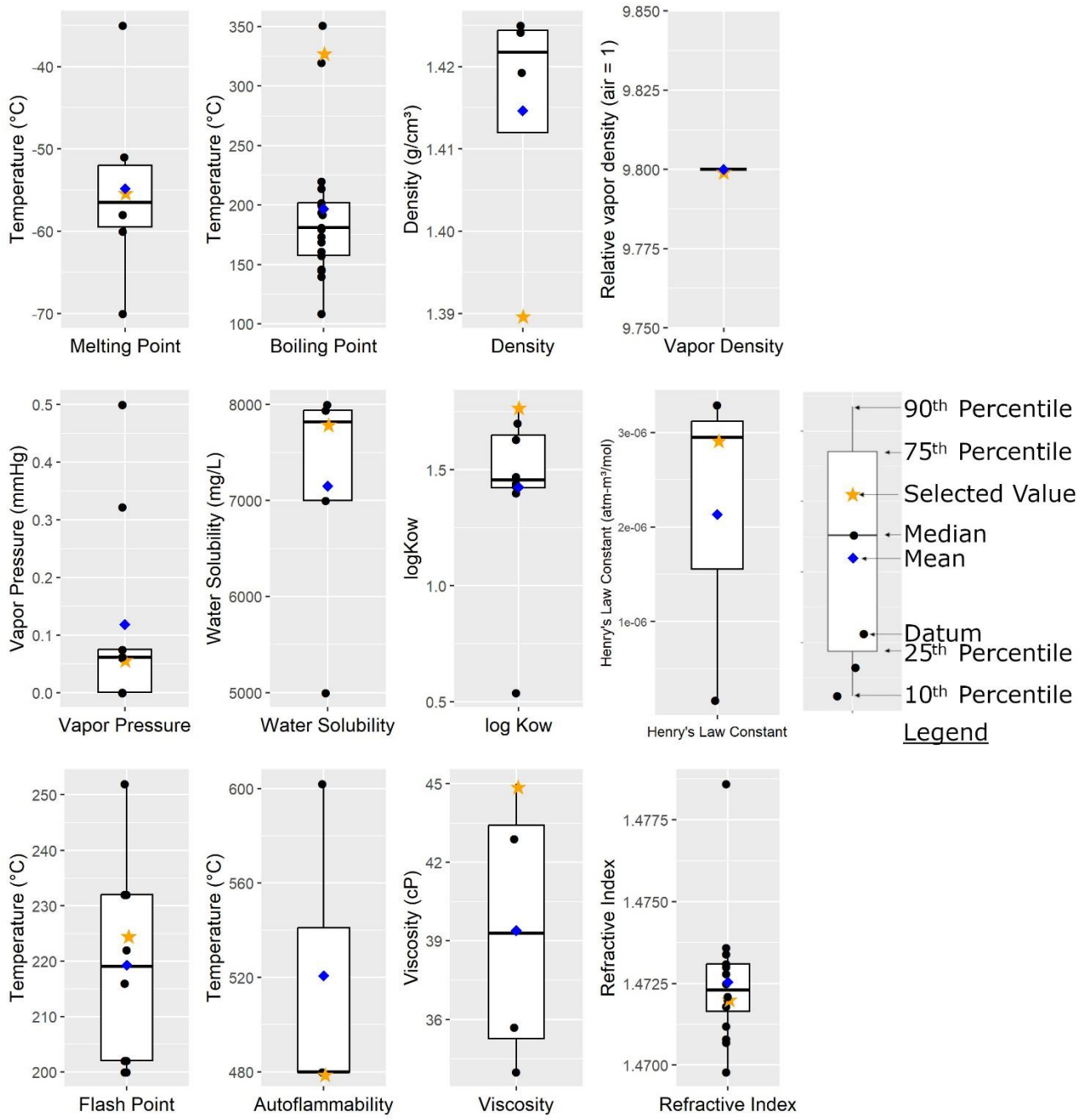
F.1 Physical and Chemical Properties Evidence Integration

The physical and chemical property values selected for use in the risk evaluation for TCEP are given in Table 2-1. These values were taken from the *Final Scope of the Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) CASRN 115-96-8* (U.S. EPA, 2020b), except for physical form, vapor density, autoflammability, flashpoint, Henry's Law constant, and logarithmic octanol:air partition coefficient (log K_{OA}). In the final scope, no vapor density, log K_{OA}, and autoflammability data were reported and a flashpoint value from a medium-quality study was provided. After the final scope was published, additional data were identified in the systematic review process.

The systematic review process identified multiple data with the same quality rating for many physical and chemical properties discussed in this document. Some of these data were duplicates that were initially extracted more than once (*e.g.*, when multiple databases cite the same study), but were later removed during data curation before any further analysis. Much of the remaining data were collected under standard environmental conditions (*i.e.*, 20 to 25 °C and 760 mm Hg).

When a specific data point is cited for a given physical and chemical property, priority is given to data from expert-curated, peer-reviewed databases that have been identified as "trusted sources" (U.S. EPA, 2021a). If no data are available from trusted databases, second preference is given to measured data from studies that implement experimental measurements according to established test guidelines or which are conducted according to scientific principles with sufficient documentation. Finally, estimated, or calculated data are only presented in the instance that no measured data is available.

A composite plot comprising box and whisker plots of reported high-, medium-, and low-quality physical and chemical property data values are shown in Figure_Apx F-1.



Figure_Apx F-1. Box and Whisker Plots of Reported Physical and Chemical Property Data Values

F.1.1 Physical Form

In the final scope (U.S. EPA, 2020b), physical state and physical properties were 2 of 17 endpoints provided. As provided in the *Final Scope of Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) Supplemental File – Data Extraction and Data Evaluation Tables for Physical and Chemical Property Studies* (U.S. EPA, 2020c), only one source was identified and evaluated as a high-quality data for the physical state endpoint. Ultimately, "liquid" was used in the risk evaluation. For physical properties, two sources were identified and evaluated as high-quality studies. The reason was not provided, but "clear, transparent liquid" was preferred and reported over "low odor." For this risk evaluation, both endpoints were combined and re-named to physical form. After the systematic review process was completed, six high-quality data were identified and extracted while a medium-quality study was excluded. TCEP is

identified as a clear, transparent liquid with slight odor (DOE, 2016; U.S. EPA, 2015b; ECB, 2009; Lewis and Hawley, 2007; Weil, 2001). These descriptions agree with the qualitative description given in the final scope (U.S. EPA, 2020b).

F.1.2 Vapor Pressure

Systematic review identified 19 high-quality vapor pressure data, including 12 data collected at 20–25 °C. However, five studies reported extrapolated vapor pressures from measured vapor pressures at higher temperatures and those studies were excluded for this risk evaluation. The remaining seven data collected under standard environmental conditions cover the range of 3.59×10^{-4} to 0.062 mm Hg at 20–25 °C. The vapor pressure of 0.0613 mm Hg at 25 °C reported by Dobry and Keller (1957) was selected for this risk evaluation because out of the 7 remaining vapor pressure data, 5 data reported the vapor pressure from Dobry and Keller (1957).

F.1.3 Vapor Density

A vapor density data was identified through systematic review. It was from a secondary source, NCBI (2020) and rated it high-quality. Therefore, the vapor density of 9.8 was included in the risk evaluation. The primary source of the data is ILO (2019).

F.1.4 Water Solubility

Systematic review identified 19 high-quality water solubility data, including 9 data collected at 20 °C. The data collected under standard environmental conditions cover the range of 7,000–7,900 mg/L at 20 °C. The water solubility value of 7,820 mg/L at 20 °C from ECB (2009) was selected for this risk evaluation because out of the 9 remaining water solubility data, 6 data reported the water solubility from ECB (2009).

F.1.5 Logarithmic Octanol:Air Partition Coefficient (log K_{OA})

Two high-quality log K_{OA} data were identified through systematic review. Okeme et al. (2020) gave a log K_{OA} range of 7.85 to 7.93. Yaman et al. (2020) gave a log K_{OA} value of 7.91. Because 7.91 is within the range of 7.85 to 7.93, the Okeme et al. (2020) data was selected for use in the risk evaluation.

F.1.6 Henry's Law Constant

A Henry's Law constant of 2.55×10^{-8} atm·m^3/mol at 25 °C was reported in the final scope (U.S. EPA, 2020b). It was calculated using the Bond method in HENRYWIN™, which is an estimation method that splits a compound into a summation of the individual bonds that comprise the compound (U.S. EPA, 2017a). However, when measured Henry's Law constant values are not available, a calculated value based on high-quality measured water solubility and vapor pressure data are typically preferred over an estimated value (Meylan and Howard, 1991). With a high-quality measured vapor pressure of 0.0613 mm Hg from Dobry and Keller (1957) and water solubility of 7,820 mg/L from ECB (2009), the revised Henry's Law constant is 2.945×10^{-6} atm·m^3/mol at 25 °C. Systematic review identified two Henry's Law constant data: one high-quality (Ekpe et al., 2020) and one medium-quality data (IPCS, 1998). Both data were not included in this risk evaluation because a calculated Henry's Law constant value based on high-quality measured water solubility and vapor pressure data were available for use in the risk evaluation.

F.1.7 Flash Point

Eight high-quality and four medium-quality flash point data were identified through systematic review. The flash point data ranged from 200 to 252 °C. In general, flash point is measured using either an open cup or closed cup technique. The closed cup technique normally gives lower values for the flash point than open cup (approximately 5–10 °C lower). The extracted flash point data include values measured

using both closed cup and open cup techniques and some sources not reporting the technique used. Four medium-quality data were excluded for this risk evaluation because high-quality flash point data are available. The 216 °C datum extracted from U.S. EPA (2015a) and Lewis and Hawley (2007) was excluded because the analytical method was not provided and there was no indication that a reliable method was used. The 202 °C datum extracted from IPCS (1998) was excluded because the data were extracted from a secondary source without peer review and did not provide a reference of the original source. The 200 °C datum extracted from U.S. EPA (2015a) was excluded because the test sample appeared to catch fire at approximately 200 °C but did not show a distinct flash point as defined by the ASTM D93 method. The 232 °C datum extracted from Toscano and Coleman (2012) and Sigma-Aldrich (2019) was excluded because the analytical method used was not reported. Between the remaining two high-quality flash point data, the 225 °C datum extracted from U.S. EPA (2015a) was selected for use in this risk evaluation because flash point is defined as "the lowest temperature at which a chemical will ignite with an ignition source."

F.1.8 Autoflammability

Three medium-quality autoflammability data were identified through systematic review. The 480 °C datum extracted from ECB (2009) and ILO (2019) was selected for use in this risk evaluation because autoflammability is defined as "the lowest temperature at which a chemical will spontaneously combust without an ignition source." Therefore, the 1,115 °F (≈602 °C) datum extracted from NTP (1992) was excluded.

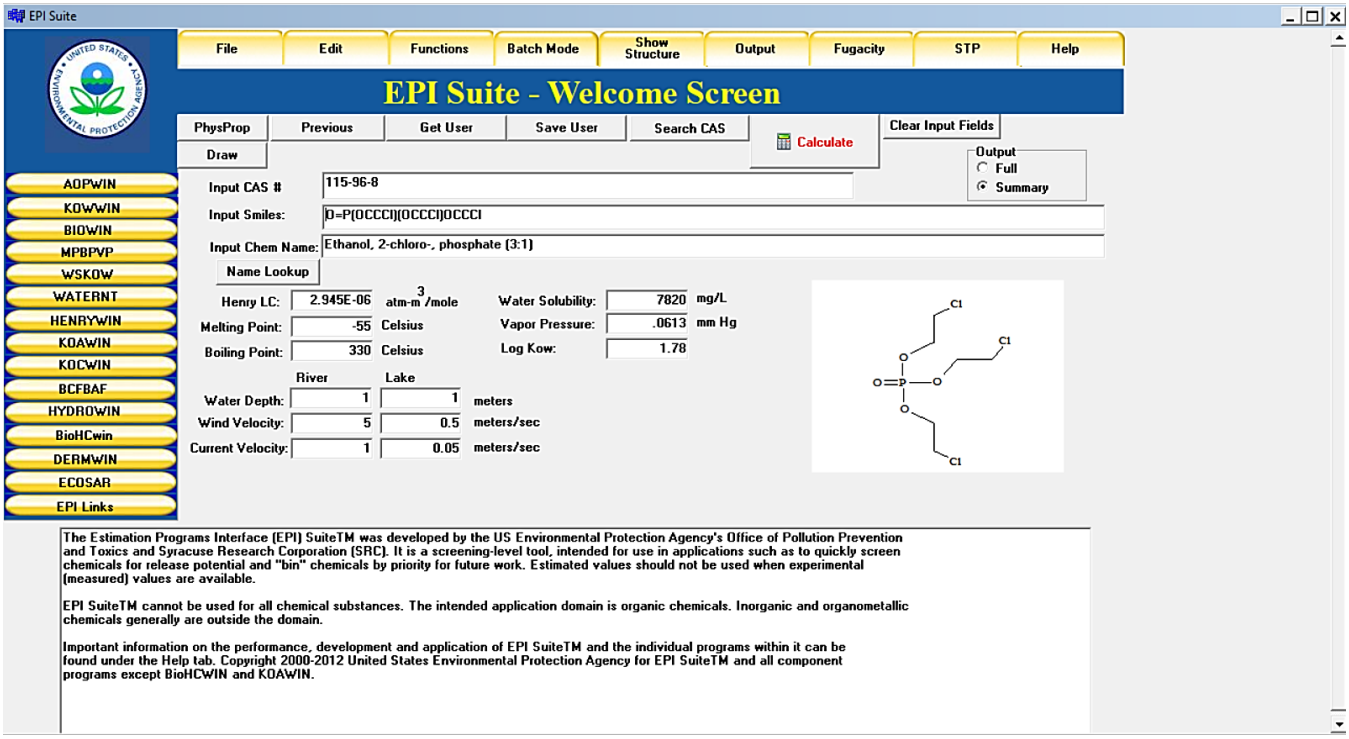
F.2 Fate and Transport

F.2.1 Approach and Methodology

EPA conducted a Tier I assessment to identify the environmental compartments (*i.e.*, water, sediment, biosolids, soil, groundwater, air) of major and minor relevance to the fate and transport of TCEP. Next, a Tier II assessment was conducted to identify the fate pathways and media most likely to cause exposure to environmental releases. Media-specific fate analyses were performed as described in Sections F.2.2, F.2.3, and F.2.4.

F.2.1.1 EPI Suite™ Model Inputs

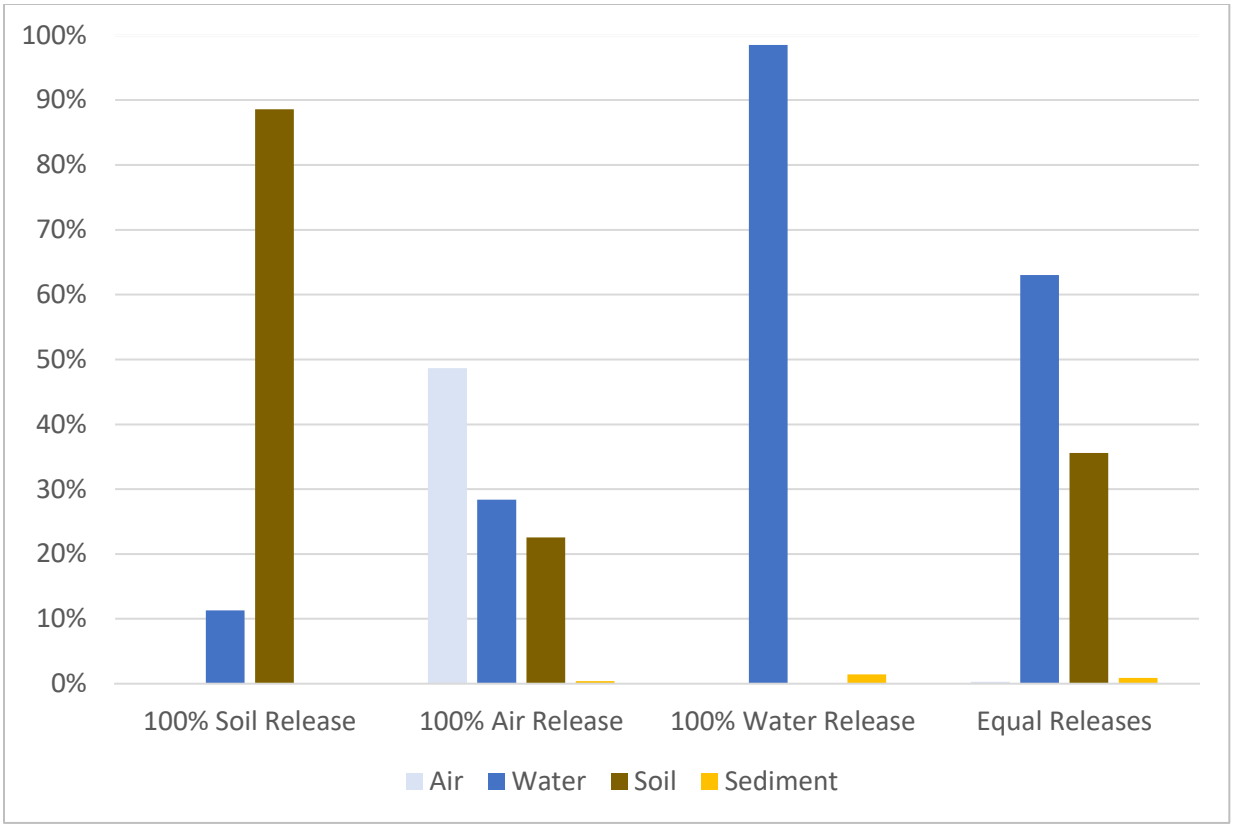
To set up EPI Suite™ for estimating fate properties of TCEP, the physical and chemical properties were input based on the values in Table 2-1. EPI Suite™ was run using default settings (*i.e.*, no other parameters were changed or input) (Figure_Apx F-2).



Figure_Apx F-2. Screen Capture of EPI Suite™ Parameters Used to Calculate Fate and Physical and Chemical Properties for TCEP

F.2.1.2 Fugacity Modeling

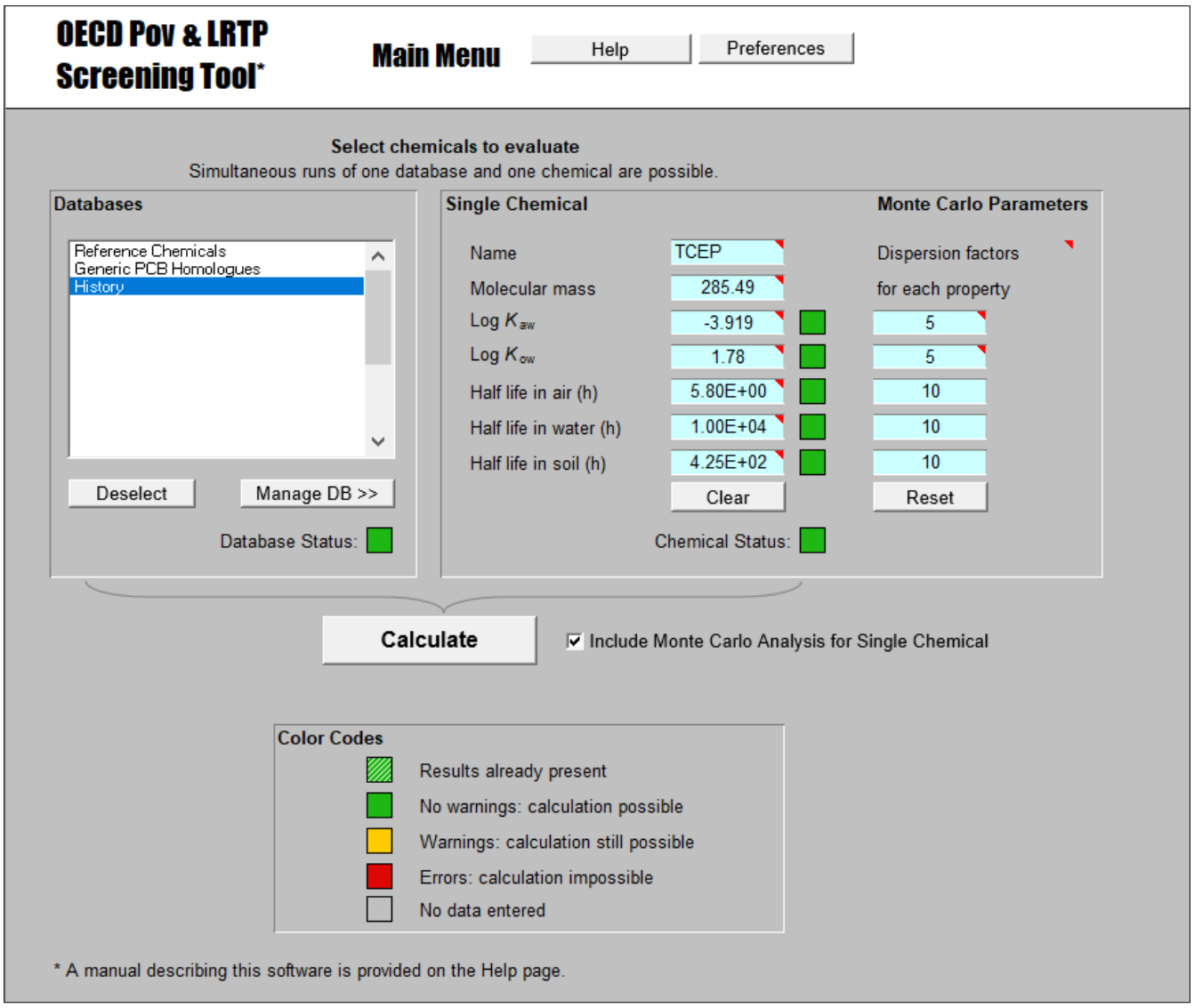
Because no current data were being reported to the TRI or DMR, TCEP releases to the environment could not be estimated. The approach described by Mackay et al. (1996) using the Level III Fugacity Model in EPI Suite™ (LEV3EPI™) was used for TCEP's Tier II analysis. LEV3EPI™ is described as a steady-state, non-equilibrium model that uses a chemical's physical and chemical properties and degradation rates to predict partitioning of the chemical between environmental compartments and its persistence in a model environment (U.S. EPA, 2017a). TCEP's physical and chemical properties were taken directly from Table 2-1. Environmental release information is useful for fugacity modeling because the emission rates will predict a real-time percent mass distribution for each medium. Instead, environmental degradation half-lives were taken from high-quality studies that were identified through systematic review to reduce levels of uncertainties. The results of the Level III Fugacity model reported releases that show that the emissions of TCEP will primarily partition to water (59%) and soil (38%) with less than 2 percent partitioning to sediment and less than 1 percent to air (Figure_Apx F-3). These results reiterate the Tier I analysis results that water and soil are expected to be an important environmental media for TCEP released to the environment. The LEV3EPI™ results were consistent with environmental monitoring data. Further discussion of TCEP partitioning can be found in Sections F.2.2, F.2.3, and F.2.4.



Figure_Apx F-3. EPI Suite™ Level III Fugacity Modeling Graphical Result for TCEP

F.2.1.3 OECD P_{OV} and LRTP Screening Tool

TCEP's long-range transport potential (LRTP) was evaluated by using OECD's Overall Environmental Persistence (P_{OV}) and LRTP Screening Tool, Version 2.2 (Wegmann et al., 2009). The OECD POV and LRTP Tool is in a spreadsheet format containing multimedia chemical fate models that were designed based on the recommendations of the OECD expert group to estimate environmental persistence and LRTP of organic chemicals at a screening level. With a chemical's physical and chemical properties, the OECD POV and LRTP Tool will be able to predict its P_{OV}, characteristic travel distance (CTD), and transfer efficiency (TE). P_{OV} is the overall persistence in the whole environment in days, CTD quantifies the distance in kilometers (km) from the point of release to the point at which the concentration has dropped to 1/e, or approximately 37 percent of its initial value, and TE estimates the percentage of emitted chemical that is deposited to surface media after transport away from the region of release. The OECD P_{OV} and LRTP Screening Tool calculates two emission scenarios specific CTD values, for emissions to air and water. Only transport in the medium that receives the emission is considered, thus CTD in air is calculated from the emission-to-air scenario and CTD in water is calculated from the emission-to-water scenario. No CTD is calculated for emissions to soil because soil is not considered to be mobile. The physical and chemical properties were input based on the values in Table 2-1 and Table 2-2 (Figure_Apx F-4). The modeling results will be discussed further in Sections F.2.2 and F.2.3.1.



Figure_Apx F-4. Screen Capture of OECD Pov and LRTP Screening Tool Parameters Used to Calculate TCEP's LRTP

F.2.1.4 Evidence Integration

A brief description of evidence integration for fate and transport is available in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Additional details on fate and transport evidence integration are provided here.

The environmental fate characteristics given in Appendix C of the *Final Scope of the Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) CASRN 115-96-8* (U.S. EPA, 2020b) were identified prior to completing the systematic review. The following sections summarize the findings and provide the rationale for selecting the environmental fate characteristics that was given in Table 2-2.

F.2.2 Air and Atmosphere

TCEP in its pure form is a liquid at environmental temperatures with a melting point of –55 °C (DOE, 2016; U.S. EPA, 2015a, b; Toscano and Coleman, 2012) and a vapor pressure of 0.0613 mm Hg at 25 °C (U.S. EPA, 2019b; Dobry and Keller, 1957). The log K_{OA} range of 7.5 to 7.98 indicates that TCEP is

expected to adsorb to the organic carbon present in airborne particles (Okeme et al., 2020; Ji et al., 2019; Wang et al., 2017b).

As an SVOC, TCEP will exist in both the gas and particle phases (Wang et al., 2020a; Okeme, 2018; TERA, 2015). Results from air monitoring studies reported concentrations of gaseous TCEP up to 6,499 pg/m^3 (Ma et al., 2021; Wu et al., 2020) and particle bound TCEP up to 2,100 pg/m^3 in North America (Wu et al., 2020; Abdollahi et al., 2017; Salamova et al., 2016; Salamova et al., 2014; Shoeib et al., 2014). Multiple studies have identified urban sources as sources of TCEP in the environment through fugitive emissions to air (Abdollahi et al., 2017; Luo et al., 2015; Möller et al., 2011). Although the exact sources of TCEP emissions from urban environment are unknown, they are likely the articles that were treated with or containing TCEP (Abdollahi et al., 2017; Luo et al., 2015; Wei et al., 2014; Möller et al., 2011; Aston et al., 1996).

Compared to outdoor air, TCEP concentrations are significantly higher in indoor air because TCEP has the potential to volatilize from treated products and diffuse into air, as well as partition onto dust due to its use as an additive (Qi et al., 2019; TERA, 2015; Liu et al., 2014; ATSDR, 2012; EC, 2009; NICNAS, 2001). In northern California, indoor air concentrations of TCEP were detected up to 15,340 pg/m^3 (Bradman et al., 2014) and dust concentrations was measured up to 6.84 µg/g (Bradman et al., 2012). In addition, TCEP is a known impurity in 2,2-bis(chloromethyl)-propane-1,3-diyltetrakis(2-chloroethyl) bisphosphate (V6) commercial mixtures that are primarily used in furniture and automobile foam. Higher concentrations of TCEP (up to 50.12 µg/g) were found in dust samples that were collected from the surfaces of the front and back seats of automobiles in Boston, MA (Fang et al., 2013).

TCEP is not expected to undergo significant direct photolysis in the atmosphere because its chemical structure does not absorb light at wavelengths greater than 290 nm (HSDB, 2015). TCEP in the gaseous phase is expected to degrade rapidly by reaction with photochemically produced hydroxyl radicals (·OH) in the atmosphere. A half-life of 5.8 hours was calculated from the AOPWIN™ module in EPI Suite™ using an estimated rate constant of 2.2×10^{-11} cm^3/molecules-second at 25 °C, assuming an atmospheric hydroxyl radical concentration of 1.5×10^6 molecules/cm^3 and a 12-hour day (U.S. EPA, 2017a). The atmospheric half-life of TCEP does not pertain to indoor environments due to lower hydroxyl radical concentrations, less mixing of air, and lower sunlight intensity.

TCEP has been detected in air and snow in remote locations such as the Arctic and Antarctica (Na et al., 2020; Wang et al., 2020a; Xie et al., 2020; Rauert et al., 2018; Li et al., 2017b; Sühring et al., 2016; Cheng et al., 2013b; Möller et al., 2012; NIVA, 2008). Particle-bound TCEP was found to be highly persistent in the atmosphere and had slower rates for the reaction with hydroxyl radicals due to the presence of atmospheric water (Wu et al., 2020; Li et al., 2017a; Liu et al., 2014). Particle-bound TCEP is primarily removed from the atmosphere by wet or dry deposition. Based on its physical and chemical properties and short half-life in the atmosphere ($t_{1/2}$ = 5.8 hours), TCEP was assumed to be not persistent in the air (U.S. EPA, 2017a). The OECD P_{OV} and LRTP Screening Tool was run to get additional information on TCEP's long-range transport potential in the air. For TCEP emissions in air, a P_{OV} of 11 days, CTD of 118 km (≈73 miles), and TE of 0.0142 percent were given using a molecular mass of 285.49 g/mol, logarithmic air:water partitioning coefficient (log K_{AW}) of –3.919, and logarithmic octanol:water partitioning coefficient (log K_{OW}) of 1.78 along with atmospheric half-life of 5.8 hours, water half-life of 10,000 hours, and soil half-life of 424.8 hours (Figure_Apx F-4). A CTD of 118 km (≈73 miles) suggests that TCEP does not have the potential to undergo long-range transport in the air and a TE of 0.0142 percent suggests that negligible fraction of TCEP emitted to air will be deposited to surface media such as water. CTD can also be calculated using the LEV3EPI™ module in EPI Suite™ without considerations for advection (U.S. EPA, 2017a; Beyer et al., 2000). After entering TCEP's

physical and chemical properties (Figure_Apx F-2), a CTD of 238 km (≈148 miles) was calculated. Particle-bound TCEP has the potential to undergo long-range atmospheric transport (LRAT) and it is likely the reason why TCEP is found in the Arctic and other remote locations with no source of releases. TCEP's LRTP could be crucially underestimated when using gaseous phase atmospheric half-life in multimedia models like the OECD P_{OV} and LRTP Screening Tool.

F.2.3 Aquatic Environments

Wastewater treatment effluent, atmospheric deposition, air-water gaseous exchange, and runoff have been identified as sources of TCEP detected in aquatic and marine environments, especially in urban areas (Ma et al., 2021; Cristale et al., 2019; Guo et al., 2017a; Kim et al., 2017).

F.2.3.1 Surface Water

TCEP is not expected to undergo abiotic degradation processes such as hydrolysis and photolysis in aquatic environments under environmentally relevant conditions. The rate of hydrolysis will be highly dependent on pH and temperature. TCEP showed no significant hydrolysis over 35 days at pH levels of 7, 9, and 11 at 20 °C, but an extensive degradation occurred when the pH level was adjusted to 13 ($t_{1/2}$ = 0.083 days) (Su et al., 2016). A hydrolysis study by Saint-Hilaire et al. (2011) observed the pH-dependent hydrolysis of TCEP between pH 8 to 13 at 50 °C and confirmed that TCEP's hydrolysis rates increased as pH levels increased. TCEP's hydrolysis half-life was estimated to be approximately 2 years at pH level of 8 at 25 °C. In addition, TCEP's hydrolysis rates also increased in the presence of reduced sulfur species. The calculated half-lives for TCEP after reacting with 5.6 mM bisulfide (HS^-) and 0.33 mM polysulfides (S_n^{2-}) were 90 and 30 days, respectively. The results also indicated that the three reduced sulfur species reacted with TCEP in a nucleophilic substitution reaction with bis(chloroethyl) phosphate (BCEP) being the major transformation product. The hydrolysis half-lives estimated by QSAR models were found to be inconsistent with experimental values. HYDROWIN™, an aqueous hydrolysis rate program in EPI Suite™, estimated TCEP's half-life to be approximately 20 days at pH 5 to 9 and approximately 17 days at pH 10 (U.S. EPA, 2017a). However, the half-life values from HYDROWIN™ were not included in this risk evaluation because the half-life values from high-quality hydrolysis studies mentioned above are available. In addition, it is unlikely for TCEP undergo indirect photolysis. No photolytic degradation was observed after exposing TCEP to natural sunlight for 15 days in lake water (Regnery and Püttmann, 2010a). Other experimental studies also observed no photolytic degradation (Chen et al., 2019b; Lee et al., 2014; Watts and Linden, 2009, 2008).

For biotic degradation in water, TCEP is not readily biodegradable under aerobic conditions. In a ready biodegradability test using the Modified Sturm test (OECD TG 301B), TCEP showed a minimal degradation after 28 days and the cumulative carbon dioxide (CO_2) production was negligible (Life Sciences Research Ltd, 1990b). In another ready biodegradability test using the Closed Bottle test (OECD TG 301D), TCEP was not readily biodegradable (Life Sciences Research Ltd, 1990c). Based on these two biodegradation studies, rapid biodegradation of TCEP is not likely when it is released to surface water.

A limited number of test results on anaerobic biodegradability of TCEP were available. Kawagoshi et al. (2002) reported that TCEP did not undergo biodegradation under anaerobic condition after 60 days using leachate from a sea-based solid waste disposal site in Japan. This study was not selected for use in the risk evaluation because it was rated as a medium-quality study because critical information on test conditions was not included and there was insufficient evidence to confirm that TCEP disappearance was not likely due to other processes. Due to lack of anaerobic biodegradation studies on TCEP, no anaerobic biodegradation data were selected for this risk evaluation.

Two studies showed that TCEP was able to undergo volatilization from oceans and had the highest water-to-air emission flux in two monitoring studies. In Li et al. (2017b), TCEP volatilization from seawater to air was seen in all samples across the North Atlantic and the Arctic, and equilibrium was reached in some samples that was caused by relatively low TCEP concentrations in seawater. A similar result was seen in another air-water gaseous exchange study on a coastal site where TCEP had the highest emission flux in water (Wang et al., 2018b). Both studies suggest that the air-water gaseous exchange is an important process for TCEP to transport between the air and water, causing a secondary pollution. TCEP's volatilization behavior did not align with its physical and chemical properties and modeling prediction. A Henry's Law constant of 2.945×10^{-6} atm·m^3/mol at 25 °C (Table 2-1) indicates that TCEP is not expected to volatilize from surface water (TERA, 2015; Toscano and Coleman, 2012; Regnery and Puettmann, 2009; Dobry and Keller, 1957). A Henry's Law constant is equivalent to a K_{AW} of 1.21×10^{-4} or log K_{AW} of –3.19 at 25 °C, which indicates that TCEP will favor water over air (U.S. EPA, 2017a). However, monitoring studies disproved that assumption and suggested that volatilization is significant. The Water Volatilization Program in EPI Suite™ estimated the volatilization half-lives of TCEP from a model river and lake and default settings were applied (see default settings in Figure_Apx F-2). TCEP's volatilization half-life from a model river was 337.6 hours (≈14 days), and 3,825 hours (≈159 days) for the model lake (U.S. EPA, 2017a). Overall, TCEP's potential to volatilize from water can be underestimated significantly if one relies solely on interpreting its physical and chemical properties or using QSAR models. Only experimental data would properly describe TCEP's volatilization behavior.

When precipitation events occur, TCEP's mobility in the environment will be greatly enhanced because rain and snow are believed to be effective scavengers of organic contaminants (Awonaike et al., 2021; Mihajlovic and Fries, 2012; Regnery and Puettmann, 2009; Lei and Wania, 2004). Atmospheric deposition has been identified as an important source of TCEP to surface water, especially in urban areas. Several studies showed that higher TCEP concentrations in precipitation were generally seen in densely populated areas with high traffic volume (Kim and Kannan, 2018; Regnery and Püttmann, 2010b; Regnery and Puettmann, 2009; Marklund et al., 2005b). In addition, storm water and urban runoff can contribute to additional emissions to surface water. The presence of TCEP in runoffs can be attributed to TCEP's use as an additive in car interiors and building materials and high water solubility. During periods without precipitation events, dry deposition is expected to occur (Na et al., 2020; Li et al., 2017b; Lai et al., 2015; Mihajlovic and Fries, 2012).

The OECD P_{OV} and LRTP Screening Tool was run to get additional information on TCEP's LRTP in water (Figure_Apx F-4). For TCEP emissions in water, a P_{OV} of 414 days, CTD of 707 km (≈439.3 miles), and TE of 0.0014 percent were estimated. A CTD of 707 km suggests that TCEP does not have the potential to undergo long-range transport. Yet, TCEP was detected in the waters of the Arctic, which is approximately 1,775 miles away from New York City (Na et al., 2020; McDonough et al., 2018; Li et al., 2017b). As previously mentioned, snow is an effective scavenger of organic contaminants, and it is possible to see the TCEP concentration in adjacent surface water spike from global warming. In addition, plastic debris, and ocean currents (*e.g.*, gyres) may have played a role in TCEP being widely distributed in aquatic and marine environments (Xie et al., 2020; Li et al., 2017b; Cheng et al., 2013a; Andresen et al., 2007). Plastic debris existing in marine environments have been found to contain various types of chemicals (Takada and Karapanagioti, 2019; Zhang et al., 2018a; Mato et al., 2001). Plastic products typically contain various additives that are used at high volume fractions in the plastic formulation such as plasticizers and flame retardants to maintain their performances (Takada and Karapanagioti, 2019). In locations where waste is uncollected or unmanaged, plastic wastes are likely to end up as litter where TCEP are released into the open environment. Extreme events such as storms, floods, cyclones, tidal waves, and tsunamis, are also a significant immediate source of land-based plastic

debris. Plastic wastes containing TCEP can potentially migrate from the plastic product to water by the weathering of microplastics (Hahladakis et al., 2018). Because TCEP has primarily been used as an additive flame retardant and plasticizers, they can easily leach from plastic wastes. Furthermore, plastic debris (*e.g.,* macroplastics, microplastics) could act as carriers for TCEP. The high specific surface areas of microplastics make them a good sorbent for hydrophobic and hydrophilic organic chemicals (Zhang et al., 2018a). Widely used plastics such as polyvinyl chloride (PVC) and polyethylene (PE) sorb organic pollutants from seawater after they are exposed to environmental conditions (Takada and Karapanagioti, 2019). In Chen et al. (2019a) TCEP was seen to sorb onto PVC and PE microplastics in seawater. When the temperature was in the range of 5 to 15 °C, the adsorption capacity of TCEP increased with increasing temperature, but when the temperature was greater than 15 °C, the adsorption capacity decreased with increasing temperature. Through adsorbing pollutants from surrounding seawater, microplastics can accumulate and increase the concentrations of pollutants up to the order of 10^6 (Mato et al., 2001). Plastic wastes are found in the ocean all over the world and they can travel long distances, especially to remote regions.

Based on the findings provided above, TCEP has the potential undergo long-range transport in water and its LRTP could be underestimated when using multimedia models like the OECD P_{OV} and LRTP Screening Tool.

F.2.3.2 Sediments

TCEP can be transported to sediment from overlying surface water by advection and dispersion of dissolved TCEP and by deposition of suspended solids containing TCEP. TCEP is expected to partition to organic matter in suspended and benthic sediments based on its measured sediment logarithmic organic carbon:water partition coefficient (log K_{OC}) values ranging from 3.23 to 3.46 (Zhang et al., 2021; Wang et al., 2018a; Zhang et al., 2018b). Higher concentrations of TCEP in sediment are expected to be found at potential source locations (*e.g.,* near urban and industrialized areas) (Chokwe and Okonkwo, 2019; Tan et al., 2019; Lee et al., 2018; Wang et al., 2018a; Cao et al., 2017; Maruya et al., 2016; Cristale et al., 2013).

A limited number of test results on anaerobic biodegradability of TCEP were available (see Appendix F.2.3.1). Systematic review did not identify anaerobic biodegradation studies for TCEP. However, systematic review identified two published risk assessments that reported an anaerobic biodegradation study. EC (2009) and U.S. EPA (2015a) reported that no degradation was observed for TCEP in an anaerobic biodegradation study after 58 days using ISO 11734, which is equivalent to OECD TG 311 (Noack, 1993). This result was not selected for use in the risk evaluation because the original study by Noack (1993) was published in German; therefore, it did not undergo the systematic review process.

After systematic review concluded, four studies observing anaerobic biodegradation of TCEP were identified. Pang et al. (2018) studied the fate of TCEP in sewage sludge with four different treatments, two of which were anaerobic digestion studies (Treatments 3 and 4). The results of this study confirm the recalcitrance of TCEP under anaerobic conditions and TCEP concentrations in both treatments were seen to increase over time. The removal rates were –0.41 and –74.8 percent for Treatments 3 and 4, respectively. The authors did not discuss the reasons for the increase in TCEP's final concentrations, but the authors did note that the final concentrations of Tri(n-butyl) phosphate (TnBP) increased during composting due to the "inhomogeneity" of the composts. Because of this uncertainty, Pang et al. (2018) is used qualitatively to support TCEP's persistence in anaerobic environment and will not be used to derive a biodegradation half-life. Yang et al. (2021) carried out biodegradation experiments using activated sludges from aerobic and anerobic ponds of the Nanjing Chendong STP in China mixed with kitchen garbage biomass and agricultural residues. The biodegradation of TCEP under anaerobic

conditions was observed to be very slow. The removal of TCEP did not reach 80 percent after 60 days. Because the media and inoculum used in Yang et al. (2021) comprised a rich nutrient profile and high microbial biomass, the direct use of the reported half-lives for estimating persistence in environmental sediments is not appropriate; therefore, this study was excluded from use in this risk evaluation. Yang et al. (2023) studied the degradation of TCEP using an anaerobic enrichment culture from end-of-life vehicles dismantling sites in Guangzhou, China. The study demonstrated that microbes, specifically *Dehalococcoides* sp., was able to anaerobically transform TCEP. However, the microbial community was highly enriched and the system highly controlled. This study will not appropriately represent TCEP's fate under anaerobic conditions and will not be considered in this risk evaluation.

When considering the results of the Pang et al. (2018) and Life Sciences Research Ltd (1990b) studies (Table 2-2), it is highly likely that TCEP will not degrade under anaerobic conditions and be persistent in the sediment compartment.

The rate of biodegradation in sediments can be estimated by extrapolation from aerobic biodegradation testing or estimated by considering that the rate of anaerobic degradation is typically at least three to four times slower (64 FR 60197, November 9, 1999). For the water compartment, TCEP did not pass a ready biodegradability test (OECD TG 301B) (Life Sciences Research Ltd, 1990b) (Table 2-2), so a default water half-life of 10,000 hours was used as described on page 4 in the *Interim Guidance for Using Ready and Inherent Biodegradability Tests to Derive Input Data for Multimedia Models and Wastewater Treatment Plants (WWT) Models (9/1/2000)* (U.S. EPA, 2000a). Considering that the rate of anaerobic degradation is 3 to 4 times slower than aerobic biodegradation, the estimated half-life of TCEP would be 30,000 to 40,000 hours in the sediment compartment.

F.2.3.3 Key Sources of Uncertainty

Several studies reported that TCEP concentrations in overlying water were higher when compared to sediment (Lee et al., 2018; Ma et al., 2017; Brandsma et al., 2015; Cao et al., 2012; Kawagoshi et al., 1999). For example, Lee et al. (2018) reported a TCEP mean concentration of 255 ng/L in the waters of Lake Shihwa in Korea, while the sediment had a mean TCEP concentration of 18.4 ng/g dry weight. TCEP concentrations in surface water and sediment may vary in locations due to the competing processes of advection, turbulence (sediment-water and air-water mixing), the bioturbation of benthic animals, and resuspension of particulate matters. In addition, sediment concentrations were observed to be correlated with sediment organic carbon content (Lee et al., 2018; Wang et al., 2018a; Xing et al., 2018; Zhong et al., 2018) and TCEP's water solubility may be a major factor.

F.2.4 Terrestrial Environments

TCEP is released to terrestrial environments via land application of biosolids, disposal of solid waste to landfills, and atmospheric deposition.

F.2.4.1 Soil

Based on its measured log K_{OC} values in soil ranging from 2.08 to 2.52 (Table 2-2), TCEP accumulation in soil is expected to be unlikely and TCEP may be mobile and eventually migrate to groundwater (see Section F.2.4.2). TCEP in the soil was seen to be vertically transported to deeper soil horizons, causing TCEP concentration in the surface soil to be lower (He et al., 2017; Bacaloni et al., 2008). Zhang et al. (2022) reported that higher levels of TCEP was found deeper in the soil (30 to 80 cm) compared to the surface soil samples (0 to 20 cm). Mihajlovic and Fries (2012) reported a similar result in its study.

The estimated log K_{OC} value for TCEP is 2.59, using the molecular connectivity index (MCI) method in KOCWIN™ (U.S. EPA, 2017a). The estimated value from EPI Suite™ is not included in this risk evaluation because the log K_{OC} values from high-quality field studies are available.

Systematic review identified two high-quality studies on TCEP degradation in soil. Hurtado et al. (2017) studied the degradation of TCEP in an agricultural soil from Spain. The soil had a sandy texture (90 percent sand, 8 percent silt, and 2 percent clay) and a total organic carbon content of 5 g/kg. After 40 days, 78 percent of TCEP degraded under aerobic conditions at test substance concentration of 50 µg/kg. A half-life of 17.7 days (Table 2-2) was estimated based on second-order kinetics. Another soil degradation study was identified, but this study was evaluated as low-quality ((ECB, 2009), citing (Brodsky et al., 1997)). The primary degradation of TCEP at a concentration of 5 mg/kg soil was conducted in a laboratory test system with standard soil for 100 days. The degradation kinetic curve was fitted to a 2nd order square root function resulting in a DT50 of 167 days and DT90 of >100 days. In addition, TCEP was seen to be slightly mobile in a leaching test. However, this study is not included in this risk evaluation because the testing conditions, inoculum information, sampling and analytical methods were not reported, and the omissions likely had an impact on the study results. Zhu et al. (2023) characterized the attenuation of TCEP in landfill humus and subsoil taken from a non-sanitary landfill that once was remediated. The study highlighted the roles of both abiotic degradation and biotic degradation contributing to the reduction of TCEP in both soil types. The authors indicate possible co-factors mediating the dechlorination of TCEP (*e.g.*, iron, sulfur, and carbon levels) pointing to the variety of conditions under which TCEP may be more readily degraded. Given the specificity of the experiments, Zhu et al. (2023) is not appropriate for deriving anaerobic biodegradation rates in natural environments.

TCEP in soil can re-volatilize from contaminated soil into the atmosphere causing a secondary pollution. A Henry's Law constant of 2.945×10^{-6} atm·m^3/mol at 25 °C, calculated based on a vapor pressure of 0.0613 mm Hg and a water solubility of 7,820 mg/L at 25 °C, indicates that TCEP is not expected to volatilize from dry soil but possibly from moist soil (ATSDR, 2012; Toscano and Coleman, 2012; Regnery and Puettmann, 2009; Dobry and Keller, 1957). Yet, there are field studies showing that TCEP underwent an air-soil exchange. In Wang et al. (2020b), the air-soil exchange behavior of TCEP varied between locations. TCEP was observed to be at an air-soil exchange equilibrium in the suburban and rural areas, but net volatilization occurred in the urban area. The highest volatilization flux was found at a site near a bus terminal. Yadav et al. (2018) reported net volatilization from soil to the air as TCEP's principal process in air-soil exchange. Han et al. (2020) reported a net volatilization in a sampling site located in the Arctic.

Also, several studies have reported that atmospheric deposition of TCEP may have contributed to soil contamination because there were no point sources nearby (Ji et al., 2019; Ren et al., 2019; Fries and Mihajlović, 2011; Mihajlović et al., 2011). In Bacaloni et al. (2008), lake water samples were collected from three remote volcanic lakes in central Italy. The three lakes were specifically chosen because there were no local contamination sources (*e.g.*, tributaries, industries, sewage treatment plants) nearby. Therefore, the possible sources of contamination would be from local anthropogenic activities, long-range transport and deposition from rainfall, or runoff processes. TCEP was detected in all three lakes at the ng/L level and the maximum concentrations occurred during the late summer to autumn months (August to October), which coincides with higher tourism activity and vehicular traffic at all three locations. In Han et al. (2020), the net deposition from air to soil was found to be predominant in four out of five sampling sites in the Arctic.

F.2.4.2 Groundwater

There are two sources of TCEP in the environment that may contaminate groundwaters. Point sources include wastewater effluents and landfill leachates and are discussed in Sections F.2.5.2 and F.2.4.3. Diffuse sources include storm water runoff and runoff from biosolids applied to agricultural land and are discussed in sections F.2.3.1 and F.2.4.4.

Municipal solid waste landfills (MSWLFs) can be a source of TCEP groundwater contamination. Historic landfills are more likely to lack the infrastructure of modern landfills, such as liners, leachate collection systems, and reactive barriers, which would prevent leachate from entering the groundwater system (Propp et al., 2021; Lapworth et al., 2012; Barnes et al., 2004).

Propp et al. (2021) assessed contaminants of emerging concern in leachate-impacted groundwater from 20 closed MSWLFs in Ontario, Canada. Those "historic" landfills had been closed for at least three decades. High concentrations of TCEP were reported in groundwater up to 2.92 µg/L. In addition, Buszka et al. (2009) collected groundwater samples from a domestic well located in a neighborhood east of the Himco Dump, which is an unlined landfill that was used for commercial, industrial, medical, and general waste disposal from 1960 to 1976 in Elkhart, Indiana. TCEP concentration ranged from 0.65 to 0.74 µg/L. Both studies suggests that TCEP in landfill impacted groundwater was resistant to biotic and abiotic degradation processes and is very persistent. Barnes et al. (2004) collected groundwater samples from a historic landfill in central Oklahoma. The landfill was unlined and built adjacent to the Canadian River in 1920, then covered with a clay cap and vegetated when it was permanently closed in 1985. TCEP concentration of 0.36 µg/L was measured in a well that was 3.28 feet away from the landfill. However, a TCEP concentration of 0.74 µg/L was measured in a well located 305 feet away from the landfill. This shows TCEP's potential to be transported away from point sources and enter groundwater.

F.2.4.3 Landfills

TCEP is not considered a hazardous waste, so it is not listed under Subtitle C of the Resource Conservation and Recovery Act (RCRA) (40 CFR 261). Solid waste containing TCEP can be disposed in MSWLFs or industrial waste landfills (*i.e.*, construction and demolition [C&D] debris landfills). MSWLFs that were built after 1991 are required to use a composite liner and a leachate collection system. The composite liner includes a minimum of 30-mil flexible membrane liner (FML) overlaying a two-foot layer of compacted soil lining the bottom and sides of the landfill (40 CFR 258.40). It is expected that solid waste containing TCEP will be disposed to a lined landfill with a leachate collection system. However, historic landfills are likely to lack the infrastructure of modern landfills, such as liners, leachate collection systems, and reactive barriers (Propp et al., 2021; Lapworth et al., 2012; Barnes et al., 2004). Leachate-impacted groundwater in historic landfills is discussed in Section F.2.4.2.

As discussed in Section 2.2.2, TCEP is primarily used as an additive plasticizer and flame retardant. When used as an additive, TCEP is added to manufactured materials via physical mixing rather than chemical bonding (Qi et al., 2019; Liu et al., 2014; ATSDR, 2012; EC, 2009; NICNAS, 2001). Consequently, it is highly likely that TCEP will be released from the solid wastes and enter the leachate. Leachates from 11 landfill sites in Japan reported TCEP concentrations in the range of 6 to 30,100 ng/L (Yasuhara et al., 1999). The maximum concentration of TCEP was reported in a landfill that consisted of waste plastics, waste combustion residue, plants, and domestic incombustible wastes. Several other studies also showed high concentrations of TCEP in leachate samples collected from MSWLFs in the United States and China (Qi et al., 2019; Deng et al., 2018a; Masoner et al., 2016; Masoner et al., 2014b).

Landfill leachate can be discharged to WWTPs and the release of TCEP to surface water from treated landfill leachate will depend on the removal of TCEP during wastewater treatment (see Section F.2.5.2). The fate and transport of TCEP entering surface water is discussed in Section F.2.3.1.

F.2.4.4 Biosolids

Sludge is defined as the solid, semi-solid, or liquid residue generated by wastewater treatment processes. The term "biosolids" refers to treated sludge that meet the EPA pollutant and pathogen requirements for land application and surface disposal (40 CFR 503).

Because TCEP is resistant to degradation in wastewater treatment, some residual concentrations of TCEP may be present in biosolids and transferred to surface soil during land application. TCEP concentrations between 78.9 to 317 ng/g dry weight were detected in sewage sludge collected from wastewater treatment plants located in the United States (Wang et al., 2019c; Kim et al., 2017). An anaerobic digestion study using sewage sludge showed that TCEP was persistent under anaerobic conditions (Pang et al., 2018). It is likely that dissolved TCEP will eventually reach surface water via runoff after the land application of biosolids due to its persistence.

F.2.4.5 Key Sources of Uncertainty

There are significant differences between the predicted and the field observed log K_{OC} values. The predicted log K_{OC} values are generally lower than the ones reported from field studies. The log K_{OC} reported in previous assessments of TCEP were in the range of 2.04 to 2.59 (TERA, 2015; ATSDR, 2012; EC, 2009; ECB, 2009; NICNAS, 2001). K_{OC} values within this range are associated with low sorption to soil and will be able to migrate to groundwater. However, a range of 2.5 to 4.3 was obtained from several field studies (Awonaike et al., 2021; Zhang et al., 2021; Wang et al., 2018a; Zhang et al., 2018b). Log K_{OC} within this range are associated with moderate to strong sorption to soil, sediment, and suspended solids.

F.2.5 Persistence Potential

Biotic and abiotic degradation studies have shown TCEP to be persistent. In the atmosphere, TCEP in the gaseous phase will be degraded by reacting with hydroxyl radicals (·OH), but particle-phase TCEP will not be degraded (see Section F.2.2). TCEP does not undergo hydrolysis under environmentally relevant conditions and is persistent in water (see Section F.2.3.1), sediment (see Section F.2.3.2), and soil (see Section F.2.4.1). Using the Level III Fugacity model in EPI Suite™ (LEV3EPI™) (see Section F.2.1.2), TCEP's overall environmental half-life was estimated to be approximately 168 days (U.S. EPA, 2017a). Therefore, TCEP is expected to be persistent in the atmosphere as well as aquatic and terrestrial environments.

F.2.5.1 Destruction and Removal Efficiency

Destruction and removal efficiency is a percentage that represents the mass of a pollutant removed or destroyed in a thermal incinerator in relative to the mass that entered the system. EPA requires that hazardous waste incineration systems destroy and remove at least 99.99 percent of each harmful chemical in the waste, including treated hazardous waste (46 FR 7684, January 23, 1981).

Only one study was identified in regard to thermal treatment and open burning of articles containing TCEP. Li et al. (2019a) reported that the articles released TCEP in the range of 9,800 to 49,000 ng/g after undergoing thermal treatment at 300 °C for 150 minutes. For open burning, the articles released TCEP in the range of 1,000 to 2,600 ng/g after being exposed to an open flame for 3 minutes at 800 to 1,350 °C. These results showed that TCEP was not completely destroyed. This finding, however, is

expected because flame retardant-containing materials are known/intended to have reduced flammability, which can result in incomplete combustion.

When undergoing thermal degradation in air at 220 °C and higher, TCEP will rapidly decompose to produce numerous toxic byproducts, including 1,2-dichloroethane ($C_2H_4Cl_2$), vinyl chloride (C_2H_3Cl), hydrogen chloride (HCl), carbon monoxide (CO), and acetaldehyde (C_2H_4O), among others (U.S. EPA, 2015a; NICNAS, 2001; Muir, 1984; Paciorek et al., 1978).

Because open burning can contribute to the emission of TCEP or other toxic byproducts to the surrounding environment (Matsukami et al., 2015), thermal treatment and open burning are not favorable options for the disposal of TCEP.

F.2.5.2 Removal in Wastewater

Wastewater treatment is performed to remove contaminants from wastewater using physical, biological, and chemical processes. Generally, municipal wastewater treatment facilities apply primary and secondary treatments. During the primary treatment, screens, grit chambers, and settling tanks are used to remove solids from wastewater. After undergoing primary treatment, the wastewater typically undergoes a secondary treatment. Secondary treatment processes can remove up to 90 percent of the organic matter in wastewater using biological treatment processes such as trickling filters or activated sludge. Sometimes an additional stage of treatment, such as tertiary treatment, is utilized to further clean water prior to release using advanced treatment techniques (*e.g.*, ozonation). A negative removal efficiency can be reported if the pollutant concentration is higher in the effluents than the pollutant concentration in the influents.

Because TCEP is not readily biodegradable under aerobic conditions based on two ready biodegradability tests (Life Sciences Research Ltd, 1990b, c), it is not expected to be removed from wastewater by biodegradation. This conclusion is supported by STPWIN™, an EPI Suite™ module that estimates chemical removal in sewage treatment plants. STPWIN™ estimated that a total of 2.23 percent of TCEP in wastewater will be removed: 0.08 percent by biodegradation, 0.17 percent by air stripping, and 1.99 percent by sorption to sludge (U.S. EPA, 2017a). STPWIN™ simulates a conventional wastewater treatment plant that uses activated sludge secondary treatment. The biodegradation half-life parameter was set to 10,000 hours for the primary clarifier, aeration vessel, and settling tank, which is a default for recalcitrant chemicals. The physical and chemical properties for TCEP given in Table 2-1 were used (Figure_Apx F-2). The results from STPWIN™ were not included in this risk evaluation because high-quality wastewater treatment studies are available.

A total of 19 wastewater treatment studies were identified during systematic review. Seven were evaluated and rated as medium-quality studies. These studies were not included in this risk evaluation. Numerous high-quality wastewater treatment studies reported either a negative removal efficiency or a removal of less than 10 percent for TCEP after undergoing primary and secondary treatments. An overall TCEP removal of –60.2 percent was calculated for a municipal wastewater treatment in Frankfurt, Germany (Fries and Puttmann, 2001). An average overall TCEP removal of −32.2 percent was calculated from the removals reported for five activated sludge treatment plants in Catalonia, Spain (Cristale et al., 2016).

An TCEP removal of −18.9 percent removal was calculated for a municipal wastewater treatment plant in Beijing, China (Liang and Liu, 2016). TCEP was not removed (0%) in two activated sludge treatment plants in western Germany (Meyer and Bester, 2004) and an activated sludge treatment plant in South Korea (Kim et al., 2007). An overall TCEP removal of 9 percent was calculated from the removals

reported for two small-, three medium-, and two large-sized municipal sewage treatment plants in Sweden (Marklund et al., 2005a). An overall TCEP removal of −19.1 percent was reported from an activated sludge plant in Albany, New York, based on measured concentrations in wastewater and suspended particle matter (Kim et al., 2017). This study was selected for use in this risk evaluation because this is the best representative of the full-scale wastewater treatment processes that are used in the United States.

Several high-quality studies observing the efficacy of advanced (tertiary) treatment techniques were identified. Cristale et al. (2016) reported a low TCEP removal rate (<38%) after a several series of advanced treatment techniques such as chlorination, ozonation, ultraviolet (UV) radiation, and UV/hydrogen peroxide (UV/H_2O_2). Liang and Liu (2016) reported an overall TCEP removal of −30.1 percent after undergoing tertiary treatment that consisted of hyperfiltration, ozonation, and chlorination. Pang et al. (2016) reported an overall TCEP removal of 0.3 percent and 12.3 percent using UV filters in two activated sludge plants in China.

Overall, because TCEP has a high water solubility and remains in treated wastewater, negligible to low accumulation of TCEP will be found in sewage sludge and will not significantly contribute to the removal of TCEP in wastewater treatments (Kim et al., 2017; Cristale et al., 2016; Liang and Liu, 2016; Marklund et al., 2005a). In addition, biodegradation and air stripping are not expected to be significant removal processes. Therefore, TCEP is expected to pass through wastewater treatment systems and be discharged into the receiving waters.

F.2.5.3 Removal in Drinking Water Treatment

In the United States, drinking water typically comes from surface water (*i.e.*, lakes, rivers, reservoirs) and groundwater. The source water then flows to a drinking water treatment plant (DWTP) where it undergoes a series of water treatment steps before being dispersed to homes and communities. In the United States, public water systems often use "conventional treatment" processes that include coagulation, flocculation, sedimentation, filtration, and disinfection, as required by law.

Five U.S. studies were identified and reviewed on the removal of TCEP in DWTPs. Those DWTPs consisted of both conventional and advanced treatment processes and used river water as the source. In all five studies, TCEP was found to be either minimally removed or not removed at all after undergoing pre-ozonation (or coagulation), flocculation, sedimentation, ozonation, filtration, and chlorination (Choo and Oh, 2020; Zhang et al., 2016a; Benotti et al., 2009; Snyder et al., 2006; Westerhoff et al., 2005; Stackelberg et al., 2004).

Several studies have demonstrated that granular activated carbon (GAC) or powdered activated carbon (PAC) enhanced the removal of TCEP when added to conventional treatment methods (Feng et al., 2023; Choo and Oh, 2020; Padhye et al., 2014; Westerhoff et al., 2005; Stackelberg et al., 2004). A South Korean drinking water treatment study reported a removal efficiency of 52 percent after undergoing coagulation and ultrafiltration. After undergoing the GAC step, 73.7 percent of TCEP was removed (Kim et al., 2007). Notably, a high level of uncertainty exists about TCEP's carbon usage rate. The higher the carbon usage rate, the more expensive the treatment costs will be to achieve high levels of TCEP removal. Higher treatment costs may determine that GAC nor PAC is not an economically feasible method for removing TCEP from drinking water. In addition, the use of activated carbon filtration, such as PAC and GAC, is not mandatory for drinking water treatment facilities in the United States.

F.2.6 Bioaccumulation Potential

Information on bioconcentration and bioaccumulation in aquatic and terrestrial organisms are important to understand the behavior of TCEP in the environment and a key component in assessing its risk to all living organisms, including humans.

Bioconcentration is the uptake and retention of a chemical by an aquatic organism from ambient water only (U.S. EPA, 2003c). Bioconcentration does not include chemical exposure through diet, but rather its uptake by respiratory and dermal surfaces (Arnot and Gobas, 2006). The bioconcentration factor (BCF) is the ratio of the concentration of a chemical in the tissue of an organism to its concentration in the ambient water once a steady state has been achieved (OECD, 2012). The resulting BCF value provides an indication of the potential for a chemical to bioconcentrate in lipids of organisms.

Three high-quality semi-static tests were identified and selected for use in the risk evaluation. Tang et al. (2019) reported steady-state BCF values of 1.0 in the muscle, 1.6 in the gill, 2.6 in the brain, 1.6 in the kidney, and 4.3 in the liver in juvenile common carp (*Cyprinus carpio*) after 28 days of exposure to TCEP at 9.1 µg/L using OECD TG 305 (OECD, 2012). Wang et al. (2017a) reported steady-state BCF values of 0.8 in the muscle, 1.9 in the gill, 2.2 in the brain, and 2.4 in liver of adult zebrafish (*Danio rerio*) after 19 days of exposure to TCEP at 893 µg/L using OECD TG 305 (OECD, 2012). The concentration of TCEP in all tissue compartments achieved steady-state in 3 days and the depuration half-life was less than 5.3 hours. Another high-quality semi-static test reporting BCF values in fish was identified and selected. Arukwe et al. (2018) reported BCF values of 0.31, 0.16, and 0.34 in the muscle in juvenile Atlantic salmon (*Salmo salar*) after 7 days of exposure to TCEP at concentrations of 0.04, 0.2, 1 mg/L, respectively.

A continuous flow-through test was identified during systematic review. Sasaki et al. (1982) reported BCF values of 1.1 and 1.3 in killifish (*Oryzias latipes*) after 5 and 11 days of exposure to TCEP at concentrations of 12.7 and 2.3 mg/L, respectively. The depuration half-life was 0.7 hour, which indicates that the killifish eliminated TCEP rapidly. This study was evaluated as a medium-quality study because insufficient information was available on the test conditions and study design. This added uncertainty on whether its BCF values would be a good representation of TCEP's bioconcentration potential and thus will not be considered in this risk evaluation.

The range of experimental BCF values provided above agrees with the calculated BCF values of 1.04 L/kg given by the BCFBAF™ module in EPI Suite™ (U.S. EPA, 2017a) and 1.29 by another QSAR model, OPEn structure-activity/property Relationship App (OPERA) (U.S. EPA, 2019c; Mansouri et al., 2018). The calculated values from EPI Suite™ and OPERA are not included in this risk evaluation because the BCF values from high-quality studies cited above are available.

Bioaccumulation is the net accumulation of a chemical by an organism by all possible routes of exposure (*e.g.*, respiration, dietary, dermal) from all surrounding environmental media (*e.g.*, air, water, sediment, and diet) (ECHA, 2008). The bioaccumulation factor (BAF) can be expressed as the steady-state ratio of the chemical concentration in an organism to the concentration in the ambient water. The concentration of a chemical in an organism can be measured and reported on wet weight, dry weight, or lipid weight basis. In order to reduce any variability and uncertainty, lipid-normalized BAFs in whole fish and fish tissues were used in this risk evaluation. Lipid weight BAF (BAF_{lw}) values were converted to wet weight BAF (BAF_{ww}) values by using Equation_Apx F-1.

Equation_Apx F-1.

$$\mathrm{BAF_{ww}} = \mathrm{BAF_{lw}} \times \left(\frac{\%\ \mathrm{lipid}}{100}\right)$$

There are multiple wet weight BAF values reported for aquatic organisms collected from water bodies that contained TCEP. A mean BAF value (L/kg wet weight) of 794 in the muscle and 1,995 in the liver, kidney, and gill, respectively, were reported for pelagic and benthic fish collected from Laizhou Bay in China (Bekele et al., 2021). A mean BAF value (L/kg wet weight) of 30.7 in the muscle and 70.7 in the liver was reported for crucian carp (*Carassius auratus*) collected from Nakdong River in South Korea (Choo et al., 2018). A mean BAF value (L/kg wet weight) of 2,198 was reported in walleye (*Sander vitreus*) collected from the Great Lakes (Guo et al., 2017b). Mean whole body BAF values (L/kg wet weight) ranging from 109 to 1,248 were reported for aquatic organisms collected from a freshwater pond containing e-waste in South China (Liu et al., 2019a). Mean BAF values of 6,310 in benthic invertebrates, 2,690 in pelagic fish, and 4,270 in benthic fish were reported for fish collected from Zhushan Bay in Lake Taihu, China (Wang et al., 2019b).

Zhang et al. (2018b) reported a median BAF value (L/kg wet weight) of 21,380 in the muscle of fishes collected from a site that was less than 1 km away from the outfall of a wastewater treatment plant located in Pearl River Delta, China. Fish species included catfish (*Clarias batrachus*), common carp (*Cyprinus carpio*), bream (*Parabramis pekinensis*), and white semiknife-carp (*Hemiculter leucisculus*). This BAF value is not included in this risk evaluation because this study was evaluated as a medium quality. Surface water samples were collected from 11 different sites, while fish samples were collected from only 1 site. Because the TCEP concentrations in surface water were reported as a range, independent calculation of the BAF could not be conducted. In addition, the reported BAF value could not be verified whether it was a lipid-normalized BAF value. Hou et al. (2017) reported a mean whole body BAF value (L/kg wet weight) of 34.7 for topmouth gudgeon, (*Pseudorasbora parva*), crucian carp (*Carassius auratus*), and loach (*Misgurnus anguillicaudatus*) collected from urban surface water in Beijing, China. Because this study was evaluated as a medium quality, these BAF data are not included in this risk evaluation. The tissue-specific values were based on average water concentrations; however, the study did not specify which of the nine rivers the tissue concentrations in the fish were from and not all loach samples have reported corresponding concentrations in several rivers, which adds uncertainty in the study's calculations. Sutton et al. (2019) measured TCEP in the blubber of harbor seals (*Phoca vitulina*) from San Francisco Bay. This study is not included in this risk evaluation because upper trophic fish are the focus of this bioaccumulation assessment.

The upper-trophic fish BAF value of 6.3 and a biotransformation half-life of 0.0798 days (≈1 hour and 55 minutes) were estimated using a log K_{OW} value of 1.78 (ECB, 2009) in the BCFBAF™ Model (U.S. EPA, 2017a). The biotransformation half-life of 0.219 days (≈5.3 hours) was estimated by OPERA (U.S. EPA, 2019c; Mansouri et al., 2018). These estimated values were not included in this risk evaluation because data from high-quality monitoring studies are available.

Bioaccumulation from soil to terrestrial or benthic organisms is expressed by the biota-sediment accumulation factor (BSAF), which is the ratio of concentrations of a chemical in the tissue of a sediment-dwelling organism to the concentration of a chemical in sediment. Wang et al. (2019b) reported a BSAF value of 2.19×10^{-3} and 1.48×10^{-3} for invertebrates and benthic fishes, respectively, from Zhushan Bay in Lake Taihu, China. Liu et al. (2019a) reported a BSAF range of 0.015 to 0.171 for aquatic organisms collected from freshwater pond polluted with e-wastes in South China. Choo et al. (2018) reported a mean BSAF value of 1.09 in the muscle and 2.49 in the liver of crucian carp (*Carassius auratus*). Zhang et al. (2018b) reported a BSAF value of 1.38×10^{-3} in fish muscles collected from a site that was less than 1 km away from the outfall of a wastewater treatment plant located in Pearl

River Delta, China. This BSAF value is not included in this risk evaluation because this study was evaluated as a medium quality. Sediment samples were collected from 11 different sites, while fish samples were collected from only 1 site. Because the TCEP concentration in sediment was reported as a range, independent calculation of BSAF could not be conducted.

Biomagnification describes the potential of a chemical to be transferred through the food web. It is defined as an increase of a chemical concentration in the tissue of an organism compared to the tissue concentration of its prey. The biomagnification potential of a chemical can be expressed as either a biomagnification factor (BMF) or trophic magnification factor (TMF). Generally, TMF is preferred over BMF because TMF represents the average value of the prey-to-predator magnification factor over a food chain rather than just a specific predator-prey relationship (Fu et al., 2020). When a trophic dilution occurs, the concentration of a pollutant decreases as the trophic level increases. It could be a result of a net balance of ingestion rate, uptake from food, internal transformation, or elimination processes favoring loss of pollutant that enters the organism via food.

In Brandsma et al. (2015), TMFs were calculated for organophosphate flame retardants (OPFRs) in two food webs (benthic and pelagic) and in total food web of Western Scheldt in Netherlands. No significant relationship was observed between TCEP and pelagic food web and total food web. It is possible that the trophic dilution in the pelagic food web occurred because TCEP was likely to be adsorbed to particles, and thus were likely to be more abundant in the sediment than in the water column. However, a TMF value of 2.6 was reported for benthic food web. It was determined that the trophic magnification in the benthic food web of TCEP was due to high levels of TCEP emission and the organisms' substantial exposure. Fu et al. (2020) studied the trophic magnification behavior of organophosphate esters in the Antarctic ecosystem that included algae (*Halymenia floresia*), archaeogastropoda (*Nacella concinna*), neogastropoda (*Trophon geversianus*), black rockcod (*Notothenia coriiceps*), and penguins (*Pygoscelis papua*). The TMF of TCEP was 5.2, which indicated that TCEP can be magnified through this food chain. Zhao et al. (2018) studied the trophic transfer of OPFRs in a lake food web from Taihu Lake, China, which included plankton, 5 invertebrate species, and 11 fish species. There was no significant correlation between TCEP and trophic level. Trophic dilution was likely to be a result of rapid metabolism in sampled fishes.

F.2.6.1 Key Sources of Uncertainty

There is a significant disparity between the BCF and BAF values reported for TCEP. It was observed that field-measured BAFs were much higher than laboratory-measured BCFs. In controlled laboratory studies, the exposure time is short, reaching equilibrium is challenging, and the exposure pathway is limited (lack of dietary intake). A field-measured BAF considers an organism's exposure to a chemical through all exposure routes in a natural aquatic ecosystem and incorporates chemical biomagnification and metabolism, making it the most direct measure of bioaccumulation (U.S. EPA, 2003c). TCEP has the ability to quickly bioaccumulate in fish tissue if it is exposed to high TCEP concentration in the surrounding water for a period of time. For example, TCEP concentration in the muscle of juvenile Atlantic salmon (*Salmo salar*) increased 10-fold when the water concentration of TCEP increased from 0.2 to 1 mg/L in 7 days (Arukwe et al., 2018).

Overall, a significantly higher concentration of TCEP was observed in liver than in the muscle (Tang et al., 2019; Choo et al., 2018; Hou et al., 2017; Wang et al., 2017a). Hou et al. (2017) showed that metabolically active tissues, such as liver and kidney, accumulate more than metabolically inactive tissue like muscle. The liver is the first tissue to be perfused by trace pollutants and it has a higher lipid contents and assimilation rate than in muscles (Kim et al., 2015; Kojadinovic et al., 2007). Several studies showed that a significant correlation was observed between lipid contents and TCEP

concentrations, indicating that lipid content is an important factor determining TCEP bioaccumulation in aquatic organisms (Bekele et al., 2019; Wang et al., 2017a; Gao et al., 2014). However, some studies showed no significant correlations between TCEP concentrations and lipid contents (Liu et al., 2019a; Liu et al., 2019b; Brandsma et al., 2015). The accumulative potential of TCEP can vary greatly due to several factors such as fish species, feeding habits, and temporal and spatial factors (U.S. EPA, 2003c). Collectively, the above studies indicate that TCEP could have the potential to bioaccumulate and biomagnify in benthic food webs.

The reported TMF reported by Brandsma et al. (2015) was reported as "tentative" because the sample size was small (n = 15). As a general rule, a number of samples between 30 and 60 are recommended to achieve statistical reliable TMFs (Borgå et al., 2012). The small sample size adds some uncertainty with the use of this TMF value in this risk evaluation.

Appendix G ENVIRONMENTAL HAZARD DETAILS

G.1 Approach and Methodology

For aquatic species, EPA estimates hazard by calculating a COC for a hazard threshold. COCs can be calculated using a deterministic method by dividing a hazard value by an AF according to EPA methods as shown below in Equation_Apx G-1 (U.S. EPA, 2016a, 2014b, 2012b).

Equation_Apx G-1.

$$COC = \frac{toxicity\ value}{AF}$$

COCs can also be calculated using probabilistic methods. For example, an SSD can be used to calculate a hazardous concentration for 5 percent of species (HC05). The HC05 estimates the concentration of a chemical that is expected to protect 95 percent of aquatic species. This HC05 can then be used to calculate a COC. For TCEP, Web-ICE, Version 3.3 (Appendix G.2.1.1) followed by SSD probabilistic method (Appendix G.2.1.4) was used to calculate the acute COC. The deterministic method was used to calculate at chronic COC

For terrestrial species, EPA estimates hazard by using a hazard value for soil invertebrates, a deterministic approach, or by calculating a TRV for mammals (Appendix G.2.20). The TRV is expressed as doses in units of mg/kg-bw/day. Although the TRV for TCEP is derived from laboratory mice and rat studies, body weight is normalized; therefore, the TRV can be used with ecologically relevant wildlife species to evaluate chronic dietary exposure to TCEP (U.S. EPA, 2007a).

G.2 Hazard Identification

G.2.1 Aquatic Hazard Data

G.2.1.1 Web-Based Interspecies Correlation Estimation (Web-ICE)

Results from the systematic review process indicated studies with empirical data meeting evaluation criteria on aquatic species for TCEP with several studies producing LC50 and EC50 endpoint data. To supplement the empirical data, EPA used a modeling approach, Web-ICE. Web-ICE predicts toxicity values for environmental species that are absent from a dataset and can provide a more robust dataset to estimate toxicity thresholds. Specifically, EPA used Web-ICE to supplement empirical data for aquatic organisms for acute exposure durations for invertebrates and vertebrates. The Agency also used Web-ICE to supplement empirical hazard data within aquatic plants. EPA also considered ECOSAR predictions. However, after comparing predictions with empirical data available for TCEP, EPA had greater confidence in the Web-ICE predictions. Therefore, Web-ICE predictions were used quantitatively during evidence integration.

G.2.1.2 Invertebrate and Vertebrate Web-ICE

Acute dose-response assays for fish and aquatic invertebrates create useful hazard endpoints for risk assessments. Calculated endpoints such as EC50 or LC50 values and associated descriptors (CI, NOEC, and LOEC values) are often comparable across taxa when standardized methodologies and statistical analysis are employed and documented. Three studies in the TCEP dataset had LC50 data for rainbow trout, zebrafish, and *Daphnia magna* (Alzualde et al., 2018; Toray Research Center, 1997a; Life Sciences Research Ltd, 1990a) and could be used as "surrogate species" for Web-ICE predictions. This dataset for aquatic organisms contained data gaps that EPA looked to fill using other lines of evidence

(*i.e.*, modeling approaches). The remaining empirical LC50 values reported in Zhang et al. (2024) represented four species (brine shrimp, Japanese seabass, Manila clam, mysid shrimp [*Neomysis awatschensis*]) that were currently not available for surrogate to predicted species but were incorporated into the SSD.

The Web-ICE application was developed by EPA and collaborators to provide interspecies extrapolation models for acute toxicity (Raimondo and Barron, 2010). Web-ICE models estimate the acute toxicity (LC50/LD50) of a chemical to a species, genus, or family with no test data (the predicted taxon) from the known toxicity of the chemical to a species with test data (the commonly tested surrogate species).

Web-ICE models are log-linear least square regressions of the relationship between surrogate and predicted taxon based on a database of acute toxicity values; that is, median effect or lethal water concentrations for aquatic species (EC50/LC50). Separate acute toxicity databases are maintained for aquatic animals (vertebrates and invertebrates), aquatic plants (algae), and wildlife (birds and mammals), with 1,440 models for aquatic taxa and 852 models for wildlife taxa currently included in Web-ICE version 3.3 (Willming et al., 2016). Open-ended toxicity values (*i.e.*, >100 mg/kg or <100 mg/kg) and duplicate records among multiple sources are not included in any of the databases.

The aquatic animal database within Web-ICE comprises 48- or 96-hour EC50/LC50 values based on death or immobility. This database is described in detail in the Aquatic Database Documentation found on the Download Model Data page of Web-ICE and describes the data sources, normalization, and quality and standardization criteria (*e.g.*, data filters) for data used in the models. Data used in model development adhered to standard acute toxicity test condition requirements of the ASTM International (ASTM, 2014) and EPA's OCSPP (*e.g.*, (U.S. EPA, 2016a)).

EPA used the 96-hour LC50 toxicity data from rainbow trout and zebrafish studies and 48-hour EC50 toxicity data from *Daphnia magna* in Table 4-2 as surrogate species to predict acute toxicity values using the Web-ICE application (Raimondo et al., 2023; Raimondo and Barron, 2010). The Web-ICE Model estimated toxicity values for 97 species. For model validation, the predicted species model results are then screened by the following criteria detailed within Willming et al. (2016) to ensure confidence in the model predictions. If a predicted species did not meet all the quality criteria listed below, the predicted hazard value from that was not included within the dataset for the SSD:

- High R^2 (> ≈0.6)
    - The proportion of the data variance that is explained by the model. The closer the R^2 value is to 1.0, the more robust the model is in describing the relationship between the predicted and surrogate taxa.
- Low mean square error (MSE; < ≈0.95)
    - An unbiased estimator of the variance of the regression line.
- High slope (> ≈0.6)
    - The regression coefficient represents the change in log10 value of the predicted taxon toxicity for every change in log10 value of the surrogate species toxicity.
- Narrow 95 percent CIs
    - Two orders of magnitude between lower and upper limit

After screening, the predicted acute toxicity values for 46 additional aquatic organisms (22 fish, 1 amphibian, 9 aquatic invertebrates, 14 benthic invertebrate species) were added from the surrogate rainbow trout, zebrafish, and *Daphnia magna* data (Table_Apx G-1). The toxicity data were then used to

calculate the distribution of species sensitivity to TCEP exposure through the SSD toolbox as shown in Figure_Apx G-4 and Table 4-6 (Etterson, 2020). The distribution of acute hazard values from vertebrates and invertebrates was examined to determine if these two groups had similar sensitivity to TCEP. The mean LC50 values for the 26 vertebrates and 27 invertebrates were 399 ± 54 mg/L (± SEM) and 197 ± 51 mg/L and were not significantly different (F = 1.87, df = 1,51, P = 0.17). Two predicted values, representing one vertebrate (cape fear shiner) and one invertebrate (tubifex worm), were each four standard deviations from their respective means. When statistical analysis was conducted with these two values omitted as outliers the two groups were not significantly different (F = 6.24, df = 1,49, P = 0.015) with mean LC50 values of 253 ± 30 mg/L and 152 ± 27 mg/L for vertebrates and invertebrates, respectively. Acute hazard values for invertebrates divided between benthic and water column species were not significantly different (F = 1.01, df = 1,25, P = 0.32) with mean LC50 values of 243 ± 90 mg/L and 138 ± 27 mg/L, respectively.

G.2.1.3 Algal Web-ICE

Two studies in the TCEP dataset contained EC50 data for two marine diatom species: *Phaeodactylum tricornutum* and *Skeletonema costatum* (Zhang et al., 2024) as well as a freshwater green alga species (*Raphidocelis subcapitata*) (Toray Research Center, 1997b) that could be applied as "surrogate species" for Web-ICE predictions.

This dataset for aquatic plants contained data gaps that EPA looked to fill using other lines of evidence (*i.e.*, modeling approaches). The remaining empirical EC50 values reported in Zhang et al. (2024) represented two marine green algae species (*Dunaliella salina* and *Platymonas subcordiformis)* currently not available for surrogate to predicted species within Web-ICE but were incorporated into the SSD.

The Web-ICE Model estimated toxicity values for 10 species. For model validation, the predicted species model results are then screened by the following criteria detailed within Willming et al. (2016) to ensure confidence in the model predictions. If a predicted species did not meet all the quality criteria previously listed in Appendix G.2.1.2, the predicted hazard value from that was not included within the dataset for the SSD. After screening, the predicted acute toxicity values for three additional species (*Desmodesmus subspicatus*, *Minutocellus polymorphus*, and *Thalassiosira pseudonana*) were added from the surrogate *Phaeodactylum tricornutum*, *Skeletonema costatum*, and *Raphidocelis subcapitata* data (Table_Apx G-1). The toxicity data were then used to calculate the distribution of species sensitivity to TCEP exposure through the SSD toolbox as shown in Figure_Apx G-4 and Table 4-6 (Etterson, 2020).

Table_Apx G-1. Invertebrate and Vertebrate Web-ICE Predicted Species that Met Model Selection Criteria

Predicted Species	Surrogate Species	LC50 (mg/L)	95% CI	R^2	MSE	Slope
	Daphnia magna	171.0				
	Rainbow trout	249.0				
	Zebrafish embryo	279.1				
Amphipod (*Grammarus fasciatus*)	*Daphnia magna*	105.70	30.27–369.10	0.75	0.77	0.86
Atlantic salmon	Rainbow trout	260.08	104.18–649.30	0.95	0.12	1.01
Beaver-tail fairy shrimp	*Daphnia magna*	102.10	64.38–161.91	0.98	0.05	0.91
Bluegill[a]	Rainbow trout	231.66	183.95–291.73	0.88	0.21	0.93
Bluegill[a]	*Daphnia magna*	68.99	45.26–105.17	0.62	0.80	0.66
Brook trout	Rainbow trout	258.83	127.67–524.75	0.94	0.11	1.02
Brown trout	Rainbow trout	252.59	117.39–543.50	0.95	0.10	0.99
Bullfrog[a]	Rainbow trout	333.43	159.02–699.15	0.97	0.15	0.88
Bullfrog[a]	*Daphnia magna*	195.05	34.49–1,102.98	0.86	0.9	0.99
Cape Fear shiner	Rainbow trout	1,443.70	318.31–6,547.86	0.99	0.01	1.18
Channel catfish	Rainbow trout	172.56	100.50–296.29	0.79	0.40	0.82
Channel catfish	Zebrafish embryo	151.86	22.10–1,043.50	0.92	0.09	0.76
Chinook salmon	Rainbow trout	229.96	123.71–427.43	0.96	0.07	0.94
Coho salmon	Rainbow trout	319.44	220.60–462.55	0.98	0.04	0.98
Common carp	Rainbow trout	304.89	104.49–889.56	0.87	0.30	0.89
Cutthroat trout	Rainbow trout	168.04	99.52–283.73	0.94	0.09	0.93
Daphnid (*Ceriodaphnia dubia*)	*Daphnia magna*	121.66	70.17–210.92	0.95	0.26	1.00
Daphnid (*Daphnia pulex*)	*Daphnia magna*	154.64	72.16–331.36	0.97	0.12	1.01
Daphnid (*Simocephalus serrulatus*)	*Daphnia magna*	176.58	15.97–1,951.88	0.88	0.21	1.00

Predicted Species	Surrogate Species	LC50 (mg/L)	95% CI	R^2	MSE	Slope
Daphnid (*Simocephalus vetulus*)	*Daphnia magna*	337.13	298.97–380.16	0.99	0.00	0.98
Fathead minnow[a]	Rainbow trout	298.23	192.71–461.53	0.83	0.31	0.86
Fathead minnow[a]	Zebrafish embryo	258.44	135.54–492.77	0.84	0.54	0.91
Fatmucket	*Daphnia magna*	57.968	20.38–164.81	0.86	0.47	0.74
Fountain darter	Rainbow trout	514.72	37.20–7,120.23	0.99	0	1.09
Goldfish	Rainbow trout	392.65	153.71–1,002.99	0.86	0.42	0.85
Green sunfish	Rainbow trout	314.51	107.19–922.85	0.94	0.13	0.92
Guppy	Rainbow trout	195.83	46.64–822.14	0.73	0.54	0.80
Isopod (*Caecidotea brevicauda*)	Rainbow trout	33.26	3.60–306.95	0.64	0.59	0.76
Lake trout	Rainbow trout	98.62	51.81–187.73	0.93	0.08	0.86
Largemouth bass	Rainbow trout	143.43	52.46–392.13	0.86	0.24	0.94
Medaka	Rainbow trout	691.06	49.03–9,739.30	0.92	0.32	1.02
Midge (*Paratanytarsus dissimilis*)	Rainbow trout	607.00	92.95–3,963.87	0.83	0.54	0.80
Midge (*Paratanytarsus parthenogeneticus*)	*Daphnia magna*	449.18	251.62–801.83	0.98	0.04	0.93
Mysid (*Americamysis bahia*)	*Daphnia magna*	25.01	12.98–48.18	0.68	0.93	0.83
Mysid (*Metamysidopsis insularis*)	*Daphnia magna*	274.59	20.01–3,777.57	0.94	0.18	0.86
Neosho mucket	*Daphnia magna*	83.59	6.28–1,112.01	0.97	0.07	0.97
Oligochaete (*Tubifex tubifex*)	*Daphnia magna*	1,356.76	196.86–9,350.73	0.87	0.5	0.86
Paper pondshell	*Daphnia magna*	76.370	42.62–136.83	0.96	0.11	0.90
Sheepshead minnow	Rainbow trout	101.20	47.13–217.30	0.65	0.56	0.75
Shortnose sturgeon	Rainbow trout	474.09	104.63–2,148.02	0.98	0.04	1.14
Swamp lymnaea	*Daphnia magna*	194.54	81.47–464.51	0.96	0.19	1.01
Tadpole physa	*Daphnia magna*	146.40	68.29–313.81	0.96	0.14	0.99

Predicted Species	Surrogate Species	LC50 (mg/L)	95% CI	R^2	MSE	Slope
Threeridge	*Daphnia magna*	31.91	13.87–73.42	0.94	0.18	0.87
Vernal pool fairy shrimp	*Daphnia magna*	105.95	45.16–248.57	0.98	0.09	0.9
Walleye	Rainbow trout	20.30	1.24–332.56	0.67	0.27	0.69
Washboard	*Daphnia magna*	64.97	32.71–129.02	0.96	0.16	0.92
Water flea (*Pseudosia ramosa*)	*Daphnia magna*	43.45	3.94–478.20	0.87	0.57	0.93
Western pearlshell	*Daphnia magna*	82.23	29.32–171.98	0.95	0.14	0.86
White heelsplitter	*Daphnia magna*	55.66	22.90–135.31	0.98	0.10	0.92
Yellow perch	Rainbow trout	201.79	78.70–517.39	0.94	0.14	0.98

[a] The geometric mean of LC50 data for multiple predictions from different surrogate species are used for the species sensitivity distribution (SSD).

Table_Apx G-2. Algal Web-ICE Predicted Species that Met Model Selection Criteria

Predicted Species	Surrogate Species	LC50 (mg/L)	95% CI	R^2	MSE	Slope
	Phaeodactylum tricornutum	76				
	Skeletonema costatum	353				
	Raphidocelis subcapitata	212				
Desmodesmus subspicatus[a]	*Phaeodactylum tricornutum*	27.6	2.03–374.60	0.93	0.72	0.95
Desmodesmus subspicatus[a]	*Skeletonema costatum*	1,153.63	296.57–4,487.48	0.95	0.58	1.01
Desmodesmus subspicatus[a]	*Raphidocelis subcapitata*	291.25	140.60–603.32	0.96	0.31	1.10
Minutocellus polymorphus	*Skeletonema costatum*	221.46	23.40–2,095.99	0.75	0.24	1.36
Thalassiosira pseudonana	*Skeletonema costatum*	411.18	103.73–1,629.93	0.92	0.11	0.7

[a] The geometric mean of LC50 data for multiple predictions from different surrogate species are used for the species sensitivity distribution (SSD)

G.2.1.4 Species Sensitivity Distribution (SSD)

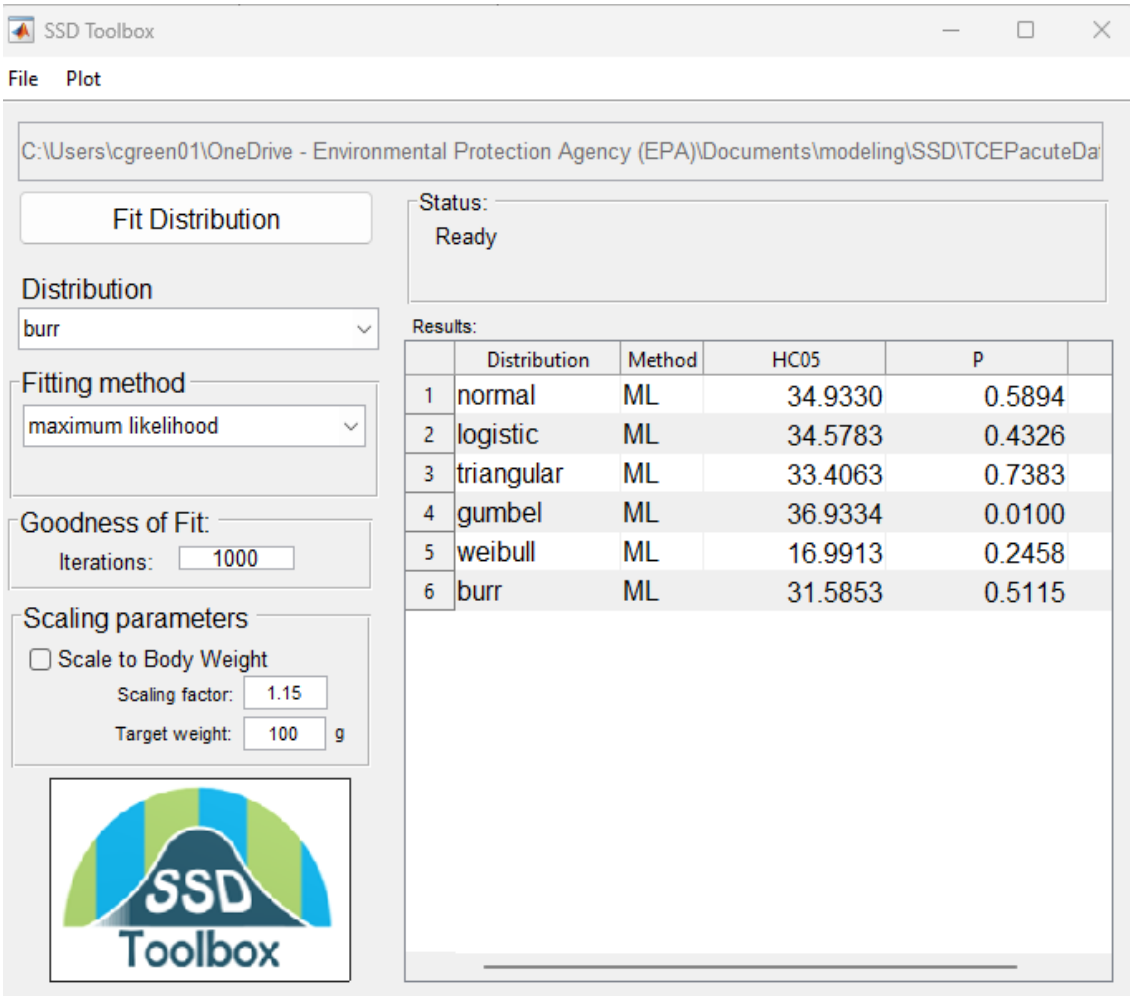
The SSD Toolbox is a resource created by EPA's Office of Research and Development (ORD) that can fit SSDs to environmental hazard data (Etterson, 2020). The SSD Toolbox runs on Matlab 2018b (9.5) for Windows 64 bit. For the TCEP Risk Evaluation, EPA calculated an SSD with the SSD Toolbox using acute LC50 hazard data from systematic review and estimated data from the Web-ICE application (Appendix G.2.1.1) that included 25 fish, 1 amphibian, 12 aquatic invertebrates, and 15 benthic invertebrates. A second SSD was performed for the algae hazard data and was applied with the 5 empirical values and 3 predicted species values detailed in the previous section. The SSD is used to calculate a HC05. The HC05 estimates the concentration of TCEP that is expected to be protective for 95 percent of species.

G.2.1.5 Vertebrate and Invertebrate SSD

The SSD toolbox contains functions for fitting six distributions (normal, logistic, triangular, Gumbel, Weibull, and Burr). Maximum likelihood was used to assess the goodness-of-fit of the data distribution based on Bayesian P-values. The larger the deviation of the *p*-value from 0.5 the greater the indication of lack of fit. The Burr distribution P value was nearest to 0.5 at P = 0.51 (Figure_Apx G-1), however, the triangular distribution (P = 0.73) demonstrated the best fit model due to with sample-size corrected Akaike Information Criterion (AICc) followed by the normal distribution (Figure_Apx G-2). Because numerical methods may lack statistical power for small sample sizes, a visual inspection of the data were also used to assess goodness-of-fit. For the Q-Q plot, the horizontal axis gives the empirical quantiles, and the vertical axis gives the predicted quantiles (from the fitted distribution). The Q-Q plot demonstrates a good model fit with the data points in close proximity to the line across the data distribution. Q-Q plots were visually used to assess the goodness-of-fit for the distributions (Figure_Apx G-3) with the Burr and logistic distributions demonstrating the best fit. The HC05 was similar among Burr, triangular, and normal at 31.6 mg/L, 33.4 mg/L, and 34.9 mg/L, respectively; however, the Burr distribution demonstrated the best fit based on Baysian P-value, and visual inspection of the Q-Q plot. The SSD plot shows the distribution of species sensitivity to TCEP exposure using Burr distribution with the calculated HC05 of 31.6 mg/L with a 95 percent CI of 16.7 mg/L to 57.0 mg/L (Figure_Apx G-4).

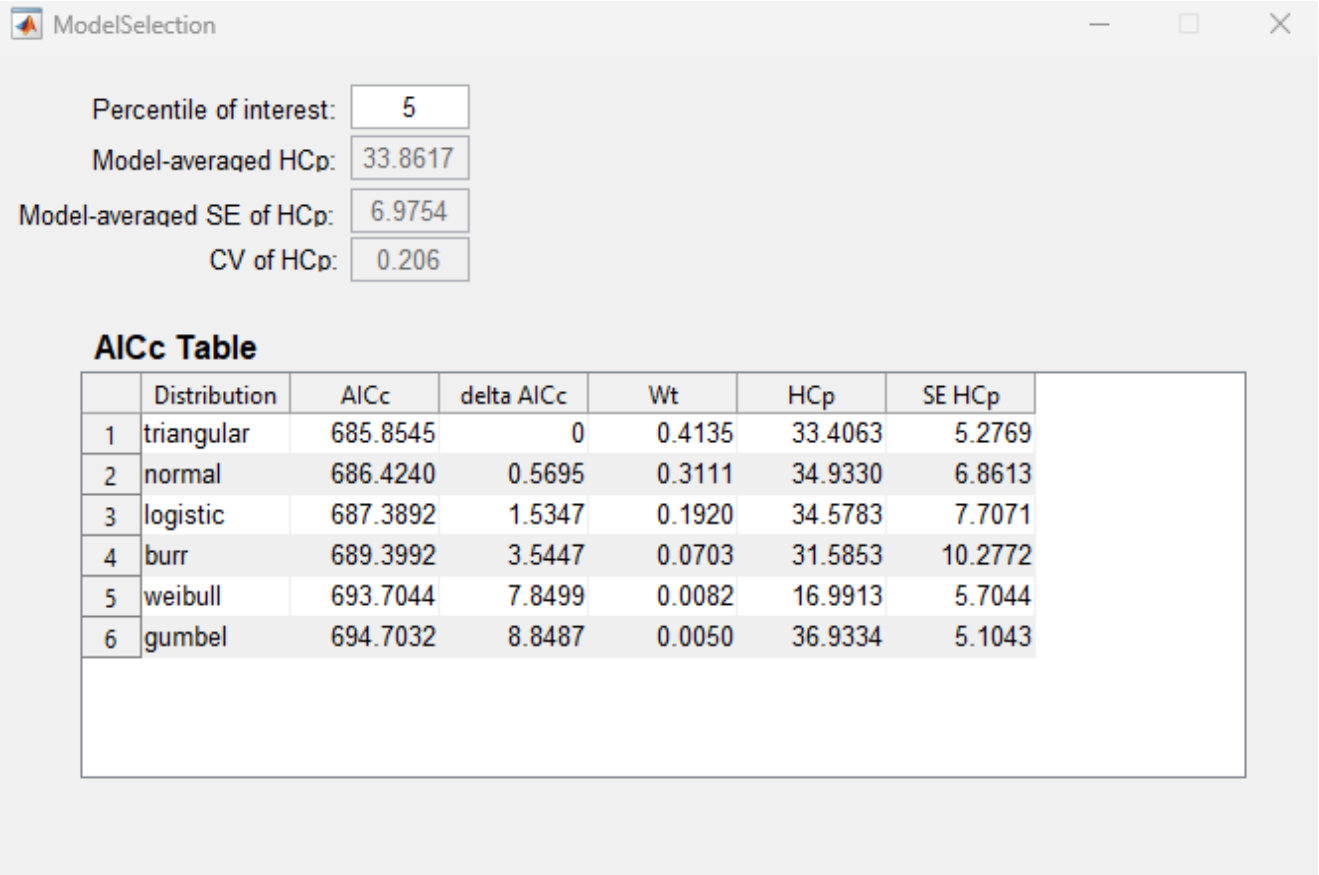
G.2.1.6 Algal SSD

The SSD toolbox contains functions for fitting six distributions (normal, logistic, triangular, Gumbel, Weibull, and Burr). Maximum likelihood was used to assess the goodness-of-fit of the data distribution based on Bayesian P-values. The larger the deviation of the p-value from 0.5 the greater the indication of lack of fit. The logistic distribution was nearest to 0.5 at P = 0.53, followed by the Weibull distribution at 0.46 and normal distribution at 0.39 (Figure_Apx G-5). The Weibull and logistic distributions had the lowest AICc at 103.6 and 105.5, respectively (Figure_Apx G-6). Q-Q plots were visually used to assess the goodness-of-fit for the distributions with both the logistic and normal demonstrating the best fit (Figure_Apx G-7). The HC05 for the logistic and normal distributions were 116.2 and 104.2, respectively, with both distributions resulted in the same lower 95 percent CI of the HC05 at 66 mg/L (Figure_Apx G-8).



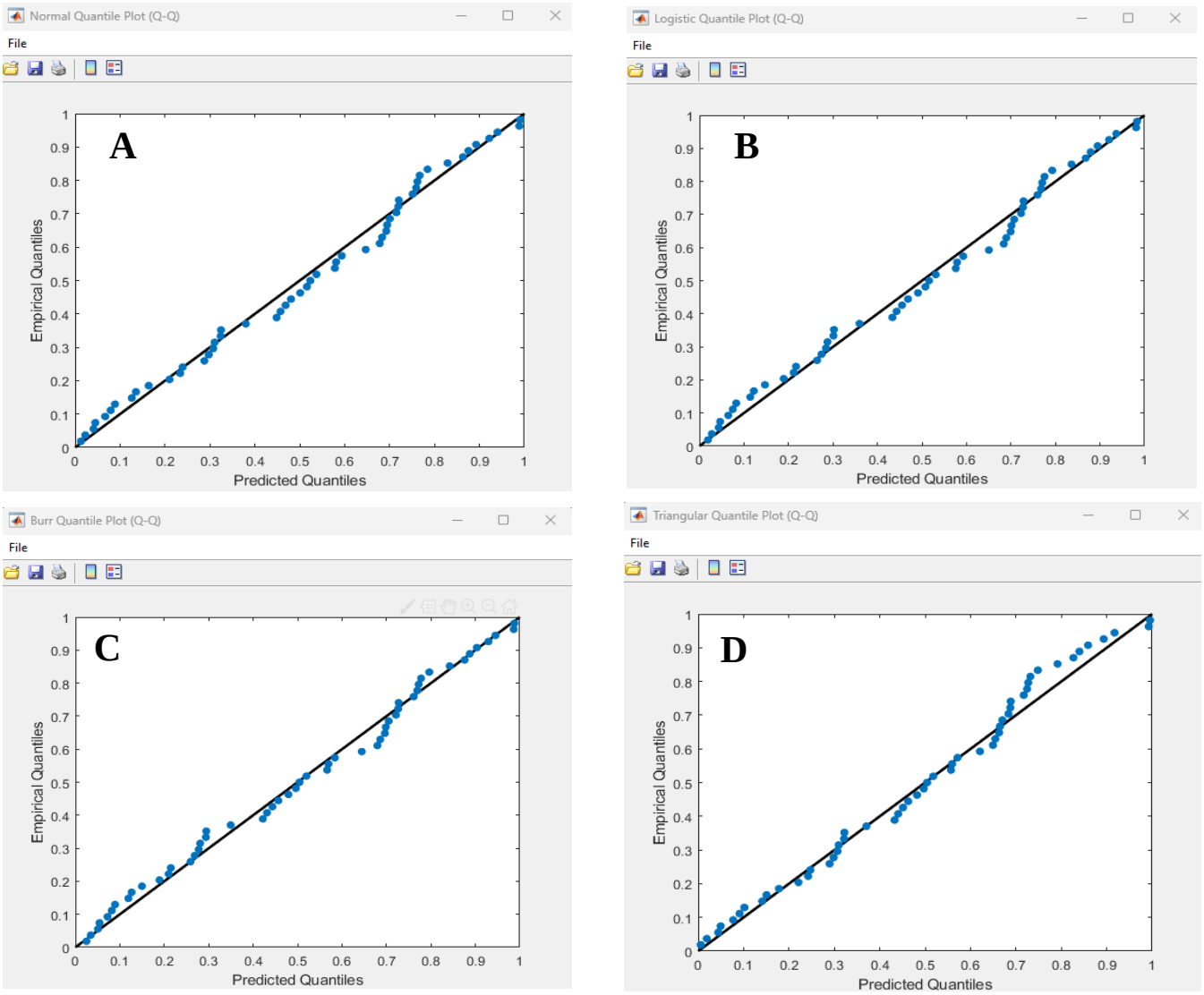
	Distribution	Method	HC05	P
1	normal	ML	34.9330	0.5894
2	logistic	ML	34.5783	0.4326
3	triangular	ML	33.4063	0.7383
4	gumbel	ML	36.9334	0.0100
5	weibull	ML	16.9913	0.2458
6	burr	ML	31.5853	0.5115

Figure_Apx G-1. SSD Toolbox Interface Showing HC05s and P-Values for Each Distribution Using Maximum Likelihood Fitting Method Using TCEP's Acute Aquatic Hazard Data for Vertebrates and Invertebrates (Etterson, 2020)

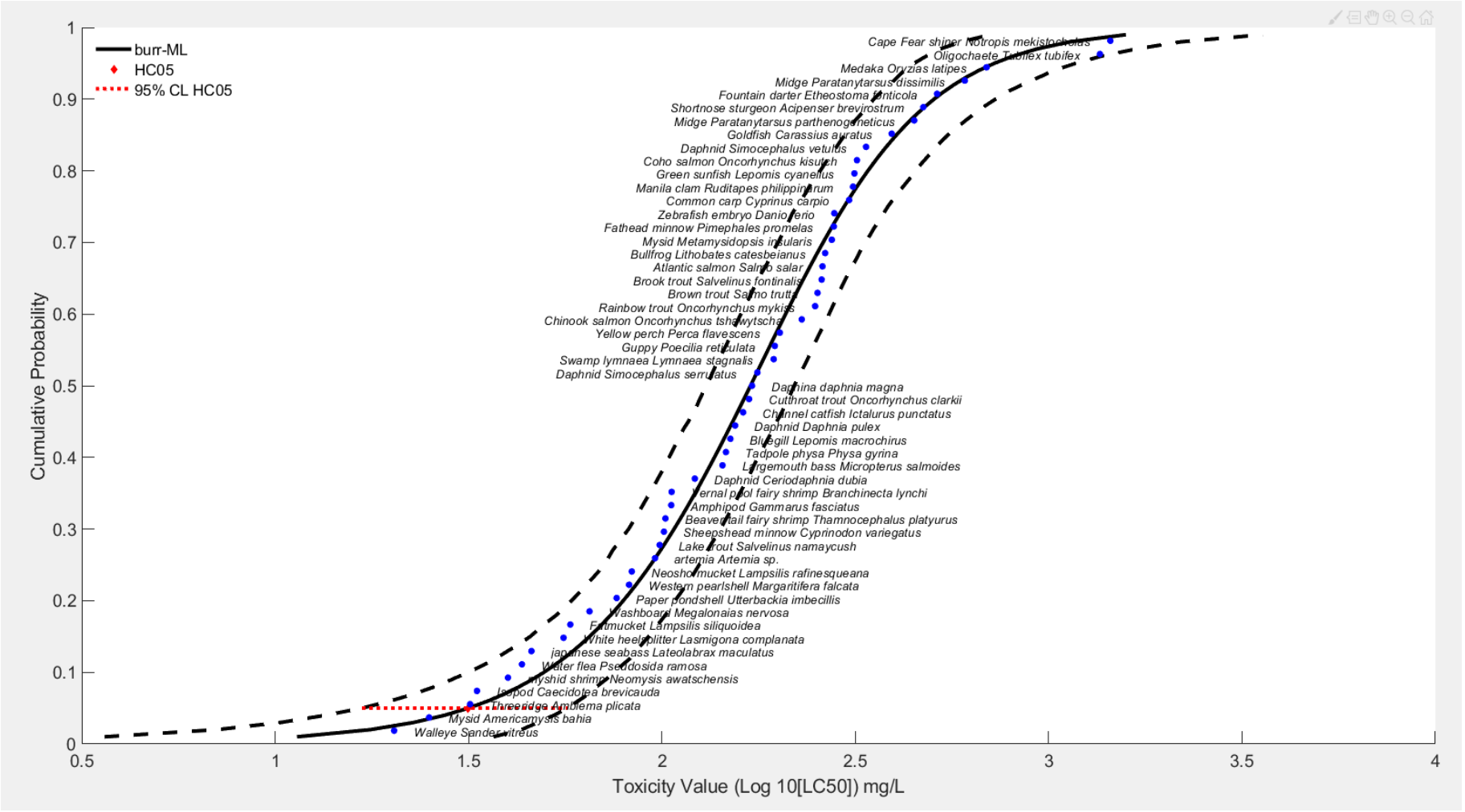


	Distribution	AICc	delta AICc	Wt	HCp	SE HCp
1	triangular	685.8545	0	0.4135	33.4063	5.2769
2	normal	686.4240	0.5695	0.3111	34.9330	6.8613
3	logistic	687.3892	1.5347	0.1920	34.5783	7.7071
4	burr	689.3992	3.5447	0.0703	31.5853	10.2772
5	weibull	693.7044	7.8499	0.0082	16.9913	5.7044
6	gumbel	694.7032	8.8487	0.0050	36.9334	5.1043

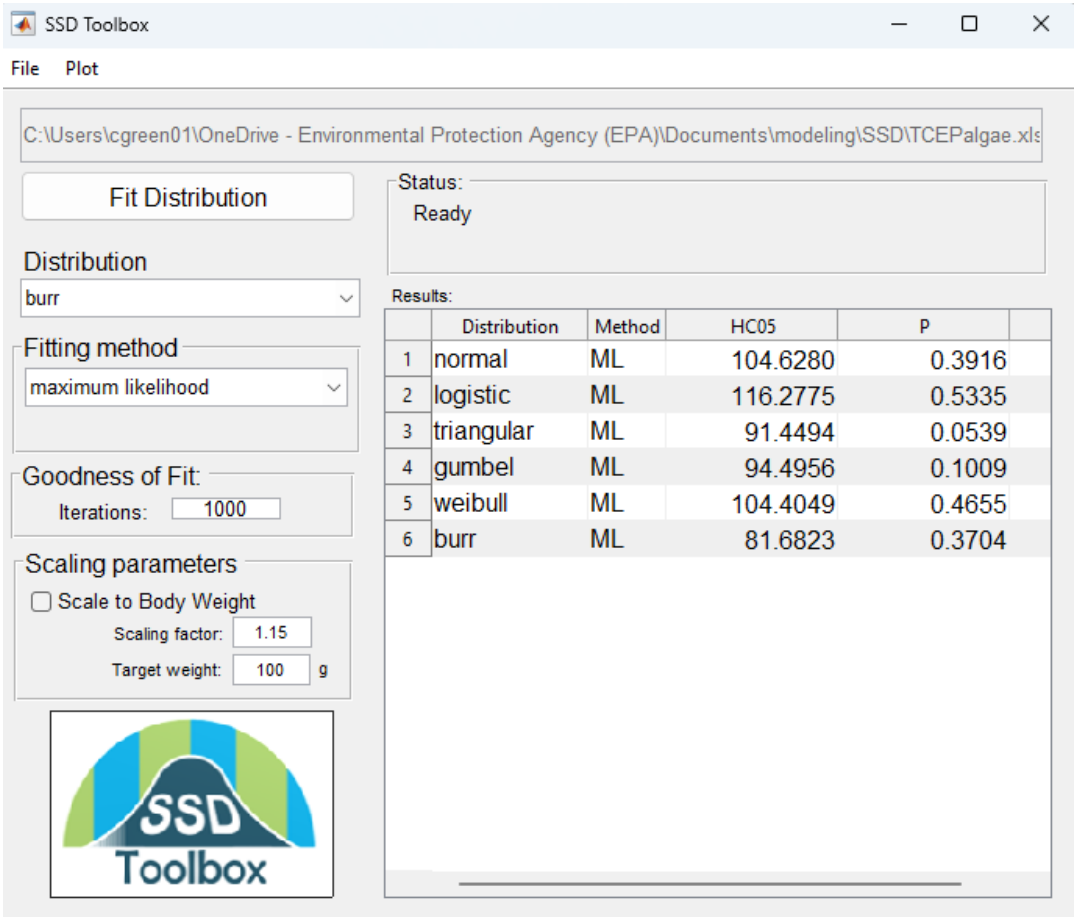
Figure_Apx G-2. AICc for the Six Distribution Options in the SSD Toolbox for TCEP's Acute Aquatic Hazard Data for Vertebrates and Invertebrates (Etterson, 2020)



Figure_Apx G-3. Q-Q Plots of TCEP Acute Aquatic Hazard Data for Vertebrates and Invertebrates with the (A) Normal, (B) Logistic, (C) Burr, and (D) Triangular Distributions (Etterson, 2020)

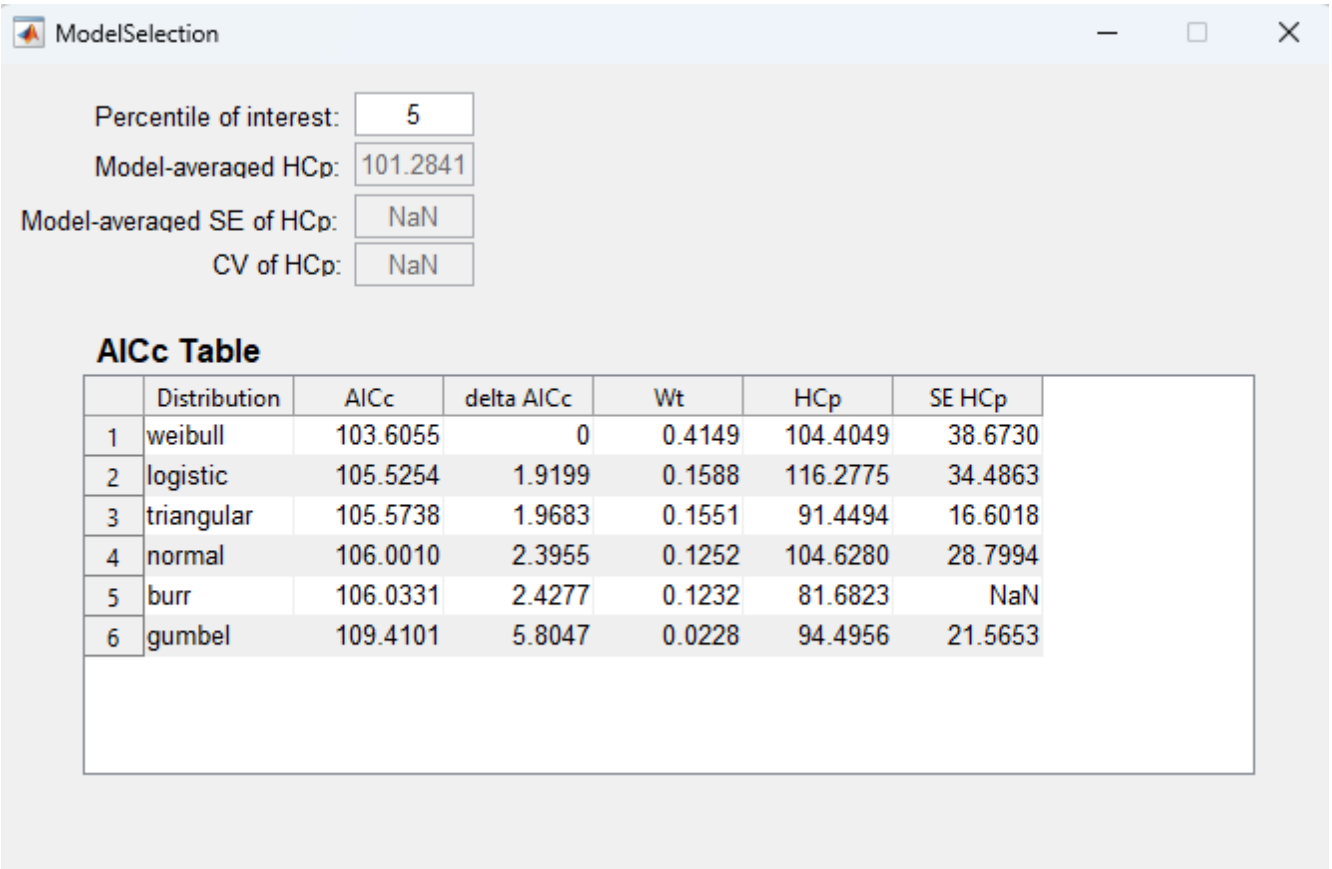


Figure_Apx G-4. SSD Distribution for TCEP's Acute Hazard Data for Invertebrates and Vertebrates (Etterson, 2020).
Aquatic invertebrate/vertebrate HC05 of 31.6 mg/L with a 95 percent CI of 16.7 mg/L to 57.0 mg/L



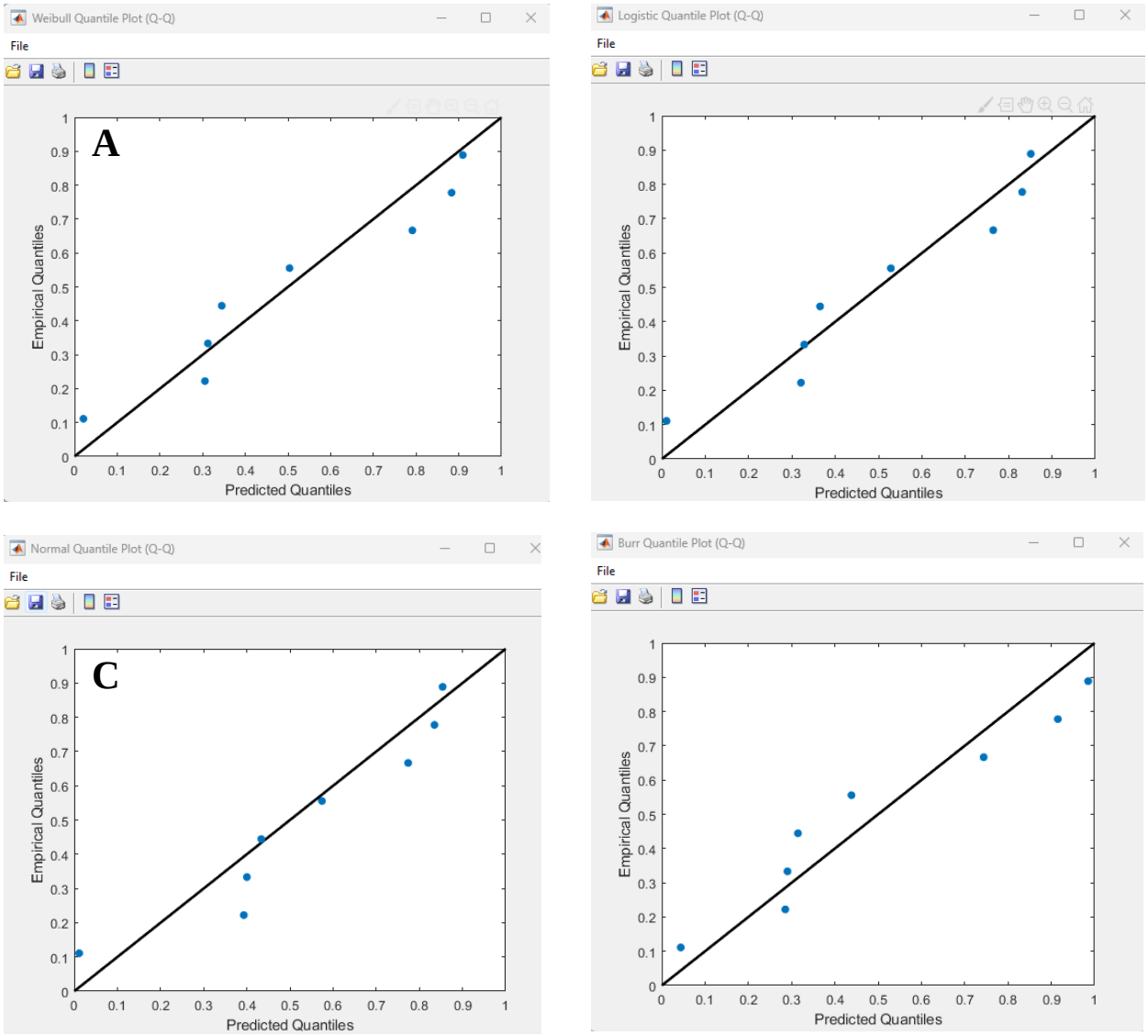
	Distribution	Method	HC05	P
1	normal	ML	104.6280	0.3916
2	logistic	ML	116.2775	0.5335
3	triangular	ML	91.4494	0.0539
4	gumbel	ML	94.4956	0.1009
5	weibull	ML	104.4049	0.4655
6	burr	ML	81.6823	0.3704

Figure_Apx G-5. SSD Toolbox Interface Showing HC05s and P-Values for Each Distribution Using Maximum Likelihood Fitting Method Using TCEP's Acute Aquatic Hazard Data for Algae (Etterson, 2020)

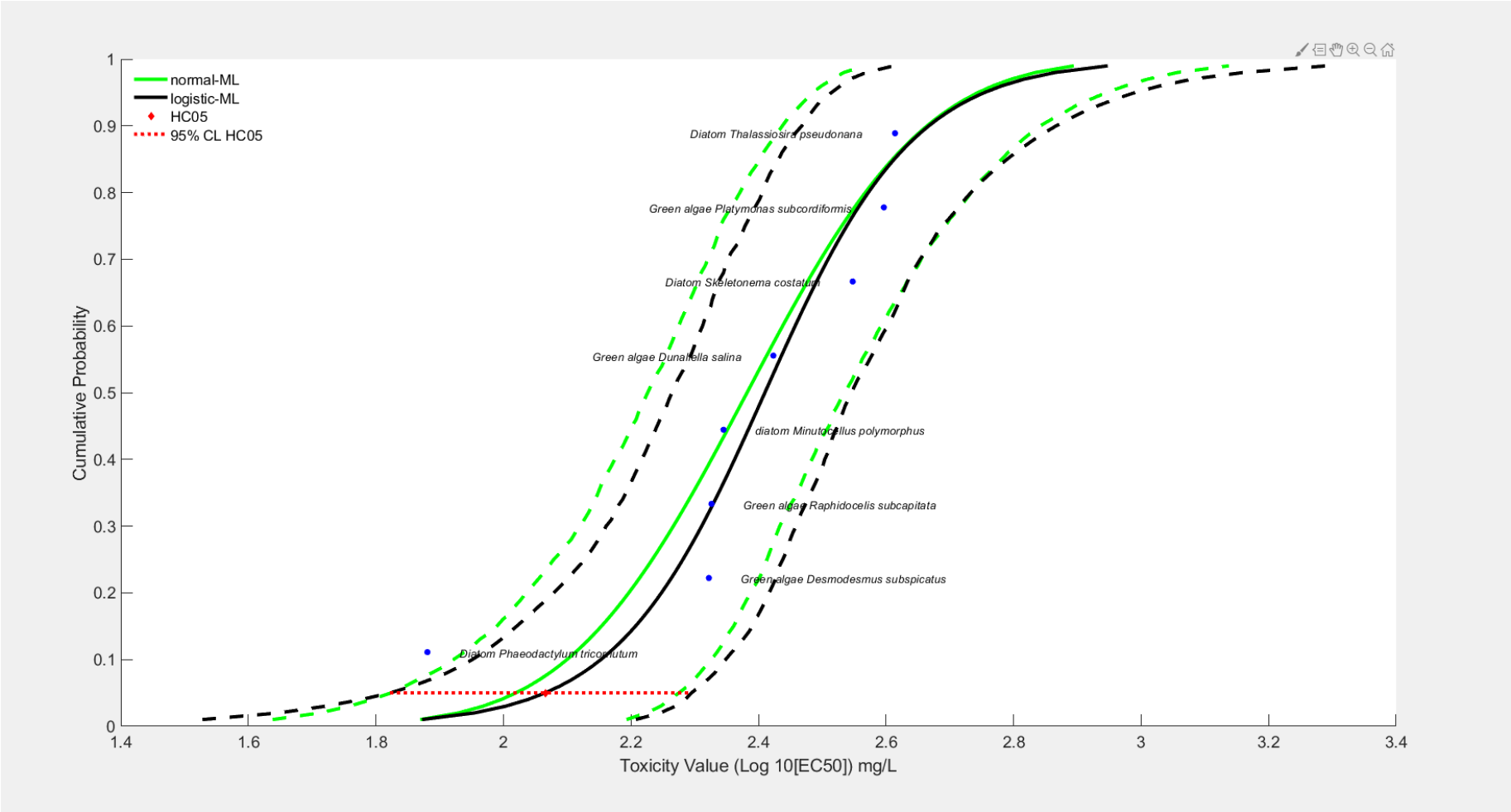


	Distribution	AICc	delta AICc	Wt	HCp	SE HCp
1	weibull	103.6055	0	0.4149	104.4049	38.6730
2	logistic	105.5254	1.9199	0.1588	116.2775	34.4863
3	triangular	105.5738	1.9683	0.1551	91.4494	16.6018
4	normal	106.0010	2.3955	0.1252	104.6280	28.7994
5	burr	106.0331	2.4277	0.1232	81.6823	NaN
6	gumbel	109.4101	5.8047	0.0228	94.4956	21.5653

Figure_Apx G-6. AICc for the Six Distribution Options in the SSD Toolbox for TCEP's Acute Aquatic Hazard Data for Algae (Etterson, 2020**)**



Figure_Apx G-7. Q-Q Plots of TCEP Acute Aquatic Hazard Data for Algae with the (A) Weibull, (B) Logistic, (C) Normal, and (D) Burr Distributions (Etterson, 2020)



Figure_Apx G-8. SSD Distribution for TCEP's Acute Hazard Data for Algae (Etterson, 2020).
Aquatic Algae HC05 for the logistic and normal distributions were 116.2 and 104.2, respectively, with both distributions resulting in a lower 95% CI of 66 mg/L.

G.2.2 Terrestrial Hazard Data

For calculation of the mammal TRV, an a priori framework for selection of the TRV value based on the results of the NOAEL and LOAEL data (Figure_Apx G-9.). The minimum dataset required to calculate a TRV consists of three results with NOEL or LOEL values for reproduction, growth, or mortality for at least two species. If these minimum results are not available, then a TRV is not calculated.

For mammalian species, EPA estimates hazard by calculating a TRV. The TRV is expressed as doses in units of mg/kg-bw/day. Although the TRV for TCEP is derived from laboratory mice and rat studies, body weight is normalized; therefore, the TRV can be used with ecologically relevant wildlife species to evaluate chronic dietary exposure to TCEP. Representative wildlife species chronic hazard threshold will be evaluated in the trophic transfer assessments using the TRV. The flow chart in Figure_Apx G-9. was used to select the data to calculate the TRV with NOEL and/or LOEL data and described below (U.S. EPA, 2007a).

Step 1: At least three results and two species tested for reproduction, growth, or mortality general end points.

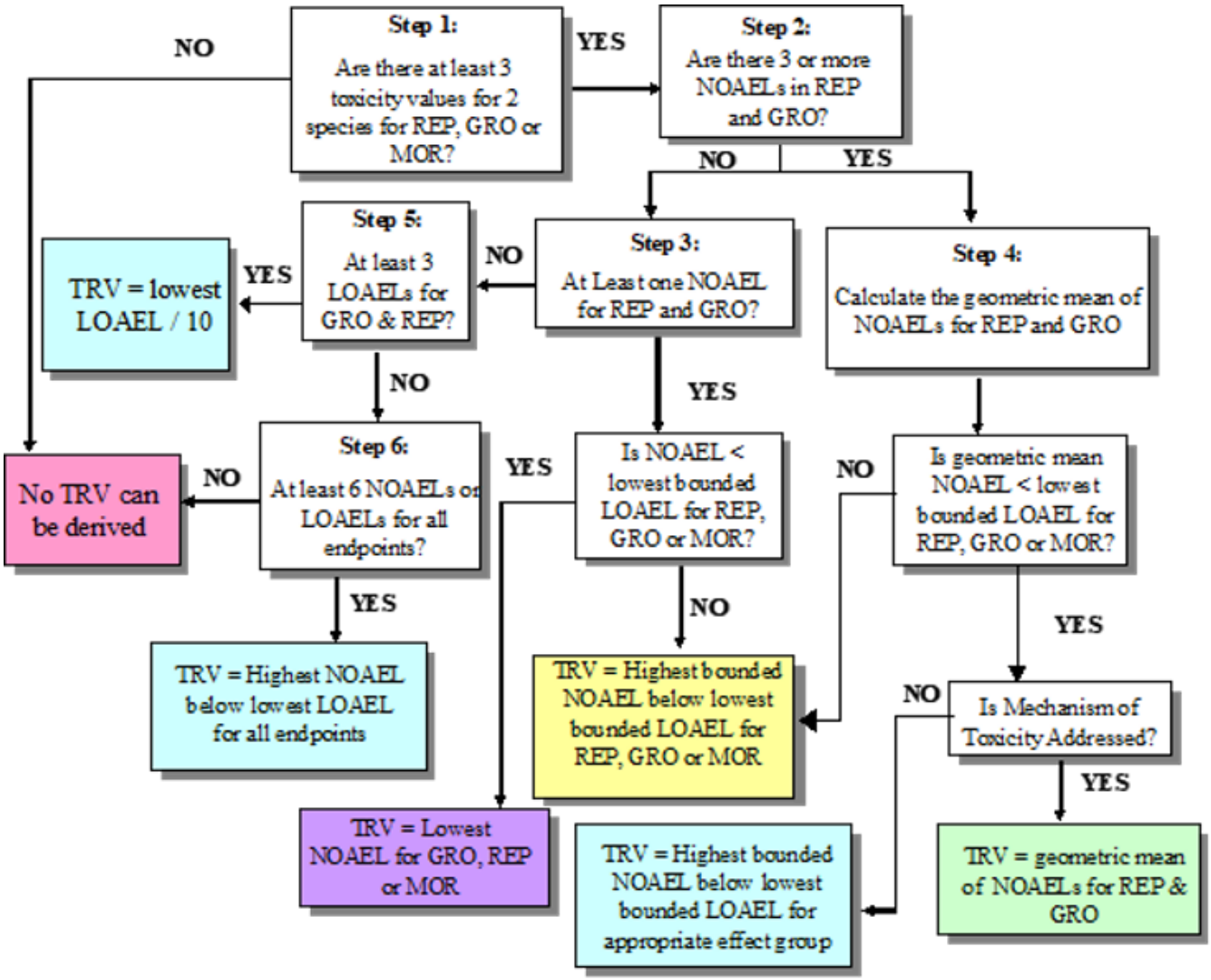
For rats, a 2-year NOEL/LOEL (NTP, 1991b), a 16-week NOEL/LOEL for males, and a 16-week NOEL/LOEL for females for mortality were used (Matthews et al., 1990).
For mice, a 16-week NOEL/LOEL for reproduction (Matthews et al., 1990) and an 8-day LOEL for mortality were used (Hazleton Laboratories, 1983).

Step 2: Are there three or more NOELs in reproduction or growth effect groups?
Because there was only a single reproduction effect result and no growth effect results, then proceed to step 3.

Step 3: If there is at least one NOEL result for the reproduction or growth effect groups?

The NOEL for reproduction is 175 mg/kg-bw/day
Then the TRV is equal to the lowest reported NOEL for any effect group (reproduction, growth, or mortality), except in cases where the NOEL is higher than the lowest bounded LOEL.
The lowest bounded LOEL for mortality is 88 mg/kg-bw/day
Then the TRV is equal to the highest bounded NOEL below the lowest bounded LOEL.
The highest NOEL below the lowest NOEL is 44 mg/kg-bw/day.

The TRV for TCEP is 44 mg/kg-bw/day.



Figure_Apx G-9. TRV Flow Chart

G.2.3 Evidence Integration

Data integration includes analysis, synthesis, and integration of information for the risk evaluation. During data integration, EPA considers quality, consistency, relevancy, coherence, and biological plausibility to make final conclusions regarding the weight of scientific evidence. As stated in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), data integration involves transparently discussing the significant issues, strengths, and limitations as well as the uncertainties of the reasonably available information and the major points of interpretation.

The general analytical approaches for integrating evidence for environmental hazard is discussed in Section 7.4 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a).

The organization and approach to integrating hazard evidence is determined by the reasonably available evidence regarding routes of exposure, exposure media, duration of exposure, taxa, metabolism and distribution, effects evaluated, the number of studies pertaining to each effect, as well as the results of the data quality evaluation.

The environmental hazard integration is organized around effects to aquatic and terrestrial organisms as well as the respective environmental compartments (*e.g.*, pelagic, benthic, soil). Environmental hazard assessment may be complex based on the considerations of the quantity, relevance, and quality of the available evidence.

For TCEP, environmental hazard data from toxicology studies identified during systematic review have used evidence that characterizes apical endpoints; that is, endpoints that could have population-level effects such as reproduction, growth, and/or mortality. Additionally, mechanistic data that can be linked to apical endpoints will add to the weight of scientific evidence supporting hazard thresholds. EPA also considered predictions from Web-ICE and ECOSAR to supplement the empirical data found during systematic review.

G.2.3.1 Weight of Scientific Evidence

After calculating the hazard thresholds that were carried forward to characterize risk, a narrative describing the weight of scientific evidence and uncertainties was completed to support EPA's decisions. The weight of scientific evidence fundamentally means that the evidence is weighed (*i.e.*, ranked) and weighted (*i.e.*, a piece or set of evidence or uncertainty may have more importance or influence in the result than another). Based on the weight of scientific evidence and uncertainties, a confidence statement was developed that qualitatively ranks (*i.e.*, robust, moderate, slight, or indeterminate) the confidence in the hazard threshold. The qualitative confidence levels are described below.

The evidence considerations and criteria detailed within U.S. EPA (2021a) guides the application of strength-of-evidence judgments for environmental hazard effect within a given evidence stream and were adapted from Table 7-10 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a).

EPA used the strength-of-evidence and uncertainties from (U.S. EPA, 2021a) for the hazard assessment to qualitatively rank the overall confidence using evidence Table 4-8 for environmental hazard. Confidence levels of robust (+ + +), moderate (+ +), slight (+), or indeterminant are assigned for each evidence property that corresponds to the evidence considerations (U.S. EPA, 2021a). The rank of the *Quality of the Database* consideration is based on the systematic review overall quality determination (High, Medium, or Low) for studies used to calculate the hazard threshold, and whether there are data gaps in the toxicity dataset. Another consideration in the *Quality of the Database* is the risk of bias (*i.e.*, how representative is the study to ecologically relevant endpoints). Additionally, because of the importance of the studies used for deriving hazard thresholds, the *Quality of the Database* consideration may have greater weight than the other individual considerations. The high, medium, and low systematic review overall quality determination ranks correspond to the evidence table ranks of robust (+ + +), moderate (+ +), or slight (+), respectively. The evidence considerations are weighted based on professional judgment to obtain the overall confidence for each hazard threshold. In other words, the weights of each evidence property relative to the other properties are dependent on the specifics of the weight of scientific evidence and uncertainties that are described in the narrative and may or may not be equal. Therefore, the overall score is not necessarily a mean or defaulted to the lowest score. The confidence levels and uncertainty type examples are described below.

Confidence Levels

- Robust (+ + +) confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the exposure or hazard estimate.

- Moderate **(+ +)** confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize exposure or hazard estimates.
- Slight (+) confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.
- Indeterminant (N/A) corresponds to entries in evidence tables where information is not available within a specific evidence consideration.

Types of Uncertainties

The following uncertainties may be relevant to one or more of the weight of scientific evidence considerations listed above and will be integrated into that property's rank in the evidence table (Table 4-8):

- *Scenario Uncertainty:* Uncertainty regarding missing or incomplete information needed to fully define the exposure and dose.
    - The sources of scenario uncertainty include descriptive errors, aggregation errors, errors in professional judgment, and incomplete analysis.
- *Parameter Uncertainty:* Uncertainty regarding some parameter.
    - Sources of parameter uncertainty include measurement errors, sampling errors, variability, and use of generic or surrogate data.
- *Model Uncertainty:* Uncertainty regarding gaps in scientific theory required to make predictions on the basis of causal inferences.
    - Modeling assumptions may be simplified representations of reality.

Table_Apx G-3 summarizes the weight of scientific evidence and uncertainties, while increasing transparency on how EPA arrived at the overall confidence level for each exposure hazard threshold. Symbols are used to provide a visual overview of the confidence in the body of evidence, while de-emphasizing an individual ranking that may give the impression that ranks are cumulative (*e.g.*, ranks of different categories may have different weights).

Table_Apx G-3. Considerations that Inform Evaluations of the Strength of the Evidence within an Evidence Stream (*i.e.*, Apical Endpoints, Mechanistic, or Field Studies)

Consideration	Increased Evidence Strength (of the Apical Endpoints, Mechanistic, or Field Studies Evidence)	Decreased Evidence Strength (of the Apical Endpoints, Mechanistic, or Field Studies Evidence)
The evidence considerations and criteria laid out here guide the application of strength-of-evidence judgments for an outcome or environmental hazard effect within a given evidence stream. Evidence integration or synthesis results that do not warrant an increase or decrease in evidence strength for a given consideration are considered "neutral" and are not described in this table (and, in general, are captured in the assessment-specific evidence profile tables).		
Quality of the database[a] (risk of bias)	• A large evidence base of *high*- or *medium*-quality studies increases strength. • Strength increases if relevant species are represented in a database.	• An evidence base of mostly *low*-quality studies decreases strength. • Strength also decreases if the database has data gaps for relevant species (*i.e.*, a trophic level that is not represented). • Decisions to increase strength for other considerations in this table should generally not be made if there are serious concerns for risk of bias; in other words, all the other considerations in this table are dependent upon the quality of the database.
Consistency	Similarity of findings for a given outcome (*e.g.*, of a similar magnitude, direction) across independent studies or experiments increases strength, particularly when consistency is observed across species, lifestage, sex, wildlife populations, and across or within aquatic and terrestrial exposure pathways.	• Unexplained inconsistency (*i.e.*, conflicting evidence; see U.S. EPA (2005b) decreases strength.) • Strength should not be decreased if discrepant findings can be reasonably explained by study confidence conclusions; variation in population or species, sex, or lifestage; frequency of exposure (*e.g.*, intermittent or continuous); exposure levels (low or high); or exposure duration.
Strength (effect magnitude) and precision	• Evidence of a large magnitude effect (considered either within or across studies) can increase strength. • Effects of a concerning rarity or severity can also increase strength, even if they are of a small magnitude. • Precise results from individual studies or across the set of studies increases strength, noting that biological significance is prioritized over statistical significance. • Use of probabilistic model (*e.g.*, Web-ICE, SSD) may increase strength.	Strength may be decreased if effect sizes that are small in magnitude are concluded not to be biologically significant, or if there are only a few studies with imprecise results.
Biological gradient/dose-response	• Evidence of dose-response increases strength. • Dose-response may be demonstrated across studies or within studies and it can be dose- or duration-dependent. • Dose response may not be a monotonic dose-response (monotonicity should not necessarily be	• A lack of dose-response when expected based on biological understanding and having a wide range of doses/exposures evaluated in the evidence base can decrease strength. • In experimental studies, strength may be decreased when effects resolve under certain experimental conditions (*e.g.*, rapid reversibility after removal of exposure).

Consideration	Increased Evidence Strength (of the Apical Endpoints, Mechanistic, or Field Studies Evidence)	Decreased Evidence Strength (of the Apical Endpoints, Mechanistic, or Field Studies Evidence)
	expected (*e.g.*, different outcomes may be expected at low vs. high doses due to activation of different mechanistic pathways or induction of systemic toxicity at very high doses). • Decreases in a response after cessation of exposure (*e.g.*, return to baseline fecundity) also may increase strength by increasing certainty in a relationship between exposure and outcome (this particularly applicable to field studies).	• However, many reversible effects are of high concern. Deciding between these situations is informed by factors such as the toxicokinetics of the chemical and the conditions of exposure, see (U.S. EPA, 1998b), endpoint severity, judgments regarding the potential for delayed or secondary effects, as well as the exposure context focus of the assessment (*e.g.*, addressing intermittent or short-term exposures). • In rare cases, and typically only in toxicology studies, the magnitude of effects at a given exposure level might decrease with longer exposures (*e.g.*, due to tolerance or acclimation). • Like the discussion of reversibility above, a decision about whether this decreases evidence strength depends on the exposure context focus of the assessment and other factors. • If the data are not adequate to evaluate a dose-response pattern, then strength is neither increased nor decreased.
Biological relevance	Effects observed in different populations or representative species suggesting that the effect is likely relevant to the population or representative species of interest (*e.g.*, correspondence among the taxa, lifestages, and processes measured or observed and the assessment endpoint).	An effect observed only in a specific population or species without a clear analogy to the population or representative species of interest decreases strength.
Physical/chemical relevance	Correspondence between the substance tested and the substance constituting the stressor of concern.	The substance tested is an analogue of the chemical of interest or a mixture of chemicals which include other chemicals besides the chemical of interest.
Environmental relevance	Correspondence between test conditions and conditions in the region of concern.	The test is conducted using conditions that would not occur in the environment.

[a] Database refers to the entire dataset of studies integrated in the environmental hazard assessment and used to inform the strength of the evidence. In this context, database does *not* refer to a computer database that stores aggregations of data records such as the ECOTOX Knowledgebase.

Appendix H ENVIRONMENTAL RISK DETAILS

H.1 Risk Estimation for Aquatic Organisms

Table_Apx H-1. Calculated RQs Based on TCEP Surface Water Concentrations (ppb) as Calculated Using Modeled Data for Annual Air Deposition to Surface Water

Exposure Scenario	Production Volume (lb/year)[a]	Meteorological Model[b]	Surface Water Concentration (ppb) at 1,000 m[c]	Chronic RQ (Hazard Value: 2.8 ppb)
Import and repackaging	2,500	MetCT	3.93E–05	1.40E–05
		MetHIGH	4.78E–05	1.71E–05
	25,000	MetCT	1.40E–04	5.00E–05
		MetHIGH	1.94E–04	6.93E–05
Incorporation into paints and coatings – 1-part coatings	2,500	MetCT	8.60E–04	3.07E–04
		MetHIGH	1.37E–03	4.89E–04
	25,000	MetCT	1.95E–03	6.96E–04
		MetHIGH	2.07E–03	7.39E–04
Incorporation into paints and coatings – 2-part reactive coatings	2,500	MetCT	2.20E–04	7.86E–05
		MetHIGH	3.18E–04	1.14E–04
	25,000	MetCT	6.05E–04	2.16E–04
		MetHIGH	9.65E–04	3.45E–04
Use in paints and coatings – spray application	2,500	MetCT	3.42E–01	1.22E–01
		MetHIGH	4.93E–01	1.76E–01
	25,000	MetCT	5.10	1.82
		MetHIGH	8.10	2.89
Formulation of TCEP-containing reactive resins (for use in 2-part systems)	2,500	MetCT	1.02E–03	3.64E–04
		MetHIGH	9.70E–04	3.46E–04
	25,000	MetCT	7.60E–04	2.71E–04
		MetHIGH	7.05E–04	2.52E–04
Processing into 2-part resin article	2,500	MetCT	2.46E–04	8.79E–05
		MetHIGH	3.55E–04	1.27E–04
	25,000	MetCT	7.20E–04	2.57E–04
		MetHIGH	1.15E–03	4.11E–04
Laboratory chemicals	2,500	MetCT	1.26E–03	4.50E–04
		MetHIGH	1.17E–03	4.18E–04
	25,000	MetCT	7.20E–04	2.57E–04
		MetHIGH	6.65E–04	2.38E–04

[a] Production volume of 2,500 lb TCEP/year uses high-end estimates (95th percentile). Production volume of 25,000 lb TCEP/yr uses central tendency estimates (median).
[b] The ambient air modeled concentrations and deposition values are presented for two meteorology conditions (Sioux Falls, South Dakota, for central tendency meteorology; and Lake Charles, Louisiana, for higher-end meteorology).
[c] Estimated annual concentrations of TCEP (90th percentile) that could be in surface water via air deposition at a community (1,000 m from the source) exposure scenario.

Table_Apx H-2. Environmental RQs by Exposure Scenario with Production Volumes of 2,500 lb/year with Central Tendency Release Estimates for Aquatic Organisms with TCEP Surface Water Concentration (ppb) Modeled by VVWM-PSC with 50% Percentile Flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Modeled Using VVWM-PSC[c]				
				Max Day Avg (ppb)[b]	COC Type	COC (ppb)	Days of Exceedance (days per year)	RQ
Import and repackaging	2,500	4	6.35	1,510	Acute	16,700	N/A	0.09
				202	Chronic	2.8	33	72.14
Incorporation into paints and coatings – 1-part coatings	2,500	6	10.23	2,960	Acute	16,700	N/A	0.18
				596	Chronic	2.8	74	212.86
Incorporation into paints and coatings – 2-part reactive coatings	2,500	1	27.12	6,930	Acute	16,700	N/A	0.41
				264	Chronic	2.8	32	94.28
Use in paints and coatings at – spray application	2,500	1	2.36	520	Acute	16,700	NA	0.03
				18.8	Chronic	2.8	29	6.71
Formulation of TCEP into 2-part reactive resins	2,500	1	25.19	7,220	Acute	16,700	N/A	0.43
				290	Chronic	2.8	39	103.57
Laboratory chemicals	2,500	193	0.88	212	Acute	16,700	N/A	1.27E–02
				212	Chronic	2.8	226	75.71

[a] Production volume of 2,500 lb TCEP/year uses central tendency estimates (50th percentile for all COUs except the laboratory chemicals COU uses the 5th percentile).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 50th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-3. Environmental RQs by Exposure Scenario with Production Volumes of 2,500 lb/year with Central Tendency Release Estimates for Aquatic Organisms with TCEP Pore Water Concentration (ppb) Modeled by VVWM-PSC with 50% Percentile Flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Benthic Pore Water Concentration (ppb)[b]	Benthic Pore Water Concentration[c]			
					COC Type	COC (ppb)	Days of Exceedance	RQ
Import and repackaging	2,500	4	6.35	98.6	Acute	16,700	N/A	5.90E−03
				79.6	Chronic	2.8	225	28.43
Incorporation into paints and coatings – 1-part coatings	2,500	6	10.23	285	Acute	16,700	N/A	1.70E−02
				232	Chronic	2.8	291	82.85
Incorporation into paints and coatings – 2-part reactive coatings	2,500	1	27.12	130	Acute	16,700	N/A	7.78E−03
				105	Chronic	2.8	242	37.5
Use in paints and coatings – spray application	2,500	1	2.36	9.29	Acute	16,700	N/A	5.60E−04
				7.48	Chronic	2.8	78	2.67
Formulation of TCEP into 2-part reactive resins	2,500	1	25.19	141	Acute	16,700	N/A	8.44E−03
				115	Chronic	2.8	249	41.07
Laboratory chemicals	2,500	193	0.88	200	Acute	16,700	N/A	1.19E−02
				199	Chronic	2.8	364	71.07

[a] Production volume of 2,500 lb TCEP/year uses central tendency estimates (50th percentile for all COUs except the laboratory chemicals COU uses the 5th percentile).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 50th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-4. Environmental RQs by Exposure Scenario with Production Volumes of 2,500 lb/year with Central Tendency Release Estimates for Aquatic Organisms with TCEP Surface Water Concentration (ppb) Modeled by VVWM-PSC with 90% Percentile Flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Modeled Using VVWM-PSC[c]				
				Max Day Avg (ppb)[b]	COC Type	COC (ppb)	Days of Exceedance (days per year)	RQ
Import and repackaging	2,500	4	6.35	8.70	Acute	16,700	N/A	5.20E−04
				1.17	Chronic	2.8	0	0.41
Incorporation into paints and coatings – 1-part coatings	2,500	6	10.23	10.60	Acute	16,700	N/A	6.30E−04
				2.12	Chronic	2.8	0	0.75
Incorporation into paints and coatings – 2-part reactive coatings	2,500	1	27.12	28.20	Acute	16,700	N/A	1.69E−03
				0.94	Chronic	2.8	0	0.33
Use in paints and coatings – spray application	2,500	1	2.36	3.30	Acute	16,700	NA	2.00E−04
				0.11	Chronic	2.8	0	0.04
Formulation of TCEP into 2-part reactive resins	2,500	1	25.19	4.00	Acute	16,700	N/A	2.40E−04
				0.13	Chronic	2.8	0	0.05
Laboratory chemicals	2,500	193	0.88	1.20	Acute	16,700	N/A	7.00E−05
				1.21	Chronic	2.8	0	0.43

[a] Production volume of 2,500 lb TCEP/year uses central tendency estimates (50th percentile for all COUs except the laboratory chemicals COU uses the 5th percentile).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 90th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-5. Environmental RQs by Exposure Scenario with Production Volumes of 2,500 lb/year with Central Tendency Release Estimates for Aquatic Organisms with TCEP Pore Water Concentration (ppb) Modeled by VVWM-PSC with 90% Percentile Flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Benthic Pore Water Concentration (ppb)[b]	Benthic Pore Water Concentration[c]			
					COC Type	COC (ppb)	Days of Exceedance	RQ
Import and repackaging	2,500	4	6.35	0.58	Acute	16,700	N/A	3.47E−05
				0.46	Chronic	2.8	0	0.16
Incorporation into paints and coatings – 1-part coatings	2,500	6	10.23	1.04	Acute	16,700	N/A	6.23E−05
				0.83	Chronic	2.8	0	0.29
Incorporation into paints and coatings – 2-part reactive coatings	2,500	1	27.12	0.48	Acute	16,700	N/A	2.87E−05
				0.38	Chronic	2.8	0	0.14
Use in paints and coatings – spray application	2,500	1	2.36	0.06	Acute	16,700	N/A	3.59E−06
				0.04	Chronic	2.8	0	1.42E−02
Formulation of TCEP into 2-part reactive resins	2,500	1	25.19	0.07	Acute	16,700	N/A	4.19E−06
				0.05	Chronic	2.8	0	1.78E−02
Laboratory chemicals	2,500	193	0.88	1.15	Acute	16,700	N/A	6.89E−05
				1.14	Chronic	2.8	84	0.41

[a] Production volume of 2,500 lb TCEP/year uses central tendency estimates (50th percentile for all COUs except the laboratory chemicals COU uses the 5th percentile).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 90th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-6. Environmental RQs by Exposure Scenario with Production Volumes of 2,500 lb/year with High-End Release Estimates for Aquatic Organisms with TCEP Surface Water Concentration (ppb) Modeled by VVWM-PSC with 90% Percentile Flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Modeled Using VVWM-PSC[c]				
				Max Day Avg (ppb)[b]	COC Type	COC (ppb)	Days of Exceedance (days per year)	RQ
Import and repackaging	2,500	4	9.88	13.60	Acute	16,700	N/A	8.10E−04
				1.81	Chronic	2.8	0	0.64
Incorporation into paints and coatings – 1-part coatings	2,500	2	35.18	36.50	Acute	16,700	N/A	2.19E−03
				2.43	Chronic	2.8	0	0.87
Incorporation into paints and coatings – 2-part reactive coatings	2,500	1	31.89	33.10	Acute	16,700	N/A	1.98E−03
				1.10	Chronic	2.8	0	0.39
Use in paints and coatings – spray application	2,500	2	23.26	32.00	Acute	16,700	NA	1.92E−03
				2.13	Chronic	2.8	0	0.76
Formulation of TCEP into 2-part reactive resins	2,500	1	31.54	5.00	Acute	16,700	N/A	3.00E−04
				0.17	Chronic	2.8	0	0.06
Laboratory chemicals	2,500	182	0.40	0.60	Acute	16,700	N/A	4.00E−05
				0.55	Chronic	2.8	0	0.2

[a] Production volume of 2,500 lb TCEP/year uses high-end estimates (95th percentile for all COUs except the laboratory chemicals COU uses the 1st percentile).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 90th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-7. Environmental RQs by Exposure Scenario with Production Volumes of 2,500 lb/year with High-End Release Estimates for Aquatic Organisms with TCEP Pore Water Concentration (ppb) Modeled by VVWM-PSC with 90% Percentile Flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Benthic Pore Water Concentration (ppb)[b]	Benthic Pore Water Concentration[c]			
					COC Type	COC (ppb)	Days of Exceedance	RQ
Import and repackaging	2,500	4	9.88	0.9	Acute	16,700	N/A	5.39E−05
				0.72	Chronic	2.8	0	0.25
Incorporation into paints and coatings – 1-part coatings	2,500	2	35.18	1.23	Acute	16,700	N/A	7.37E−05
				0.97	Chronic	2.8	0	0.34
Incorporation into paints and coatings – 2-part reactive coatings	2,500	1	31.89	0.56	Acute	16,700	N/A	3.35E−05
				0.44	Chronic	2.8	0	0.16
Use in paints and coatings – spray application	2,500	2	23.26	1.08	Acute	16,700	N/A	6.47E−05
				0.85	Chronic	2.8	0	0.3
Formulation of TCEP into 2-part reactive resins	2,500	1	31.54	0.08	Acute	16,700	N/A	4.79E−06
				0.7	Chronic	2.8	0	0.25
Laboratory chemicals	2,500	182	0.40	0.52	Acute	16,700	N/A	3.11E−05
				0.52	Chronic	2.8	0	0.19

[a] Production volume of 2,500 lb TCEP/year uses high-end estimates (95th percentile for all COUs except the laboratory chemicals COU uses the 5th percentile).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 90th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-8. Environmental RQs by Exposure Scenario with Production Volumes of 25,000 lb/year with Central Tendency Release Estimates for Aquatic Organisms with TCEP Surface Water Concentration (ppb) Modeled by VVWM-PSC with 50% Percentile Flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Modeled Using VVWM-PSC[c]				
				Max Day Avg (ppb)[b]	COC Type	COC (ppb)	Days of Exceedance (days per year)	RQ
Import and repackaging	25,000	39	7.13	1,710	Acute	16,700	N/A	0.10
				1700	Chronic	2.8	156	607.14
Incorporation into paints and coatings – 1-part coatings	25,000	57	10.98	3,220	Acute	16,700	N/A	0.19
				3,210	Chronic	2.8	237	1146.43
Incorporation into paints and coatings – 2-part reactive coatings	25,000	4	65.90	19,100	Acute	16,700	N/A	1.14
				2560	Chronic	2.8	158	914.29
Use in paints and coatings – spray application	25,000	1	2.31	509	Acute	16,700	N/A	0.03
				18.4	Chronic	2.8	29	6.57
Formulation of TCEP into 2-part reactive resins	25,000	3	45.51	15,500	Acute	16,700	N/A	0.93
				1570	Chronic	2.8	139	560.71

[a] Production volume of 25,000 lb TCEP/year uses central tendency estimates (50th percentile for all COUs).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 50th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-9. Environmental RQs by Exposure Scenario with Production Volumes of 25,000 lb/year with central tendency release estimates for Aquatic Organisms with TCEP Benthic Pore Water Concentration (ppb) Modeled by VVWM-PSC with 50% percentile flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Benthic Pore Water Concentration (ppb)[b]	Benthic Pore Water Concentration[c]			
					COC Type	COC (ppb)	Days of Exceedance	RQ
Import and repackaging	25,000	39	7.13	825	Acute	16,700	N/A	0.04
				724	Chronic	2.8	364	258.57
Incorporation into paints and coatings – 1-part coatings	25,000	57	10.98	1990	Acute	16,700	N/A	0.11
				1800	Chronic	2.8	364	642.85
Incorporation into paints and coatings – 2-part reactive coatings	25,000	4	65.90	1240	Acute	16,700	N/A	0.07
				1010	Chronic	2.8	364	360.71
Use in paints and coatings – spray application	25,000	1	2.31	9.1	Acute	16,700	N/A	5.4E−04
				7.3	Chronic	2.8	77	2.61
Formulation of TCEP into 2-part reactive resins	25,000	3	45.51	759	Acute	16,700	N/A	0.04
				619	Chronic	2.8	349	221.07

[a] Production volume of 25,000 lb TCEP/year uses central tendency estimates (50th percentile for all COUs).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 50th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-10. Environmental RQs by Exposure Scenario with Production Volumes of 25,000 lb/year with central tendency release estimates for Aquatic Organisms with TCEP Surface Water Concentration (ppb) Modeled by VVWM-PSC with 90% percentile flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Modeled Using VVWM-PSC[c]				
				Max Day Avg (ppb)[b]	COC Type	COC (ppb)	Days of Exceedance (days per year)	RQ
Import and repackaging	25,000	39	7.13	9.80	Acute	16,700	N/A	5.90E−04
				9.80	Chronic	2.8	51	3.5
Incorporation into paints and coatings – 1-part coatings	25,000	57	10.98	11.40	Acute	16,700	N/A	6.80E−04
				11.40	Chronic	2.8	71	4.07
Incorporation into paints and coatings – 2-part reactive coatings	25,000	4	65.90	68.40	Acute	16,700	N/A	4.10E−03
				9.12	Chronic	2.8	30	3.25
Use in paints and coatings at – spray application	25,000	1	2.31	3.20	Acute	16,700	NA	1.9E−04
				0.11	Chronic	2.8	0	0.04
Formulation of TCEP into 2-part reactive resins	25,000	3	45.51	7.20	Acute	16,700	N/A	4.30E−04
				0.72	Chronic	2.8	0	0.25

[a] Production volume of 25,000 lb TCEP/year uses central tendency estimates (50th percentile for all COUs).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 90th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

Table_Apx H-11. Environmental RQs by Exposure Scenario with Production Volumes of 25,000 lb/year with Central Tendency Release Estimates for Aquatic Organisms with TCEP Benthic Pore Water Concentration (ppb) Modeled by VVWM-PSC with 90% Percentile Flow of the 7Q10

Exposure Scenario	Production Volume (lb/year)[a]	Days of Release	Release (kg/day)	Benthic Pore Water Concentration (ppb)[b]	Benthic Pore Water Concentration[c]			
					COC Type	COC (ppb)	Days of Exceedance	RQ
Import and repackaging	25,000	39	7.13	4.79	Acute	16,700	N/A	2.90E−04
				4.17	Chronic	2.8	51	1.49
Incorporation into paints and coatings – 1-part coatings	25,000	57	10.98	7.11	Acute	16,700	N/A	4.30E−04
				6.4	Chronic	2.8	94	2.28
Incorporation into paints and coatings – 2-part reactive coatings	25,000	4	65.90	4.55	Acute	16,700	N/A	2.70E−04
				3.6	Chronic	2.8	25	1.29
Use in paints and coatings at – spray application	25,000	1	2.31	0.05	Acute	16,700	N/A	3.00E−06
				0.04	Chronic	2.8	0	1.42E−02
Formulation of TCEP into 2-part reactive resins	25,000	3	45.51	0.36	Acute	16,700	N/A	2.20E−05
				0.29	Chronic	2.8	0	0.10

[a] Production volume of 25,000 lb TCEP/year uses central tendency estimates (50th percentile for all COUs).
[b] Max day average represents the maximum concentration over a 1- or 30-day average period corresponding with the acute or chronic COC used for the RQ estimate.
[c] Flow inputs for PSC represent the 90th percentile 7Q10 flows as the lowest expected weekly flow over a 10-year period.
N/A = Days of exceedance are modeled for the application of chronic COCs and do not apply for acute COCs and corresponding RQs.

H.2 Risk Estimation for Terrestrial Organisms

Table_Apx H-12. Calculated RQs Based on TCEP Soils Concentrations (mg/kg) as Calculated Using Modeled Data for Air Deposition to Soil

Exposure Scenario	Production Volume (lb/year)[a]	Meteorological Model[b]	Soil Concentration (mg/kg) at 1,000 m[c]	Chronic RQ (Hazard Value: 612 mg/kg)
Import and Repackaging	2,500	MetCT	1.49E–06	2.43E–09
		MetHIGH	1.92E–06	3.14E–09
	25,000	MetCT	5.43E–06	8.87E–09
		MetHIGH	7.59E–06	1.24E–08
Incorporation into paints and coatings – 1-part coatings	2,500	MetCT	3.33E–05	5.44E–08
		MetHIGH	5.67E–05	9.27E–08
	25,000	MetCT	7.59E–05	1.24E–07
		MetHIGH	8.24E–05	1.35E–07
Incorporation into paints and coatings – 2-part reactive coatings	2,500	MetCT	1.11E–05	1.82E–08
		MetHIGH	2.41E–05	3.94E–08
	25,000	MetCT	2.19E–05	3.59E–08
		MetHIGH	3.68E–05	6.01E–08
Use in paints and coatings at – spray application	2,500	MetCT	3.97E–03	6.49E–06
		MetHIGH	5.58E–03	9.11E–06
	25,000	MetCT	5.59E–02	9.14E–05
		MetHIGH	8.65E–02	1.41E–04
Formulation of TCEP-containing reactive resins (for use in 2-part systems)	2,500	MetCT	3.89E–05	6.35E–08
		MetHIGH	3.85E–05	6.30E–08
	25,000	MetCT	2.93E–05	4.79E–08
		MetHIGH	2.82E–05	4.60E–08
Processing into 2-part resin article	2,500	MetCT	1.21E–05	1.97E–08
		MetHIGH	2.57E–05	4.20E–08
	25,000	MetCT	2.71E–05	4.42E–08
		MetHIGH	4.58E–05	7.48E–08
Laboratory chemicals	2,500	MetCT	4.84E–05	7.90E–08
		MetHIGH	4.65E–05	7.59E–08
	25,000	MetCT	2.75E–05	4.50E–08
		MetHIGH	2.68E–05	4.37E–08

[a] Production volume of 2,500 lb TCEP/yr uses high-end estimates (95th percentile). Production volume of 25,000 lb TCEP/yr uses central tendency estimates (median).
[b] The ambient air modeled concentrations and deposition values are presented for two meteorology conditions (Sioux Falls, South Dakota, for central tendency meteorology; and Lake Charles, Louisiana, for higher-end meteorology).
[c] Estimated concentrations of TCEP (90th percentile) that could be in soil via air deposition at a community (1,000 m from the source) exposure scenario.

H.3 Trophic Transfer Analysis Results

Table_Apx H-13. RQs Based on Potential Trophic Transfer of TCEP in Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs (Equation 4-1)

Exposure Scenario	PV (lb/year)[a]	Model[b]	Soil Concentration (mg/kg) at 1,000 m[c]	Nematode		Mammal		Short-Tailed Shrew	
				TCEP in biota (mg/kg/day)	RQ	TCEP in Biota (mg/kg/day)	RQ	TCEP in Biota (mg/kg/day)	RQ
Import and Repackaging	2,500	MetCT	1.49E–06	1.5E–06	2.4E–09	1.2E–06	2.7E–08	1.2E–06	1.8E–06
		MetHIGH	1.92E–06	1.9E–06	3.1E–09	1.5E–06	3.5E–08	1.5E–06	2.3E–06
	25,000	MetCT	5.43E–06	5.4E–06	8.9E–09	4.3E–06	9.8E–08	4.3E–06	6.5E–06
		MetHIGH	7.59E–06	7.6E–06	1.2E–08	6.0E–06	1.4E–07	6.0E–06	9.1E–06
Incorporation into paints and coatings – 1-part coatings	2,500	MetCT	3.33E–05	3.3E–05	5.4E–08	2.6E–05	6.0E–07	2.6E–05	4.0E–05
		MetHIGH	5.67E–05	5.7E–05	9.3E–08	4.5E–05	1.0E–06	4.5E–05	6.8E–05
	25,000	MetCT	7.59E–05	7.6E–05	1.2E–07	6.0E–05	1.4E–06	6.0E–05	9.1E–05
		MetHIGH	8.24E–05	8.2E–05	1.3E–07	6.5E–05	1.5E–06	6.5E–05	9.9E–05
Incorporation into paints and coatings - 2-part reactive coatings	2,500	MetCT	1.11E–05	1.1E–05	1.8E–08	8.8E–06	2.0E–07	8.8E–06	1.3E–05
		MetHIGH	2.41E–05	2.4E–05	3.9E–08	1.9E–05	4.4E–07	1.9E–05	2.9E–05
	25,000	MetCT	2.19E–05	2.2E–05	3.6E–08	1.7E–05	4.0E–07	1.7E–05	2.6E–05
		MetHIGH	3.68E–05	3.7E–05	6.0E–08	2.9E–05	6.6E–07	2.9E–05	4.4E–05
Use in paints and coatings–spray application	2,500	MetCT	0.004	0.004	6.4E–06	0.003	6.8E–05	0.003	0.005
		MetHIGH	0.006	0.0056	9.0E–06	0.004	9.8E–05	0.004	0.007
	25,000	MetCT	0.056	0.059	9.6E–05	0.044	1.0E–03	0.044	0.067
		MetHIGH	0.086	0.086	1.4E–04	0.068	1.5E–03	0.068	0.103
Formulation of TCEP-containing reactive resins (for use in 2-part systems)	2,500	MetCT	3.89E–05	3.9E–05	6.4E–08	3.1E–05	7.0E–07	3.1E–05	4.7E–05
		MetHIGH	3.85E–05	3.9E–05	6.3E–08	3.1E–05	7.0E–07	3.1E–05	4.6E–05
	25,000	MetCT	2.93E–05	2.9E–05	4.8E–08	2.3E–05	5.3E–07	2.3E–05	3.5E–05
		MetHIGH	2.82E–05	2.8E–05	4.6E–08	2.2E–05	5.1E–07	2.2E–05	3.4E–05

Exposure Scenario	PV (lb/year)[a]	Model[b]	Soil Concentration (mg/kg) at 1,000 m[c]	Nematode		Mammal		Short-Tailed Shrew	
				TCEP in biota (mg/kg/day)	RQ	TCEP in Biota (mg/kg/day)	RQ	TCEP in Biota (mg/kg/day)	RQ
Processing into 2-part resin article	2,500	MetCT	1.21E–05	1.2E–05	2.0E–08	9.6E–06	2.2E–07	9.6E–06	1.5E–05
		MetHIGH	2.57E–05	2.6E–05	4.2E–08	2.0E–05	4.6E–07	2.0E–05	3.1E–05
	25,000	MetCT	2.71E–05	2.7E–05	4.4E–08	2.2E–05	4.9E–07	2.2E–05	3.3E–05
		MetHIGH	4.58E–05	4.6E–05	7.5E–08	3.6E–05	8.3E–07	3.6E–05	5.5E–05
Laboratory chemicals	2,500	MetCT	4.84E–05	4.8E–05	7.9E–08	3.8E–05	8.7E–07	3.8E–05	5.8E–05
		MetHIGH	4.65E–05	4.6E–05	7.6E–08	3.7E–05	8.4E–07	3.7E–05	5.6E–05
	25,000	MetCT	2.75E–05	2.8E–05	4.5E–08	2.2E–05	5.0E–07	2.2E–05	3.3E–05
		MetHIGH	2.68E–05	2.7E–05	4.4E–08	2.1E–05	4.8E–07	2.1E–05	3.2E–05

[a] PV = Production volume of 2,500 lb TCEP/yr uses high-end estimates (95th percentile); PV of 25,000 lb TCEP/yr uses central tendency estimates (median).
[b] The ambient air modeled concentrations and deposition values are presented for two meteorology conditions (Sioux Falls, South Dakota, for central tendency meteorology; and Lake Charles, Louisiana, for higher-end meteorology).
[c] Estimated concentrations of TCEP (90th percentile) that could be in soil via air deposition at a community (1,000 m from the source) exposure scenario.

Table_Apx H-14. RQs Based on Potential Trophic Transfer of TCEP from Fish to American Mink as a Model Aquatic Predator Using EPA's Wildlife Risk Model for Eco-SSLs (Equation 4-1)

Scenario Name	Production Volume (lb/year)[a]	Release Distribution	SWC[b] (µg/L)	Fish Concentration (mg/kg)	American Mink	
					TCEP in Biota (mg/kg/day)	RQ
Import and repackaging	2,500	High-end	2,370	0.81	0.51	0.02
Incorporation into paints and coatings – 1-part coatings	2,500	High-end	10,300	3.50	2.21	0.08
Incorporation into paints and coatings – 2-part reactive coatings	2,500	High-end	9,340	3.18	2.01	0.07
Use in paints and coatings – spray application	2,500	High-end	5,580	1.90	1.20	0.04
Formulation of TCEP containing reactive resin	2,500	High-end	10,900	3.71	2.34	0.08
Laboratory chemicals	2,500	High-end	96	3.2E–02	0.02	7.0E–04
Import and repackaging	25,000	Central tendency	1,720	0.58	0.37	0.01
Incorporation into paints and coatings – 1-part coatings	25,000	Central tendency	3,230	1.10	0.69	0.02
Incorporation into paints and coatings – 2-part reactive coatings	25,000	Central tendency	19,300	6.56	4.15	0.14
Use in paints and coatings – spray application	25,000	Central tendency	555	0.19	0.12	4.1E–03
Processing into 2-part resin article	25,000	Central tendency	15,800	5.37	3.39	0.12
Laboratory chemicals	25,000	Central tendency	663	0.23	0.14	5.0E–03

[a] Production volume of 2,500 lb TCEP/yr uses high-end estimates (95th percentile for all COUs except the laboratory chemicals COU uses the 1st percentile). Production volume of 25,000 lb TCEP/yr uses central tendency estimates (median).
[b] TCEP Surface Water Concentration (SWC) calculated using VVWM-PSC.

Appendix I GENERAL POPULATION EXPOSURE DETAILS

I.1 Exposure Factors

Table_Apx I-1. Body Weight by Age Group

Age Group[a]	Mean Body Weight (kg)[b]
Infant (<1 year)	7.83
Young toddler (1 to <2 years)	11.4
Toddler (2 to <3 years)	13.8
Small child (3 to <6 years)	18.6
Child (6 to <11 years)	31.8
Teen (11 to <16 years)	56.8
Adults (16 to <78 years)	80.0

[a] Age group weighted average
[b] U.S. EPA (2011a), Table 8-1

Table_Apx I-2. Fish Ingestion Rates by Age Group

Age Group	Fish Ingestion Rate (g/kg-day)[a]	
	50th Percentile	90th Percentile
Infant (<1 year)[b]	N/A	N/A
Young toddler (1 to <2 years)[b]	0.053	0.412
Toddler (2 to <3 years)[b]	0.043	0.341
Small child (3 to <6 years)[b]	0.038	0.312
Child (6 to <11 years)[b]	0.035	0.242
Teen (11 to <16 years)[b]	0.019	0.146
Adult (16 to <78 years)[c]	0.063	0.277
Subsistence fisher (adult)[d]	1.78	

[a] Age group weighted average, using body weight from Table_Apx I-1 above
[b] U.S. EPA (2014a), Table 20a
[c] U.S. EPA (2014a), Table 9a
[d] U.S. EPA (2000b)

I.2 Water Pathway

I.2.1 Surface Water and Groundwater Monitoring Database Retrieval and Processing

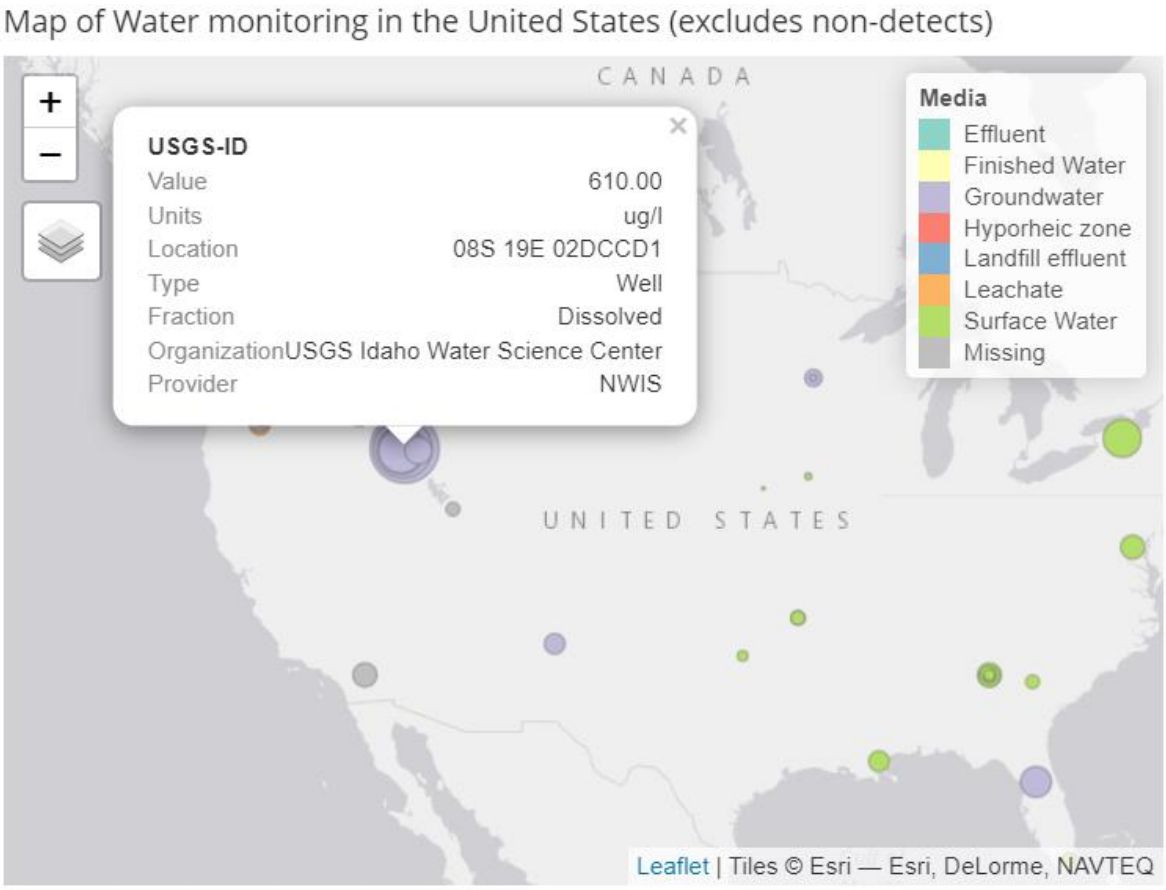
The complete set of TCEP monitoring results stored in the WQP was retrieved in March 2023, with no filters applied other than the chemical name (NWQMC, 2022). This raw dataset included 17,521 samples. To filter down to only the desired surface water samples to include in this analysis, only samples with the "ActivityMediaSubdivisionName" attribute of "Surface Water" were kept. The dataset removed values that that were below the detection limit.

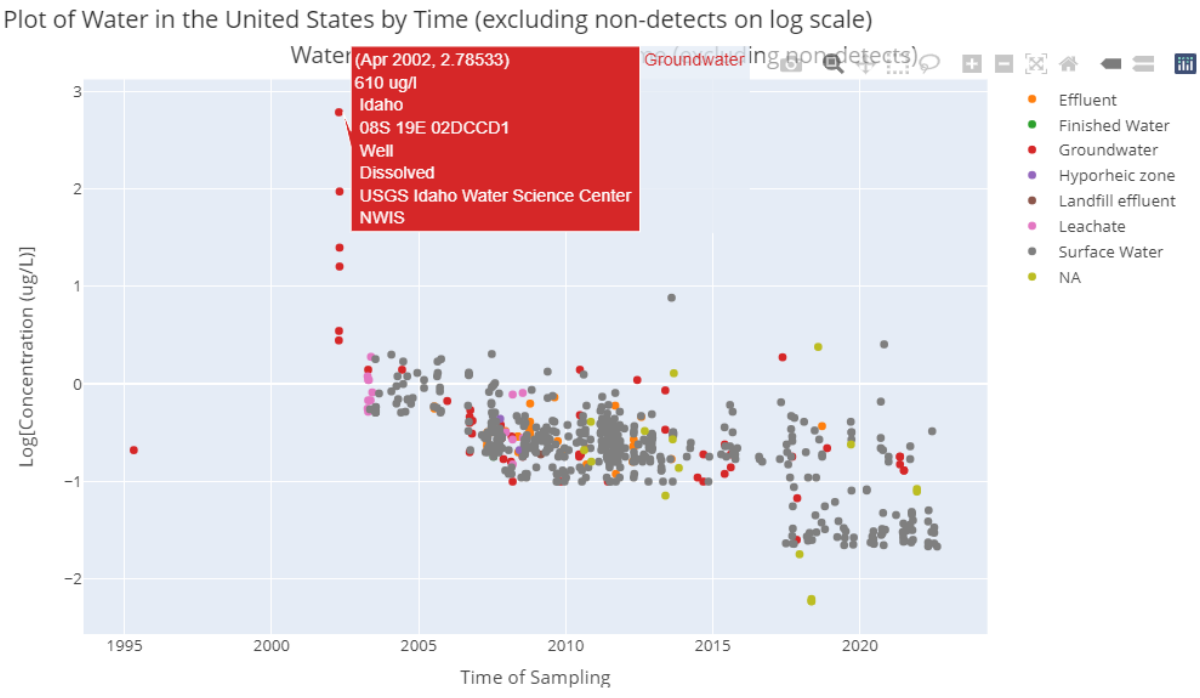
After these steps, a total of 466 surface water samples and 51 groundwater samples remained in the dataset. This monitoring dataset is attached as the *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Water Quality Portal Processed Water Data* (U.S. EPA, 2024o).

I.2.1.1 Water Plots and Figures Generated in R

Exploratory analysis of the WQP data were conducted in R. An Rmarkdown file summarizing the steps taken to explore, wrangle and visualize this dataset is available at (U.S. EPA, 2024b).

The Water Media Maps and Time Series Graphs are interactive plots made with the leaflet and plotly packages. Clicking on the points in the water media maps displays summary information of the associated data point. Similarly hovering over the data points in the Time Series Graphs provides summary information of the plotted data point. Media can be selected and de-selected in the legend to display and remove select media from the figures. The tiles to the left in the media maps allow for different map layers (Esri.WorldGrayCanvas, OpenStreetMap, Esri.WorldTopoMap) and allows users to select and deselect the underlying datasets.





Figure_Apx I-1. Example Tooltips from Media Maps and Time Series Graphs

I.2.2 Methodology for Obtaining New Flow Data (2015 to 2020)

The following steps were utilized to retrieve more recent flow data for the TCEP environmental assessment (flow values for the 2015 to 2020 are summarized in *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: E-FAST Modeling Results* (U.S. EPA, 2024g):

1. SIC codes assigned to TCEP were provided: 2851, 4952, 2821, 2823, 2824.
2. Wastewater discharge facility information was obtained for all facilities assigned to each of the SIC codes using the "echoWaterGetFacilityInfo" function in the echor package in R. This results in ≈47,000 facilities.
3. A data field was added to categorize the SIC codes into new industrial sector names as described in Table 3 of Versar's "Facility and Stream Flow Database" document. These include "Paint Formulation," "POTWs—All facilities," and "Adhesives, Sealants, Plastics, Resins, Rubber, and Manufacturing."
4. For the 4952 SIC code, only facilities with a "POTW" indicator in the permit component data field were included. This results in a list of ≈19,000 facilities. This step was taken in parallel to one described in EPA Contractor Versar's "Facility and Stream Flow Database" document, where instead of acquiring facilities with a 4,952 SIC designation, all NPDES with a POTW permit component were retrieved from the water facility search tool in ECHO. Note: Versar also created a subset "Industrial POTW" category by extracting NPDES permits with a "Y" pre-treatment indicator from the "POTW—All facilities" category, using the ICI-NPDES database on the ECHO website.
5. Any duplicate NPDESs were excluded.
6. Four hundred facilities were selected at random without replacement from each industrial sector group. This step was taken because 19,000 facilities is too many to acquire NHD flow information for in a timely manner.
7. NHD 14-digit reach codes were retrieved from the ECHO "dmr_rest_services.get_facility_report" backend server for each unique NPDES/permit that was active between 2015 to 2020, thus narrowing the facilities to only those with active permits during this time.
8. Facilities where a NPDES identifier could not be matched with a NHD reach code were excluded. 877 facilities had active permits during this time period and which also included reported NHD reach codes.
9. For each unique NPDES-reach code combination, mean and monthly average flow data were retrieved from the NHD flowline database. Exposure related flow metrics (*e.g.,* 7Q10 and 30Q5) were then calculated using methods established by the 1,4-D and 1,1-DCA teams.
10. The distribution of flows was plotted.
11. A summary statistics table was created for each of the industrial SIC categories.

I.2.3 E-FAST: Predicted Flowing Surface Water Concentrations (First Tier Modeling)

EPA's E-FAST, Version 2.0, was specifically developed to support EPA assessments of potential environmental exposures. The E-FAST Model contains default parameter values that allow for exposure estimations of a chemical in the surface water after a source emits the chemical into a water body considering simple dilution. EPA uses H-1 to estimate surface water concentrations in E-FAST.

Equation_Apx I-1.

$$SWC = \frac{R \times CF1 \times \left(1 - \frac{T}{100}\right)}{SF \times CF2}$$

Where:

SWC	=	Surface water concentration in µg/L
R	=	Release kg/site/day
$CF1$	=	Conversion factor (10^9 µg/kg)
T	=	Percent removal, typically from wastewater treatment
SF	=	Flow of receiving river (MLD)
$CF2$	=	Conversion factor (10^6 L/day/MLD)

Inputs

Release (kg/site/day): As discussed in Section 3.2, the daily release values (kg/site/day) were calculated using a production volume of 2,500 lb/year, 25,000 lb/yr, emission factors (kg TCEP released/kg TCEP handled), and number of release days per year. Refer to Table 3-3 for a summary of the release values by COU, and for sub-scenario-specific release values.

Removal from Wastewater Treatment (%): Removal from wastewater treatment is the percentage of the chemical removed from wastewater during treatment before discharge to a body of water. Although removal from wastewater treatment for TCEP was estimated as 0 percent. This is a conservative estimate relative to what is indicated in Table 2-2 that indicates wastewater removal to be 5 percent for primary treatment and 19.1 percent for complete treatment (Kim et al., 2017). EPA assumed that "on-site WWT," "POTW" release types and direct releases to water did not receive wastewater treatment and no wastewater treatment removal was applied. This is a conservative assumption that results in the total amount of TCEP released to wastewater treatment at a direct discharging site being released to surface water. It reflects the uncertainty of the type of wastewater treatment that may be in use at a direct discharging facility and the TCEP removal efficiency in that treatment.

Flow of Receiving River (Million L/Day): E-FAST requires the selection of a receiving stream flow from the E-FAST 2014 database. For site-specific assessments, the stream flow is selected by searching for a facility's NPDES permit number, name, or the known discharging waterbody reach code. As no specific facilities were identified for the TCEP assessment for water releases, stream flows were selected using the "SIC Code Option" within E-FAST. This option uses the 10th and 50th percentile stream flows of all facilities in a given industry sector, as defined by the SIC codes of the industry sector. The associated SIC Codes for the COU/OES are organized as presented in Table_Apx I-3 below.

Table_Apx I-3. Crosswalk of COU and OES, Abbreviations, and Relevant SIC Codes

COU	OES	Abbreviation	SIC Code
Manufacturing – Import – Import	Repackaging of import containers	MFG-IMP	POTW All
Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	PAINT-WB	Paint Formulation
Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Paint and coating manufacturing	Incorporation into paints and coatings – 2-part reactive coatings	PAINT-SB	Paint Formulation
Commercial use – Paints and coatings	Use in paints and coatings at job sites	COM	POTW All
Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Polymers	Formulation of TCEP containing reactive resin	PROC	Plastic Resins and Synthetic Fiber Manufacture
Use of laboratory chemicals	Wastewater to on-site treatment or discharge to POTW (with or without pretreatment)	LAB	POTW All

These SIC Code stream flows were selected because they were thought to best represent the industrial activity associated with the COUs and release type.

The flow of rivers is highly variable and is dependent on many factors such as weather patterns and effluent released from different facilities. The volume of a river varies over time with different flows expected seasonally and from year to year. The 50th percentile 7Q10 flows represent the lowest expected weekly flow over a 10-year period and were selected for use in the ecological risk assessment. The flows for the selected industry sector/SIC Code are shown in Table_Apx I-4. Although not used in the ecological assessment, harmonic means are also shown because they were used to calculate surface water concentrations for the scenario specific fish ingestion scenario in the highly exposed human exposure assessment. Harmonic mean flow values represent long-term average flow conditions.

Table_Apx I-4. Harmonic Mean, 30Q5, 7Q10, and 1Q10 50th Percentile Flows for Relevant TCEP SIC Codes

Sector within E-FAST	Year(s)	Harmonic Mean Flow MLD (50th Percentile)	30Q5 Flow MLD (50th Percentile)	7Q10 Flow MLD (50th Percentile)	1Q10 Flow MLD (50th Percentile)
SIC Code – POTW – All Facilities	2009	1.11E01	1.94	1.06	9.60E−01
	2015–2020	1.15E01	7.23	4.13	3.47
SIC Code – Paint Formulation	2009	3.54E01	1.25E01	7.29	6.10
	2015–2020	9.21	5.95	3.38	2.84
SIC Code – Plastic Resins and Synthetic Fiber Manufacture	2009	4.45E01	1.37E01	8.02	7.44
	2015–2020	6.51	5.05	2.85	2.40

Outputs

The supplemental document entitled *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Exposure EFAST 2014 Surface Water Modeling Inputs, Flow Data, and General Population Exposure Estimates and Risk Calculations* (U.S. EPA, 2024g) provides the inputs, outputs, and equations that were utilized for calculating surface water concentrations of TCEP, drinking water estimates, diluted drinking water estimates, incidental oral ingestion estimates from swimming and incidental dermal absorption estimates from swimming.

Advantages to the E-FAST 2014 model are that it requires minimal input parameters, and it has undergone extensive peer review by experts outside of EPA. The limitations associated with use of the E-FAST 2014 model relate to the assumptions made regarding use of sector-based flow information as a surrogate for site-specific flow information, as well as lack of partitioning (between dissolved and suspended sediment within the water column or between the water column and the benthic environment) and degradation parameters that were employed in the PSC model. Additionally, note that low-flow stream inputs combined with high-release estimates may yield overly conservative surface water concentrations greater than the water solubility of TCEP.

I.2.3.1 E-FAST 2014 Exposure Activity Parameters

Table_Apx I-5. Incidental Dermal (Swimming) Modeling Parameters

Input	Description (Units)	Adult (≥21 years)	Youth (11–15 years)	Child (6–10 years)	Notes	Reference
BW	Body weight (kg)	80	56.8	31.8	EPA *Exposure Factors Handbook* Chapter 8 (2011), Table 8-1 mean body weight	(U.S. EPA, 2011)
SA	Skin surface area exposed (cm^2)	19,500	15,900	10,800	U.S. EPA Swimmer Exposure Assessment Model (SWIMODEL), 2015	(U.S. EPA, 2015)
ET	Exposure time (hr/day)	3	2	1	High-end default short-term duration from U.S. EPA Swimmer Exposure Assessment Model (SWIMODEL), 2015.	(U.S. EPA, 2015)
ED	Exposure duration (years for ADD)	33	5	5	Number of years in age group, up to the 95th percentile residential occupancy period. EPA *Exposure Factors Handbook* Chapter 16 (2011), Table 16-5.	(U.S. EPA, 2011)
AT	Averaging time (years for ADD)	33	5	5	Number of years in age group, up to the 95th percentile residential occupancy period. EPA *Exposure Factors Handbook* Chapter 16 (2011), Table 16-5.	(U.S. EPA, 2011)
K_p	Permeability coefficient (cm/hr)	2.20E−03			CEM 3.2 estimated aqueous K_p based on log K_{OW} of 1.25	(Abdallah et al., 2016)

Table_Apx I-6. Incidental Oral Ingestion (Swimming) Modeling Parameters

Input	Description (Units)	Adult (≥ 21 years)	Youth (11–15 years)	Child (6–10 years)	Notes	Reference
IR_{inc}	Ingestion rate (L/hr)	0.092	0.152	0.096	EPA *Exposure Factors Handbook* Chapter 3 (2019), Table 3-7, upper percentile ingestion while swimming.	(U.S. EPA, 2019)
BW	Body weight (kg)	80	56.8	31.8	EPA *Exposure Factors Handbook* Chapter 8 (2011), Table 8-1 mean body weight.	(U.S. EPA, 2011)
ET	Exposure time (hr/day)	3	2	1	High-end default short-term duration from U.S. EPA Swimmer Exposure Assessment Model (SWIMODEL), 2015; based on competitive swimmers in the age class.	(U.S. EPA, 2015)
$IR_{inc\text{-}daily}$	Incidental daily ingestion rate (L/day)	0.276	0.304	0.096	Calculation: ingestion rate × exposure time	
IR/BW	Weighted incidental daily ingestion rate (L/kg-day)	0.0035	0.0054	0.0030	Calculation: ingestion rate/body weight	
ED	Exposure duration (years for ADD)	33	5	5	Number of years in age group, up to the 95th percentile residential occupancy period. EPA *Exposure Factors*	(U.S. EPA, 2011)

Input	Description (Units)	Adult (≥ 21 years)	Youth (11–15 years)	Child (6–10 years)	Notes	Reference
					Handbook Chapter 16 (2011), Table 16-5.	
AT	Averaging time (years for ADD)	33	5	5	Number of years in age group, up to the 95th percentile residential occupancy period. EPA *Exposure Factors Handbook* Chapter 16 (2011), Table 16-5.	(U.S. EPA, 2011)
CF1	Conversion factor (mg/µg)	1.00E−03				
CF2	Conversion factor (days/year)	365				

I.2.4 VVWM-PSC: Predicted Flowing Surface Water Concentrations (Second Tier Modeling)

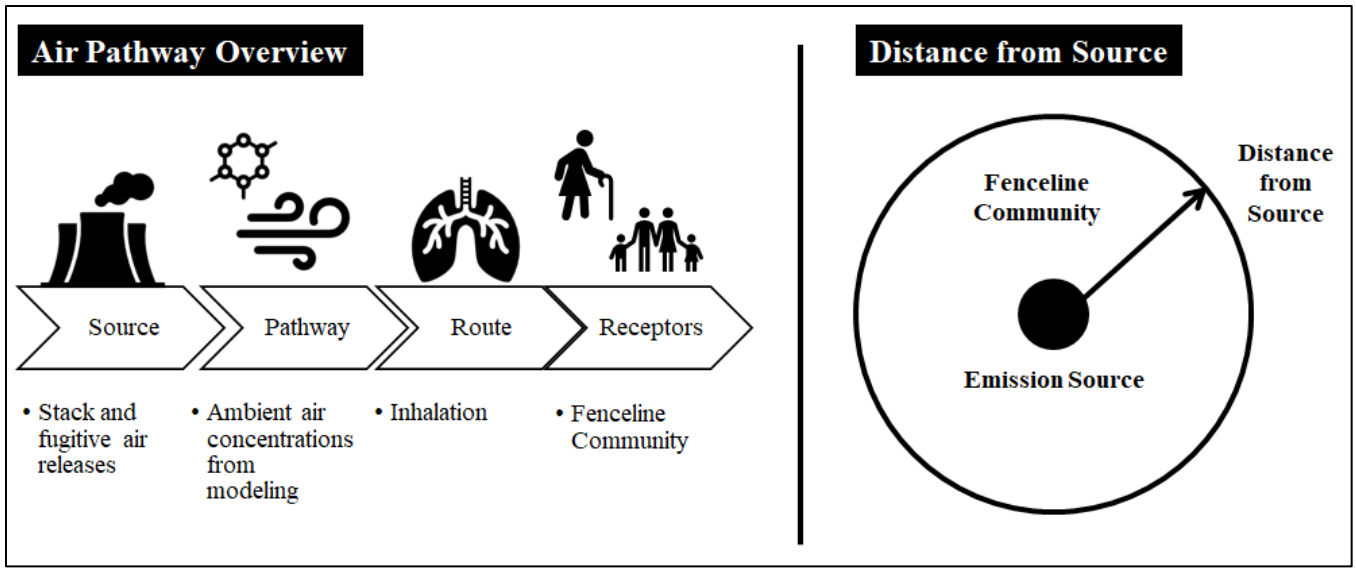
Site-specific parameters influence how partitioning occurs over time. For example, the concentration of suspended sediments, water depth, and weather patterns all influence how a chemical may partition between compartments. Physical and chemical properties of the chemical itself also influence partitioning and half-lives into environmental media. TCEP has a K_{OC} greater than 100, indicating a high potential to sorb to suspended particles in the water column and settled sediment in the benthic environment.

EPA conducted higher tier modeling with PSC-VVWM to estimate benthic concentrations (porewater and sediment).

I.3 Ambient Air Pathway

This section provides an overview of EPA's screening-level methodology for the ambient air pathway. Where reasonably available, fugitive and stack air release data from the 2019 TRI are used to quantify environmental releases. No TRI data were available for TCEP. EPA used estimated releases from a hypothetical facility using TCEP for the COUs (Figure_Apx I-2).

AERMOD is used to estimate ambient air concentrations and exposures to human populations at various distances from the emission source. Distances of up to 10,000 m are evaluated to capture potential exposures and associated risks to fenceline communities. A distance of 10,000 m is used for this methodology to capture populations nearer to releasing facilities than may otherwise be evaluated under other EPA administered laws. Additionally, professional knowledge and experience regarding exposures associated with the ambient air pathway find risks frequently occur out to approximately 1,000 m from a releasing facility and quickly decrease farther out. Although 10,000 m is an order of magnitude farther out than where risks are expected to occur, 10,000 m provides an opportunity to capture other factors related to potential exposure and associated potential risks via the ambient air pathway (like multiple facilities impacting a single individual) providing flexibility for screening-level analyses for future risk evaluations. While 10,000 m is used for the outer distance in the screening-level analysis, the methodology is not limited to 10,000 m. If risks are identified out to 10,000 m, then additional analysis using the screening-level methodology can be extended to farther distances for purposes of identifying where risks may fall below levels of concern.



Figure_Apx I-2. Overview of EPA's Screening-Level Ambient Air Pathway Methodology

I.3.1 Modeling Approach for Estimating Concentrations in Ambient Air

EPA applied a tiered approach to estimate ambient air concentrations and exposures for members of the general population that are in proximity (between 10 to 10,000 m) to emissions sources emitting the chemicals being evaluated to the ambient air. All exposures were assessed for the inhalation route only. For TCEP, multi-year release data were not available.

Step 1: Ambient Air: IIOAC Methodology
Methodology is scenario-specific. Analysis evaluates ambient air concentrations and associated exposures/risks resulting from facility-specific releases at three pre-defined distances (100, 100 to 1,000, and 1,000 m) from a releasing facility.

Step 2: Ambient Air: AERMOD Methodology
Methodology is scenario-specific. Analysis evaluates ambient air concentrations and associated exposures/risks, and deposition concentrations to land and water, resulting from facility-specific releases at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and two area distances (30 to 60 m and 100 to 1,000 m) from each releasing facility (or generic facility for alternative release estimates).

I.3.2 Ambient Air: Screening Methodology

The Ambient Air: IIOAC Methodology identifies, at a high level, if there are inhalation exposures to select human populations from a chemical undergoing risk evaluation that indicates a potential risk. This methodology inherently includes both estimates of exposures as well as estimates of risks to inform the need, or potential need, for further analysis. If findings from the Ambient Air: IIOAC Methodology indicate any potential risk (acute non-cancer, chronic non-cancer, or cancer) for a given chemical above (or below as applicable) typical Agency benchmarks, EPA generally will conduct a higher tier analysis of exposures and associated risks for that chemical. If findings from the Ambient Air: IIOAC Methodology do not indicate any potential risks for a given chemical above (or below as applicable) typical agency benchmarks, EPA would not expect a risk would be identified with higher tier analyses, but may still conduct a limited higher tier analysis at select distances to ensure potential risks are not missed (*e.g.*, at distances <100 m to ensure risks do not appear very near a facility where populations may be exposed).

Model

EPA's IIOAC model[45] was used to estimate high-end and central tendency (mean) exposures to select human populations at three pre-defined distances from a facility releasing a chemical to the ambient air (100, 100 to 1,000, and 1,000 m). IIOAC is a spreadsheet-based tool that estimates indoor and outdoor air concentrations using pre-run results from a suite of dispersion scenarios run in a variety of meteorological and land-use settings within EPA's AERMOD. As such, IIOAC is limited by the parameterizations utilized for the pre-run scenarios within AERMOD (meteorologic data, stack heights, distances, populations, etc.) and any additional or new parameterization would require revisions to the model itself. Readers can learn more about the IIOAC model, equations within the model, detailed input and output parameters, pre-defined scenarios, default values used, and supporting documentation by reviewing the IIOAC users guide (U.S. EPA, 2019f).

Releases

EPA modeled exposures for the following list of COUs/OES that had air releases. EPA ran two scenarios for each release scenario:

1. Central Tendency (50th percentile) Estimate for High Production Volume (25,000 lb) – HIGH-CT; and
2. High-End (95th percentile) Estimate for Low Production Volume (2,500 lb) – LOW-HE.

Table_Apx I-7. Ambient Air Release Inputs Utilized for Ambient Air Modeling: IIOAC and AERMOD Methodology for TCEP

Scenario Name	Production Volume	Estimate	Fugitive/ Stack	Release Duration (hours/day)	Release Frequency (days/year)	Release Amount (kg/site/day)
COM-Paints-USE	LOW	HE	Fugitive	8 h/day (8–4 p.m.)	2	1.14E02
IND-LabChem-USE	LOW	HE	Fugitive	8 h/day (8–4 p.m.)	235	2.32E−04
IND-LabChem-USE	LOW	HE	Stack	8 h/day (8–4 p.m.)	235	2.32E−04
MFG-Repack	LOW	HE	Fugitive	1 h/day (12–1 p.m.)	4	3.43E−04
MFG-Repack	LOW	HE	Stack	1 h/day (1 p.m.)	4	3.43E−04
PROC-Article-PROC-twopart-resin	LOW	HE	Fugitive	8 h/day (8–4 p.m.)	109	4.22E−04
PROC-Article-PROC-twopart-resin	LOW	HE	Stack	8 h/day (8–4 p.m.)	109	4.22E−04
PROC-Paints-INC-2-part reactive coatings	LOW	HE	Fugitive	8 h/day (8–4 p.m.)	1	7.90E−03
PROC-Paints-INC-2-part reactive coatings	LOW	HE	Stack	8 h/day (8–4 p.m.)	1	1.99E−02
PROC-Paints-INC-1-part	LOW	HE	Fugitive	8 h/day (8–4 p.m.)	4	9.60E−03
PROC-Paints-INC-1-part	LOW	HE	Stack	8 h/day (8–4 p.m.)	4	9.60E−03
PROC-Polymer-FORM-reactive-resin	LOW	HE	Fugitive	8 h/day (8–4 p.m.)	1	8.83E−03

[45] The IIOAC website is available at https://www.epa.gov/tsca-screening-tools/iioac-integrated-indoor-outdoor-air-calculator.

Scenario Name	Production Volume	Estimate	Fugitive/ Stack	Release Duration (hours/day)	Release Frequency (days/year)	Release Amount (kg/site/day)
PROC-Polymer-FORM-reactive-resin	LOW	HE	Stack	8 h/day (8–4 p.m.)	1	2.07E−02
COM-Paints-USE	HIGH	CT	Fugitive	8 h/day (8–4 p.m.)	1	1.23E01
IND-LabChem-USE	HIGH	CT	Fugitive	8 h/day (8–4 p.m.)	230	1.35E−04
IND-LabChem-USE	HIGH	CT	Stack	1 h/day (1 p.m.)	230	1.35E−04
MFG-Repack	HIGH	CT	Fugitive	1 h/day (12–1 p.m.)	39	1.88E−04
MFG-Repack	HIGH	CT	Stack	1 hr/day (1 p.m.)	39	1.88E−04
PROC-Article-PROC-twopart-resin	HIGH	CT	Fugitive	8 h/day (8–4 p.m.)	231	1.43E−04
PROC-Article-PROC-twopart-resin	HIGH	CT	Stack	8 h/day (8–4 p.m.)	231	1.43E−04
PROC-Paints-INC-2-part reactive coatings	HIGH	CT	Fugitive	8 h/day (8–4 p.m.)	4	6.77E−03
PROC-Paints-INC-2-part reactive	HIGH	CT	Stack	8 h/day (8–4 p.m.)	4	5.63E−03
PROC-Paints-INC-1-part	HIGH	CT	Fugitive	8 h/day (8–4 p.m.)	52	1.63E−03
PROC-Paints-INC-1-part	HIGH	CT	Stack	8 h/day (8–4 p.m.)	52	1.63E−03
PROC-Polymer-FORM-reactive-resin	HIGH	CT	Fugitive	8 h/day (8–4 p.m.)	6	5.36E−03
PROC-Polymer-FORM-reactive-resin	HIGH	CT	Stack	8 h/day (8–4 p.m.)	8	3.72E−03

Exposure Scenarios

EPA modeled exposure scenarios for two source types: stack (point source) and fugitive (area source) releases. These source types have different plume and dispersion characteristics accounted for differently within the IIOAC model. All COUs had stack and fugitive emissions except for the commercial use of paints and coatings (COM-Paints-USE).

The topography represents an urban or rural population density and certain boundary layer effects (like heat islands in an urban setting) that can affect turbulence and resulting concentration estimates at certain times of the day. EPA ran both urban and rural population density for all scenarios.

IIOAC includes 14 pre-defined climate regions (each with a surface station and upper-air station). Because release data used for the Ambient Air: IIOAC Methodology was not facility- or location-specific, EPA selected 1 of the 14 climate regions to represent a high-end (South [Coastal]) climate region. This selection was based on a sensitivity analysis of the average concentration and deposition predictions. This climate regions selected represents the meteorological dataset that tended to provide high-end concentration estimates relative to the other stations within IIOAC. The meteorological data within the IIOAC Model are from years 2011 to 2015 as that is the meteorological data utilized in the suite of pre-run AERMOD exposure scenarios during development of the IIOAC model (see (U.S. EPA,

2019f)). While this is older meteorological data, sensitivity analyses related to different years of meteorological data found that although the data does vary, the variation is minimal across years so the impacts to the model outcomes remain relatively unaffected.

The release scenarios were informed by the release duration and release frequency that were provided in Section 3.2.

Results

The supplemental document entitled *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: IIOAC Modeling Inputs and Results* (U.S. EPA, 2024l) presents the overall inputs and outputs for IIOAC. In IIOAC, all calculated air concentrations of fine and coarse particles are capped by an upper limit equal to the National Ambient Air Quality Standards (NAAQS) for particulate matter (PM) (U.S. EPA, 2016c). These limits are 35 and 150 $\mu g/m^3$ for fine and coarse particles (*i.e.,* the NAAQS for PM2.5 and PM10), respectively. For the IIOAC results, these limits were met for all the COU/OES releases with stack emissions. In addition, this limit reach was reached for the fine, fugitive emissions, LOW-HE release scenario for the commercial use of paints and coatings.

A further limitation of IIOAC is that it does not model for gaseous deposition. Due to the inability to model gaseous deposition, and due to the initial screening results meeting the NAAQS caps, EPA decided to run a higher tier model (AERMOD) for the ambient air pathway.

I.3.3 Ambient Air: AERMOD Methodology

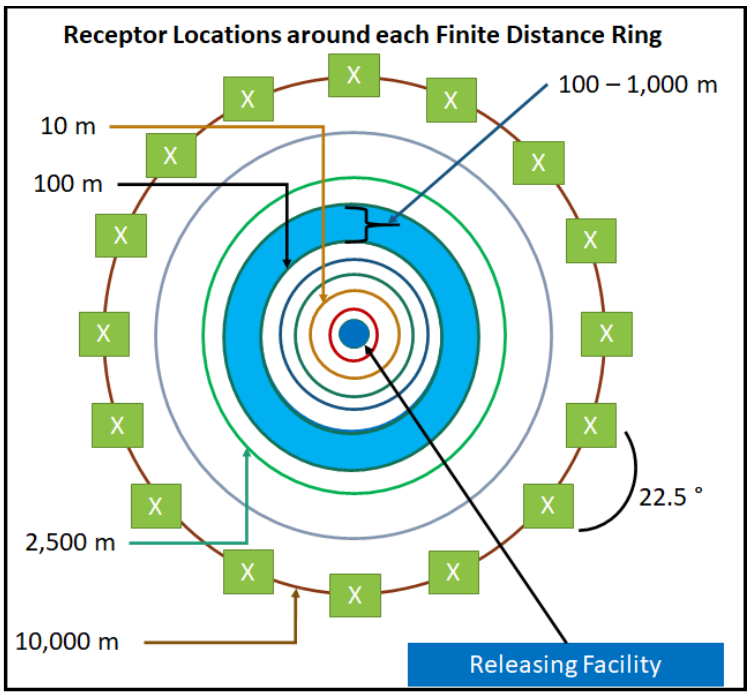
The Ambient Air: AERMOD Methodology was developed to allow EPA to conduct a higher tier analysis of releases, exposures, and associated risks to human populations around releasing facilities at multiple distances when EPA has site-specific data like reported releases, facility locations (for local meteorological data), source attribution, and other data when reasonably available. This methodology can also incorporate additional site-specific information like stack parameters (stack height, stack temperature, plume velocity, etc.), building characteristics, release patterns, different terrains, and other parameters when reasonably available. AERMOD can be performed independent of the Tier 1 modeling described above, provides a more thorough analysis, can include wet and dry deposition estimates, and allows EPA to fully characterize identified risks for chemicals undergoing risk evaluation. The application of this methodology can be applied to single or multiple years of data. TCEP had no TRI or NEI data. Thus, air releases from the release assessment were used to estimated ambient air concentrations for a single year.

Model

The Ambient Air: AERMOD Methodology for this risk evaluation utilizes AERMOD to estimate TCEP exposures to fenceline communities at user defined distances from a facility releasing TCEP. AERMOD is a steady-state Gaussian plume dispersion model that incorporates air dispersion based on planetary boundary layer turbulence structure and scaling concepts, including treatment of both surface and elevated sources and both simple and complex terrain. AERMOD can incorporate a variety of emission source characteristics, chemical deposition properties, complex terrain, and site-specific hourly meteorology to estimate air concentrations and deposition amounts at user-specified population distances and at a variety of averaging times. Readers can learn more about AERMOD, equations within the model, detailed input and output parameters, and supporting documentation by reviewing the AERMOD Users Guide (U.S. EPA, 2018).

Releases
EPA modeled exposures using the release data developed as described in Section 3.2. Release data were provided (and modeled) on a COU-by-COU basis as no facility information was available for TCEP.

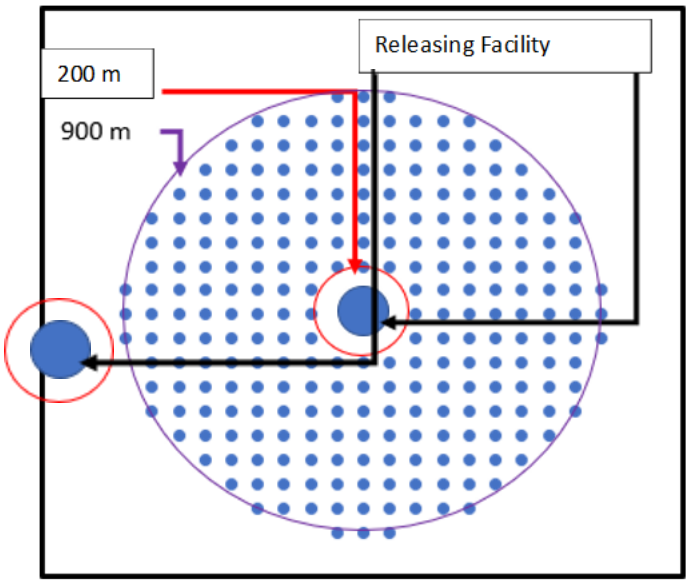
Exposure Points
The Ambient Air: AERMOD Methodology evaluated exposures to exposure points at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and two area distances (30 to 60 m, and 100 to 1,000 m) from each releasing facility (or generic facility for alternative release estimates). Exposure points for each of the eight finite distances were placed in a polar grid every 22.5 degrees around the respective distance ring. This results in a total of 16 exposure points around each finite distance ring for which exposures are modeled. Figure_Apx I-3 provides a visual depiction of the placement of exposure points around a finite distance ring. Although the visual depiction only shows exposure points locations around a single finite distance ring, the same placement of exposure points occurred for all eight finite distance rings.



Figure_Apx I-3. Modeled Exposure Points Locations for Finite Distance Rings

Exposure points for the area distance 30 to 60 m evaluated were placed in a cartesian grid at equal distances between 40 and 50 m around each releasing facility (or generic facility for alternative release estimates) were placed at 10-meter increments.

Exposure points for the area distance 100 to 1,000 m evaluated were placed in a cartesian grid at equal distances between 200 and 900 m around each releasing facility (or generic facility for alternative release estimates) were placed at 100-meter increments. This results in a total of 456 exposure points for which exposures are modeled. Figure_Apx I-4 provides a visual depiction of the placement of exposure points (each dot) around the 100 to 1,000 m area distance ring.



Figure_Apx I-4. Modeled Exposure Points for Area Distance

All exposure points were at 1.8 m above ground, as a proximation for breathing height for ambient air concentration estimations. A duplicate set of exposure points was at ground level (0 m) for deposition estimations.

Meteorological Data

Meteorological data for EPA estimated releases (where TRI or city data were not available) were modeled with the two meteorological stations utilized in the pre-screen methodology (Sioux Falls, South Dakota, for central tendency meteorology; Lake Charles, Louisiana, for higher-end meteorology). These two meteorological stations represent meteorological datasets that tended to provide high-end and central tendency concentration estimates relative to the other stations within IIOAC based on a sensitivity analysis of the average concentration and deposition predictions conducted in support of IIOAC development. These two meteorological stations are based on 5 years of meteorological data (2011 to 2015) and provide high-end and central tendency exposure concentrations utilized for risk calculation purposes to identify potential risks. The "ADJ_U*" option was not used for the 2011 to 2015 data as this could lead to model overpredictions of ambient concentrations during those particular conditions.

All processing also used automatic substitutions for small gaps in data for cloud cover and temperature.

Urban/Rural Designations

Urban/rural designations of the area around a facility are relevant when considering possible boundary layer effects on concentrations.

Air emissions taking place in an urbanized area are subject to the effects of urban heat islands, particularly at night. When sources are set as urban in AERMOD, the model will modify the boundary layer to enhance nighttime turbulence, often leading to higher nighttime air concentrations. AERMOD uses urban-area population as a proxy for the intensity of this effect.

Where TRI or city data were not available for a facility requiring modeling, there was no way for EPA to determine an appropriate urban or rural designation. Instead, EPA modeled each such facility once as

urban and once as not urban.[46] There is no recommended default urban population for AERMOD modeling, so for these facilities EPA assumed an urban population of 1 million people, which is consistent with the estimated populations used with IIOAC. Although slightly higher, the assumed urban population is close to the average of all the urban populations used for the TRI reporting facilities (which was 847,906 people).

For the TCEP risk evaluation EPA selected the urban air concentrations vs. rural air concentrations as urban concentrations were generally more conservative. Rural air concentrations may be relevant for facilities located in rural areas, and because TCEP has long range transport potential. However due to lack of site-specific information for facilities, this risk evaluation used the more conservative urban air estimates from AERMOD.

Physical Source Specifications for Alternative Release Estimates

EPA estimated releases (where TRI or city data were not available) were modeled centering all emissions on one location and using IIOAC default physical parameters. Stack emissions were modeled from a point source at 10 meters above ground from a 2-meter inside diameter, with an exit gas temperature of 300 Kelvin and an exit gas velocity of 5 m/sec (see Table 6 of the IIOAC User Guide). Fugitive emissions were modeled at 3.05 m above ground from a square area source of 10 m on a side (see Table 7 of the IIOAC User Guide).

Deposition Parameters

AERMOD was used to model daily ($g/m^2/day$) and annual ($g/m^2/year$) deposition rates from air to land and water at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and two area distances (30 to 60 m and 100 to 1,000 m) from each releasing facility (or generic facility for alternative release estimates).

AERMOD can model both gaseous and particle deposition. For TCEP, EPA considered both gaseous and particle deposition. There is conflicting literature on whether TCEP is present in particulates vs. gas. Section 3.3.1.2.1discusses these differences. Input parameter values for AERMOD deposition modeling are shown in Table_Apx I-8.

EPA provided the parameter values and settings for AERMOD deposition modeling, as indicated in Table_Apx I-8 and Table_Apx I-9. The particle deposition utilized the "METHOD_2" option in AERMOD, which is recommended when particle size distributions are not well known and when less than 10 percent of particles (by mass) are 10 µm or larger. Note that we modeled each scenario twice—once with gaseous deposition utilizing land cover of "suburban area, forested" and once with "bodies of water."

[46] Although this may be viewed as a potential double counting of these releases, EPA only utilized the highest estimated releases from a single exposure scenario from the suite of exposure scenarios modeled for surrogate/estimated facility releases as exposure estimates and for associated risk calculations.

Table_Apx I-8. Settings for Gaseous Deposition

Parameter	Value	Source
Diffusivity in air	5.67E−02 cm^2/sec	Utilizing www.envmodels.com with the chemical properties from Table 1 of Shin et al. (2014)
Diffusivity in water	2.70E−05 cm^2/sec	Page 2310 of Melnikova et al. (2019)
Henry's Law constant	2.95E−06 Pa m^3/mol	Not specified
rcl: Cuticular resistance to uptake by lipids for individual leaves	3.26E03 sec/cm	Based on vapor pressure (Vp = 8.13 Pa), empirical relationships described by Welke et al. (1998) and Kerler and Schoenherr (1988) and the values of rcl and of Vp available for numerous chemicals in Wesely et al. (2002) —together, these imply a relationship of log(rcl) = 0.4892*log(Vp in Pa) + 3.0682
Seasons	DJF = winter with no snow; MAM = transitional spring with partial green coverage or short annuals; JJA = Midsummer with lush vegetation; SON. = Autumn with unharvested cropland	Assumption
Land Cover	Option 1: Suburban areas, forested; Option 2: Bodies of water	A limited set of AERMOD tests suggested suburban-forest was a reasonable and appropriately health-protective default land-cover selection when land-cover analysis is not possible. Bodies of water typically led to the highest deposition values (ICF unpublished data).
Pa = Pascal; mol = mole; DJF = December–February; MAM = March–May; JJA = June–August; SON = September–November.		

Table_Apx I-9. Settings for Particle Deposition

Parameter	Value	Source
Mass fraction 2.5 µm or smaller	0.4 µm	Based on ranges found for phosphates in (Delumyea and Petel, 1979) and (Lee and Patterson, 1969)
Mass-mean diameter	2.2 µm	Based on a default for phosphates (source not specified)

Cuticular Resistance

The cuticular resistance (r_{cl}) value represents the resistance of a chemical to uptake by individual leaves in a vegetative canopy. For TCEP, r_{cl} was not readily available in literature. For chemicals for which the r_{cl} value is not readily available in literature, EPA developed three methods to estimate the r_{cl} value. For TCEP, EPA used r_{cl} value estimated using Method 2.

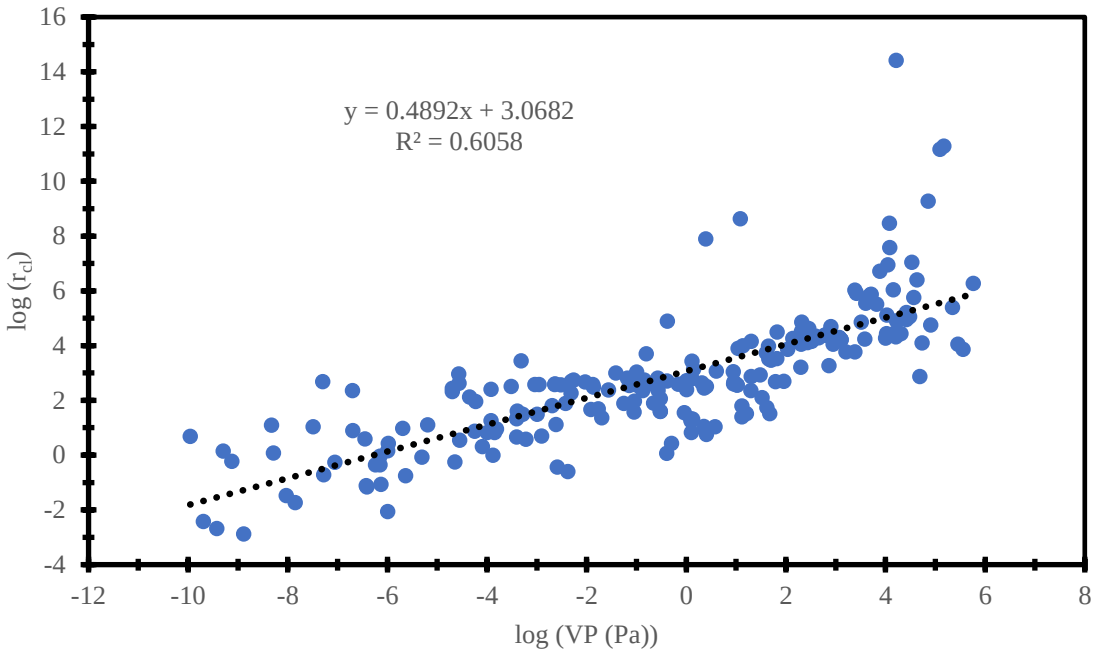
Method 1: Approximation of R_{cl} Value as a Function of Vapor Pressure: Data from the literature indicate that r_{cl} value varies as a function of the vapor pressure (VP, units of Pa) of a chemical (Welke et al., 1998; Kerler and Schoenherr, 1988). A high VP indicates that chemical has a high propensity for the vapor phase relative to the condensed phase, and therefore, would have high resistance to uptake from

the atmosphere into leaves (*i.e.*, high r_{cl}). Furthermore, Wesely et al. (2002) provides a large database of VP and r_{cl} values.

Analysis of the that study data (Welke et al., 1998) reveals that there is a linear correlation between log(VP) and log(r_{cl}), as illustrated in Figure_Apx I-5 and Equation_Apx I-2. The unit of VP is in Pascals (Pa). Linear regression yields r_{cl} as a function of VP ($R^2 = 0.606$):

Equation_Apx I-2.

$$log(r_{cl}) = 0.489\, log(VP) + 3.068$$
$$\therefore\ r_{cl} = 1170\, VP^{0.498}$$



Figure_Apx I-5. Cuticular Resistance as a Function of Vapor Pressure

Method 2: Empirical Calculation of Cuticular Resistance: Method 2 estimates r_{cl} value using various empirical equations found in literature. This method assumes the vapor pressure of the chemical at 20 to 25 °C is equal to the saturation vapor pressure. For VOCs, using the equations collectively provided under Equation_Apx I-3 (Welke *et al.*,) the polymer matrix-air partition coefficient (K_{Mxa}) can be calculated as follows:

Equation_Apx I-3.

$$\log(K_{MXa}) = 6.290 - 0.892 \log(VP)$$

Next, K_{Mxa} can be converted to the cuticular membrane-air partition coefficient, K_{Cma}:

$$K_{CMa} = 0.77\, K_{MXa}$$

Welke, *et al.* also provide an empirical relationship between the polymer matric-water partition coefficient and the air-water partition coefficient, K_{Mxw}. Recognizing the air-water partition coefficient is the Henry's law constant, HLC (unitless), yields,

$$K_{MXw} = K_{MXa}\, HLC$$

This relationship can be generalized from the polymer matrix to the cuticular membrane.

$$K_{CMw} = K_{CMa}\, HLC$$

In a separate study, Kerler and Schoenherr (1988) have developed an empirical relationship that equates K_{CMw} to the permeance coefficient for cuticular membranes, P_{CM}. However, this relationship was developed using data for non-volatile chemicals. Consequently, applying it to volatile organic chemicals introduces a large amount of uncertainty to the analysis and may not be scientifically justifiable.

$$\log(P_{CM}) = 238 \left(\frac{\log\left(K_{CMw}\right)}{MV}\right) - 12.48$$

In the above equation, MV is the molecular volume of the chemical in question, which can be calculated from the molar mass, m (units of g/mol), and density, d (units of g/cm^3):

$$MV = \frac{m}{d}$$

Finally, r_{cl} is understood to be the inverse of P_{CM}. The above relationships can be put together and simplified to yield a single equation for rcl as a function of vapor pressure, molar mass, and density:

$$r_{cl} = \left(\frac{HLC \times 1.501 \times 10^6}{VP^{0.892}}\right)^{\frac{-238\, d}{m}} \times 10^{12.48}$$

Method 3: Read-Across of Cuticular Resistance from an Analog: This method assumes that chemicals that have structural similarity, physical and chemical similarity, and exhibit similar vapor pressures will also exhibit similar r_{cl} values. Available data in literature (Wesely et al., 2002) can be used as a crosswalk for read-across determination of r_{cl}. The unknown r_{cl} value is then assumed to be equal to the r_{cl} of the analog.

Ambient Air Exposure Concentration Outputs

Hourly-average concentration outputs were provided from AERMOD for each exposure points around each distance ring (each of 16 exposure points around a finite distance ring or each exposure points within the area distance ring). Daily and period averages were then calculated from the modeled hourly data. Daily averages for the finite distance rings were calculated as arithmetic averages of all hourly data for each day modeled for each v around each ring. Daily averages for the area distance ring were calculated as the arithmetic average of the hourly data for each day modeled across all exposure points within the area distance ring. This results in the following number of daily average concentrations at each distance modeled.

1. Daily averages for EPA estimated releases: Average concentrations for each of 365 (or 366) days for each of 16 exposure points around each finite distance ring.

Period averages were calculated from all the daily averages for each exposure points for each distance ring over 1 year for facilities where releases were estimated. This results in a total of 16 period average concentration values for each finite distance ring. This is derived from either averaging the daily averages across the single year of meteorological data used for TRI reporting facilities or across the multi-year meteorological data used for EPA estimated releases.

Daily and period average Outputs were stratified by different source scenarios, such as urban/not urban setting or emission-strengths where needed. Outputs from AERMOD are provided in units of micrograms per cubic meter ($\mu g/m^3$) requiring conversion to parts per million (ppm) for purposes of calculating risk estimates for 1,4-dioxane. The following formula was used for this conversion:

Equation_Apx I-4.

$$C_{ppm} = (24.45*(C_{AERMOD})/1{,}000)/MW$$

Where:

C_{ppm}	=	Concentration (ppm)
24.45	=	Molar volume of a gas at 25 °C and 1 atmosphere pressure
C_{AERMOD}	=	Concentration from AERMOD ($\mu g/m^3$)
MW	=	Molecular weight of the chemical of interest (g/mole)

Post-processing scripts were used to extract and summarize the output concentrations for each facility, release, and exposure scenario. The following statistics for daily- and period-average concentrations were extracted or calculated from the results for each of the modeled distances (*i.e.,* each ring or grid of exposure points) and scenarios:

- Minimum;
- Maximum;
- Average;
- Standard deviation; and
- 10th, 25th, 50th, 75th, and 95th percentiles.

Table_Apx I-10. Description of Daily or Period Average and Air Concentration Statistics

Statistic	Description
Minimum	The minimum daily or period average concentration estimated at any exposure point on any day at the modeled distance.
Maximum	The maximum daily or period average concentration estimated at any exposure point on any day at the modeled distance.
Average	Arithmetic mean of all daily or period average concentrations estimated at all exposure points locations on all days at the modeled distance. This incorporates lower values (from days when the exposure point largely was upwind from the facility) and higher values (from days when the exposure point largely was downwind from the facility).
Percentiles	The daily or period average concentration estimate representing the numerical percentile value across the entire distribution of all concentrations at all exposure point locations on any day at the modeled distance. The 50th percentile represents the median of the daily or period average concentration across all concentration values for all exposure point locations on any day at the modeled distance.

Deposition from Ambient Air to Soil and Water Exposure Concentration Outputs

As previously mentioned, AERMOD was used to model daily (g/m^2/day) and annual (g/m^2/year) deposition rates (*i.e.,* deposition flux) from air releases to water body catchment areas. EPA quantitatively evaluated the risk to aquatic (pelagic and benthic) and terrestrial organisms from exposure to soil, surface water bodies and sediment via air deposition resulting from the manufacturing, processing, use, or disposal of TCEP. The following equations and parameters are based on the generic

farm pond scenario from models, such as the GENEEC2 (Generic Estimated Environmental Concentration) and EXAM (Exposure Analysis Modeling System) used by EPA. Total deposition for each media (soil, water body, and sediment) were derived using the deposition rate modeled by AERMOD to calculate media (soil, water body, and sediment) concentrations using the generic farm pond parameters for area, mixing depths, and densities, respectively:

Soil:
Equation_Apx I-5.

$$Total\ Deposition\ to\ Soil\ Catchment\ (ug) = Deposition\ flux\ x\ Area\ x\ CF$$

Where:

Deposition flux	=	Annual deposition flux to water body catchment (g/m^2)
Area	=	Area of soil catchment (area of water body catchment – area of water body) or 100,000 m^2 – 10,000 m^2 = 90,000 m^2
CF	=	g to µg; 1,000,000

$$Soil\ Catchment\ Concentration\ \left(\frac{ug}{kg}\right) = \frac{(Total\ Deposition\ to\ Soil\ Catchent)}{(Area\ of\ soil\ catchment\ x\ mix\ depth\ x\ soil\ density)}$$

Where:

Area	=	90,000 m^2
Mix depth	=	0.1 m
Soil density	=	1,700 kg/m^3

Water Body:
Equation_Apx I-6.

$$Total\ Deposition\ to\ Water\ Body\ (ug) = Deposition\ flux\ x\ Area\ x\ CF$$

Where:

Deposition flux	=	Annual deposition flux to water body catchment (g/m^2)
Area	=	Area of water body; 10,000 m^2
CF	=	g to ug; 1,000,000

$$Water\ Body\ Concentration\ \left(\frac{ug}{L}\right) = \frac{Total\ Deposition\ to\ Water\ Body}{(Area\ x\ Pond\ Depth\ x\ CF)}$$

Where:

Area	=	area of water body; 10,000 m^2
Pond depth	=	2 m
CF	=	m^3 to L; 1,000

Sediment:
Equation_Apx I-7.

$$Sediment\ Concentration\ \left(\frac{ug}{kg}\right) = \frac{Total\ Deposition\ to\ Water\ Body}{(Area\ x\ mix\ depth\ x\ sediment\ density)}$$

Where:

Area	=	Area of water body; 10,000 m^2

Mix depth	=	0.1 m
Sediment density	=	1,300 kg/m^3

AERMOD Air Concentrations and Deposition Results

Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Exposure Air Concentration Risk Calculations (U.S. EPA, 2024j) includes the ambient air concentrations, deposition concentrations (soil, water body, and sediment) for all OESs, and the associated risk calculations.

I.4 Fish Ingestion Pathway

I.4.1 Exposure Estimates

Table_Apx I-11. Adult General Population Fish Ingestion Doses by Scenario Based on a Production Volume of 2,500 lb/year, High-End Release Distribution, and Modeled Surface Water Concentrations Based on 90th Percentile Flow of Harmonic Mean

Scenario Name	SWC[a] (µg/L)	ADR[b] (mg/kg-day)		ADD[b] (mg/kg-day)				LADD[b] (mg/kg-day)			
		BAF 2,198	BAF 109	BAF 2,198		BAF 109		BAF 2,198		BAF 109	
		HE		CT	HE	CT	HE	CT	HE	CT	HE
Import and Repackaging	5.5	3.38E−03	1.68E−04	7.68E−04	3.38E−03	3.81E−05	1.68E−04	6.11E−04	2.69E−03	3.03E−05	1.33E−04
Incorporation into Paints and Coatings – 1-Part Coatings	15.4	9.39E−03	4.66E−04	2.14E−03	9.39E−03	1.06E−04	4.66E−04	1.70E−03	7.46E−03	8.42E−05	3.70E−04
Incorporation into Paints and Coatings – 2-Part Reactive Coatings	14.0	8.51E−03	4.22E−04	1.94E−03	8.51E−03	9.60E−05	4.22E−04	1.54E−03	6.77E−03	7.63E−05	3.35E−04
Use in Paints and Coatings at Job Sites	13.1	7.95E−03	3.94E−04	1.81E−03	7.95E−03	8.97E−05	3.94E−04	1.44E−03	6.32E−03	7.13E−05	3.13E−04
Formulation of TCEP Containing Reactive Resin	2.1	1.29E−03	6.37E−05	2.92E−04	1.29E−03	1.45E−05	6.37E−05	2.32E−04	1.02E−03	1.15E−05	5.07E−05
Laboratory Chemicals	77.1	4.69E−02	2.33E−03	1.07E−02	4.69E−02	5.29E−04	2.33E−03	8.48E−03	3.73E−02	4.21E−04	1.85E−03

[a] Surface water concentrations based on 90th percentile flow of harmonic mean flow conditions.
[b] ADR calculated using the 90th percentile fish ingestion rate (22.2 g/day). ADD and LADD were calculated using both the mean (CT) and 90th percentile (HE) fish ingestion rates, 5.04 g/day and 22.2 g/day respectively. An ADD based on the 90th percentile ingestion rate is the same as an ADR.

Table_Apx I-12. Adult Subsistence Fisher Doses by Scenario Based on a Production Volume of 2,500 lb/year, High-End Release Distribution, and Modeled Surface Water Concentrations Based on 90th Percentile Flow of Harmonic Mean

Scenario Name	SWC[a] (µg/L)	ADD, ADR (mg/kg-day) BAF 2,198	ADD, ADR (mg/kg-day) BAF 109	LADD (mg/kg-day) BAF 2,198	LADD (mg/kg-day) BAF 109
Import and repackaging	5.5	2.17E−02	1.08E−03	1.73E−02	8.56E−04
Incorporation into paints and coatings – 1-part reactive coatings	15.4	6.03E−02	2.99E−03	4.79E−02	2.38E−03
Incorporation into paints and coatings – 2-part reactive coatings	14.0	5.47E−02	2.71E−03	4.35E−02	2.16E−03
Use in paints and coatings at job sites	13.1	5.11E−02	2.53E−03	4.06E−02	2.01E−03
Formulation of TCEP containing reactive resin	2.1	8.26E−03	4.10E−04	6.56E−03	3.26E−04
Laboratory chemicals	77.1	3.01E−01	1.49E−02	2.40E−01	1.19E−02

[a] Surface water concentrations based on 90th percentile flow of harmonic mean flow conditions.

Table_Apx I-13. Adult Tribal Fish Ingestion Doses by Scenario Based on a Production Volume of 2,500 lb/year, High-End Release Distribution, Modeled Surface Water Concentrations Based on 90th Percentile Flow, and Two Fish Ingestion Rates

Scenario Name	SWC[a] (µg/L)	ADD, ADR (mg/kg-day) BAF 2,198	ADD, ADR (mg/kg-day) BAF 109	LADD (mg/kg-day) BAF 2,198	LADD (mg/kg-day) BAF 109
Current mean fish ingestion rate reported by the Suquamish Tribe (216 g/day)					
Import and repackaging	5.5	3.29E−02	1.63E−03	2.62E−02	1.30E−03
Incorporation into paints and coatings – 1-part reactive coatings	15.4	9.15E−02	4.54E−03	7.27E−02	3.61E−03
Incorporation into paints and coatings – 2-part reactive coatings	14.0	8.30E−02	4.11E−03	6.59E−02	3.27E−03
Use in paints and coatings at job sites	13.1	7.75E−02	3.84E−03	6.16E−02	3.06E−03
Formulation of TCEP containing reactive resin	2.1	1.25E−02	6.21E−04	9.96E−03	4.94E−04
Laboratory chemicals	77.1	4.57E−01	2.27E−02	3.63E−01	1.80E−02
Heritage fish ingestion rate (1,646 g/day)					
Import and repackaging	5.5	2.51E−01	1.24E−02	2.00E−01	9.89E−03
Incorporation into paints and coatings – 1-part reactive coatings	15.4	6.97E−01	3.46E−02	5.54E−01	2.75E−02
Incorporation into paints and coatings – 2-part reactive coatings	14.0	6.32E−01	3.14E−02	5.03E−01	2.49E−02
Use in paints and coatings at job sites	13.1	5.91E−01	2.93E−02	4.70E−01	2.33E−02
Formulation of TCEP containing reactive resin	2.1	9.55E−02	4.73E−03	7.59E−02	3.76E−03

Scenario Name	SWC[a] (µg/L)	ADD, ADR (mg/kg-day) BAF 2,198	ADD, ADR (mg/kg-day) BAF 109	LADD (mg/kg-day) BAF 2,198	LADD (mg/kg-day) BAF 109
Laboratory chemicals	77.1	3.49	1.73E−01	2.77	1.37E−01
[a] Surface water concentrations based on 90th percentile flow of harmonic mean flow conditions.					

I.4.2 Risk Estimates

Table_Apx I-14. Acute Fish Ingestion Non-cancer Risk Summary Based on 90th Percentile Flow of Harmonic Mean

COU		OES	Acute Oral Non-cancer MOEs *UFs = 30*							
			General Population		Subsistence Fishers		Tribes (Current IR)[a]		Tribes (Heritage IR)[b]	
Life Cycle Stage/Category	Subcategory		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
Manufacturing/ Import	Import	Repackaging	2,800	56,460	436	8,786	287	5,792	38	760
Processing/ Processing – Incorporation into formulation, mixture, or reaction product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	1,008	20,321	157	3,162	103	2,085	14	274
		Incorporation into paints and coatings – 2-part reactive coatings	1,112	22,414	173	3,488	114	2,300	15	302
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	1,189	23,986	185	3733	122	2,461	16	323
Commercial use	Laboratory chemicals	Use of laboratory chemicals	7,361	148,441	1,146	23,100	755	15,229	99	1,998
	Paints and coatings	Use of paints and coatings at job sites	202	4,066	31	633	21	417	3	55

[a] Current fish consumption rate at 216 g/day based on survey of Suquamish Indian Tribe in Washington (see Section 5.1.3.4.4).
[b] Heritage fish consumption rate at 1,646 g/day based on study of Kootenai Tribe in Idaho (see Section 5.1.3.4.4).

Table_Apx I-15. Chronic Fish Ingestion Non-cancer Risk Summary Based on 90th Percentile Flow of Harmonic Mean

COU		OES	Gen Pop				Subsistence Fishers[b]		Tribes (Current)[c]		Tribes (Heritage)[d]	
Life Cycle Stage/Category	Subcategory		BAF 2,198[a]		BAF 109[a]		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
			CT[e]	HE	CT[e]	HE						
Manufacturing/ Import	Import	Repackaging	3,553	808	71,639	16,293	126	2,536	83	1,672	11	219
Processing/ Processing – Incorporation into Formulation, Mixture, or Reaction Product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	1,279	291	25,784	5,864	45	913	30	602	4	79
		Incorporation into paints and coatings – 2-part reactive coatings	1,410	321	28,440	6,468	50	1,007	33	664	4	87
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	9,340	2,124	188,349	42,838	331	6,666	218	4,395	29	577
Processing/ Processing – Incorporation into Article	Aerospace equipment and products and automotive articles and replacement parts containing TCEP	Processing into 2-part resin article	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Commercial Use	Laboratory chemicals	Use of laboratory chemicals	256	58	5,160	1,173	9	183	6	120	1	16
	Paints and coatings	Use of paints and coatings at job sites	1,509	343	30,435	6,922	53	1,077	35	710	5	93

[a] GP exposure estimates based on general population fish ingestion rate of 22.2 g/day.
[b] SF exposure estimates based on subsistence fisher ingestion rate of 142.2 g/day.
[c] Current fish consumption rate at 216 g/day based on survey of Suquamish Indian Tribe in Washington (see Section 5.1.3.4.4).
[d] Heritage fish consumption rate at 1,646 g/day based on study of Kootenai Tribe in Idaho (see Section 5.1.3.4.4).
[e] Exposure estimates based on a general population mean fish ingestion rate of 5.04 g/day.

Table_Apx I-16. Lifetime Cancer Risk Summary for Fish Consumption Based on 90th Percentile Flow of Harmonic Mean

COU		OES	Lifetime Cancer Oral Risk Estimates									
Life Cycle Stage/Category	Subcategory		Adult Fish Ingestion General Population[a]				Adult Subsistence Fisher		Tribes (Current IR)		Tribes (Heritage IR)	
			BAF 2,198		BAF 109		BAF 2,198	BAF 109	BAF 2,198	BAF 109	BAF 2,198	BAF 109
			CT[b]	HE	CT[b]	HE						
Manufacturing/ Import	Import	Repackaging	1.50E−05	6.58E−05	7.42E−07	3.26E−06	4.23E−04	2.10E−05	6.41E−04	3.18E−05	4.89E−03	2.42E−04
Processing/ Processing – Incorporation into Formulation, Mixture, or Reaction Product	Flame retardant in: paint and coating manufacturing	Incorporation into paints and coatings – 1-part coatings	4.16E−05	1.83E−04	2.06E−06	9.07E−06	1.17E−03	5.83E−05	1.78E−03	8.84E−05	1.36E−02	6.74E−04
		Incorporation into paints and coatings – 2-part reactive coatings	3.77E−05	1.66E−04	1.87E−06	8.22E−06	1.07E−03	5.28E−05	1.62E−03	8.01E−05	1.23E−02	6.11E−04
	Polymers used in aerospace equipment and products	Formulation of TCEP containing reactive resin	5.69E−06	2.50E−05	2.82E−07	1.24E−06	1.61E−04	7.98E−06	2.44E−04	1.21E−05	1.86E−03	9.22E−05
Commercial Use	Laboratory chemicals	Use of laboratory chemicals	2.08E−04	9.14E−04	1.03E−05	4.53E−05	5.87E−03	2.91E−04	8.90E−03	4.42E−04	6.79E−02	3.37E−03
	Paints and coatings	Use of paints and coatings at job sites	3.52E−05	1.55E−04	1.75E−06	7.68E−06	9.95E−04	4.94E−05	1.51E−03	7.49E−05	1.15E−02	5.71E−04

[a] Cancer risk estimates for the adult general population are based on the high-end fish ingestion rate of 22.2 g/day.
[b] Exposure estimates are based on a general population mean fish ingestion rate of 5.04 g/day.

I.5 Human Milk Pathway

TCEP is predicted to passively accumulate in human milk because it has a small mass (285.48 Da), is slightly lipophilic (Log P = 1.78), and is a weak base (thus, less likely to be ionized or protein bound). The key chemical characteristics of TCEP are shown below in Table_Apx I-17. Furthermore, biomonitoring data confirmed TCEP's presence in human milk (He et al., 2018a; Kim et al., 2014; Sundkvist et al., 2010). Because of TCEP's potential to transfer to human milk and infants' susceptibility to its health effects, a quantitative analysis of the milk pathway is necessary to predict potential risks to infants. TCEP concentrations in milk were estimated based on the maternal doses using a multi-compartment physiologically based pharmacokinetic (PBPK) model identified by EPA as the best available model (Verner et al., 2009; Verner et al., 2008), hereafter referred to as the Verner Model.

Table_Apx I-17. Key Chemical Characteristics of TCEP

Key Question or Decision	Result	Chemical Property or Population	Current Value Used for Analysis	Reference(s)
Is the chemical lipophilic (log P > 1) and less than 800 Da?	Yes	Average mass	285.49 Da	CompTox Dashboard (epa.gov) | Tris(2-chloroethyl) phosphate
		Log K_{OW} (Log P) from Scoping review (Measured)	1.78	U.S. EPA (2020b)
		Log K_{OW} (Log P) from other EPA sources	1.44, 1.78, 0.54–1.4	EPA, personal communication
		Log K_{OW} (Log P, Predicted)	1.44108	CompTox Dashboard (epa.gov) | Tris(2-chloroethyl) phosphate
Is the chemical hydrophilic and less than 200 Da?	No	Average mass	285.49 Da	CompTox Dashboard (epa.gov) | Tris(2-chloroethyl) phosphate
		Water solubility (measured)	7,820 mg/L at 20 °C	U.S. EPA (2020b)
Is the chemical a weak base?	Neutral pH according to reference	pKa	TCEP does not have a pKa because it does not have any ionizable groups. It may have a pKb, but those are rarely reported.	PubChem (nih.gov) | compound/8295
		Phosphorus esters hydrolysis rates available	NR	U.S. EPA (2020b)
Passive Diffusion Prediction	Yes	Also supported by topological polar surface area (calculated) [a]	44.8 Å	PubChem (nih.gov) | compound/8295
Is there evidence of passive diffusion in peer-reviewed literature?	No	N/A	NR	N/A
Active Transport Prediction	No	N/A	NR	N/A
Is there evidence of active transport?	No	N/A	NR	N/A
Has the chemical been detected in human milk?	Yes	United States	Range: ND to 0.8 ng/mL	Ma et al. (2019)

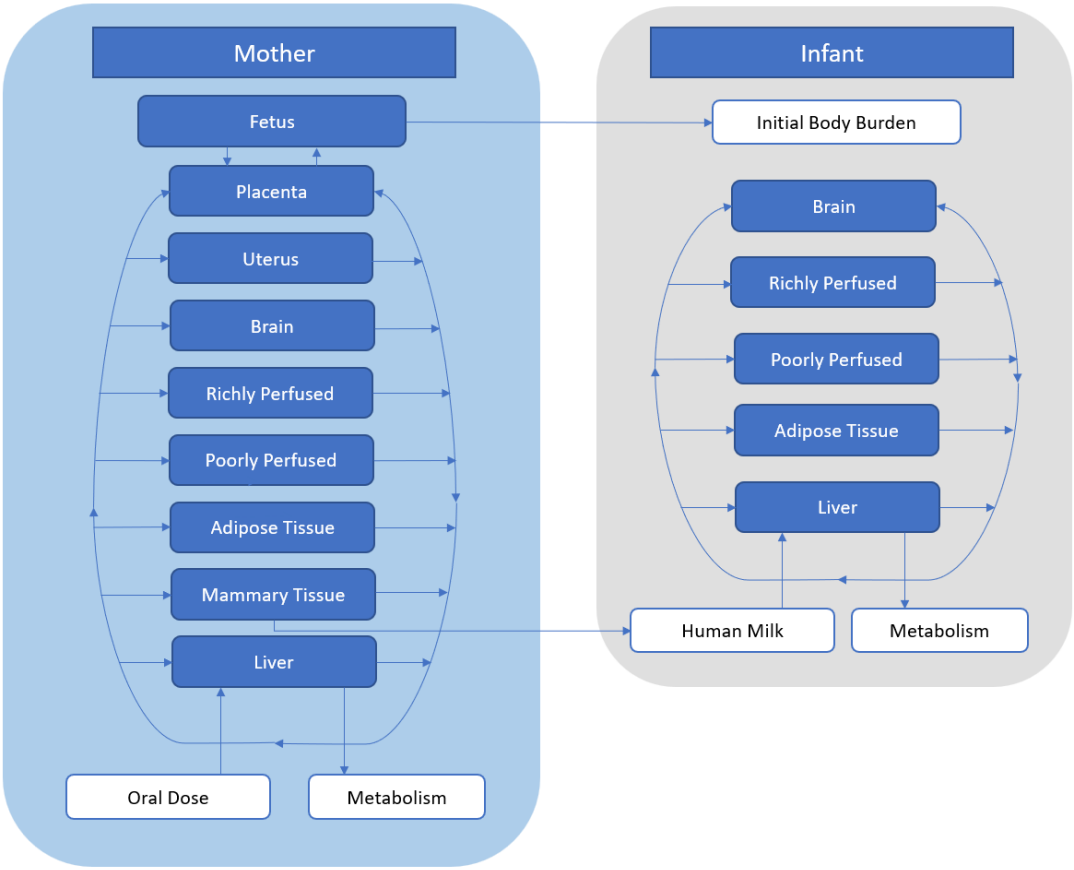
Key Question or Decision	Result	Chemical Property or Population	Current Value Used for Analysis	Reference(s)
		Women in Australia, Japan, Philippines, Vietnam, and Sweden	Range: ND to 0.47 ng/mL	He et al. (2018a)
			Central tendency: 0.14 ng/g to 42 ng/g lw	Kim et al. (2014)
			Central tendency: 4.9 ng/g lw	Sundkvist et al. (2010)
Is there a measured value for human milk partition coefficient?	No	N/A	NR	N/A

[a] The topological polar surface area of a molecule is defined as the surface sum over all polar atoms in a molecule. Membrane permeability is typically limited when polar surface area (PSA) exceeds 140 $Å^2$. (Matsson and Kihlberg, 2017).
NR – not reported.

I.5.1 Verner Model

The solubility of TCEP in the water of tissue and blood must be considered because it is slightly lipophilic (log P = 1.78). EPA identified the Verner Model, a multi-compartment PBPK model that distributes a chemical between different tissue compartments, as appropriate for evaluating infant exposure to less lipophilic chemicals like TCEP. The Verner Model accounts for every female lifestage and includes data on maternal height, weight, and age. It also integrates several concurrent physiologic events that are relevant to infant exposure from milk (*e.g.*, pre- and postpartum changes in maternal physiology, lactation, infant growth) and inputs physiological parameters, including organ volume, composition, and blood flow throughout a woman's entire life. Note that the Verner Model was validated using only data on persistent organic pollutants levels measured in mothers and infants from a Northern Québec Inuit population (Verner et al., 2009). It was not validated using data on TCEP, which were not available.

The Verner model describes the period from the beginning of the mother's life to the first year of the infant's life. As shown in Figure_Apx I-6, the model consists of a total of 14 compartments: 9 maternal (uterus, brain, richly perfused tissue, poorly perfused tissue, adipose tissue, mammary tissue, liver, placenta, and fetus) and 5 infantile (brain, richly perfused tissue, poorly perfused tissue, adipose tissue, and liver). Distribution of the chemical is driven by blood flow and the partitioning between the blood and the tissues.



Figure_Apx I-6. Compartments and Exposure Routes for Verner Model
Adapted from (Verner et al., 2009).

EPA implemented the Verner Model in the R programming language to enable running the model using modern R packages. The model was written as three systems of ordinary differential equations (ODEs), corresponding to preconception, pregnancy, and breastfeeding. The number of compartments included in preconception, pregnancy, and breastfeeding are 7, 9, and 12, respectively. In addition, the following additional updates were introduced into the R code:

- Discontinuities related to physiological terms at ages 3 and 18 were corrected.
- Mass balance tables were introduced for quality assurance evaluation.
- Brain volume parameters were added (personal communication) (Verner et al., 2008).
- A batch version of the code was developed to run several exposure scenarios consecutively.
- Graphics were elaborated to visualize three key stages: conception, birth, and lactation.
- Milk intake rates updated using EPA's *Exposure Factors Handbook* (U.S. EPA, 2011a).
- Model output expanded to include daily infant dose.
- Model computes peak and average infant dose for each age group within the first year of life.

The model inputs are shown in Table_Apx I-18 below.

Table_Apx I-18. Data Input Requirements for the Multi-compartment Model

Input	Organs or Data	Data Source(s)
Blood flow	Mother: fetus, placenta, uterus, brain, richly perfused tissue, poorly perfused tissue, adipose tissue, mammary tissue, liver, heart Infant: brain, richly perfused tissue, poorly perfused tissue, adipose tissue, liver, heart	Calculated from equations in (Verner et al., 2009; Verner et al., 2008); blood flow to brain was not published and estimated based on correspondences with author
Organ volume	Mother: fetus, placenta, uterus, brain, richly perfused tissue, poorly perfused tissue, adipose tissue, mammary tissue, liver Infant: brain, richly perfused tissue, poorly perfused tissue, adipose tissue, liver	Calculated from equations in (Verner et al., 2009; Verner et al., 2008). Changes made to skeletal muscles (part of poorly perfused tissue) and extra fat, mammary, and uterine volume at end of pregnancy to keep parameters continuous
Fraction of lipid or water in tissue	Mother: blood, brain, liver, adipose tissue, richly perfused tissue, poorly perfused tissue, mammary tissue, uterus, placenta Infant: blood, adipose tissue, liver, richly perfused tissue, poorly perfused tissue, brain	(Verner et al., 2009; Verner et al., 2008; Price et al., 2003; White et al., 1991)
Tissue:blood partition coefficients	Mother: fetus, placenta, uterus, brain, richly perfused tissue, poorly perfused tissue, adipose tissue, mammary tissue, liver Infant: brain, richly perfused tissue, poorly perfused tissue, adipose tissue, liver	Calculated from K_{OW}, fraction of lipid or water in tissue of interest, and equation in (Verner et al., 2008)
Milk:blood partition coefficient	Same formula used for tissue:blood coefficients	Calculated from K_{OW}, fraction of lipid or water in milk, and equations in (Verner et al., 2008)
Fraction of lipids in milk	Function of number of days post-partum, or age of the child	(Verner et al., 2008)
Half-life (TCEP)	17.64 hours Half-life is used to calculate a hepatic extraction ratio that varies by age because it considers blood and tissue volumes that change by age.	Half-life value estimated from a one-compartment model https://comptox.epa.gov/dashboard/chemical/adme-ivive-subtab/DTXSID5021411
Oral dose	Default/User input	Derived from occupational, consumer, and general population doses adjusted for body weight representative of women of reproductive age
Duration of breastfeeding	Default/user input	One year is the default.
Volume of breastfeeding	Default/user input	(Verner et al., 2009)

Description of Absorption, Distribution, and Excretion Parameters

The model is composed of three different stages: pre-conception, pregnancy, and breastfeeding. Each model solves the rate of change of the amount $\frac{dA_t}{dt}$ of the chemical in compartment t (tissue) as listed in,

Table_Apx I-19 where A_t denotes the amount of chemical in the tissue. These rates of change are given in terms of the blood flow to the tissue Q_t, the compartment concentration C_t, the tissue:blood partition coefficient $P_{t:b}$, and the arterial blood concentration C_a, as collectively defined under Equation_Apx I-8 below. The distribution of the chemical can be described by mass balance equations for tissue t as described in Verner et al. (2008) as

Equation_Apx I-8.

$$\frac{dA_t}{dt} = Q_t\left(C_a - \frac{C_t}{P_{t:b}}\right).$$

The arterial blood concentration is computed as

$$C_a = \sum_t \frac{Q_t C_{vt}}{Q_c},$$

with this sum being taken over all tissues. Here, Q_c denotes the cardiac blood flow and C_{vt} denotes the tissue venous blood concentration. The tissue:blood partition coefficients can be computed according to Verner et al. (2008) by

$$P_{t:b} = \frac{K_{OW} \cdot Fl_t + Fw_t}{K_{OW} \cdot Fl_b + Fw_b},$$

where K_{OW} denotes the octanol-water partition coefficient of the chemical under consideration, Fl_t and Fw_t denote the time-varying percentages of lipid and water, respectively, in compartment t. Fl_b and Fw_b denote the percentages of lipid and water, respectively, in blood.

The mass balance equation for the liver compartment has a slightly different form, as it has an absorption and metabolism term. It is given by Verner et al. (2008) as

$$\frac{dA_l}{dt} = Intake + Q_l\left(C_a - \frac{C_l}{P_{l:b}}\right) - RAM$$

where Q_l is the blood flow to the liver and RAM represents the metabolism in µg/day. To compute this, the volume of distribution is first calculated.

$$Vd_{age} = V_{blood} + P_{rp:b} \cdot V_{rp} + P_{pp:b} \cdot V_{pp} + P_{u:b} \cdot V_u + P_{f:b} \cdot V_f + P_{l:b} \cdot V_l + P_{mam:b} \cdot V_{mam} + P_{brain:b} \cdot V_{brain},$$

where V_{blood} denotes the volume of blood in the mother, computed according to the Nadler equation (Sharma and Sharma, 2023), *rp* is richly perfused, *pp* is poorly perfused, *u* is uterus, *f* is fat, *l* is liver, and *mam* is mammary tissue. This is used to compute additional parameters defined in (Verner et al., 2008). The clearance is

$$CL_{age} = \left(\frac{\ln(2)}{HL}\right) \cdot Vd_{age},$$

where HL denotes the half-life of the chemical in days. This is used to compute the quantity Eh_{age} as

$$Eh_{age} = \frac{CL_{age}}{Ql},$$

which in turn is used to compute the intrinsic clearance value

$$CLint_C = \frac{1}{Vl} \cdot \left(\frac{Eh_{age} \cdot Ql}{1 - Eh_{age}}\right).$$

From here, the hepatic extraction is computed by

$$Eh = \frac{CLint_C \cdot Vl}{CLint_C \cdot Vl + Ql},$$

which is used to compute the metabolism rate measured in µg/day.

$$RAM = Ql \cdot Eh \cdot Ca,$$

To solve this system of differential equations, organ volumes and blood flows are required for all times. The system is solved numerically using the ODE function in the deSolve package in R. The output of the model is a chemical amount and concentration in each organ compartment, as well as the TCEP concentrations in milk for the entire time period of the simulation.

I.5.2 Milk Ingestion Rates by Age

Milk ingestion rates by age are provided in Table 15-1 of the *Exposure Factors Handbook* (U.S. EPA, 2011a) and presented in Table_Apx I-19.

Table_Apx I-19. Mean and Upper Milk Ingestion Rates by Age

Age Group	Milk Ingestion (mL/kg day)	
	Mean	Upper (95th percentile)
Birth to <1 month	150	220
1 to <3 month	140	190
3 to <6 month	110	150
6 to <12 month	83	130
Birth to <1 year	104.8	152.5

I.5.3 Modeled TCEP Concentrations in Milk

Four biomonitoring studies demonstrated the presence of TCEP in human milk. One U.S. study measured a mean wet weight concentration of 0.036 ng/mL among 100 samples, with a range from 0–0.8 ng/mL (Ma et al., 2019). Ma et al. (2019) set non-detects to 0 and concentrations below the limit of quantification (LOQ) to half the value of the LOQ. Among non-U.S. studies, only one measured wet weight concentrations from three milk samples collected in Australia, and concentrations ranged from non-detect (<0.13 ng/mL) to 0.47 ng/mL (He et al., 2018a). He et al. (2018a) assigned half the value of the method detection limit for all non-detects. Two other non-US studies measured lipid weight concentrations that ranged from non-detect to 512 ng/g (average 0.14–42 ng/g) in (Kim et al., 2014) and 2.1 to 8.2 ng/g (median 4.9 ng/g) in (Sundkvist et al., 2010). These two studies' treatment of non-detects are not discussed because they report lipid weight concentrations that cannot be compared to the outputs from the Verner Model. The Verner Model estimates wet weight concentrations, thus modeled

concentrations can only be compared with measured concentrations by (Ma et al., 2019) and (He et al., 2018a). The range of the wet weight concentrations across each COU/OES for each maternal group is presented in Table_Apx I-17. In general, the lower and upper bound of the modeled concentrations are three orders of magnitudes below and five orders of magnitudes above measured concentrations, respectively.

Table_Apx I-20. Comparison of the Range of Measured and Modeled TCEP Concentrations in Human Milk

Maternal Group	Modeled Concentrations (mg/mL)	Measured Concentrations (mg/mL)
Consumer	3.96E−08 to 2.62E−04	0 to 8E−07 (based on 100 samples collected in U.S.) <1.3E−07 to 4.7E−07 (based on three samples collected in Australia)
Occupational	1.96E−10 to 1.13E−03	
General population	1.83E−10 to 5.22E−04	
Tribal populations	1.79E−06 to 2.93E−02	

I.5.4 Infant Exposure Estimate

Table_Apx I-21. Average Infant Doses via Human Milk Exposure from Maternal Consumer Use Scenarios

COU Subcategory and Consumer Exposure Scenarios	Maternal Dose (mg/kg-day) [a] [b]	Milk Intake Rate Type	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Fabric textile, leather products not covered elsewhere (carpet back coating)	6.12E−03	Mean	1.01E−04	1.03E−04	9.23E−05	8.39E−05	9.06E−05
Fabric textile, leather products not covered elsewhere (textile for children's play structures)	9.02E−02	Mean	1.49E−03	1.52E−03	1.36E−03	1.24E−03	1.33E−03
Building/construction materials not covered elsewhere (roofing insulation)	1.74	Mean	2.87E−02	2.92E−02	2.62E−02	2.38E−02	2.57E−02
Building/construction materials not covered elsewhere (acoustic ceiling)	1.40E−01	Mean	2.31E−03	2.35E−03	2.10E−03	1.91E−03	2.07E−03
Foam seating and bedding product (foam automobile)	6.85E−03	Mean	1.13E−04	1.15E−04	1.03E−04	9.39E−05	1.01E−04
Foam seating and bedding product (foam living room)	1.53E−02	Mean	2.53E−04	2.58E−04	2.31E−04	2.10E−04	2.27E−04
Foam seating and bedding product (mattress)	1.95E−03	Mean	3.22E−05	3.27E−05	2.93E−05	2.67E−05	2.88E−05
Foam seating and bedding product (foam – other – toy block)	1.06E−03	Mean	1.75E−05	1.78E−05	1.59E−05	1.45E−05	1.56E−05
Building/construction materials – wood and engineered wood products (wood flooring)	1.51	Mean	2.50E−02	2.55E−02	2.28E−02	2.07E−02	2.24E−02
Building/construction materials – wood and engineered wood products (wooden TV stand)	1.01E−01	Mean	1.68E−03	1.70E−03	1.53E−03	1.39E−03	1.50E−03
Fabric textile, leather products not covered elsewhere (carpet back coating)	6.12E−03	Upper	1.48E−04	1.39E−04	1.25E−04	1.30E−04	1.32E−04
Fabric textile, leather products not covered elsewhere (textile for children's play structures)	9.02E−02	Upper	2.18E−03	2.05E−03	1.85E−03	1.92E−03	1.95E−03
Building/construction materials not covered elsewhere (roofing insulation)	1.74	Upper	4.20E−02	3.95E−02	3.56E−02	3.70E−02	3.75E−02
Building/construction materials not covered elsewhere (acoustic ceiling)	1.40E−01	Upper	3.38E−03	3.18E−03	2.86E−03	2.98E−03	3.02E−03

COU Subcategory and Consumer Exposure Scenarios	Maternal Dose (mg/kg-day) [a] [b]	Milk Intake Rate Type	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Foam seating and bedding product (foam automobile)	6.85E−03	Upper	1.66E−04	1.56E−04	1.40E−04	1.46E−04	1.48E−04
Foam seating and bedding product (foam living room)	1.53E−02	Upper	3.71E−04	3.49E−04	3.14E−04	3.27E−04	3.31E−04
Foam seating and bedding product (mattress)	1.95E−03	Upper	4.71E−05	4.43E−05	3.99E−05	4.15E−05	4.20E−05
Foam seating and bedding product (foam – other – toy block)	1.06E−03	Upper	2.56E−05	2.41E−05	2.17E−05	2.25E−05	2.28E−05
Building/construction materials – wood and engineered wood products (wood flooring)	1.51	Upper	3.67E−02	3.45E−02	3.10E−02	3.23E−02	3.27E−02
Building/construction materials – wood and engineered wood products (wooden TV stand)	1.01E−01	Upper	2.45E−03	2.31E−03	2.07E−03	2.16E−03	2.19E−03

[a] Consumer maternal doses were combined across oral, dermal, and inhalation routes. For inhalation, CEM 3.2 already calculates a dose in mg/kg-day, as shown in Section 5.1.2.3 for consumers.
[b] Chronic maternal doses are the most relevant durations for building and construction materials, fabric and textile products, and foam seating and bedding products because they are typically used over a longer time frame than other types of consumer products with direct applications (*e.g.*, household cleaners, solvents).

Table_Apx I-22. Average Infant Doses from Maternal Workers Based on Mean Milk Intake Rate

OES	Route	Maternal Exposure Duration	Maternal Dose (mg/kg-day)	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Import and repackaging	Dermal, Inhalation (High-end)	Chronic	1.57E−01	2.59E−03	2.63E−03	2.36E−03	2.15E−03	2.32E−03
Incorporation into paints and coatings – 1-part coatings		Chronic	8.38E−01	1.39E−02	1.41E−02	1.26E−02	1.15E−02	1.24E−02
Incorporation into paints and coatings – 2-part reactive coatings		Chronic	8.53E−02	1.41E−03	1.43E−03	1.29E−03	1.17E−03	1.26E−03
Processing – Formulation of TCEP into 2-part reactive resins		Chronic	1.73E−01	2.86E−03	2.90E−03	2.60E−03	2.37E−03	2.56E−03
Processing – Processing into 2-part resin article		Chronic	2.18	3.60E−02	3.66E−02	3.28E−02	2.98E−02	3.22E−02
Processing – Recycling electronics		Chronic	1.37E−04	2.26E−06	2.30E−06	2.06E−06	1.87E−06	2.03E−06
Commercial use – Paints and coatings – Spray (1-part, 250-day application)		Chronic	1.45	2.40E−02	2.44E−02	2.18E−02	1.99E−02	2.14E−02
Commercial use – Paints and coatings – Spray (2-part reactive, 250-day application)		Chronic	7.25	1.20E−01	1.22E−01	1.09E−01	9.93E−02	1.07E−01
Laboratory chemicals		Chronic	4.35	7.20E−02	7.32E−02	6.56E−02	5.96E−02	6.44E−02
Industrial/Commercial use – Installation of articles, chronic, inhalation	Inhalation (High-end)	Chronic	1.35E−06	2.23E−08	2.27E−08	2.04E−08	1.85E−08	2.00E−08
Import and repackaging	Dermal, Inhalation (High-end)	Intermediate	1.86	3.07E−02	3.12E−02	2.80E−02	2.55E−02	2.75E−02
Incorporation into paints and coatings – 1-part coatings		Intermediate	5.84	9.65E−02	9.81E−02	8.79E−02	8.00E−02	8.64E−02
Incorporation into paints and coatings – 2-part reactive coatings		Intermediate	5.74E−01	9.50E−03	9.65E−03	8.65E−03	7.87E−03	8.50E−03
Processing – Formulation of TCEP into 2-part reactive resins		Intermediate	1.63	2.70E−02	2.75E−02	2.46E−02	2.24E−02	2.42E−02
Processing – Processing into 2-part resin article		Intermediate	2.33	3.86E−02	3.92E−02	3.51E−02	3.19E−02	3.45E−02
Processing – Recycling electronics		Intermediate	1.47E−04	2.42E−06	2.46E−06	2.21E−06	2.01E−06	2.17E−06
Commercial use – Paints and coatings – Spray (1-part, 250-day application)		Intermediate	1.55	2.57E−02	2.61E−02	2.34E−02	2.13E−02	2.30E−02
Commercial use – Paints and coatings – Spray (2-part reactive, 250-day application)		Intermediate	7.76	1.28E−01	1.30E−01	1.17E−01	1.06E−01	1.15E−01
Laboratory chemicals		Intermediate	5.83	9.63E−02	9.79E−02	8.78E−02	7.98E−02	8.62E−02
Industrial/Commercial use – Installation of aerospace products, chronic, inhalation	Inhalation (High-end)	Intermediate	1.45E−06	2.39E−08	2.43E−08	2.18E−08	1.98E−08	2.14E−08

Table_Apx I-23. Average Infant Doses from Maternal Workers Based on Upper Milk Intake Rate

OES	Route	Maternal Exposure Duration	Maternal Dose (mg/kg-day)	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Import and repackaging	Dermal, Inhalation (High-end)	Chronic	1.57E−01	3.79E−03	3.56E−03	3.21E−03	3.34E−03	3.38E−03
Incorporation into paints and coatings – 1-part coatings		Chronic	8.38E−01	2.03E−02	1.91E−02	1.72E−02	1.79E−02	1.81E−02
Incorporation into paints and coatings – 2-part reactive coatings		Chronic	8.53E−02	2.06E−03	1.94E−03	1.75E−03	1.82E−03	1.84E−03
Processing – Formulation of TCEP into 2-part reactive resins		Chronic	1.73E−01	4.18E−03	3.93E−03	3.54E−03	3.68E−03	3.73E−03
Processing – Processing into 2-part resin article		Chronic	2.18	5.27E−02	4.96E−02	4.46E−02	4.64E−02	4.70E−02
Processing – Recycling electronics		Chronic	1.37E−04	3.31E−06	3.11E−06	2.80E−06	2.92E−06	2.95E−06
Commercial use – Paints and coatings – Spray (1-part, 250-day application)		Chronic	1.45	3.51E−02	3.30E−02	2.97E−02	3.09E−02	3.13E−02
Commercial use – Paints and coatings – Spray (2-part reactive, 250-day application)		Chronic	7.25	1.75E−01	1.65E−01	1.48E−01	1.54E−01	1.56E−01
Laboratory chemicals		Chronic	4.35	1.05E−01	9.91E−02	8.92E−02	9.28E−02	9.40E−02
Industrial/Commercial use – Installation of articles, chronic, inhalation	Inhalation (High-end)	Chronic	1.35E−06	3.27E−08	3.08E−08	2.77E−08	2.88E−08	2.92E−08
Import and repackaging	Dermal, Inhalation (High-end)	Intermediate	1.86	4.50E−02	4.23E−02	3.81E−02	3.96E−02	4.62E−02
Incorporation into paints and coatings – 1-part coatings		Intermediate	5.84	1.41E−01	1.33E−01	1.20E−01	1.24E−01	1.45E−01
Incorporation into paints and coatings - 2-part reactive coatings		Intermediate	5.74E−01	1.39E−02	1.31E−02	1.18E−02	1.22E−02	1.43E−02
Processing - Formulation of TCEP into 2-part reactive resins		Intermediate	1.63	3.95E−02	3.72E−02	3.35E−02	3.48E−02	4.07E−02
Processing – Processing into 2-part resin article		Intermediate	2.33	5.65E−02	5.31E−02	4.78E−02	4.97E−02	5.80E−02
Processing – Recycling electronics		Intermediate	1.47E−04	3.55E−06	3.33E−06	3.00E−06	3.12E−06	3.65E−06
Commercial use – Paints and coatings – Spray (1-part, 250-day application)		Intermediate	1.55	3.76E−02	3.53E−02	3.18E−02	3.31E−02	3.86E−02
Commercial use – Paints and coatings – Spray (2-part reactive, 250-day application)		Intermediate	7.76	1.88E−01	1.77E−01	1.59E−01	1.65E−01	1.93E−01
Laboratory chemicals		Intermediate	5.83	1.41E−01	1.33E−01	1.19E−01	1.24E−01	1.45E−01
Industrial/commercial use – Installation of articles, chronic, inhalation	Inhalation (High-end)	Intermediate	1.45E−06	3.50E−08	3.29E−08	2.96E−08	3.08E−08	3.60E−08

Table_Apx I-24. Average Infant Doses via Human Milk Exposure from Maternal General Population Oral Exposures Based on Mean Milk Intake Rate

COUs/OES	Route	Maternal Dose (mg/kg-day)	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Import and repackaging	Gen Pop Fish Ingestion, High BAF	6.37E−01	1.05E−02	1.07E−02	9.60E−03	8.73E−03	9.43E−03
Incorporation into paints and coatings – 1-part coatings	Gen Pop Fish Ingestion, High BAF	2.82	4.66E−02	4.74E−02	4.25E−02	3.86E−02	4.17E−02
Incorporation into paints and coatings – 2-part reactive coatings	Gen Pop Fish Ingestion, High BAF	2.56	4.23E−02	4.30E−02	3.86E−02	3.51E−02	3.79E−02
Use in paints and coatings at job sites	Gen Pop Fish Ingestion, High BAF	1.50	2.48E−02	2.52E−02	2.26E−02	2.05E−02	2.22E−02
Formulation of TCEP containing reactive resin	Gen Pop Fish Ingestion, High BAF	3.58	5.92E−02	6.02E−02	5.39E−02	4.90E−02	5.30E−02
Laboratory chemicals	Gen Pop Fish Ingestion, High BAF	2.55E−02	4.22E−04	4.29E−04	3.84E−04	3.49E−04	3.77E−04
Import and repackaging	Gen Pop Fish Ingestion, Low BAF	3.16E−02	5.23E−04	5.31E−04	4.76E−04	4.33E−04	4.68E−04
Incorporation into paints and coatings – 1-part coatings	Gen Pop Fish Ingestion, Low BAF	1.40E−01	2.32E−03	2.35E−03	2.11E−03	1.92E−03	2.07E−03
Incorporation into paints and coatings - 2-part reactive coatings	Gen Pop Fish Ingestion, Low BAF	1.27E−01	2.10E−03	2.13E−03	1.91E−03	1.74E−03	1.88E−03
Use in paints and coatings at job sites	Gen Pop Fish Ingestion, Low BAF	7.44E−02	1.23E−03	1.25E−03	1.12E−03	1.02E−03	1.10E−03
Formulation of TCEP containing reactive resin	Gen Pop Fish Ingestion, Low BAF	1.78E−01	2.94E−03	2.99E−03	2.68E−03	2.44E−03	2.63E−03
Laboratory chemicals	Gen Pop Fish Ingestion, Low BAF	1.27E−03	2.10E−05	2.13E−05	1.91E−05	1.74E−05	1.88E−05
Import and repackaging	Undiluted Drinking Water	3.16E−05	5.23E−07	5.31E−07	4.76E−07	4.33E−07	4.68E−07
Incorporation into paints and coatings – 1-part coatings	Undiluted Drinking Water	1.40E−04	2.31E−06	2.35E−06	2.11E−06	1.92E−06	2.07E−06
Incorporation into paints and coatings – 2-part reactive coatings	Undiluted Drinking Water	1.26E−04	2.08E−06	2.12E−06	1.90E−06	1.73E−06	1.86E−06
Use in paints and coatings at job sites	Undiluted Drinking Water	7.42E−05	1.23E−06	1.25E−06	1.12E−06	1.02E−06	1.10E−06
Formulation of TCEP containing reactive resin	Undiluted Drinking Water	1.77E−04	2.93E−06	2.97E−06	2.67E−06	2.42E−06	2.62E−06
Laboratory chemicals	Undiluted Drinking Water	1.26E−06	2.08E−08	2.12E−08	1.90E−08	1.73E−08	1.86E−08

Table_Apx I-25. Average Infant Doses via Human Milk Exposure from Maternal General Population Oral Exposures Based on Upper Milk Intake Rate

COUs/OES	Route	Maternal Dose (mg/kg-day)	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Import and repackaging	Gen Pop Fish Ingestion, High BAF	6.37E−01	1.54E−02	1.45E−02	1.30E−02	1.36E−02	1.38E−02
Incorporation into paints and coatings – 1-part coatings	Gen Pop Fish Ingestion, High BAF	2.82	6.83E−02	6.42E−02	5.78E−02	6.01E−02	6.09E−02
Incorporation into paints and coatings – 2-part reactive coatings	Gen Pop Fish Ingestion, High BAF	2.56	6.20E−02	5.83E−02	5.24E−02	5.45E−02	5.53E−02
Use in paints and coatings at job sites	Gen Pop Fish Ingestion, High BAF	1.50	3.63E−02	3.41E−02	3.07E−02	3.20E−02	3.24E−02
Formulation of TCEP containing reactive resin	Gen Pop Fish Ingestion, High BAF	3.58	8.66E−02	8.15E−02	7.33E−02	7.63E−02	7.73E−02
Laboratory chemicals	Gen Pop Fish Ingestion, High BAF	2.55E−02	6.17E−04	5.80E−04	5.22E−04	5.43E−04	5.51E−04
Import and repackaging	Gen Pop Fish Ingestion, Low BAF	3.16E−02	7.65E−04	7.19E−04	6.47E−04	6.73E−04	6.82E−04
Incorporation into paints and coatings – 1-part coatings	Gen Pop Fish Ingestion, Low BAF	1.40E−01	3.39E−03	3.19E−03	2.87E−03	2.98E−03	3.02E−03
Incorporation into paints and coatings – 2-part reactive coatings	Gen Pop Fish Ingestion, Low BAF	1.27E−01	3.07E−03	2.89E−03	2.60E−03	2.71E−03	2.74E−03
Use in paints and coatings at job sites	Gen Pop Fish Ingestion, Low BAF	7.44E−02	1.80E−03	1.69E−03	1.52E−03	1.59E−03	1.61E−03
Formulation of TCEP containing reactive resin	Gen Pop Fish Ingestion, Low BAF	1.78E−01	4.31E−03	4.05E−03	3.65E−03	3.79E−03	3.84E−03
Laboratory chemicals	Gen Pop Fish Ingestion, Low BAF	1.27E−03	3.07E−05	2.89E−05	2.60E−05	2.71E−05	2.74E−05
Import and repackaging	Undiluted Drinking Water	3.16E−05	7.65E−07	7.19E−07	6.47E−07	6.73E−07	6.82E−07
Incorporation into paints and coatings –1-part coatings	Undiluted Drinking Water	1.40E−04	3.39E−06	3.19E−06	2.87E−06	2.98E−06	3.02E−06
Incorporation into paints and coatings – 2-part reactive coatings	Undiluted Drinking Water	1.26E−04	3.05E−06	2.87E−06	2.58E−06	2.68E−06	2.72E−06
Use in paints and coatings at job sites	Undiluted Drinking Water	7.42E−05	1.80E−06	1.69E−06	1.52E−06	1.58E−06	1.60E−06
Formulation of TCEP containing reactive resin	Undiluted Drinking Water	1.77E−04	4.28E−06	4.03E−06	3.63E−06	3.77E−06	3.82E−06
Laboratory chemicals	Undiluted Drinking Water	1.26E−06	3.05E−08	2.87E−08	2.58E−08	2.68E−08	2.72E−08

Table_Apx I-26. Average Infant Doses via Human Milk Exposure from Maternal Tribal Fish Ingestion Based on Mean Milk Intake Rate

COUs/OES	Route	Maternal Dose (mg/kg-day)	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Import and repackaging	Current IR, High BAF	6.21	1.03E−01	1.04E−01	9.36E−02	8.51E−02	9.19E−02
Incorporation into paints and coatings – 1-part coatings	Current IR, High BAF	2.75E01	4.55E−01	4.62E−01	4.14E−01	3.77E−01	4.07E−01
Incorporation into paints and coatings – 2-part reactive coatings	Current IR, High BAF	2.49E01	4.12E−01	4.18E−01	3.75E−01	3.41E−01	3.68E−01
Use in paints and coatings at job sites	Current IR, High BAF	1.46E01	2.41E−01	2.45E−01	2.20E−01	2.00E−01	2.16E−01
Formulation of TCEP containing reactive resin	Current IR, High BAF	3.49E01	5.77E−01	5.87E−01	5.26E−01	4.78E−01	5.16E−01
Laboratory chemicals	Current IR, High BAF	2.49E−01	4.12E−03	4.18E−03	3.75E−03	3.41E−03	3.68E−03
Import and repackaging	Current IR, Low BAF	3.08E−01	5.09E−03	5.18E−03	4.64E−03	4.22E−03	4.56E−03
Incorporation into paints and coatings – 1-part coatings	Current IR, Low BAF	1.36	2.25E−02	2.29E−02	2.05E−02	1.86E−02	2.01E−02
Incorporation into paints and coatings – 2-part reactive coatings	Current IR, Low BAF	1.24	2.05E−02	2.08E−02	1.87E−02	1.70E−02	1.83E−02
Use in paints and coatings at job sites	Current IR, Low BAF	7.25E−01	1.20E−02	1.22E−02	1.09E−02	9.93E−03	1.07E−02
Formulation of TCEP containing reactive resin	Current IR, Low BAF	1.73	2.86E−02	2.91E−02	2.61E−02	2.37E−02	2.56E−02
Laboratory chemicals	Current IR, Low BAF	1.23E−02	2.03E−04	2.07E−04	1.85E−04	1.68E−04	1.82E−04
Import and repackaging	Heritage IR, High BAF	4.73E01	7.82E−01	7.95E−01	7.13E−01	6.48E−01	7.00E−01
Incorporation into paints and coatings –1-part coatings	Heritage IR, High BAF	2.10E02	3.47	3.53	3.16	2.88	3.11
Incorporation into paints and coatings – 2-part reactive coatings	Heritage IR, High BAF	1.91E02	3.16	3.21	2.88	2.62	2.83
Use in paints and coatings at job sites	Heritage IR, High BAF	1.11E02	1.84	1.87	1.67	1.52	1.64
Formulation of TCEP containing reactive resin	Heritage IR, High BAF	2.66E02	4.40	4.47	4.01	3.64	3.94

COUs/OES	Route	Maternal Dose (mg/kg-day)	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Laboratory chemicals	Heritage IR, High BAF	1.89	3.13E−02	3.18E−02	2.85E−02	2.59E−02	2.80E−02
Import and repackaging	Heritage IR, Low BAF	2.34	3.87E−02	3.93E−02	3.53E−02	3.21E−02	3.46E−02
Incorporation into paints and coatings –1-part coatings	Heritage IR, Low BAF	1.04E01	1.72E−01	1.75E−01	1.57E−01	1.42E−01	1.54E−01
Incorporation into paints and coatings – 2-part reactive coatings	Heritage IR, Low BAF	9.43	1.56E−01	1.58E−01	1.42E−01	1.29E−01	1.40E−01
Use in paints and coatings at job sites	Heritage IR, Low BAF	5.52	9.13E−02	9.28E−02	8.32E−02	7.56E−02	8.17E−02
Formulation of TCEP containing reactive resin	Heritage IR, Low BAF	1.32E01	2.18E−01	2.22E−01	1.99E−01	1.81E−01	1.95E−01
Laboratory chemicals	Heritage IR, Low BAF	9.41E−02	1.56E−03	1.58E−03	1.42E−03	1.29E−03	1.39E−03

Table_Apx I-27. Average Infant Doses via Human Milk Exposure from Maternal Tribal Fish Ingestion Based on Upper Milk Intake Rate

COUs/OES	Route	Maternal Dose (mg/kg-day)	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Import and repackaging	Current IR, High BAF	6.21	1.50E−01	1.41E−01	1.27E−01	1.32E−01	1.34E−01
Incorporation into paints and coatings – 1-part coatings	Current IR, High BAF	2.75E01	6.66E−01	6.26E−01	5.63E−01	5.86E−01	5.94E−01
Incorporation into paints and coatings – 2-part reactive coatings	Current IR, High BAF	2.49E01	6.03E−01	5.67E−01	5.10E−01	5.31E−01	5.38E−01
Use in paints and coatings at job sites	Current IR, High BAF	1.46E01	3.53E−01	3.32E−01	2.99E−01	3.11E−01	3.15E−01
Formulation of TCEP containing reactive resin	Current IR, High BAF	3.49E01	8.45E−01	7.94E−01	7.15E−01	7.44E−01	7.53E−01
Laboratory chemicals	Current IR, High BAF	2.49E−01	6.03E−03	5.67E−03	5.10E−03	5.31E−03	5.38E−03
Import and repackaging	Current IR, Low BAF	3.08E−01	7.45E−03	7.01E−03	6.31E−03	6.56E−03	6.65E−03
Incorporation into paints and coatings – 1-part coatings	Current IR, Low BAF	1.36	3.29E−02	3.10E−02	2.79E−02	2.90E−02	2.94E−02
Incorporation into paints and coatings – 2-part reactive coatings	Current IR, Low BAF	1.24	3.00E−02	2.82E−02	2.54E−02	2.64E−02	2.68E−02

COUs/OES	Route	Maternal Dose (mg/kg-day)	Birth to <1 Month (mg/kg-day)	1 to <3 Month (mg/kg-day)	3 to <6 Month (mg/kg-day)	6 to 12 Month (mg/kg-day)	Birth to 12 Month (mg/kg-day)
Use in paints and coatings at job sites	Current IR, Low BAF	7.25E−01	1.75E−02	1.65E−02	1.48E−02	1.54E−02	1.57E−02
Formulation of TCEP containing reactive resin	Current IR, Low BAF	1.73	4.19E−02	3.94E−02	3.54E−02	3.69E−02	3.73E−02
Laboratory chemicals	Current IR, Low BAF	1.23E−02	2.98E−04	2.80E−04	2.52E−04	2.62E−04	2.66E−04
Import and repackaging	Heritage IR, High BAF	4.73E01	1.14	1.08	9.69E−01	1.01	1.02
Incorporation into paints and coatings –1-part coatings	Heritage IR, High BAF	2.10E02	5.08	4.78	4.30	4.47	4.53
Incorporation into paints and coatings – 2-part reactive coatings	Heritage IR, High BAF	1.91E02	4.62	4.35	3.91	4.07	4.12
Use in paints and coatings at job sites	Heritage IR, High BAF	1.11E02	2.69	2.53	2.27	2.37	2.40
Formulation of TCEP containing reactive resin	Heritage IR, High BAF	2.66E02	6.44	6.05	5.45	5.67	5.74
Laboratory chemicals	Heritage IR, High BAF	1.89	4.57E−02	4.30E−02	3.87E−02	4.03E−02	4.08E−02
Import and repackaging	Heritage IR, Low BAF	2.34	5.66E−02	5.33E−02	4.79E−02	4.99E−02	5.05E−02
Incorporation into paints and coatings –1-part coatings	Heritage IR, Low BAF	1.04E01	2.52E−01	2.37E−01	2.13E−01	2.22E−01	2.25E−01
Incorporation into paints and coatings – 2-part reactive coatings	Heritage IR, Low BAF	9.43	2.28E−01	2.15E−01	1.93E−01	2.01E−01	2.04E−01
Use in paints and coatings at job sites	Heritage IR, Low BAF	5.52	1.34E−01	1.26E−01	1.13E−01	1.18E−01	1.19E−01
Formulation of TCEP containing reactive resin	Heritage IR, Low BAF	1.32E01	3.19E−01	3.00E−01	2.70E−01	2.81E−01	2.85E−01
Laboratory chemicals	Heritage IR, Low BAF	9.41E−02	2.28E−03	2.14E−03	1.93E−03	2.00E−03	2.03E−03

I.5.5 Infant Risk Estimates

Table_Apx I-28. Infant Risks via Human Milk Exposure from Maternal Consumer Use Scenarios

COU Subcategory and Consumer Exposure Scenarios	Milk Intake Rate Type	Intermediate	Chronic	Cancer
Fabric textile, leather products not covered elsewhere (carpet back coating)	Mean	26959	30129	2.85E−08
Fabric textile, leather products not covered elsewhere (textile for children's play structures)	Mean	1830	2045	4.19E−07
Building/construction materials not covered elsewhere (roofing insulation)	Mean	95	106	8.07E−06
Building/construction materials not covered elsewhere (acoustic ceiling)	Mean	1182	1321	6.49E−07
Foam seating and bedding product (foam automobile)	Mean	24083	26916	3.19E−08

COU Subcategory and Consumer Exposure Scenarios	Milk Intake Rate Type	Intermediate	Chronic	Cancer
Foam seating and bedding product (foam living room)	Mean	10,771	12,037	7.12E−08
Foam seating and bedding product (mattress)	Mean	84,787	94,757	9.05E−09
Foam seating and bedding product (foam - other - toy block)	Mean	156,138	174,500	4.91E−09
Building/construction materials – Wood and engineered wood products (wood flooring)	Mean	109	122	7.04E−06
Building/construction materials – Wood and engineered wood products (wooden TV stand)	Mean	1,630	1,821	4.71E−07
Fabric textile, leather products not covered elsewhere (carpet back coating)	Upper	18,419	20,649	4.15E−08
Fabric textile, leather products not covered elsewhere (textile for children's play structures)	Upper	1,250	1,402	6.12E−07
Building/construction materials not covered elsewhere (roofing insulation)	Upper	65	73	1.18E−05
Building/construction materials not covered elsewhere (acoustic ceiling)	Upper	808	905	9.47E−07
Foam seating and bedding product (foam automobile)	Upper	16,455	18,447	4.65E−08
Foam seating and bedding product (foam living room)	Upper	7,359	8,250	1.04E−07
Foam seating and bedding product (mattress)	Upper	57,930	64,944	1.32E−08
Foam seating and bedding product (foam – other – toy block)	Upper	106,680	119,597	7.17E−09
Building/construction materials – Wood and engineered wood products (wood flooring)	Upper	74	83	1.03E−05
Building/construction materials – Wood and engineered wood products (wooden TV stand)	Upper	1,114	1,248	6.87E−07

Table_Apx I-29. Infant Risks via Human Milk Exposure from Maternal Occupational Use Scenarios Based on Mean Milk Intake Rate

OES	Route	Maternal Exposure Duration	Intermediate	Chronic	Cancer
Import and repackaging	Dermal, Inhalation (High-end)	Chronic	1,054	1,178	7.28E−07
Incorporation into paints and coatings – 1-part coatings		Chronic	197	220	3.89E−06
Incorporation into paints and coatings – 2-part reactive coatings		Chronic	1,935	2163	3.97E−07
Processing – Formulation of TCEP into 2-part reactive resins		Chronic	956	1068	8.03E−07
Processing – Processing into 2-part resin article		Chronic	76	85	1.01E−05
Processing – Recycling Electronics		Chronic	1,206,274	1,348,128	6.36E−10
Commercial use – Paints and coatings – Spray (1-part, 250-day application)		Chronic	114	127	6.74E−06
Commercial use – Paints and coatings – Spray (2-part reactive, 250-day application)		Chronic	23	25	3.37E−05
Laboratory chemicals		Chronic	38	42	2.02E−05
Industrial/Commercial use – Installation of articles	Inhalation (High-end)	Chronic	122,184,772	136,553,356	6.28E−12
Import and repackaging	Dermal, Inhalation (High-end)	Intermediate	89	99	8.64E−06
Incorporation into paints and coatings – 1-part coatings		Intermediate	28	32	2.71E−05
Incorporation into paints and coatings – 2-part reactive coatings		Intermediate	287	321	2.67E−06
Processing – Formulation of TCEP into 2-part reactive resins		Intermediate	101	113	7.59E−06
Processing – Processing into 2-part resin article		Intermediate	71	79	1.08E−05
Processing – Recycling electronics		Intermediate	1,126,657	1259,148	6.81E−10
Commercial use – Paints and coatings – Spray (1-part, 250-day application)		Intermediate	106	119	7.21E−06
Commercial use – Paints and coatings – Spray (2-part reactive, 250-day application)		Intermediate	21	24	3.61E−05
Laboratory chemicals		Intermediate	28	32	2.71E−05
Industrial/Commercial use – Installation of articles	Inhalation (High-end)	Intermediate	114,120,319	127,540,532	6.72E−12

Table_Apx I-30. Infant Risks via Human Milk Exposure from Maternal Occupational Use Scenarios Based on Upper Milk Intake Rate

OES	Route	Maternal Exposure Duration	Intermediate	Chronic	Cancer
Import and repackaging	Dermal, Inhalation (High-end)	Chronic	720	808	1.06E−06
Incorporation into paints and coatings – 1-part coatings		Chronic	135	151	5.68E−06
Incorporation into paints and coatings – 2-part reactive coatings		Chronic	1,322	1,482	5.78E−07
Processing – Formulation of TCEP into 2-part reactive resins		Chronic	653	732	1.17E−06
Processing – Processing into 2-part resin article		Chronic	52	58	1.48E−05
Processing – Recycling electronics		Chronic	824,172	923,965	9.28E−10
Commercial use – Paints and coatings – Spray (1-part, 250-day application)		Chronic	78	87	9.83E−06
Commercial use – Paints and coatings – Spray (2-part reactive, 250-day application)		Chronic	16	17	4.91E−05
Laboratory chemicals		Chronic	26	29	2.95E−05
Industrial/Commercial use – Installation of articles	Inhalation (High-end)	Chronic	83,481,270	93,493,151	9.17E−12
Import and repackaging	Dermal, Inhalation (High-end)	Intermediate	61	59	1.45E−05
Incorporation into paints and coatings – 1-part coatings		Intermediate	19	19	4.56E−05
Incorporation into paints and coatings – 2-part reactive coatings		Intermediate	196	191	4.49E−06
Processing – Formulation of TCEP into 2-part reactive resins		Intermediate	69	67	1.28E−05
Processing – Processing into 2-part resin article		Intermediate	48	47	1.82E−05
Processing – Recycling electronics		Intermediate	769,775	748,688	1.15E−09
Commercial use – Paints and coatings – Spray (1-part, 250-day application)		Intermediate	73	71	1.21E−05
Commercial use – Paints and coatings – Spray (2-part reactive, 250-day application)		Intermediate	15	14	6.06E−05
Laboratory chemicals		Intermediate	19	19	4.55E−05
Industrial/Commercial use – Installation of articles	Inhalation (High-end)	Intermediate	77,971,323	75,835,431	1.13E−11

Table_Apx I-31. Infant Risks via Human Milk Exposure from Maternal General Population Oral Exposures Based on Mean Milk Intake Rate

COUs/OESs	Route	Intermediate	Chronic	Cancer
Import and repackaging	Gen Pop Fish Ingestion, High BAF	259	290	2.96E−06
Incorporation into paints and coatings – 1-part coatings	Gen Pop Fish Ingestion, High BAF	59	65	1.31E−05
Incorporation into paints and coatings – 2-part reactive coatings	Gen Pop Fish Ingestion, High BAF	64	72	1.19E−05
Use in paints and coatings at job sites	Gen Pop Fish Ingestion, High BAF	110	123	6.97E−06
Formulation of TCEP containing reactive resin	Gen Pop Fish Ingestion, High BAF	46	52	1.66E−05
Laboratory chemicals	Gen Pop Fish Ingestion, High BAF	6,474	7,235	1.19E−07
Import and repackaging	Gen Pop Fish Ingestion, Low BAF	5,224	5,839	1.47E−07
Incorporation into paints and coatings – 1-part coatings	Gen Pop Fish Ingestion, Low BAF	1,179	1,318	6.51E−07
Incorporation into paints and coatings – 2-part reactive coatings	Gen Pop Fish Ingestion, Low BAF	1,300	1,453	5.90E−07
Use in paints and coatings at job sites	Gen Pop Fish Ingestion, Low BAF	2,219	2,480	3.46E−07
Formulation of TCEP containing reactive resin	Gen Pop Fish Ingestion, Low BAF	927	1,037	8.27E−07
Laboratory chemicals	Gen Pop Fish Ingestion, Low BAF	129,993	145,279	5.90E−09
Import and repackaging	Undiluted Drinking Water	5,224,384	5,838,756	1.47E−10
Incorporation into paints and coatings – 1-part coatings	Undiluted Drinking Water	1,179,218	1,317,891	6.51E−10
Incorporation into paints and coatings - 2-part reactive coatings	Undiluted Drinking Water	1,310,242	1,464,323	5.86E−10
Use in paints and coatings at job sites	Undiluted Drinking Water	2,224,940	2,486,586	3.45E−10
Formulation of TCEP containing reactive resin	Undiluted Drinking Water	932,715	1042,399	8.23E−10
Laboratory chemicals	Undiluted Drinking Water	131,024,263	146,432,358	5.86E–12

Table_Apx I-32. Infant Risks via Human Milk Exposure from Maternal General Population Oral Exposures Based on Upper Milk Intake Rate

COUs/OESs	Route	Intermediate	Chronic	Cancer
Import and repackaging	Gen Pop Fish Ingestion, High BAF	177	199	4.32E−06
Incorporation into paints and coatings – 1-part coatings	Gen Pop Fish Ingestion, High BAF	40	45	1.91E−05
Incorporation into paints and coatings – 2-part reactive coatings	Gen Pop Fish Ingestion, High BAF	44	49	1.74E−05
Use in paints and coatings at job sites	Gen Pop Fish Ingestion, High BAF	75	84	1.02E−05
Formulation of TCEP containing reactive resin	Gen Pop Fish Ingestion, High BAF	32	35	2.43E−05
Laboratory chemicals	Gen Pop Fish Ingestion, High BAF	4,423	4,959	1.73E−07
Import and repackaging	Gen Pop Fish Ingestion, Low BAF	3,569	4,002	2.14E−07
Incorporation into paints and coatings – 1-part coatings	Gen Pop Fish Ingestion, Low BAF	806	903	9.49E−07
Incorporation into paints and coatings – 2-part reactive coatings	Gen Pop Fish Ingestion, Low BAF	888	996	8.61E−07
Use in paints and coatings at job sites	Gen Pop Fish Ingestion, Low BAF	1,516	1,700	5.05E−07
Formulation of TCEP containing reactive resin	Gen Pop Fish Ingestion, Low BAF	634	710	1.21E−06
Laboratory chemicals	Gen Pop Fish Ingestion, Low BAF	88,816	99,570	8.61E−09
Import and repackaging	Undiluted Drinking Water	3,569,498	4,001,703	2.14E−10
Incorporation into paints and coatings – 1-part coatings	Undiluted Drinking Water	805,687	903,241	9.49E−10
Incorporation into paints and coatings – 2-part reactive coatings	Undiluted Drinking Water	895,207	1,003,602	8.54E−10
Use in paints and coatings at job sites	Undiluted Drinking Water	1,520,163	1,704,229	5.03E−10
Formulation of TCEP containing reactive resin	Undiluted Drinking Water	637,266	714,428	1.20E−09
Laboratory chemicals	Undiluted Drinking Water	89,520,729	100,360,171	8.54E–12

Table_Apx I-33. Infant Risks via Human Milk Exposure from Tribal Maternal Fish Exposures Based on Mean Milk Intake Rate

COUs/OESs	Route	Intermediate	Chronic	Acute Based on Intermediate Dose	Acute Based on Chronic Dose	Cancer
Import and repackaging	Current IR, High BAF	27	30	92	103	2.89E−05
Incorporation into paints and coatings – 1-part coatings	Current IR, High BAF	6	7	21	23	1.28E−04
Incorporation into paints and coatings – 2-part reactive coatings	Current IR, High BAF	7	7	23	26	1.16E−04
Use in paints and coatings at job sites	Current IR, High BAF	11	13	39	44	6.79E−05
Formulation of TCEP containing reactive resin	Current IR, High BAF	5	5	16	18	1.62E−04
Laboratory chemicals	Current IR, High BAF	663	741	NA	NA	1.16E−06
Import and repackaging	Current IR, Low BAF	536	599	NA	NA	1.43E−06
Incorporation into paints and coatings – 1-part coatings	Current IR, Low BAF	121	136	NA	NA	6.32E−06
Incorporation into paints and coatings – 2-part reactive coatings	Current IR, Low BAF	133	149	NA	NA	5.76E−06
Use in paints and coatings at job sites	Current IR, Low BAF	228	254	NA	NA	3.37E−06
Formulation of TCEP containing reactive resin	Current IR, Low BAF	95	107	NA	NA	8.04E−06
Laboratory chemicals	Current IR, Low BAF	13,422	15,000	NA	NA	5.72E−08
Import and repackaging	Heritage IR, High BAF	3	4	12	14	2.20E−04
Incorporation into paints and coatings – 1-part coatings	Heritage IR, High BAF	1	1	3	3	9.76E−04
Incorporation into paints and coatings – 2-part reactive coatings	Heritage IR, High BAF	1	1	3	3	8.88E−04
Use in paints and coatings at job sites	Heritage IR, High BAF	1	2	5	6	5.16E−04
Formulation of TCEP containing reactive resin	Heritage IR, High BAF	1	1	2	2	1.24E−03
Laboratory chemicals	Heritage IR, High BAF	87	98	303	338	8.78E−06
Import and repackaging	Heritage IR, Low BAF	71	79	NA	NA	1.09E−05
Incorporation into paints and coatings – 1-part coatings	Heritage IR, Low BAF	16	18	55	61	4.83E−05
Incorporation into paints and coatings – 2-part reactive coatings	Heritage IR, Low BAF	18	20	61	68	4.38E−05
Use in paints and coatings at job sites	Heritage IR, Low BAF	30	33	NA	NA	2.57E−05
Formulation of TCEP containing reactive resin	Heritage IR, Low BAF	13	14	43	48	6.13E−05
Laboratory chemicals	Heritage IR, Low BAF	1,754	1,961	NA	NA	4.37E−07

Table_Apx I-34. Infant Risks via Human Milk Exposure from Tribal Maternal Fish Exposures Based on Upper Milk Intake Rate

COUs/OESs	Route	Intermediate	Chronic	Acute Based on Intermediate Dose	Acute Based on Chronic Dose	Cancer
Import and repackaging	Current IR, High BAF	18	20	63	71	4.21E−05
Incorporation into paints and coatings – 1-part coatings	Current IR, High BAF	4	5	14	16	1.86E−04
Incorporation into paints and coatings – 2-part reactive coatings	Current IR, High BAF	5	5	16	18	1.69E−04
Use in paints and coatings at job sites	Current IR, High BAF	8	9	27	30	9.90E−05
Formulation of TCEP containing reactive resin	Current IR, High BAF	3	4	11	13	2.37E−04
Laboratory chemicals	Current IR, High BAF	453	508	NA	NA	1.69E−06
Import and repackaging	Current IR, Low BAF	366	411	NA	NA	2.09E−06
Incorporation into paints and coatings – 1-part coatings	Current IR, Low BAF	83	93	NA	NA	9.22E−06
Incorporation into paints and coatings – 2-part reactive coatings	Current IR, Low BAF	91	102	NA	NA	8.41E−06
Use in paints and coatings at job sites	Current IR, Low BAF	156	174	NA	NA	4.92E−06
Formulation of TCEP containing reactive resin	Current IR, Low BAF	65	73	NA	NA	1.17E−05
Laboratory chemicals	Current IR, Low BAF	9,170	10,281	NA	NA	8.34E−08
Import and repackaging	Heritage IR, High BAF	2	3	8	9	3.21E−04
Incorporation into paints and coatings – 1-part coatings	Heritage IR, High BAF	1	1	2	2	1.42E−03
Incorporation into paints and coatings – 2-part reactive coatings	Heritage IR, High BAF	1	1	2	2	1.30E−03
Use in paints and coatings at job sites	Heritage IR, High BAF	1	1	4	4	7.53E−04
Formulation of TCEP containing reactive resin	Heritage IR, High BAF	0	0	1	2	1.80E−03
Laboratory chemicals	Heritage IR, High BAF	60	67	NA	NA	1.28E−05
Import and repackaging	Heritage IR, Low BAF	48	54	NA	NA	1.59E−05
Incorporation into paints and coatings – 1-part coatings	Heritage IR, Low BAF	11	12	38	42	7.05E−05
Incorporation into paints and coatings – 2-part reactive coatings	Heritage IR, Low BAF	12	13	41	46	6.39E−05
Use in paints and coatings at job sites	Heritage IR, Low BAF	20	23	71	NA	3.74E−05
Formulation of TCEP containing reactive resin	Heritage IR, Low BAF	9	10	30	33	8.95E−05
Laboratory chemicals	Heritage IR, Low BAF	1,199	1,344	NA	NA	6.38E−07

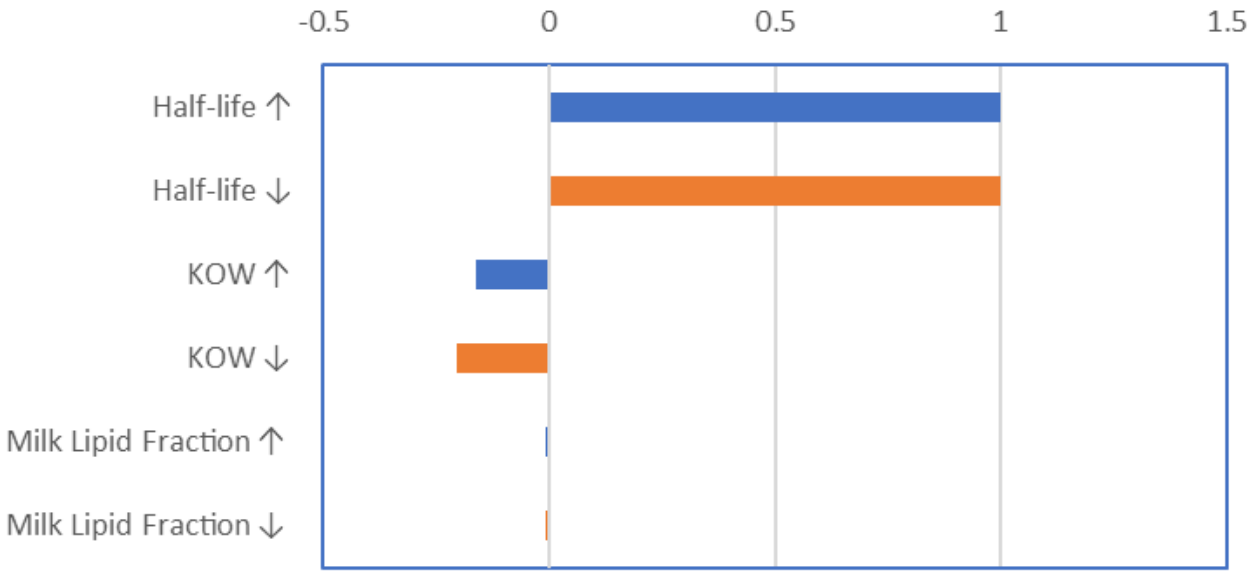
I.5.6 Sensitivity Analysis

EPA conducted a sensitivity analysis for TCEP to evaluate the effect of chemical and biological considerations on modeled TCEP concentrations in milk, as shown in Table_Apx I-35. Sensitivity was measured using elasticity, which is defined as the ratio of percent change in each result to the corresponding percent change in model input. A positive elasticity means that an increase in the model parameter resulted in an increase in the model output, whereas a negative elasticity had an associated decrease in the model output. Figure_Apx I-7 shows the results of the sensitivity analysis.

Table_Apx I-35. Variables and Values Used in Sensitivity Analysis

Variable	Base/Default Values	Sensitivity Values
Half-Life	17.64	15.87, 19.40 (increased and decreased from base value by 10%)
K_{OW}[a]	60.26	66.28 and 54.23 (increased and decreased from base value by 10%)
Lipid fraction in milk	0.038 + 0.000095*age	Multiplied the function by 1.1 and 0.9 to increase and decrease from base value by 10%, respectively
Age at pregnancy	25	40 (increased to reflect an alternate scenario)

[a] The analysis varied K_{OW} rather than log K_{OW} because the partition coefficient equations used are based on K_{OW}. K_{OW} is not used elsewhere in the model equations.



Figure_Apx I-7. Sensitivity Analysis of Model Inputs Measured as Elasticity

The elasticity for half-life is close to one. A ±10 percent change in half-life reflected a near equivalent percent change in the infant milk dose. In contrast, a ±10 percent change to K_{OW} resulted in a smaller change in the infant milk dose. Half-life and K_{OW} parameters are independent values in the model. The half-life is used to estimate the liver compartment's elimination rate while K_{OW} is used to estimate the partition coefficients. For a slightly lipophilic compound like TCEP, an increase in K_{OW} (and calculated partition coefficient) leads to a relatively larger increase in the lipid:blood partition coefficient than for

other compartments such as mammary tissue. Thus, more TCEP will be stored in lipids and less in the mammary tissue, causing a decrease in infant milk dose. If half-life increases, more TCEP is available in the body and each compartment at a given time, including the mammary tissue, causing an increase in infant milk dose. TCEP infant doses were insensitive to alterations of milk lipid fractions. TCEP concentrations in milk were similarly insensitive (data not shown). This insensitivity may reflect the relatively low K_{OW} for TCEP.

Although the model treats K_{OW} and half-life independently, these parameters are linked from a toxicokinetic perspective. The K_{OW} of the chemical likely influences both the partition coefficient (the lipid compartments in particular) and the half-life. More lipophilic compounds tend to have larger lipid:blood partition coefficient and longer half-lives than less lipophilic compounds. Thus, a 10 percent change in K_{OW} might also cause a percent change in the half life, and that correlation is not captured in the model or sensitivity analysis.

Neither maternal age nor infant sex (results not shown) affected milk doses, indicating this model is not sensitive to these parameters for TCEP. For infant sex, the only parameter differentiating male and females in this model are growth curves, which are considered in the dose calculation.

Metabolic Rate

EPA conducted a similar sensitivity analysis for the metabolic rate of an infant compared to an adult. The Verner model adjusts the intrinsic hepatic clearance rate such that the half-life of TCEP remains constant throughout all lifestages irrespective of changes from birth through adulthood. This adjustment does not consider that the metabolic potential of an infant is lower than an adult, and it is well known that the levels of cytochrome P450 (CYP450) enzymes vary with age. Since the metabolic pathways for TCEP is not fully characterized and it has a much shorter half-life than persistent organic pollutants (*i.e.*, hours vs. years), parameterizing the Verner model to vary metabolic rates by age, body weight, or another physiological factor could be speculative and not even result in noticeable differences. Nonetheless, EPA still aimed to characterize CYP450 metabolism differences by varying TCEP's half-life as a proxy.

The CYP450 metabolism of an infant was estimated at approximately 30 percent of an adult's based on peer review comments and supported by (Ginsberg et al., 2004). To be conservative, EPA assumed the infant's metabolism remains at 30 percent throughout the first year of life, even though it increases steadily during that time. TCEP's half-life was then increased by 3.3-fold (100/30) from 17.6 hours to 58.8 hours. EPA then estimated exposure and risks for worst-case scenarios: COUs with the highest maternal doses and the upper milk intake rate. Both exposure and risk estimates for the nursing infant still remain below that of the mother. The results are shown in Table_Apx I-36 and Table_Apx I-37:

Table_Apx I-36. Average Infant Doses Using a Longer TCEP Half-life

Population Group	COU Subcategory	Route	Maternal Exposure Duration	Maternal Dose (mg/kg-day)	Average from Birth to <1 Month (mg/kg-day)	Average from 1 to <3 Month (mg/kg-day)	Average from 3 to <6 Month (mg/kg-day)	Average from 6 to 12 Month (mg/kg-day)	Overall Average from Birth to 12 Month (mg/kg-day)
Consumer	Building/construction materials - wood and engineered wood products (wood flooring)	Dermal, Oral, Inhalation	Chronic	1.80	1.45E−01	1.36E−01	1.22E−01	1.27E−01	1.29E−01
Occupational	Commercial Use – Paints & Coatings – Spray (resin, 250-day application, 2-part coating)	Dermal, Inhalation	Chronic	7.25	5.83E−01	5.48E−01	4.93E−01	5.12E−01	5.19E−01
Occupational	Commercial Use – Paints & Coatings – Spray (resin, 250-day application, 2-part coating)		Intermediate	7.76	6.24E−01	5.87E−01	5.28E−01	5.48E−01	5.56E−01
Tribal	Formulation of TCEP containing reactive resin, Current IR, High BAF	Oral	Chronic	3.49E01	1.62E01	1.52E01	1.37E01	1.42E01	1.44E01
Tribal	Formulation of TCEP containing reactive resin, Current IR, Low BAF		Chronic	1.73	1.39E−01	1.31E−01	1.18E−01	1.22E−01	1.24E−01
Tribal	Formulation of TCEP containing reactive resin, Heritage IR, High BAF		Chronic	2.66E02	2.14E01	2.01E01	1.81E01	1.88E01	1.91E01
Tribal	Formulation of TCEP containing reactive resin, Heritage IR, Low BAF		Chronic	1.32E01	1.06	9.98E−01	8.98E−01	9.32E−01	9.46E−01

Table_Apx I-37. Infant Risk Estimate Using a Longer TCEP Half-life

Population Group	COU Subcategory and Consumer Exposure Scenarios	Intermediate	Chronic	Acute Based on Intermediate	Acute Based on Chronic	Cancer
Consumer	Building/construction materials – wood and engineered wood products (wood flooring)	19	21	65	73	4.04E−05
Occupational	Commercial Use – Paints & Coatings – Spray (resin, 250-day application, 2-part coating)	5	5	16	18	1.63E−04
Occupational	Commercial Use – Paints & Coatings – Spray (resin, 250-day application, 2-part coating)	4	5	15	17	1.75E−04
Tribal	Formulation of TCEP containing reactive resin, Current IR, High BAF	0.2	0.2	0.6	0.7	4.52E−03
Tribal	Formulation of TCEP containing reactive resin, Current IR, Low BAF	20	22	68	76	3.89E−05
Tribal	Formulation of TCEP containing reactive resin, Heritage IR, High BAF	0.1	0.1	0.4	0.5	5.99E−03
Tribal	Formulation of TCEP containing reactive resin, Heritage IR, Low BAF	3	3	9	10	2.97E−04

I.6 Landfill Analysis Using DRAS

DRAS is an efficient tool developed by EPA Region 6 to provide a multipath risk assessment for the evaluation of Resource Conservation and Recovery Act (RCRA) hazardous waste delisting. For the TCEP Risk Evaluation, DRAS was specifically applied to model groundwater concentration estimates from disposing TCEP to a hypothetical RCRA Subtitle D landfill at a range of loading rates and leachate concentrations. A comprehensive description of the assumptions and calculations applied in DRAS can be found in the Technical Support Document for the Hazardous Waste Delisting Risk Assessment Software.

Because DRAS derives calculations based on a survey of drinking water wells located downgradient from waste management units (U.S. EPA, 1988), the model may provide the closest estimate to real world scenarios available. Although there is some uncertainty inherent to applying the model as an assessment tool under TSCA for risk evaluations, few other tools are available to effectively address this pathway. This appendix will provide the input variables and calculations used to apply the model determine potential groundwater concentrations. Table_Apx I-38 and Table_Apx I-39 provide the input values used for each parameter in the model. Note that loading volumes were based on the range of estimated production volumes (2,500 to 2,500,000 lb) and were calculated based on the density of TCEP (1.39 g/cm^3). For each loading volume, the range of leachate concentrations was applied.

Table_Apx I-38. Input Variables for Chemical of Concern

Input Variable for Chemical of Concern	Value
Chem Name	TCEP
CASRN	115-96-8
Maximum Contaminant Level	0
Oral Slope Cancer Factor	0.1[a]
Inhalation Slope Cancer Factor (1/mg kg day)	0.018[a]
Oral Reference Dose (mg/kg day)	0.03[a]
Inhalation Reference Dose (mg/kg day)	0.03[a]
Bioconcentration Factor (l/kg)	0
Soil Saturation Level	0
Toxicity Regulatory Rule regulatory level (mg/L)	0[a]
Henry's Law constant (atm -m^3/mol)	2.95E−06
Diffusion Coefficient in Water (cm^2/s)	5.07E−06
Diffusion Coefficient in Air (cm^2/s)	0.044[a]
Water Solubility (mg/L)	7,820
Landfill Dilution Attenuation Factor	15.4
Surface Impoundment Dilution Attenuation Factor	3.18
Time to Skin Attenuation (hr/event)	0
Skin Permeability Constant (cm/hr)	0.00022[a]
Lag Time (hr)	0.28[a]
Bunge Constant	4.1E−05[a]
Organic	Yes

Input Variable for Chemical of Concern	Value
Bioaccumulation Factor (L/kg)	6,016[a]
Chronic Ecological Value (mg/L)	85[a]
Carcinogen	No
Molecular Weight (g/mol)	285.49
Vapor Pressure (atm)	8.07E−5
Suspended sediment-surface water partitioning coefficient (mg/L)	298.725
log K_{OW} (log[mg/L])	1.78
Chemical Class	SVOC[a]
Analytical Method	8,260D[a]
Version Description	None[a]
Create Date	None[a]
Creator	None[a]
Cancer Risk Level	1.00E−06[a]
Hazard Quotient	1[a]
[a] Input variables do not directly or indirectly affect groundwater concentrations	

Table_Apx I-39. Waste Management Unit (WMU) Properties

Input Variable for WMU Properties	Value(s)
Waste Management Unit Type	Landfill
Loading Volume (m^3)	8.17E−01
	8.17
	8.17E01
	8.17E02
Cancer Risk Level	1.00E−06
Hazard Quotient	1.0
Detection Limit	0.5
Waste Management Active Life (years)	20
TCLP Concentration (mg/L)/Total Concentration (mg/kg)	0.0001
	0.001
	0.01
	0.1
	1
	10
	100
	1000

Once the model was executed for each loading rate and leachate concentration scenario, the groundwater concentration was calculated using the leachate concentration and the 90th percentile weight-adjusted dilution attenuation factor using

Equation_Apx I-9.

$$GW_c = \frac{Leachate\ Concentration}{Weight-Adjusted\ DAF},$$

Where:

GW_c	=	Groundwater concentration
Leachate concentration	=	Input variable for the waste management unit
Weight-Adjusted DAF =		Weight- adjusted dilution attenuation factor

The results of these analyses are provided in Table 3-8.

Appendix J CONSUMER EXPOSURE DETAILS

J.1 Approach and Methodology

EPA evaluated TCEP exposure resulting from the use of consumer products and industrial processes. The Agency utilized a modeling approach to evaluate exposure because chemical-specific personal monitoring data attributable to the COUs was not identified for consumers during data gathering and literature searches performed as part of systematic review using the evaluation strategies described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and in the TCEP Systematic Review Protocol (U.S. EPA, 2024p).

There are a limited number of consumer articles that still contain TCEP, because many manufacturers have reformulated them to remove TCEP. Consumer products containing TCEP are readily available via the internet as finished articles (*e.g.*, furniture and foam products). Use of these products can result in exposures of the consumer user to TCEP during and after article use. Consumer exposure can occur via inhalation, dermal, and oral routes.

Consumer products containing TCEP were identified through review and searches of a variety of sources, including the National Institutes of Health (NIH) Household Products Database, various government and trade association sources for products containing TCEP, company websites for safety data sheets (SDSs), *Kirk-Othmer Encyclopedia of Chemical Technology*, and the internet. In general, information on the consumer uses of TCEP was sparse and many manufacturers reported changes in formulation and ceasing the use of TCEP in favor of other chemicals.

Identified consumer products (see Table 1-1) were then categorized into six consumer use groups considering (1) consumer use patterns, (2) information reported in SDSs, (3) product availability to the public, and (4) potential risk to consumers.

Readers are referred to each model's user guide and associated user guide appendices for details on each model, as well as information related to equations used within the models, default values, and the basis for default values. Each model is peer reviewed. Default values within CEM are a combination of high-end and mean or central tendency values derived from EPA's *Exposure Factors Handbook* (U.S. EPA, 2017d), literature, and other studies.

J.1.1 Consumer Exposure Model (CEM)

CEM is a deterministic model that utilizes user provided input parameters and various assumptions (or defaults) to generate exposure estimates. In addition to pre-defined scenarios, which align well with the consumer uses identified in Table 1-1, CEM is peer reviewed, provides flexibility to the user allowing modification of certain default parameters when chemical-specific information is available and does not require chemical-specific emissions data (which may be required to run more complex indoor/consumer models).

CEM predicts indoor air concentrations from consumer product use through a deterministic, mass-balance calculation derived from emission calculation profiles within the model. There are six emission calculation profiles within CEM (E1–E6) that are summarized in the CEM users guide and associated appendices. If selected, CEM provides a time series air concentration profile for each run. These are intermediate values produced prior to applying pre-defined activity patterns.

CEM uses a two-zone representation of the building of use when predicting indoor air concentrations. Zone 1 represents the room where the consumer product is used. Zone 2 represents the remainder of the building. Each zone is considered well-mixed. CEM allows further division of Zone 1 into a near field and far field to accommodate situations where a higher concentration of product is expected very near the product user when the product is used. Zone 1-near field represents the breathing zone of the user at the location of the product use while Zone 1-far field represents the remainder of the Zone 1 room. Inhalation exposure is estimated in CEM based on zones and pre-defined activity patterns. The simulation run by CEM places the product user within Zone 1 for the duration of product use while the bystander is placed in Zone 2 for the duration of product use. Following the duration of product use, the user and bystander follow one of three pre-defined activity patterns established within CEM, based on modeler selection. The selected activity pattern takes the user and bystander in and out of Zone 1 and Zone 2 for the period of the simulation. The user and bystander inhale airborne concentrations within those zones, which will vary over time, resulting in the overall estimated exposure to the user and bystander.

CEM contains two methodologies for estimating dermal exposure to chemicals in products—the permeability method (P-DER1) and the fraction absorbed method (A-DER1). Each of these methodologies further has two model types, one designed for dermal exposure from use of a product (P-DER1a and A-DER1a) and the other designed for dermal exposure from use of an article (P-DER1b and A-DER1b). Each methodology has associated assumptions, uncertainties, and data input needs within the CEM model. Both methodologies factor in the dermal surface area to body weight ratio and weight fraction of chemical in a consumer product.

The permeability model is based on the ability of a chemical to penetrate the skin layer once contact occurs. The permeability model assumes a constant supply of chemical, directly in contact with the skin, throughout the exposure duration. The ability to use the permeability method can be beneficial when chemical-specific skin permeability coefficients are available in the scientific literature. However, the permeability model within CEM does not consider evaporative losses when it estimates dermal exposure and therefore may be more representative of a dermal exposure resulting from a constant supply of chemical to the skin due to a barrier or other factor that may restrict evaporation of the chemical of interest from the skin such as a product soaked rag against the hand while using a product), or immersion of a body part into a pool of product. Either of these examples has the potential to cause an increased duration of dermal contact and permeation of the chemical into the skin resulting in dermal exposure.

The fraction absorbed method is based on the absorbed dose of a chemical. This method essentially measures two competing processes, evaporation of the chemical from the skin and penetration of the chemical deeper into the skin. This methodology assumes the application of the chemical of concern occurs once to an input thickness and then absorption occurs over an estimated absorption time. The fraction absorbed method can be beneficial when chemical specific fractional absorption measurements are available in the scientific literature. The consideration of evaporative losses by the fraction absorbed method within CEM may make this model more representative of a dermal exposure resulting from scenarios that allow for continuous evaporation and typically would not involve a constant supply of product for dermal permeation. Examples of such scenarios include spraying a product onto a mirror and a small amount of mist falling onto an unprotected hand. For TCEP, literature values for fraction absorbed were used from Abdallah et al. (2016), rather than the faction absorbed estimation via CEM.

J.1.2 Inputs

J.1.2.1 CEM and Sensitivity Analysis

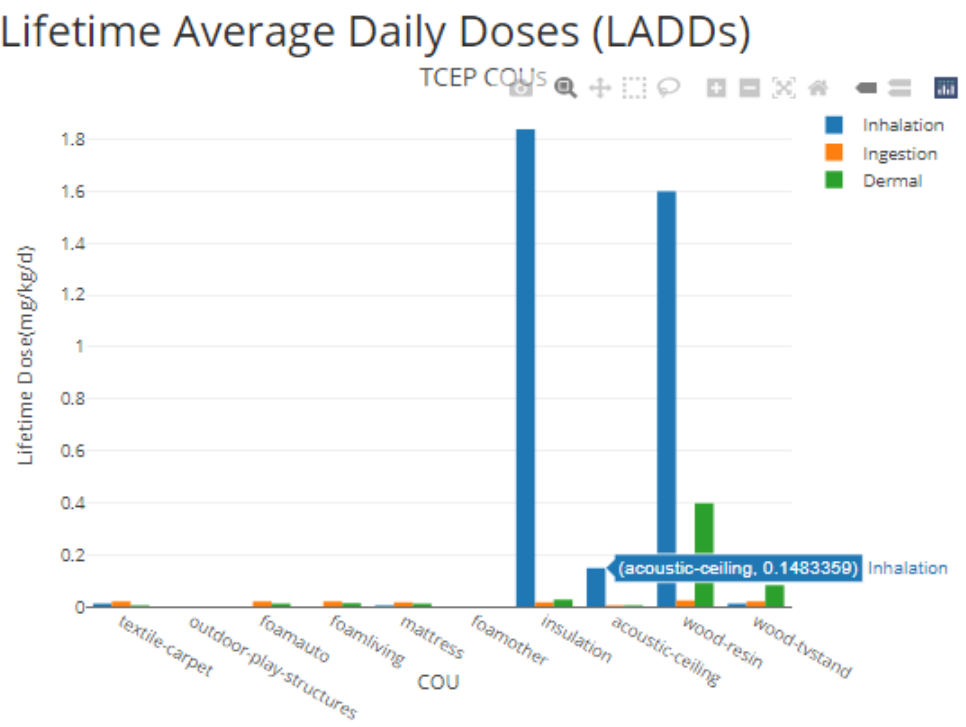
Inputs for the each of the CEM 3.0 base and sensitivity runs are provide in *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Consumer Exposure Modeling Inputs* (U.S. EPA, 2024e). Where available, EPA relied on the *Exposure Factors Handbook* (U.S. EPA, 2017d) and information identified in systematic review to inform input parameters. For article-specific parameters (*e.g.,* product density, thickness of article surface layer, surface area) that were unavailable in the handbook or the peer-reviewed or gray literature, EPA used professional judgment to determine whether the CEM default values were appropriate, or whether there should be an alternative value for the parameter based on professional judgment. All the input parameters and their rationale are provided in the *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Consumer Exposure Modeling Inputs* (U.S. EPA, 2024e). Inputs for the sensitivity analysis are provided in the "Sensitivity Analysis" tab of the *Consumer Exposure Modeling Inputs* (U.S. EPA, 2024e).

J.1.3 Results

J.1.3.1 Raw Consumer Modeling Results

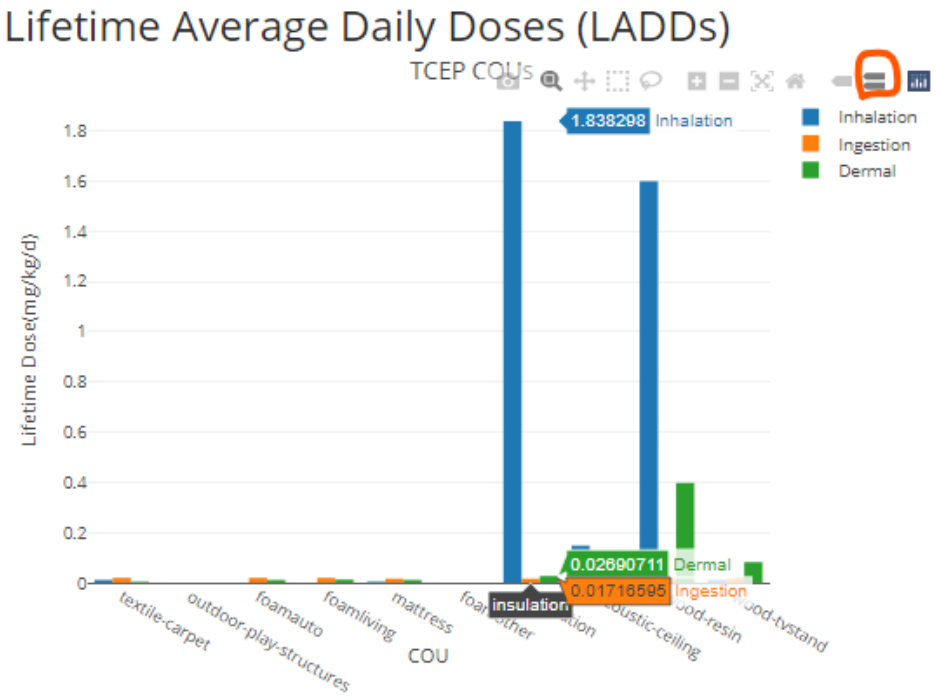
Modeling results are available in pdf and xlsx format in TCEP_Consumer_Modeling_Results.zip (U.S. EPA, 2024b). Results from the consumer modeling have been visualized in bar charts, and risk tables in the Navigating Supplemental Consumer Modeling Results Consumer Modeling Results were tabulated in R and have been displayed in an "Rmarkdown file." The associated R script uses a workflow that loads the input data from the consumer modeling results, cleans, filters, and wrangles the relevant data, and displays the modeling results in the form of bar plots and risk tables.

Bar plots are interactive, and reviewers are able to pan and select certain data fields to help compare the results from the various consumer COUs (see Figure_Apx J-1 through Figure_Apx J-4). Hovering over the data bars in the link above provides a tool tip that indicates the value of the bar.



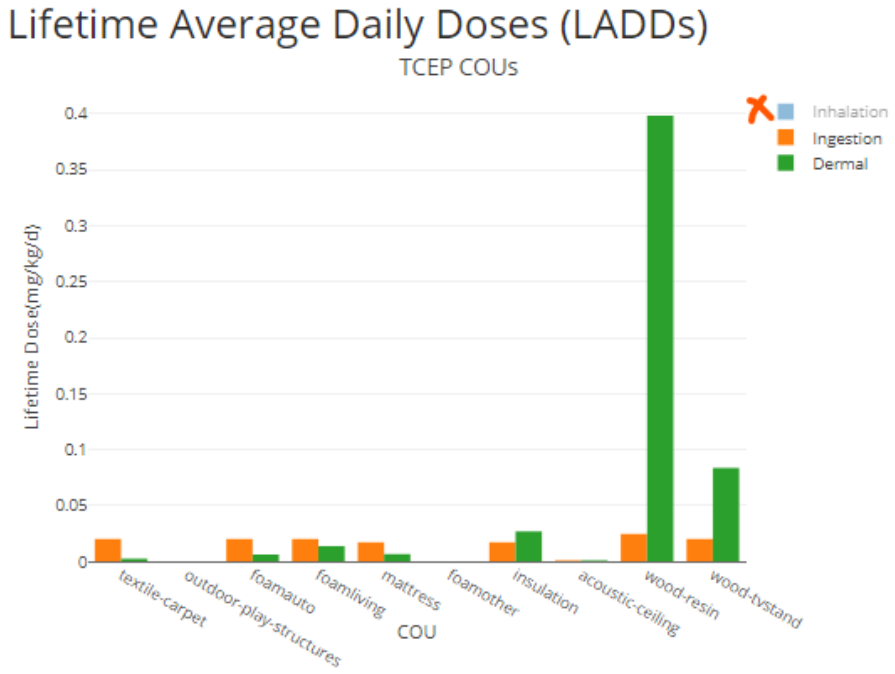
Figure_Apx J-1. Screenshot of Lifetime Average Daily Doses (LADDs) Bar Chart Displaying Tool Tip for Acoustic Ceiling, Inhalation Estimate

The toolbar at the top also has various functionalities that can allow for more exploration of the data. For example, simply hover and select the outlined double bars to compare data.



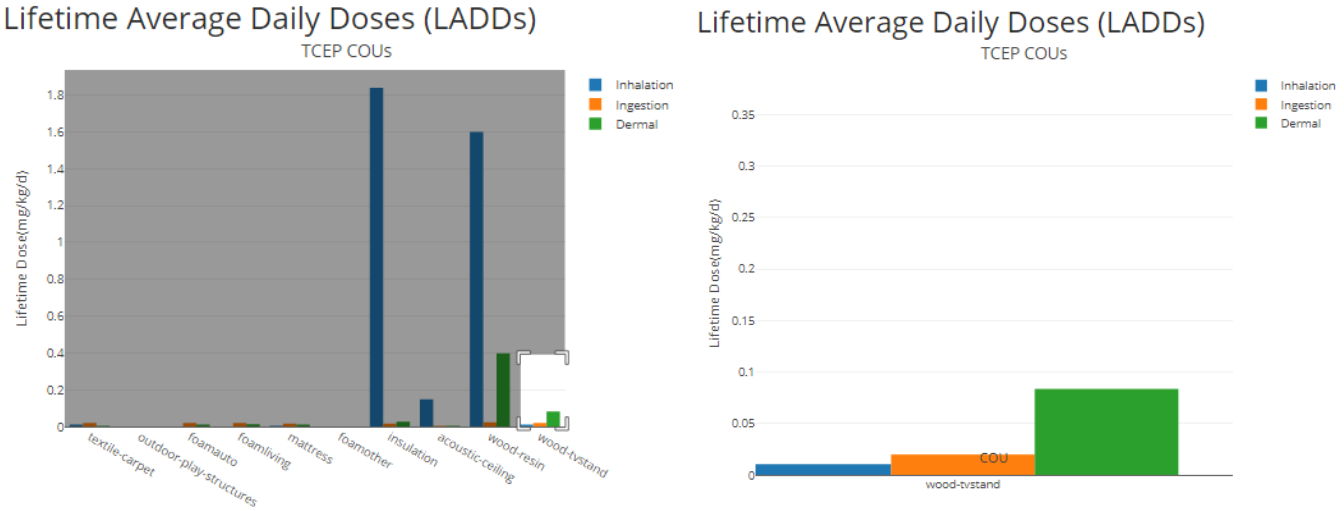
Figure_Apx J-2. Screenshot of Lifetime Average Daily Doses (LADDs) Bar Chart Displaying Function to Compare Data on Hover, for Insulation Estimates

Or to select and deselect data, the viewer can click the legend to remove data from the accompanying bar plot.



Figure_Apx J-3. Screenshot of Lifetime Average Daily Doses (LADDs) Bar Chart Displaying Bar Chart that Deselects Inhalation Estimate and Selects Ingestion and Dermal Estimates

Or the viewer can drag and select a certain section of the plot to view it in greater detail:



Figure_Apx J-4. Screenshots of Lifetime Average Daily Doses (LADDs) Bar Chart Displaying a Cropped Subsection of the Figure

J.1.3.1 CEM 3.2 User Guide and Appendices

The CEM 3.0 User Guide and appendices provide the underlying equations and default parameters that are used in CEM 3.2. The *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Consumer Exposure Modeling Inputs* (U.S. EPA, 2023) gives the inputs and assumptions used for consumer modeling.

Appendix K HUMAN HEALTH HAZARD DETAILS

K.1 Toxicokinetics and PBPK Models

K.1.1 Absorption

EPA did not identify *in vivo* human studies that evaluated absorption, distribution, metabolism, or elimination (ADME) of TCEP by any route of exposure.

Oral

Following oral exposures to radiolabeled TCEP, *in vivo* ADME studies in rats and mice found that TCEP is rapidly and extensively absorbed. More than 90 percent of ^{14}C-labeled TCEP was absorbed based on radioactivity found in urine, feces, volatiles, and CO_2 after 2 hours post-dose (Burka et al., 1991; Herr et al., 1991). In 5-week-old male Wistar rats, ^{14}C-labeled TCEP concentrations were measured in urine, feces, expired air, and body after oral exposure. Almost 100 percent of the 50 µmol/kg dose was recovered in urine, feces, expired air, and body (Minegishi et al., 1988). For input to the risk evaluation, EPA will assume that absorption is 100 percent.

Inhalation

EPA did not identify any *in vivo* animal data for absorption of TCEP by the inhalation route of exposure. For input to the risk evaluation, EPA will assume that absorption is 100 percent, equivalent to oral exposure.

Dermal

EPA did not locate any *in vivo* studies of dermal absorption in humans or animals but identified an *in vitro* study using excised human skin that evaluated the dermal absorption of TCEP (Abdallah et al., 2016).

Although no dermal *in vivo* toxicokinetic studies are available, EPA identified Abdallah et al. (2016), which measured dermal absorption using excised human skin in multiple *in vitro* experiments conducted according to OECD TG 428, *Skin Absorption: In Vitro Method.* The experiments used exposures of either 24 or 6 hours; acetone or 20 percent Tween 80 in water as the vehicle; 500 or 1,000 ng/cm^2 application to skin; and finite (depletable) or infinite dose. EPA gave each of the finite dose experiments overall quality determinations of medium. For the experiment that claimed to investigate an infinite dose, EPA assigned a low overall quality determination scenario, because conditions for infinite dosing (use of neat or large body of material) were not met and the results did not reflect steady-state flux throughout the experiment (*e.g.,* applied dose was depletable).

EPA used the 500 ng/cm^2 24-hour finite dose application in acetone (0.005 percent solution) to estimate absorption for workers because this was the only experiment for which the authors reported absorption at multiple time points. Because EPA assumes workers wash their hands after an 8-hour shift, EPA used the value of 16.5 percent, which is the amount of TCEP absorbed at 8 hours. In accordance with OECD Guidance Document 156 (OECD, 2022), EPA also added the quantity of material remaining in the skin (6.8%) at the end of the experiment as potentially absorbable.[47] Therefore, EPA assumes workers absorb 23.3 percent TCEP through skin and used this value to calculate risks for workers (see Section 5.1.1.3).

[47] EPA used 6.8 percent (the total amount remaining in skin after washing) because the authors did not conduct tape stripping.

For consumer exposures and exposure to soil scenarios that assume hand washing does not occur for 24 hours, EPA used the value at 24 hours (28.3%) plus the amount remaining in skin (6.8%) from the same experiment used for workers (500 ng/cm^2 24-hour finite dose application in acetone); total absorption was 35.1 percent absorption and was used to calculate risks (see Sections 5.1.2.2.3 and 5.1.3.3.2).

The estimates identified above apply to finite exposure scenarios for which the TCEP dose is depleted over time. For exposure scenarios such as swimming in which a maximum absorption rate is expected to be maintained (*i.e.*, the dose is not depletable during the exposure duration), EPA used the dermal permeability coefficient (K_p) of 2.2×10^{-2} cm/h derived by Abdallah et al. (2016) from the experiment that used the 24-hour 1,000 ng/cm^2 TCEP skin application to calculate risks (see Section 5.1.3.3.1).

U.S. EPA (2024s) presents quality determinations for individual experiments conducted by Abdallah et al. (2016), with EPA comments for each of the data quality metrics. Data extraction tables with details on methods and results of the experiments are also presented in U.S. EPA (2024s).

K.1.2 Distribution

Oral

TCEP distributes widely throughout the body. At 2 hours following the oral exposure, there was TCEP-derived ^{14}C in all brain regions of male and female rats. Also, the increasing levels of TCEP-derived ^{14}C were observed with increasing TCEP doses. There were no significant differences in TCEP-derived ^{14}C levels in blood and brain (including cerebellum, brainstem, caudate, hypothalamus, cortex, hippocampus, and midbrain) in male and female rats and 24 hours following a single dose. The concentration of ^{14}C-labeled TCEP in blood was significantly more increased with dose in males than females after 2 hours ($p < 0.05$). However, there was no significant difference in the amount of TCEP present in blood and all brain regions after 24 hours of exposure (Burka et al., 1991; Herr et al., 1991). Oral administration studies in rats by NTP found that TCEP produced sex-specific seizures and lesions in the hippocampal brain regions in some animals receiving the higher doses (NTP, 1991b). Results reported by Herr et al. (1991) observed similar sex-specific clinical signs of toxicity in animals receiving the higher doses. In an earlier study (Minegishi et al., 1988), rats treated orally with TCEP had the highest radioactivity in the kidney at 3, 6, and 12 hours, with similar or higher values in liver up to 168 hours (Minegishi et al., 1988).

Inhalation

No *in vivo* animal data evaluating the distribution of TCEP following inhalation route exposures were identified.

Dermal

EPA did not identify *in vivo* animal data that evaluated the distribution of TCEP following dermal route exposures.

K.1.3 Metabolism

Oral

TCEP is predominantly metabolized in the liver in laboratory animals and urinary excretion is the primary route of elimination for metabolites. In the liver, two pathways are involved in the metabolism of TCEP (Burka et al., 1991; Herr et al., 1991). First pass biotransformation occurs via oxidative and hydrolytic pathways. Some oxidative metabolites can undergo secondary biotransformation via the glucuronidation and alcohol dehydrogenase pathways. Burka et al. (1991) conducted a study to detect variations in metabolism of TCEP between male mice and male and female rats. The results showed that TCEP underwent extensive metabolism in all three groups. TCEP was excreted primarily in the form of

metabolites in urine and feces of both species and were identified as bis(2-chloroethyl) hydrogen phosphate (BCHP), the glucuronide of bis(2-chloroethyl) 2-hydroxyethyl phosphate (BCGP), and bis(2-chloroethyl) carboxymethyl phosphate (BCCP) (Burka et al., 1991). A synonym of BCHP is bis(2-chlorethyl) phosphate (BCEP), which is used in various sections of this risk evaluation (*e.g.*, 4.1.3.1 Measured Concentrations in Terrestrial Species and 5.1.3.5 Exposure Reconstruction Using Human Biomonitoring Data and Reverse Dosimetry). In other toxicological studies in rats and mice, TCEP has been shown to cause neurotoxicity at lower doses in females than in males (Yang et al., 2018a; NTP, 1991b; Matthews et al., 1990). Burka et al. (1991) examined whether there was any relationship between acute neurotoxicity and metabolism. Male and female rats were pretreated with aldehyde dehydrogenase inhibitors to alter the urinary metabolic profile. The relative amount of the hydrolytic metabolite (BCHP) was increased compared to the oxidative metabolite (BCCP). Because aldehyde dehydrogenase inhibitors interfere with the metabolic pathway leading to the oxidative metabolite (BCCP), increased levels of the reactive metabolite may possibly account for increased neurotoxicity (Burka et al., 1991).

Inhalation
No *in vivo* animal data for metabolism of TCEP by the inhalation route of exposure was identified.

Dermal
EPA did not identify *in vivo* animal data that evaluated metabolism of TCEP by the dermal route of exposure.

K.1.4 Elimination

Oral
TCEP is primarily eliminated in the urine following oral exposure. Burka et al. (1991) and Herr et al. (1991) reported that more than 75 percent of ^{14}C-labeled TCEP was eliminated in 24 hours for both rats and mice, with less than 10 percent excreted in feces (Burka et al., 1991). There was little to no sex-specific difference in the rate of elimination of TCEP for rats. However, male mice eliminated TCEP at 3 times the rate observed for rats during the first 8 hours (Burka et al., 1991). Urinary excretion is the primary route of elimination for metabolites (Burka et al., 1991; Herr et al., 1991). Minegishi et al. (1988) also reported that almost 100 percent of ^{14}C-labeled TCEP was eliminated, with 93.5 percent excreted in urine.

Inhalation
No *in vivo* animal data for metabolism of TCEP by the inhalation route of exposure was identified.

Dermal
EPA did not identify *in vivo* animal data that evaluated elimination of TCEP by the dermal route of exposure.

K.1.5 PBPK Modeling Approach

EPA did not identify any PBPK models specific to TCEP but is using the Verner Model (Verner et al., 2009; Verner et al., 2008) to predict TCEP concentrations in milk used to assess infant exposure through ingestion of human milk. The model is described in Appendix I.5.1.

K.2 Detailed Mode of Action Information

EPA has determined that TCEP is likely to cause tumors in kidneys under exposure circumstances relevant to human health. For blood cancer (mononuclear cell leukemia); thyroid cancer (follicular cell adenoma or carcinoma); Harderian gland cancer (adenoma or carcinoma); and liver cancer (hepatocellular adenomas or carcinomas), evidence of carcinogenicity is slight. EPA summarizes

biochemical, cellular, and mechanistic data that may be relevant to induction of kidney tumors—the target organ with the strongest weight of scientific evidence conclusion.

Although EPA did not specifically investigate other possible mechanisms related to other tumor types following TCEP exposure, conclusions for induction of kidney tumors may be relevant for induction of other tumors.

K.2.1 Mutagenicity

EPA did not identify *in vivo* studies that evaluated any of the following relevant effects specifically in kidneys, the target of tumors likely to be caused by TCEP: (1) oncogene or tumor suppressor gene mutations, (2) other gene mutations and chromosomal aberrations, (3) DNA adducts, or (4) DNA damage. However, one *in vivo* micronucleus assay in Chinese hamsters via intraperitoneal (i.p.) administration did identify the presence of micronuclei in bone marrow (Sala et al., 1982) and EPA considered this to be equivocal/weakly positive.[48] Also, the Agency did not identify any additional *in vivo* studies that evaluated DNA damage, DNA adducts or other measures of DNA damage and/repair in surrogate tissues.

Most bacterial reverse mutation assays using *Salmonella typhimurium* strains showed that TCEP was negative for direct gene mutations (Follmann and Wober, 2006; NTP, 1991b; Haworth et al., 1983; Prival et al., 1977; Simmon et al., 1977). TCEP was also negative in a study of forward gene mutations in Chinese hamster lung fibroblasts (Sala et al., 1982).[49]

However, Nakamura et al. (1979) identified positive dose-response trends in two *S. typhimurium* strains: in TA100, the response was less than 2-fold higher than the negative control at the highest non-toxic dose, but in TA1535 (with metabolic activation), TCEP induced an increase of more 4- to 7-fold over controls. It is not clear why the results of Nakamura et al. (1979) differed from other studies, but Nakamura et al. (1979) used Kanechlor 500 to induce enzymes in the S9 fraction whereas other studies used Aroclor 1254 or did not use a method to induce enzymes.

Two studies of TCEP induction of SCEs identified equivocal results in Chinese hamster ovary cells (positive in one of two trials with S9, negative without S9) and positive results without a dose-response in Chinese hamster lung fibroblasts (Galloway et al., 1987; Sala et al., 1982), suggesting some genetic damage. These results are not definitive for direct mutagenic effects because there is a lack of understanding of SCEs mechanism(s) of action (OECD, 2017).

TCEP was not considered to be an alkylating agent in an *in vitro* DNA binding assay (Lown et al., 1980).

Bukowski et al. (2019) conducted *in vitro* comet assays in peripheral mononuclear blood cells (PMBCs) and identified DNA damage at the highest concentration tested (1 mM); however, there is uncertainty regarding whether cytotoxicity occurred at this concentration. Another comet assay did not identify DNA damage in Chinese hamster fibroblasts at TCEP concentrations up to 1 mM with or without metabolic activation (Follmann and Wober, 2006).

[48] Two additional micronucleus tests in mice (one via the oral route and one via i.p.) were negative (Beth-Hubner, 1999) but the studies were not available for review by EPA.

[49] Beth-Hubner (1999) reported negative results in a reverse gene mutation assay using *Saccharomyces cerevisiae D4* and in two mouse lymphoma assays (using the thymidine kinase locus).

Sala et al. (1982) identified a high level of cell transformation in Syrian hamster embryo (SHE) cells but a lower level using C3H10T1/2 cells with metabolic activation. On page 24, OECD (2007) states that "cell transformation has been related to structural alterations and changes in the expression of genes involved in cell cycle control, proliferation and differentiation." The genomic changes may result from direct or indirect genetic interactions or non-genotoxic mechanisms.

EPA did not identify *in vitro* studies of DNA adducts.

Although there is uncertainty regarding reasons for equivocal/weakly positive results, EPA concludes that TCEP is not likely to induce tumors via a mutagenic MOA.

K.2.2 Other Modes of Action

Biochemical and mechanistic information that may suggest TCEP could act via MOAs other than a mutagenic MOA. Several *in vivo* and *in vitro* studies have evaluated tissue changes, gene transcription, and protein activities among other activities that identified tumor precursors or possible key events in mechanisms of tumor induction.

Taniai et al. (2012a) dosed male F344/NSIc rats daily via oral gavage with 0 or 350 mg/kg-bw/day TCEP and examined effects on proximal tubular epithelial cells of the outer stripe of the outer medulla (OSOM) of the kidney as well as the whole cortex. TCEP exposure resulted in scattered proximal tubular regeneration, likely associated with cells in the quiescent G0-phase of the cell cycle. TCEP did not induce karyomegaly (enlarged nuclei) in the tubular epithelia. TCEP also led to a significant increase in Ki-67 immunoreactive cells vs. controls ($p < 0.01$); Ki-67 nuclear antigen is a marker of cell proliferation expressed in cells in the G1 to M phase of the cell cycle. However, TCEP exposure did not result in aberrant expression of cell cycle-related molecules except for topoisomerase IIα (Topo IIα), which acts from the late S to G2 and M phase; TCEP significantly increased Topo IIα-immunoreactive cells in the cortex and OSOM ($p < 0.01$), which may signify increased cell proliferation (Taniai et al., 2012a). It is also possible that DNA damage may have been a precipitating factor in the increase of Topo IIα (Taniai et al., 2012a).

Using the same protocol (*i.e.*, male rats dosed via oral gavage at 0 or 350 mg/kg-day TCEP for 28 days), Taniai et al. (2012b) observed that TCEP exposure increased cells immunoreactive for markers of cell proliferation (Mcm3), apoptosis (Ubd), and deregulation of the G2/M phase of the cell cycle (TUNEL) ($p < 0.01$). Carcinogens that increase cell proliferation may increase cell populations undergoing M phase disruption that leads to chromosomal instability linked to cancer (Taniai et al., 2012b).

In vitro studies show that TCEP exposure of primary rabbit renal proximal tubule cells (PTCs) resulted in cytotoxicity, reduced DNA synthesis, altered expression of cell cycle regulatory proteins, and inhibition of ion- and non-ion-transport functions. Increased expression of pro-apoptotic regulatory proteins and decreased expression of proteins that inhibit apoptosis were also observed (Ren et al., 2012; Ren et al., 2009, 2008).
Additional *in vivo* and *in vitro* studies identified several biochemical changes in tissues and cell of other organs. Male ICR mice exposed to TCEP in the diet for 35 days exhibited increased markers of oxidative stress (hepatic antioxidant enzyme activities and their gene expression) in livers (Chen et al., 2015a). Liver cells or cell lines cultured with TCEP exhibited reduced viability, cell cycle arrest, cellular and mitochondrial oxidative stress, impaired mitochondrial function, and perturbation of cell signaling pathways (Mennillo et al., 2019; Zhang et al., 2017b; Zhang et al., 2017a; Zhang et al., 2016c; Zhang et al., 2016b). TCEP exposure of human peripheral blood mononuclear cells resulted in cytotoxicity (Mokra et al., 2018) and decreased DNA methylation (Bukowski et al., 2019).

In NTP (1991b), the authors reported no hyperplasia in rats at the 66-week interim sacrifice in the narrative (data tables not included). Although focal hyperplasia was observed and can be expected to be a precursor to tumors, the only related finding regarding kidney tumors at the 66-week sacrifice was a single renal tubule adenoma seen in a female rat. Therefore, evidence of temporal progression from hyperplasia to adenoma and then carcinoma is not available. At 2-years, hyperplasia was observed in male rats but incidence was slightly lower (0, 2, and 24) than adenomas (1, 5, and 24) compared with hyperplasia at 0, 44, and 88 mg/kg-day. The lack of temporality and limited information on pre-cursor lesions and their relationship with tumors leads to uncertainty regarding dose-response progression from hyperplasia to adenomas and carcinomas in males. Female rats did have higher rates of hyperplasia (0, 3, 16) than adenomas (0, 2, 5), at 0, 44, and 88 mg/kg-day, respectively.

K.2.3 Mode of Action Conclusions

EPA concluded that a mutagenic MOA is not likely from exposure to TCEP. Several studies have investigated biochemical and cellular changes in kidneys or renal cells that may be associated with steps in other MOAs for kidney cancer. However, EPA has not performed a formal analysis on postulated MOAs (*e.g.*, as in Sonich-Mullin et al. (2001)).

There is sparse information on temporality and dose-response of potential pre-cursor events within the *in vivo* studies and no clear NOAEL regarding tumor response to be able to model tumor incidence with a non-linear/threshold dose response analysis.

U.S. EPA's PPRTV (U.S. EPA, 2009) concluded that the overall weight of evidence for mutagenicity is negative and that no mechanistic data identify specific potential key events in an MOA for kidney or other tumors induced by TCEP exposure other than a general association with known proliferative and preneoplastic lesions.

K.3 Dose-Response Derivation

EPA evaluated data for health outcomes with the strongest weight of scientific evidence and from studies with sufficient sensitivity and adequate quantitative information to characterize the dose-response relationships of TCEP (see Section 5.2.5.1).

K.3.1 Adjustments for All PODs (Non-cancer and Cancer)

For TCEP, all data considered for PODs are obtained from oral animal toxicity studies in rats or mice. For consistency and easier comparison of sensitivity across health effects, EPA converted all doses to daily doses before conducting benchmark dose (BMD) modeling. For example, if the toxicity study dosed animals via gavage for five days per week at 22 mg/kg-day, EPA multiplied that value by 5/7 to obtain an equivalent daily value of 15.7 mg/kg-day. Studies in which animals were dosed every day did not require conversion. Any adjustments for different frequency of exposure (*e.g.*, 5 days per week for workers) are made in the exposure calculations specific to exposure scenarios.

Because toxicity values for TCEP are from oral animal studies, EPA must use an extrapolation method to estimate equivalent human doses (HEDs) and CSFs. The preferred method would be to use chemical-specific information for such an extrapolation. However, there are no TCEP-specific PBPK models and EPA did not locate other TCEP information to conduct a chemical-specific quantitative extrapolation. In the absence of such data, EPA relied on the guidance from U.S. EPA (2011b), which recommends scaling allometrically across species using the three-quarter power of body weight ($BW^{3/4}$) for oral data. Allometric scaling accounts for differences in physiological and biochemical processes, mostly related to kinetics.

For application of allometric scaling in risk evaluations, EPA uses dosimetric adjustment factors (DAFs), which can be calculated using Equation_Apx K-1.

Equation_Apx K-1. Dosimetric Adjustment Factor (DAF)

$$DAF = \left(\frac{BW_A}{BW_H}\right)\text{^}1/4$$

Where:

DAF	=	Dosimetric adjustment factor (unitless)
BW_A	=	Body weight of species used in toxicity study (kg)
BW_H	=	Body weight of adult human (kg)

U.S. EPA (2011b) presents DAFs for extrapolation to humans from several species. However, because those DAFs used a human body weight of 70 kg, EPA has updated the DAFs using a human body weight of 80 kg for the TCEP risk evaluation (U.S. EPA, 2011a). The Agency used the body weights of 0.025 and 0.25 kg for mice and rats, respectively, as presented in U.S. EPA (2011b). The resulting DAFs for mice and rats are 0.133 and 0.236, respectively.

For this risk evaluation, EPA assumes absorption for oral and inhalation routes is 100 percent and no adjustment was made when extrapolating to the inhalation route. This is supported by oral toxicokinetics data that shows greater than 90 percent absorption via the oral route (Burka et al., 1991).

K.3.2 Non-cancer Dose-Response Modeling

EPA concluded that TCEP *likely* causes neurotoxicity, reproductive, developmental, and kidney effects in humans under relevant exposure circumstances. For these outcomes (as well as *suggestive* evidence integration conclusions), EPA conducted BMD modeling (U.S. EPA, 2024c) and compared PODs among these two categories of evidence integration conclusion categories to determine the sensitivity of individual health affects (U.S. EPA, 2024k). Although EPA conducted BMD modeling for the non-cancer hazard outcomes with *suggestive* evidence integration conclusions, the focus of the evaluation was on the likely endpoints. Section 5.2.5.1 describes how EPA chose the sensitive studies and individual health effects within these health outcome categories for the non-cancer HED and HEC derivations.

As noted above, EPA converted doses for each study to daily doses before conducting BMD modeling. If data were not amenable to BMD modeling (*e.g.*, there was only one treatment group) or data did not fit BMD models, NOAELs or LOAELs were also converted to daily values, as needed.

K.3.2.1 Calculating Daily Oral Human Equivalent Doses (HEDs)

Use of allometric scaling for oral animal toxicity data to account for differences among species allows EPA to decrease the default intraspecies uncertainty factor (UF_A) used to set the benchmark margin of exposure (MOE); the default value of 10 can be decreased to 3, which accounts for any toxicodynamic differences that are not covered by use of $BW^{3/4}$. Using the appropriate DAF from Equation_Apx K-1, EPA adjusts the POD to obtain the daily HED as follows:

Equation_Apx K-2. Daily Oral HED

$$HED_{Daily} = POD_{Daily} \times DAF$$

Where:

HED_{Daily}	=	Human equivalent dose assuming daily doses (mg/kg-day)

POD_{Daily}	=	Oral POD assuming daily doses (mg/kg-day)
DAF	=	Dosimetric adjustment factor (unitless)

K.3.2.2 Use of Oral HED as Dermal HED

U.S. EPA (2011b) recommends the $BW^{3/4}$ approach only for oral PODs, and there is no established guidance for dosimetric adjustments of dermal PODs. However, EPA only extrapolated *between* species from oral animal toxicity values because the only acceptable data were from oral studies. EPA extrapolated to the dermal HED from the oral HED after the oral species extrapolation and accounted for differences in absorption in the dermal exposure estimate, not within the HEDs.

EPA used a value of 23.3 percent (hand washing after 8 hours) for workers as described in Section 5.1.1.3. EPA used a value of 35.1 percent (no handwashing for 24 hours) for dermal absorption in calculations of consumer exposure and exposure to soil, which are described in Sections 5.1.2.2.3 and 5.1.3.3.2, respectively. For dermal exposure from swimming (a nondepletable source), EPA uses the dermal permeability coefficient (Kp) of 2.2×10^{-2} cm/h as described in Section 5.1.3.3.1. The same uncertainty factors are used in the benchmark MOE for both oral and dermal scenarios.

K.3.2.3 Extrapolating to Inhalation Human Equivalent Concentrations (HECs)

For the inhalation route, EPA extrapolated the daily oral HEDs to inhalation human equivalent concentrations (HECs) using a human body weight and breathing rate relevant to a continuous exposure of an individual at rest, as follows:

Equation_Apx K-3. Extrapolating from Oral HED to Inhalation HEC

$$HEC_{Daily,\ continuous} = HED_{Daily} \times \left(\frac{BW_H}{IR_R * ED_C}\right)$$

Where:

$HEC_{Daily,continuous}$	=	Inhalation HEC based on continuous daily exposure (mg/m^3)
HED_{Daily}	=	Oral HED based on daily exposure (mg/kg-day)
BW_H	=	Body weight of adult humans (kg) = 80
IR_R	=	Inhalation rate for an individual at rest (m^3/h) = 0.6125
ED_C	=	Exposure duration for a continuous exposure (h/day) = 24

Based on information from U.S. EPA (2011a), EPA assumes an at rest breathing rate of 0.6125 m^3/hr. Adjustments for different breathing rates required for individual exposure scenarios are made in the exposure calculations, as needed.

It is often necessary to convert between ppm and mg/m^3 due to variation in concentration reporting in studies and the default units for different OPPT models. Therefore, EPA presents all PODs in equivalents of both units to avoid confusion and errors. Equation_Apx K-4 presents the conversion of the HEC from mg/m^3 to ppm.

Equation_Apx K-4. Converting Units for HECs (mg/m^3 to ppm)

$$X\ ppm = Y\ \frac{mg}{m^3} \times \frac{24.45}{MW}$$

Where:

24.45 = Molar volume of a gas at standard temperature and pressure (L/mol), default
MW = Molecular weight of the chemical

K.3.2.4 TCEP Non-cancer HED and HEC Calculations for Acute Exposures

Moser et al. (2015) identified neurotoxicity in pregnant female rats at 125 mg/kg-day via oral gavage in a prenatal study. The POD is based on a NOAEL of 40 mg/kg-day (tremors within a few days of dosing). EPA used Equation_Apx K-1 to determine a DAF specific to rats (0.236), which was in turn used in the following calculation of the daily HED using Equation_Apx K-2:

$$9.46\ \frac{mg}{kg-day} = 40\ \frac{mg}{kg-day} \times 0.236$$

EPA then calculated the continuous HEC for an individual at rest using Equation_Apx K-3:

$$51.5\ \frac{mg}{m^3} = 9.46\ \frac{mg}{kg-day} \times \left(\frac{80\ kg}{0.6125\ \frac{m^3}{hr} * 24\ hr}\right)$$

Equation_Apx K-4 was used to convert the HEC from mg/m^3 to ppm:

$$4.41\ ppm = 51.5\ \frac{mg}{m^3} \times \frac{24.45}{285}$$

K.3.2.5 TCEP Non-cancer HED and HEC Calculations for Intermediate and Chronic Exposures

Chen et al. (2015a) identified decreased numbers and degeneration of seminiferous tubules in male mice in a 35-day study in which TCEP was administered in the diet. This endpoint is directly applicable to intermediate exposure scenarios and because it is more sensitive than endpoints from the chronic studies, EPA also uses it for chronic exposure scenarios. The POD is based on a $BMDL_5$ of 21.0 mg/kg-day. EPA used Equation_Apx K-1 to determine a DAF specific to rats, which was in turn used in the following calculation of the daily HED using Equation_Apx K-2:

$$2.79\ \frac{mg}{kg} = 21.0\ \frac{mg}{kg} \times 0.133$$

EPA then calculated the continuous HEC for an individual at rest using Equation_Apx K-3:

$$15.2\ mg/m^3 = 2.79\ mg/kg \times \left(\frac{80\ kg}{0.6125\ \frac{m^3}{hr} * 24\ hr}\right)$$

Equation_Apx K-4 was used to convert the HEC from mg/m^3 to ppm:

$$1.30\ ppm = 15.2\ \frac{mg}{m^3} \times \frac{24.45}{285}$$

K.3.3 Cancer Dose-Response Modeling

EPA concludes that TCEP is *likely to be carcinogenic to humans* based on considerations outlined in U.S. EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b). EPA modeled the dose

response for the target organ with the most robust data—kidney tumors. For tumors in several other target organs, see the evidence integration tables in Appendix L.

K.3.3.1 Calculating Daily Oral CSFs

Like non-cancer data, all cancer data are obtained from oral animal toxicity studies (NTP, 1991b). Because an MOA has not been established for TCEP, EPA assumed linear low dose extrapolation (U.S. EPA, 2005b). EPA conducted BMD modeling of kidney tumors for both male and female rats to obtain the CSF for TCEP (U.S. EPA, 2024c). EPA adjusted the CSF using the DAF (see Equation_Apx K-1) to account for allometric scaling between species. Equation_Apx K-5 shows the calculation to obtain the DAF-adjusted CSF:

Equation_Apx K-5.

$$CSF_{Human,Daily} = CSF_{Animal,Daily}/DAF$$

Where:

$CSF_{Human,Daily}$	=	Human equivalent daily oral cancer slope factor (mg/kg-day^{-1})
$CSF_{Animal,\ Daily}$	=	Animal daily oral cancer slope factor (mg/kg-day^{-1})
DAF	=	Dosimetric adjustment factor (unitless)

Because EPA has not concluded that TCEP acts via a mutagenic MOA, an age-dependent adjustment factor (ADAF) (U.S. EPA, 2005c) was not applied. The Agnecy did not use CSFs for combined tumors (across multiple target organs) for the risk evaluation but focused on the tumors with the most robust evidence from the animal data.

K.3.3.2 Use of Oral CSF as Dermal CSF

The $BW^{3/4}$ approach is only recommended for oral toxicity data extrapolation, and there is no established guidance for dosimetric adjustments of dermal PODs. In the absence of available guidance, and when the dermal CSFs are extrapolated from oral CSFs that incorporated $BW^{3/4}$ scaling, EPA uses the oral CSF for the dermal route of exposure because it has already been converted to a human dose. The Agency accounts for dermal absorption in the dermal exposure estimate, which can then be directly compared to this HED. Sections 5.1.2.2.3 and 5.1.3.3.2 describe how EPA uses dermal absorption in calculations of consumer exposure and exposure to soil, respectively; Section 5.1.1.3 describes dermal exposure for workers; and Section 5.1.3.3.1 describes dermal exposure from swimming (an infinite, nondepletable source).

K.3.3.3 Extrapolating to Inhalation Unit Risks (IURs)

For the inhalation route, EPA extrapolated the daily oral HEDs to inhalation HECs using a human body weight and breathing rate relevant to a continuous exposure of an individual at rest. For this risk evaluation, EPA assumes absorption for oral and inhalation routes is equivalent and no adjustment was made when extrapolating from the oral to the inhalation route. The equation to convert to the inhalation route is as follows:

Equation_Apx K-6. Extrapolating from the Oral CSF to an Inhalation IUR

$$IUR_{Human,continuous} = CSF_{Human,daily} \times \left(\frac{IR_R * ED_C}{BW_H}\right)$$

Where:

$IUR_{Human,\ continuous}$	=	Human equivalent continuous daily inhalation unit risk ((mg/m^3)$^{-1}$)
$CSF_{Human,\ daily}$	=	Human equivalent daily oral cancer slope factor (mg/kg-day^{-1})

IR_R	=	Inhalation rate for an individual at rest (m^3/hr) = 0.6125
ED_C	=	Exposure duration for a continuous exposure (hr/day) = 24
BW_H	=	Body weight of adult humans (kg) = 80

Based on information presented in U.S. EPA (2011a), EPA assumes an at rest breathing rate of 0.6125 m^3/hr.

EPA may need to convert between mg/m^3 and ppm due to variation in concentration reporting in studies and the default units for different OPPT models. Therefore, all PODs are presented in equivalents of both units to avoid confusion and errors. Equation_Apx K-7 identifies how to convert the IUR from $(mg/m^3)^{-1}$ to $(ppm)^{-1}$.

Equation_Apx K-7. Converting Units for IURs (mg/m³ to ppm)

$$X\ per\ ppm = Y\ per\ \frac{mg}{m^3} \times \frac{MW}{24.45}$$

Where:

24.45	=	Molar volume of a gas at standard temperature and pressure (L/mol), default
MW	=	Molecular weight of the chemical

K.3.3.4 CSF and IUR Calculations for Lifetime Exposures

The most sensitive CSF was estimated as a risk of 0.0058 per mg/kg-day using BMD modeling software to model the dose-response for renal tubule adenomas and carcinomas in male rats from the NTP (1991b) 2-year cancer bioassay. EPA then used this CSF and the rat-specific DAF (0.24) (Equation_Apx K-1) to obtain a human relevant CSF using Equation_Apx K-5. The calculations specific to TCEP are as follows:

$$0.0245\ per\ \frac{mg}{kg} = 0.0058\ per\ \frac{mg}{kg}\ /0.236$$

Using Equation_Apx K-6, EPA converted the oral CSF to an IUR:

$$0.00451\ per\ \frac{mg}{m3} = 0.0245\ per\ mg/kg \times \left(\frac{0.6125\ m^3/hr * 24\ hr}{80\ kg}\right)$$

EPA used Equation_Apx K-7 to convert the IUR from units of mg/m^3 to ppm:

$$0.0526\ per\ ppm = 0.00451\ per\ \frac{mg}{m^3} \times \frac{285}{24.45}$$

Appendix L EVIDENCE INTEGRATION FOR HUMAN HEALTH OUTCOMES

This appendix presents evidence integration tables for the major health outcomes associated with TCEP (see Table_Apx L-1 through Table_Apx L-6). It also presents a section with short evidence integration summaries for health outcomes with limited data (Section L.2).

L.1 Evidence Integration Tables for Major Human Health Hazard Outcomes

Table_Apx L-1. Evidence Integration for Neurotoxicity

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Evidence integration summary judgment on neurotoxicity				
Evidence in studies of exposed humans considered for deriving toxicity values				*Overall judgment for neurological/ behavioral effects based on integration of information across evidence streams:* Evidence indicates that TCEP likely causes neurological/ behavioral effects in humans under relevant exposure circumstances.
• Associations between BCEP urine levels (metabolite of TCEP) in pregnancy and birth outcomes: Percy et al. (2021) examined whether prenatal exposure of suspected neurotoxicants was associated with any child cognition measures. Maternal urinary BCEP was associated with a modest increase in child full scale IQ. Percy et al. (2022) examined impairment in cognitive abilities a longitudinal cohort study. Urinary BCEP concentrations at ages 1-5 years and cognitive abilities at 8 years were assessed for in. The BCEP concentrations association with IQ was small and positive. Hernandez-Castro et al. (2023a) evaluated associations of prenatal exposures to organophosphate esters and child neurobehavior. The authors did not find an association between urinary BCEP levels and neurobehavioral outcomes. Foster et al. (2024) examined whether pregnant women exposed to organophosphate esters that also included tris(2-chloroethyl) phosphate (TCEP) in home dust was associated with higher depression and stress levels across prenatal and postpartum time periods. The authors observed small increases in maternal perceived stress levels, but not depression, after adjusting for covariates.	Consistency: Prenatal exposure to TCEP is associated with impairment in cognitive abilities for multiple measures of low SES. Biological plausibility and relevance to humans: Neurodevelopmental effects measured in humans	Consistency: Increase and decrease of IQ The epidemiological studies evaluated participants' single spot urine samples	*Key findings:* Available epidemiological studies resulted in some changes in cognitive functions in children associated with maternal urine concentrations of BCEP as well as TCEP in home dust. Overall judgment for neurological effects based on human evidence: • Slight	
Evidence from *in vivo* mammalian animal studies considered for deriving toxicity values				

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
NTP studies (Matthews et al., 1993; NTP, 1991b; Matthews et al., 1990). Rats and mice exposed by gavage; evaluated brain/hippocampal lesions, clinical signs of toxicity, serum cholinesterase activity. Overall quality determination: High Brain/hippocampal lesions (histopathology) (16 weeks, and two years [rats only]) • Female rats: brain weight decrease observed at the highest dose. • Male rats: necrosis of the neurons of the hippocampus, • Female rats: necrosis of the neurons of the hippocampus. Neuronal necrosis was also observed in the thalamus. • Female rats: in over 40% of female rats receiving the highest dose showed focal gliosis, hemorrhage, mineralization, and pigmentation, and hemosiderin in the brain stem and cerebellum after 2 years. Clinical signs of toxicity (16 days, and16 weeks) • Female rats: occasionally appeared hyperactive and exhibited resistance to handling. Seizures were observed during week 12 of dosing. • Male rats: no clinical signs of toxicity were observed in male rats. • Male and female mice exhibited convulsive movements and reduced ability to keep balance during the first three days of dosing at the two highest doses. Serum cholinesterase activity • Female rats: serum cholinesterase activity was decreased at the highest doses after 14 days. • Female rats: serum cholinesterase activity in female rats receiving the higher were 75% and 59%, respectively, of the control animals. The 88	Effect size/precision: • Histopathology, serum cholinesterase activity, behavioral changes in female rats were significantly increased over controls. Dose-response gradient: • Decrease in serum cholinesterase activity appears to increase with dose in female rats. Incidences of brain histopathology findings increased with dose in male and female rats. Consistency: • Brain weight, brain/hippocampal lesions, clinical signs of toxicity, serum cholinesterase activity, and behavioral findings were observed in female rats across different studies. Coherence across endpoints: • Signs of neurotoxicity and neurobehavioral effects corresponded to histopathology changes in female rats.	Consistency: • Effects seen primarily in female rats in some studies. • Neurodevelopmental limited effects seen only in male offspring at the highest dose but only in the presence of severe maternal toxicity.	*Key findings*: Results across available animal toxicological studies showed neurotoxicity in female rats in a dose-response manner and limited effects in male rat offspring showed effects at the highest dose in a prenatal study. *Overall judgment for neurotoxicity based on animal evidence:* • Robust	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
mg/kg/day animals were decreased 9.3% compared to control animals. • There were no treatment-related effects on serum cholinesterase activity in both male and female mice Tilson et al. (1990). 1-day gavage study in rats; evaluated hippocampal lesions and behavioral findings. Overall quality determination: High • Treatment produced consistent damage to CA1 pyramidal cells with lesser damage to CA4, CA3, and CA2 pyramidal cells. Significant damage was also seen in dentate granule cells. • Treated rats were mildly impaired in the acquisition of the water maze task that had a reference memory component. However, in the repeated acquisition task, the rats were clearly deficient. Yang et al. (2018a). 60-day gavage study in rats; evaluated clinical signs of toxicity hippocampal lesions, and behavioral findings. Overall quality determination: High Clinical signs of toxicity • Occasional periods of hyperactivity and periodic convulsions in female rats. There were not treatment-related effects observed in male rats Behavioral findings • Remarkably higher escape latencies to find the hidden platform than the vehicle controls ($p <$ 0.01). Significantly shorter cumulative distances from the original platform than the controls. Significantly fewer cross-times were noted in the highest dose for female rats. Male rats were not tested.				

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Hazleton Laboratories (1983), a single dose during GD 7-14. Overall quality determination: High • There was a low incidence of maternal animals with clinical signs of OP toxicity (up to 2/50 animals on GD 7-14). *Developmental Neurotoxicity.* Moser et al. (2015) Overall quality determination: High Assessment of neurobehavioral and related hormonal responses after dosing pregnant Long-Evans rats from GD 10 through PND 22 via oral gavage up to 90 mg/kg-day. No TCEP-related adverse effects in T3, T4, brain or serum AChE in dams or offspring. In addition, no effects on brain weight in offspring at PND 6 and sporadic behavioral changes do not suggest biologically relevant adverse outcomes or developmental toxicity. Kawashima et al. (1983) Overall quality determination: Medium Pregnant Wistar rats gavaged with 0, 50, 100, or 200 mg/kg-day from GD 7 through 15. Twenty-three percent of 30 dams at 200 mg/kg-day died between GD 10 and 14. At 200 mg/kg-day, male offspring exhibited decreased numbers of rearings (9.8 vs. 19.3 in controls; $p < 0.01$) and took longer during the learning ability test (water maze performance) in the last of four trials ($p < 0.05$). However, this dose group was associated with significant maternal toxicity. No effects were observed in female offspring in these tests or in either sex in other functional/neurobehavioral tests.				
Evidence in mechanistic studies and supplemental information				
In vivo: Yang et al. (2018a). Compared to those in the control, the major metabolites that had increased in the aqueous	• None	• None	*Overall judgment for neurotoxicity based on mechanistic evidence:*	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
phase of TCEP-treated groups were N-acetyl aspirate (NAA), glutamine (GLU), glutamic acid, glucose, taurine, choline, creatine, and myo-inositol levels, whereas those that had decreased were lactate, g-amino butyric acid (GABA), glycine, and two unknown compounds. In the lipid phase, the major metabolites that were different between the control and TCEP-treated groups were cholesterol ester and glycerol, which were increased, whereas free cholesterol, total cholesterol, lipid (CH2CH2CO), fatty acid, polyunsaturated fatty acid, and phosphatidylcholine levels were decreased.			• Indeterminate	

Table_Apx L-2. Evidence Integration for Reproductive Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Evidence integration summary judgment on reproductive effects				
Evidence in studies of exposed humans considered for deriving toxicity values (none)				*Overall judgment for reproductive effects based on integration of information across evidence streams:* Evidence indicates that TCEP exposure likely causes reproductive effects in humans under relevant exposure circumstances.
Evidence from apical endpoints in *in vivo* mammalian animal studies considered for deriving toxicity values				
• Short-term, subchronic, and chronic gavage studies in male and female rats and mice and a subchronic dietary study in male mice examined testes weight and/or histology of the reproductive organs NTP (1991b) and Chen et al. (2015a). Overall quality determination: High • The Reproductive Assessment by Continuous Breeding (RACB) Protocol[50] was used to evaluate fertility, litters/pair, live pups/litter, proportion of pups born alive, sex of live pups, pup weights at birth, sperm morphology, vaginal cytology, and/or reproductive organ weights and histology in mice treated via gavage (NTP, 1991a). Overall quality determination: High.	• Biological gradient/dose-response: The magnitude and severity of histological changes in the testes (changes in the number and appearance of seminiferous tubules) increased with increasing dose in the subchronic dietary study in ICR mice. • Fertility index, number of litters/pair decreased in a dose-related manner during the continuous F0 breeding phase of the RACB. Consistency: • Decreased testes weight was observed in gavage and dietary subchronic studies in mice. • Decreased fertility index was observed during continuous F0 breeding and crossover mating phases of the RACB. • Sperm effects (decreases on sperm concentration and percent motile sperm, increased sperm abnormalities) identified during crossover mating correlated with	Consistency: • Changes in testes histology were observed in a subchronic dietary study in ICR mice, but no histological changes to reproductive organs were observed in short-term, subchronic, or chronic gavage studies in F344 rats and CD-1 and B6C3F1 mice. Quality of the database: • Testes weights were assessed in subchronic, but not chronic, NTP studies in rats and mice.	*Key findings*: Available animal toxicological studies showed decreased testes weight, histological changes in the testes of ICR mice, sperm effects, and/or reduced fertility and fecundity. *Overall judgment for reproductive effects based on animal evidence:* • Moderate	

[50] The RACB protocol consists of 4 phases: (1) dose range-finding, (2) continuous (F0) breeding, (3) crossover mating; and (4) assessment of fertility in F1 offspring.

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
	decreased fertility index when treated males were bred with untreated females. • Mechanistic changes from *in vivo* and *in vitro* studies (decreased testicular testosterone, altered gene expression related to steroidogenesis, and decreased testosterone secretion) are consistent with observed effects on testes and sperm. Quality of the database: • Effects were observed in high-quality studies.			
Evidence in mechanistic studies and supplemental information				
• A subchronic dietary study in male mice evaluated testicular testosterone and gene expression related to testosterone synthesis (Chen et al., 2015a). • An *in vitro* study using TM3 Leydig cells evaluated testosterone secretion and gene expression related to steroidogenesis and oxidative stress (Chen et al., 2015b). • Three *in vitro* studies evaluated estrogenic, anti-estrogenic, androgenic, and/or anti-androgenic activity using a yeast reporter assay or human (endometrial, prostate and breast) cancer cell lines (Krivoshiev et al., 2016; Reers et al., 2016; Follmann and Wober,	Biological gradient/dose-response: • *In vivo* data showed decreased testicular testosterone and altered gene expression related to testosterone synthesis at the dose in which decreased testes weight and testicular damage were observed. • An *in vitro* study showed decreased testosterone secretion and/or changes in gene expression related to steroidogenesis and oxidative stress at both tested concentrations. Consistency: • Altered gene expression related to steroidogenesis	Consistency: • There was inconsistency across studies with respect to estrogen receptor and androgen receptor agonist and/or antagonist activity in human (endometrial, prostate, and breast) cancer cell lines. Quality of the database: • Few potential mechanisms were investigated in available studies. Biological plausibility/relevance to humans: • Oxidative stress is a nonspecific mechanism.	*Key findings*: Limited available mechanistic data indicate that TCEP may induce oxidative stress and endocrine disruption via altered expression of genes involved in steroidogenesis. *Overall judgment for reproductive effects based on mechanistic evidence:* • Slight	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
2006).	correlated with decreased testosterone *in vivo* and *in vitro*. Biological plausibility/relevance to humans: • Endocrine disruption, via altered expression of genes involved in testosterone synthesis, is a plausible mechanism for infertility, sperm effects, and testicular damage that is relevant to humans.			
GD = gestation day				

Table_Apx L-3. Evidence Integration for Developmental Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Evidence integration summary judgment on developmental effects				
Evidence in studies of exposed humans considered for deriving toxicity values (none)				*Overall judgment for developmental effects based on integration of information across evidence streams:* Evidence suggests but is not sufficient to conclude that TCEP causes developmental effects in humans under relevant exposure circumstances.
• Associations between BCEP urine levels (metabolite of TCEP) in pregnancy and birth outcomes: -U.S. cohort of 6,646 mother/ infant pairs that investigated gestational age (GA), birthweight (BW) (Oh et al., 2024) Overall quality determinations: Medium -Los Angeles, CA cohort of 421 mother/infant pairs – evaluation of GA, BW (Hernandez-Castro et al., 2023b) Overall quality determinations: Medium - Cincinnati, Ohio cohort of 340 mom/infant pairs and GA, pre-term birth, BW, length, ponderal index, head circumference (Yang et al., 2022) Overall quality determinations: Medium - Rhode Island cohort 56 mom/infant pairs and skinfold thickness, BW, length, head and abdominal circumferences, feeding behavior at birth and 6 wks (Crawford et al., 2020) Overall quality determinations: High	Quality of the database: High/medium Consistency: Males had lower incidence of being small for their gestational age and had increased scapular skinfold thickness, which could both suggest some increase in fat. Two studies showed statistically significant effects on gestational age and/or birthweight and length (increased GA for both sexes, decreased for females). Biological gradient/dose-response: Individual measures showed dose-response gradients; Yang et al. (2022) identified differences in multiple measures for each log10 unit increase in BCEP Crawford et al. (2020) showed increases in some skinfold thickness measures. Biological plausibility and relevance to humans: The types of outcomes are seen in humans across a variety of populations.	Consistency: One of the larger cohorts (Hernandez-Castro et al., 2023b) did not show effects on gestational age or birth weight. Others showed inconsistent effects between males, females, and when both sexes were evaluated together. Crawford et al. (2020) also showed only changes in some skinfold thickness measures but not infant length or weight, whereas females showed changes in weight and length in another study (Yang et al., 2022). Differences in effects among sexes were observed. Biological gradient/dose-response: The dose-response was not always maintained across high, low, and non-detect categories (*e.g.*, SGA for males, which showed a slightly greater change between the low dose group and controls vs. high dose and controls for males) and the females showed only a statistically significant change for low dose vs. control for LGA – large for gestational age (Oh et al., 2024).	*Key findings*: Available epidemiological studies resulted in some changes in gestational age and growth. *Overall judgment for developmental effects based on animal evidence:* • Slight	
Evidence from apical endpoints in *in vivo* mammalian animal studies considered for deriving toxicity values				

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
• An oral gavage study evaluated uterine parameters, number of pups, pup weight, and viability following gestational exposure (GDs 7-14) in female mice (Hazleton Laboratories, 1983). Overall quality determination: High • Assessment of neurobehavioral and related hormonal responses after dosing pregnant Long-Evans rats from GD 10 through PND 22 via oral gavage up to 90 mg/kg-day. No adverse effects in T3, T4, brain or serum AChE in dams or offspring. No effects on brain weight in offspring at PND 6. Sporadic behavioral changes do not suggest biologically relevant adverse outcomes or developmental toxicity. Moser et al. (2015). Overall quality determination: High • Assessment of malformations and variations Kawashima et al. (1983) Overall quality determination: Medium • Pregnant Wistar rats gavaged with 0, 50, 100, or 200 mg/kg-day from GD 7 through 15. Twenty-three percent of 30 dams at 200 mg/kg-day died between GD 10 and 14. At 200 mg/kg-day, male offspring exhibited some neurotoxicity but this was associated with significant maternal toxicity. No other developmental effects (*e.g.*, malformations, variations, growth effects) were observed.	• Biological gradient/dose-response: number of litters/pair and number of live pups/litter decreased in a dose-related manner during the continuous F0 breeding phase of the RACB. • *Supporting reproductive effects:* Magnitude and severity of testes histological changes increased with dose in the subchronic dietary study in ICR mice. Consistency: • Decreased numbers of live pups/litter were observed during continuous F0 breeding and crossover mating phases of the RACB. • Decreased number of live pups/litter was observed at the same dose in F0 and F1 breeding phases of the RACB, with greater severity in the second generation. Consistency of supporting reproductive effects: • Decreased testes weight was observed in gavage and dietary subchronic studies in mice. • Sperm effects identified during crossover mating correlated with decreased fertility index when treated males were bred with untreated females. • Mechanistic changes from *in vivo* and *in vitro* studies	Consistency: • The developmental neurotoxicity study in rats (Moser et al., 2015) did not result in effects in offspring. The pre-natal study of malformations and variations in addition to neurotoxicity (Kawashima et al., 1983) did not result in any significant effects in offspring. **Note:** It is possible that exposure prior to gestation and in male parents is needed for effects to be observed. Magnitude and precision: • The developmental gavage studies in mice used only one dose group and no developmental effects were observed (Hazleton Laboratories, 1983).	*Key findings*: Available animal toxicological studies resulted in decreased live pups per litter. *Overall judgment for developmental effects based on animal evidence:* • Slight	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
	(decreased testosterone, altered steroidogenesis gene expression) consistent with effects on testes and sperm. Quality of the database: • Effects were observed in high-quality studies.			
Evidence in mechanistic studies and supplemental information				
• Yonemoto et al. (1997) evaluated inhibitory concentrations for cell proliferation (IP_{50}) and differentiation (ID_{50}) in rat embryo limb bud cells. • *Reproductive:* A subchronic dietary study in male mice evaluated testicular testosterone and gene expression related to testosterone synthesis (Chen et al., 2015a). • An *in vitro* study using TM3 Leydig cells evaluated testosterone secretion and gene expression related to steroidogenesis and oxidative stress (Chen et al., 2015b). • Three *in vitro* studies evaluated estrogenic, anti-estrogenic, androgenic, and/or anti-androgenic activity using a yeast reporter assay or human (endometrial, prostate and breast) cancer cell lines (Krivoshiev et al., 2016; Reers et al., 2016; Follmann and Wober, 2006).	Biological gradient/dose-response (reproductive effects): • *In vivo* data showed decreased testicular testosterone and altered gene expression related to testosterone synthesis at the dose in which decreased testes weight and testicular damage were observed. • An *in vitro* study showed decreased testosterone secretion and/or changes in gene expression related to steroidogenesis and oxidative stress at both tested concentrations. Consistency (Reproductive): • Altered gene expression related to steroidogenesis correlated with decreased testosterone *in vivo* and *in vitro*. Biological plausibility/relevance to humans: • Yonemoto et al. (1997) identified an IP_{50} of 3600 µM of TCEP using rat embryo	Consistency (Reproductive): • There was inconsistency across studies with respect to estrogen receptor and androgen receptor agonist and/or antagonist activity in human (endometrial, prostate, and breast) cancer cell lines. Quality of the database: • Few potential mechanisms were investigated in available studies. Biological plausibility/relevance to humans: • Oxidative stress is a possible nonspecific mechanism.	*Key findings*: Limited available mechanistic data indicate that TCEP may induce a ratio of inhibition of proliferation and differentiation resulting in concern for development; oxidative stress; and endocrine disruption via altered expression of genes involved in steroidogenesis. *Overall judgment for developmental effects based on mechanistic evidence:* • Slight	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
	limb bud cells. The ID_{50} was 1570 µM; the ratio of concentrations suggested possible developmental toxicity. • *Reproductive:* Endocrine disruption, via altered expression of genes involved in testosterone synthesis, is a plausible mechanism for infertility, sperm effects, and testicular damage that is relevant to humans.			
GD = gestation day				

Table_Apx L-4. Evidence Integration Table for Kidney Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Evidence integration summary judgment on kidney effects				
				Overall judgment for renal effects based on integration of information across evidence streams: Evidence indicates that TCEP exposure likely causes kidney effects in humans under relevant exposure circumstances.
•Associations between BCEP urine levels (metabolite of TCEP) in pregnancy and birth outcomes: Kang et al. (2019) examined whether exposure to organophosphate esters which also included TCEP was associated with chronic kidney disease. There was association between BCEP levels and chronic kidney disease related parameters (eGFR and ACR).	None	None	Key findings: Based on one epidemiological study resulted in some changes in chronic kidney disease related parameters. Overall judgment for kidney effects based on human evidence: • Slight	
Evidence from *in vivo* mammalian animal studies considered for deriving toxicity values				
NTP (1991b): Rats and mice exposed by gavage; evaluated kidney weights and histopathology. Overall quality determination: High Kidney weights (16 days, 16 weeks, and 66 weeks [rats only]) • Male rats: increased kidney weights at all time points. • Female rats: no change after 16 days, dose-related increases in kidney weights after 16 weeks, and no change after 66 weeks. • Male mice: no change after 16 days and decreased kidney weight after 16 weeks. • Female mice: increased kidney weight after 16 days and no change after 16 weeks. Histopathology (16 days, 16 weeks, and 104 weeks) • No changes in rats or mice after 16 days or in rats after 16 weeks.	Effect size/precision: • Histopathology changes in rats and mice of both sexes were significantly increased over controls by both pairwise and trend tests. Dose-response gradient: Incidences of kidney histopathology findings increased with dose in rats and mice of both sexes. Temporality: Histopathology findings were more prevalent and occurred at lower doses as	Inconsistency Kidney weight changes did not occur at all time points in female rats or mice of either sex. Incoherence: Kidney weight changes did not correspond to histopathology changes in female rats or mice of either sex. Imprecision: • Dosing errors occurred in 16-week studies in rats and mice. • Treatment-related deaths occurred in 16-week study in rats.	*Key findings*: Results across available animal toxicological studies showed renal toxicity in rats and mice. *Overall judgment for renal effects based on animal evidence:* • Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
• Male rats: renal tubule hyperplasia and renal tubule adenomas after 104 weeks at 88 mg/kg/day; one adenoma occurred as early as 66 weeks at 88 mg/kg/day; increase in combined adenomas or carcinomas at 88 mg/kg/day (see also Table_Apx L-6 for cancer endpoints). • Female rats: renal tubule hyperplasia and renal tubule adenomas after 104 weeks at 88 mg/kg/day (see also Table_Apx L-6 for cancer endpoints). • Male mice: epithelial cytomegaly after 16 weeks at 700 mg/kg-day; karyomegaly after 104 weeks at ≥ 175 mg/kg-day; one adenocarcinoma and three adenomas at 350 mg/kg-day (see also Table_Apx L-6 for cancer endpoints). • Female mice: epithelial cytomegaly after 16 weeks at 700 mg/kg-day; karyomegaly after 104 weeks at ≥ 175 mg/kg-day. Taniai et al. (2012a) 28-day gavage study in rats; evaluated histopathology. Overall quality determination: Medium Histopathology Male rats: scattered proximal tubular regeneration in the cortex and outer stripe of the outer medulla (OSOM) at 350 mg/kg-day.	exposure duration increased. Consistency: Renal histopathology changes were observed in rats and mice of both sexes and in studies in two different laboratories. Coherence across endpoints: Kidney weight changes corresponded to histopathology changes in male rats.	• Survival was decreased at the high dose in both sexes of rat in 104-week study.		
Evidence in mechanistic studies and supplemental information				
In vivo: Markers for cell proliferation and apoptosis (Taniai et al., 2012b) and regenerating tubules (Taniai et al., 2012a) were increased in kidneys (OSOM and cortex) of rats after 28 days (gavage) *In vitro*: TCEP exposure of primary rabbit renal proximal tubule cells (PTCs) resulted in cytotoxicity, reduced DNA	Dose response gradient: Across the *in vitro* studies, dose-related changes in the endpoints were observed. Consistent with related apical endpoints: Results from mechanistic studies	Imprecision/Inconsistency: • There are few studies of mechanistic endpoints in the kidneys. • *In vitro* studies used only one cell model and	*Key findings*: Apoptosis and altered cell cycle regulation may contribute to renal effects of TCEP in animals. *Overall judgment for renal effects based on*	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
synthesis, altered expression of cell cycle regulatory proteins, and inhibition of ion- and non-ion-transport functions. Increased expression of pro-apoptotic regulatory proteins and decreased expression of proteins that inhibit apoptosis were also observed (Ren et al., 2012; Ren et al., 2009, 2008).	are consistent with *in vivo* histopathology findings in the renal tubules.	all were conducted in the same laboratory.	*mechanistic evidence:* • Slight	

Table_Apx L-5. Evidence Integration Table for Liver Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Evidence integration summary judgment on liver effects				
Evidence in studies of exposed humans considered for deriving toxicity values (none)			• Indeterminate	*Overall judgment for liver effects based on integration of information across evidence streams:* Evidence suggests but is not sufficient to conclude that TCEP causes hepatic effects in humans under relevant exposure circumstances.
Evidence from apical endpoints in *in vivo* mammalian animal studies for deriving toxicity values				
• NTP (1991b): Subchronic and chronic gavage studies in rats and mice that examined liver weights, clinical chemistry, and histopathology. Overall quality determination: High • One 35-day dietary exposure study in male mice that examined liver weights (Chen et al., 2015a). Overall quality determination: High	Biological gradient/dose-response: • A dose-related trend in hepatocellular adenoma was observed in male mice in the chronic study. • Increases in liver weights in male rats occurred at lower doses as duration increased. • Dose-related increases in liver weights were seen in female rats and female mice at 16 weeks and in male rats at 66 weeks. Quality of the database:	Magnitude and precision: • The incidence of eosinophilic foci in male mice was statistically significantly increased at only the top dose after 2 years. Consistency: • There were no histopathology findings in rats or female mice, including no hypertrophy. • Liver weight increases were seen in female rats after 16 days and 16 weeks, but not 66 weeks of exposure. • Increased liver weight was not seen in the 35-day study.	*Key findings*: Available animal toxicological studies showed increased liver weights in rats and mice in the absence of relevant clinical chemistry findings; histopathology changes in the liver were observed only in male mice. *Overall judgment for liver effects based on animal evidence:*	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
	• Effects observed in high-quality studies.	• No biologically relevant changes in serum enzymes were seen in the 2-year bioassay and not measured in shorter studies. Quality of the database: • Liver weights were not assessed in mice exposed longer than 16 weeks.	• Slight	
Evidence in mechanistic studies and supplemental information				
• One *in vivo* 35-day dietary exposure study in male mice examining markers of oxidative stress (Chen et al., 2015a). • Five *in vitro* studies examining viability, cell cycle, cellular and mitochondrial oxidative stress, mitochondrial function, and cell signaling pathways in liver cells and/or cell lines (Mennillo et al., 2019; Zhang et al., 2017b; Zhang et al., 2017a; Zhang et al., 2016c; Zhang et al., 2016b).	Biological gradient/dose-response: • *In vivo* data showed induction of hepatic oxidative stress occurring earlier than apical endpoints. • Across the *in vitro* studies, dose-related changes in viability, oxidative stress, and impaired mitochondrial functioning were observed. Biological plausibility/relevance to humans: • Oxidative stress is a plausible mechanism for eosinophilic foci and tumor formation that is relevant to humans.	Quality of the database: • Few potential mechanisms were investigated in available studies. Biological gradient/dose response: • Oxidative stress was demonstrated *in vivo* at higher doses than those associated with liver lesions in chronic study. Biological plausibility/relevance to humans: • Oxidative stress is a nonspecific mechanism and was seen only at doses higher than those associated with liver lesions.	*Key findings*: Limited available mechanistic data indicate that TCEP may induce oxidative stress, alter cellular energetics, and/or influence cell signaling related to proliferation, growth, and survival in the liver. *Overall judgment for liver effects based on mechanistic evidence:* • Slight	

Table_Apx L-6. Evidence Integration Table for Cancer

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Evidence integration summary judgment on cancer				
Evidence in studies of exposed humans considered for deriving toxicity values				*Overall judgment for cancer effects based on integration of information across evidence streams:* EPA concludes that TCEP is likely to be carcinogenic to humans using guidance from U.S. EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b).
Hoffman et al. (2017) Case-control study of thyroid cancer and TCEP in household dust. Overall quality determination: High • Significant increase in adjusted OR (2.42) for TCEP (in dust) above median level among papillary thyroid cancer cases compared to controls. TCEP in dust in homes associated with more aggressive tumors (n = 70 cases, 70 controls) Liu et al. (2022) case-control study of thyroid cancer and TCEP in serum. Overall quality determination: Medium • No significant associations between TCEP in serum and papillary thyroid cancer when comparing quartiles of exposure. Li et al. (2020) association between TCEP in plasma and prevalence of gastro-intestinal and colorectal cancers (all stages). Overall quality determination: Medium • TCEP was detected in GI and colorectal cancer patients more than controls but there were no other associations (n = 34 cases GI cancer, 40 cases colorectal cancer, 62 controls). Liu et al. (2021) TCEP in plasma and female-related cancers. Overall quality determination: Low • No positive associations between TCEP and cancers vs. benign tumors (n= 45 benign breast tumors, n= 73 breast cancer,	Biological Plausibility • Thyroid cancers also reported in female rats exposed to TCEP orally.	Quality of the database: • One epidemiological study of cancer (high-quality); no studies of renal cancers in humans. Biological gradient/dose-response: • Exposure was measured after outcome • TCEP concentrations higher for benign breast tumors compared with breast cancer Magnitude and Precision • Dust used as proxy for TCEP exposure; in on study, corresponding biological samples were not collected to match with dust samples Consistency • Studies using different TCEP exposure measures and in different geographic regions showed different results for papillary thyroid cancer. • Studied cancers not consistently associated with TCEP	*Key findings*: Available epidemiological study of cancer was limited. *Overall judgment for cancer effects based on human evidence:* • Indeterminate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
n = 62 for benign uterine tumors, n = 78 for cervical cancer)				
Evidence from apical endpoints in in vivo mammalian animal studies				
Kidney cancer				
NTP (1991b): F344 rats and B6C3F1 mice exposed by gavage for 104 weeks. Overall quality determination: High • Increased incidences of adenomas and adenomas or carcinomas in male rats (one adenoma occurred at week 66) and increased incidences of adenomas in female rats.	Quality of the database: • Evidence in high-quality study in rats and mice Magnitude and precision: • Significant pairwise comparisons in male and female rats. • Renal tubule tumors are rare in F344/N rats and B6C3F1 mice. Biological gradient/dose-response: • Significant dose-related trends in male and female rats. Consistency: • Effects seen in both sexes of rat.	Magnitude and precision: • Survival was decreased at the high dose in both sexes of rat in 104-week study. Consistency: • No significantly increased incidence of tumors was seen in two strains of female mice or in male B6C3F1 mice.	*Key findings*: Dose-related increased renal tumor incidences demonstrated in a high-quality study in rats of both sexes *Overall judgment for kidney cancer effects based on animal evidence:* • Robust	
Mononuclear cell leukemia				
NTP (1991b): Overall quality determination: High • Increased incidence of mononuclear cell leukemia (MNCL) in male and female rats • No increased incidence of MNCL or other hematologic cancer in male or female mice	Quality of the database: • Evidence in high-quality studies in rats and mice. Magnitude and precision: • Significant pairwise comparisons in male and female rats. Biological gradient/dose-response:	Magnitude and precision: MNCL is common in F344 rats, its spontaneous incidence varies widely, and incidences in male rats exposed to TCEP were within historical controls. Biological plausibility/relevance to humans:	*Key findings*: Dose-related increases in MNCL incidences demonstrated in a high-quality study in rats of both sexes, but this is a common spontaneous cancer in rats and only the	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
	• Significant dose-related trends in male and female rats. Consistency: • Evidence in two sexes.	Occurrence of MNCL is rare in mice and other strains of rats (Thomas et al., 2007). MNCL may be similar to large granular lymphocytic leukemia (LGLL) in humans (Caldwell et al., 1999; Caldwell, 1999; Reynolds and Foon, 1984), particularly an aggressive form of CD3- LGL leukemia known as aggressive natural killer cell leukemia (ANKCL) (Thomas et al., 2007). However, Maronpot *et al.* (2016) note that ANKCL is extremely rare with less than 98 cases reported worldwide, and the authors contend that ANKCL has an etiology related to infection with Epstein-Barr virus, not chemical exposure.	incidence in high dose female rats was outside the historical control range. *Overall judgment for hematopoietic system cancer effects based on animal evidence:* • Slight	
Thyroid cancer				
NTP (1991b): Overall quality determination: High • Nonsignificant increase in incidence of follicular cell adenoma or carcinoma in male rats. • Significantly increased incidences of follicular cell carcinomas and adenoma or carcinoma in female rats. • No increased incidence of thyroid tumors in male or female mice.	Quality of the database: • Evidence in high-quality studies in rats and mice. Magnitude and precision: • Significant pairwise comparison in female rats. Biological gradient/dose-response: • Significant dose-related trend in female rats; borderline significant trend in males.	Magnitude and precision: • Survival was decreased at the high dose in both sexes of rat in 104-week study. Consistency: • Effect seen in only one species (rats). Biological plausibility/relevance to humans: U.S. EPA (1998a) and Dybing and Sanner (1999) concluded that rodents are more sensitive	*Key findings*: Dose-related increases in thyroid follicular cell tumor incidences were demonstrated in a high-quality study in female rats. Rodents may be more sensitive than humans to thyroid follicular cell tumors.	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
	Consistency: Effect seen in both sexes of rats.	than humans to thyroid follicular tumors induced by thyroid-pituitary gland disruption and thyroid stimulating hormone (TSH) hyperstimulation. NTP (1991b) did not measure TSH in the chronic rat study.	*Overall judgment for thyroid cancer effects based on animal evidence:* • Slight	
Harderian gland cancer				
NTP (1991b): Overall quality determination: High • Increased incidence of adenoma or carcinoma in female mice (when interim sacrifice groups included); no increased incidence of Harderian gland tumors in rats or male mice.	Quality of the database: • Evidence in high-quality studies in rats and mice.	Magnitude and precision: • Increased incidence of tumors in female B6C3F1 mice was statistically significant only when interim sacrifice groups were included. Biological gradient/dose-response: • Increased incidence in female B6C3F1 mice occurred only at highest tested dose. Consistency • No increased incidence of tumors in male B6C3F1 mice, or rats of either sex.	*Key findings*: Increased tumor incidence was only seen in one sex of one species (female B6C3F1 mice). *Overall judgment for Harderian gland cancer effects based on animal evidence:* • Slight	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Liver cancer				
NTP (1991b): Overall quality determination: High • Dose-related trend for adenomas, borderline significant increase in male mice at high dose; no effects on female mice or rats of either sex.	Quality of the database: • Evidence in high-quality studies in rats and mice. Biological gradient/dose-response: • Significant dose-related trend in male B6C3F1 mice.	Magnitude and precision: • Increased incidence of adenomas in male B6C3F1 mice was not statistically significant by pairwise comparison. Consistency • No increase in liver tumor incidence in female mice or in rats of either sex.	*Key findings*: Dose-related trend in tumor incidence was seen only in one sex of one species (male B6C3F1 mice). *Overall judgment for liver cancer effects based on animal evidence:* • Slight	
Evidence in mechanistic studies and supplemental information				

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
Genotoxicity *In vivo*: • Weakly positive/equivocal for micronucleus induction in Chinese hamsters (Sala et al., 1982). *In vitro*: • Positive for bacterial mutagenicity in one *S. typhimurium* strains, and weakly positive in another (Nakamura et al., 1979). • Negative for bacterial mutagenicity in several studies using multiple strains of *S. typhimurium* with and without metabolic activation (Follmann and Wober, 2006); negative for mutagenicity and DNA strand breaks in hamster V79 cells (Follmann and Wober, 2006; Sala et al., 1982). • Positive for SCEs in hamster V79 cells (Sala et al., 1982) and DNA strand breaks in human PBMCs (Bukowski et al., 2019). • Positive/weak positive for cell transformation (may not be a genotoxic mechanism) in two cell types (Sala et al., 1982)	Quality of the database: • Tests of bacterial mutagenicity in multiple strains, large concentration range, and assays with and without metabolic activation.	Quality of the database: • Few studies in mammalian cells and limited *in vivo* data. Magnitude and precision/ Biological gradient/dose-response: • Few positive findings, lack of information on cytotoxicity in at least one and weak/equivocal in one. Consistency: • DNA strand break findings were not consistent across studies/cell types.	*Key findings*: Available data indicate that TCEP has little genotoxic potential. Limited available data indicate that TCEP may induce oxidative stress, alter cellular energetics, and/or influence cell signaling related to proliferation, growth, and survival in kidney, liver, and blood cells. *Overall judgment for cancer effects based on mechanistic evidence:* • Slight	
Other (non-genotoxic) mechanistic studies[a] *Kidney*: • Markers for cell proliferation and apoptosis (Taniai et al., 2012b) and regenerating tubules (Taniai et al., 2012a) were increased in kidneys (OSOM and cortex) of rats after 28 days (gavage) • TCEP exposure of primary rabbit renal proximal tubule cells (PTCs) resulted in cytotoxicity, reduced DNA synthesis, altered expression of cell cycle regulatory proteins, and inhibition of ion- and non-	Biological gradient/dose-response: • Across the *in vitro* studies, dose-related changes were observed.	Quality of the database: • There are few studies in relevant tissue types and only two *in vivo* studies. • Available studies were not directly focused on cancer mechanisms.		

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and within-Stream Strength of the Evidence Judgment	Inferences across Evidence Streams and Overall Evidence Integration Judgment
ion-transport functions. Increased expression of pro-apoptotic regulatory proteins and decreased expression of proteins that inhibit apoptosis were also observed (Ren et al., 2012; Ren et al., 2009, 2008). *Hematopoietic:* • TCEP exposure of human peripheral blood mononuclear cells resulted in cytotoxicity (Mokra et al., 2018) and decreased DNA methylation (Bukowski et al., 2019). *Liver:* • Markers of oxidative stress (hepatic antioxidant enzyme activities and their gene expression) were increased in the livers of male ICR mice after 35 days of dietary exposure to TCEP (Chen et al., 2015a). • Liver cells and/or cell lines cultured with TCEP exhibited reduced viability, cell cycle arrest, cellular and mitochondrial oxidative stress, impaired mitochondrial function, and perturbation of cell signaling pathways (Mennillo et al., 2019; Zhang et al., 2017b; Zhang et al., 2017a; Zhang et al., 2016c; Zhang et al., 2016b).				
[a] No tissue-specific mechanistic data related to harderian gland or thyroid follicular cell cancers were identified in the available literature.				

L.2 Evidence Integration Statements for Health Outcomes with Limited Data

Skin and Eye Irritation

The human evidence is indeterminate for skin and eye irritation. The two readily available dermal irritation studies in animals showed inconsistent results and the single eye irritation study of medium-quality showed that TCEP is not irritating; these studies are indeterminate. Although one study was uninformative, EPA considered that these results are not affected by the lack of statistical analysis. Overall, the currently available evidence is inadequate to assess whether TCEP causes irritation in humans.

Mortality

Human evidence is indeterminate for mortality because there are no human epidemiological studies. There is modest evidence in animal studies that shows higher mortality in rats than mice in oral studies at the same doses and uncertain potential for mortality via the dermal route given conflicting results. Overall, evidence suggests but is not sufficient to conclude that TCEP exposure causes mortality in humans under relevant exposure circumstances. This conclusion is based on oral studies in rats and mice that assessed dose levels between 12 and 700 mg/kg-day and dermal studies in rabbits at approximately 279 and 556 mg/kg-day.

Immune/Hematological

The human evidence for immune effects based on associations between mothers' exposure during gestation and children's respiratory symptoms is slight. Animal studies did not identify histopathological changes in immune-related organs or in hematological parameters. A statistically significant increased trend in mononuclear cell leukemia with increasing dose was seen in rats. In mechanistic studies, TCEP was associated with decreases in an inflammatory cytokine and altered gene expression of inflammatory proteins in two studies, but a third study identified inflammatory changes only after co-exposure with benzo[a]pyrene.

Available evidence is indeterminate and therefore, is inadequate to assess whether TCEP may cause immunological or hematological effects in humans under relevant exposure circumstances.

Thyroid

Hoffman et al. (2017) identified an association between TCEP exposure and thyroid cancer in humans and NTP (1991b) identified increased incidences of thyroid neoplasms in rats in a 2-year cancer bioassay but with uncertainty regarding its association with TCEP exposure. However, Moser et al. (2015) found no changes in serum thyroid hormone levels in rat dams and offspring in a prenatal/postnatal study. Based on these data, human evidence for thyroid effects is slight and animal evidence is indeterminate. Overall, the currently available evidence is inadequate to assess whether TCEP may cause thyroid changes in humans under relevant exposure circumstances.

Endocrine (Other)

Based on indeterminate human and animal evidence and lack of mechanistic support, the currently available evidence is inadequate to assess whether TCEP may cause endocrine changes other than thyroid and reproductive hormones in humans.

Lung/Respiratory

Two human studies that examined TCEP's association with lung function measures showed inconsistent results, and the human evidence is indeterminate. In addition, animal data are indeterminate (no relevant

histopathological effects, lung weight changes in studies with high and uninformative overall data quality determinations) based on high-quality studies. Therefore, the currently available evidence is inadequate to assess whether TCEP may cause lung or respiratory effects in humans under relevant exposure circumstances.

Body Weight

EPA identified no human studies that had information on body weight changes and therefore, human evidence is indeterminate. In animal toxicity studies, TCEP effects on body weight were not consistent across multiple studies. When body weight changes were observed, they were not consistently increased or decreased. Therefore, the animal data are indeterminate. Overall, the currently available evidence is inadequate to assess whether TCEP may cause changes in body weight in humans under relevant exposure circumstances.

Appendix M GENOTOXICITY DATA SUMMARY

Table_Apx M-3 summarizes the database of studies on chromosomal aberrations, gene mutations, and other genotoxicity endpoints for TCEP. Although EPA did not evaluate these studies using formal data quality criteria, selected studies were reviewed by comparing against current OECD test guidelines and important deviations are noted below. When interpreting the results of these studies, EPA also consulted OECD (2017).

EPA did not retrieve all original studies for one or more of the following reasons: (1) they were not readily available, (2) they were in a foreign language, (3) they evaluated effects other than chromosomal aberrations or gene mutations, and (4) there were multiple studies of the same type (*e.g.*, bacterial reverse mutation assays). EPA also referred to some studies cited in the 2009 *European Union Risk Assessment Report* (ECB, 2009) and Beth-Hubner (1999) for some studies that were not obtained.

M.1.1 Chromosomal Aberrations

EPA located one *in vivo* micronucleus assay using Chinese hamsters (Sala et al., 1982) that was equivocal/weakly positive for micronuclei. Two additional *in vivo* micronucleus studies in mice cited in ECB (2009) and Beth-Hubner (1999) were not readily available. EPA also identified an *in vitro* assay that did not find chromosomal aberrations to be associated with TCEP exposure in Chinese hamster ovary cells (Galloway et al., 1987).

M.1.1.1 *In Vivo* Data

Sala et al. (1982) report results of an *in vivo* micronucleus assay in which Chinese hamsters were treated with a single i.p. dose at 0, 62.5, 125, or 250 mg/kg bw and bone marrow was evaluated for presence of micronuclei. The authors conducted a Student's T-test to determine whether the means differed between dose groups and the DMSO negative control. In females, the two lowest doses exhibited a statistically significant increase in micronuclei compared with controls. Males had increased micronuclei at the highest dose. However, only two hamsters per sex per dose were used, which would have made statistical significance difficult to detect. When results for both sexes were combined, the two highest doses showed differences from controls (see Table_Apx M-1). The authors also conducted linear regression to evaluate the dose response but did not report those results. The authors describe the results as a slight effect that is difficult to interpret due to different responses between sexes and "variation with the doses." EPA conducted a comparison of the means of each sex for each of the doses and considered the dose-response for the combined sexes to be valid.

The study methods deviated from OECD TG 474 (OECD, 2016b) in several ways. Specifically, the authors used an exposure route that is not recommended and scored fewer erythrocytes than recommended (2,000 vs. 4,000). Furthermore, the study did not provide information to ensure that the test substance reached the bone marrow, although positive effects suggest TCEP likely reached the target tissue (Sala et al., 1982). In addition, when using both sexes, the guidelines recommend using five animals per sex, not two per sex. Despite these deviations, some of which might decrease the ability to detect a response (*e.g.*, numbers of animals/sex and number of erythrocytes scored, lack of verification that the chemical reached the bone marrow), the results are consistent with an equivocal/ weak positive response.

The 2009 *European Union Risk Assessment Report* (ECB, 2009) and Beth-Hubner (1999) reference two additional micronucleus studies that reported negative results. The cited studies were an oral study using NMRI mice with dosing for one time at 1,000 mg/kg and an i.p. injection study with doses up to 700 mg/kg using CD-1 mice (ECB, 2009).

Table_Apx M-1. Results of *In Vivo* Micronucleus Test

Dose (mg/kg-bw)	Mean (Standard Deviation)[b c d]		
	Males	Females	Both Sexes
0[a]	4 (1.3)	3 (0.58)	3.5 (1.0)
62.5	4 (0.82)	6.5 (1.4)*	5.25 (1.4)
125	6.25 (1.1)	7.0 (1.3)**	6.63 (1.1)***
250	7.25 (0.35)*	6.75 (3.0)	7.0 (2.0)**

[a] DMSO solvent control (2,200 mg/kg-bw); * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$
[b] Standard deviation is in parentheses is equal to the standard error reported in the study × square-root of n (2/sex/dose for individual sexes and 4/dose for combined sexes)
[c] Number of micronuclei per 1,000 polychromatic erythrocytes
[d] Comparison of sexes for each does was done with the following program that compared means: https://www.medcalc.org/calc/comparison_of_means.php; the *p* values for 0, 62.5, 125, and 250 mg/kg were 0.4252, 0.1612, 0.5969, and 0.8367, demonstrating that outcomes were not significantly different between the sexes and the results could be combined.
Source: Sala et al. (1982)

M.1.1.2 *In Vitro* Data

Galloway et al. (1987) evaluated chromosomal aberrations in Chinese hamster ovary cells. Many study methods were consistent with OECD TG 473 (OECD, 2016a), except that the authors scored only 100 cells per concentration compared with the recommended 300 per concentration needed to conclude that a test is clearly negative. Aberrations at 0, 160, 500 and 1,600 µg/mL were observed in 6, 10, 10 and 9 percent of cells without activation, respectively, and 4, 10, 7 and 8 percent with activation. Neither trend test was statistically significant ($p \leq 0.05$).

M.1.2 Gene Mutations

A forward gene mutation study using Chinese hamster lung fibroblasts (Sala et al., 1982) and multiple bacterial reverse gene mutation assays (Follmann and Wober, 2006; Haworth et al., 1983; BIBRA, 1977; Prival et al., 1977; Simmon et al., 1977) were all negative for the induction of gene mutations. Beth-Hubner (1999) also reported negative results in a reverse gene mutation assay yeast and in two mouse lymphoma assays. A single study (Nakamura et al., 1979) induced a 4- to 7-fold increase in gene mutations in one *Salmonella typhimurium* strain with metabolic activation and less than a doubling in a second strain.

M.1.2.1 *In Vitro* Studies

Sala et al. (1982) evaluated the effect of TCEP exposure in a forward gene mutation assay that measured induction of 6-thioguanine-resistant mutants using Chinese hamster lung fibroblasts (V79 cells) in the presence and absence of metabolic activation. The authors used a negative control (acetone) as well as two positive controls. Although the incubation times and solvents followed the recommendations of OECD TG 476 (OECD, 2016c), the experiment did not report use of an enzyme-inducing agent for the S9 fraction and the S9 fraction was used at 20 percent (vs. ≤10% as recommended by OECD TG 476). The experiment also employed three instead of a recommended four concentrations. Furthermore, it is not clear whether the OECD TG 476 recommended 20×10^6 cells were grown by the time the cells were treated with TCEP. The positive control run without S9 was not one of the OECD TG 476 recommended controls. TCEP exposure did not result in increased mutations with or without S9; the authors noted that the results were confirmed in several independent experiments.

TCEP tested negative for gene mutations in many bacterial reverse mutation assays using multiple *S. typhimurium* strains (Follmann and Wober, 2006; Haworth et al., 1983; Prival et al., 1977; Simmon et al., 1977) (see Table_Apx M-3). Beth-Hubner (1999) references two additional studies that reported negative results in reverse mutation assays using *S. typhimurium* strains TA98, TA100, TA1535, TA1537, and TA1538.

A single study (Nakamura et al., 1979) identified increased mutations using *S. typhimurium* TA100 both with and without metabolic activation and for TA1535 in the presence of metabolic activation (Table_Apx M-3). In *S. typhimurium* TA100, none of the concentrations showed a doubling of revertants compared with the negative control response. However, the TA1535 response was approximately 4 times greater than controls at 3 µM (≈860 µg/plate) and more than 7 times higher at 10 µM (≈2,900 µg/plate) (Nakamura et al., 1979). The study did not present statistical analyses. Therefore, EPA modeled the dose-response to confirm the findings. It is not clear why the Nakamura et al. (1979) results were inconsistent with other studies. Concentrations were comparable to other studies that showed negative results. One difference in this study compared with others is in the method of enzyme induction used to prepare the S9 fraction; Nakamura et al. (1979) used a mixture of PCBs (Kanechlor 500) for this induction, whereas others used Aroclor 1254 or did not appear to induce enzymes in the S9 fractions.

Table_Apx M-2. Results of Bacterial Reverse Mutation Test in *Salmonella typhimurium*

Concentration (µMol)	His+ Revertants/Plate			
	TA100		TA1535	
	−S9	+S9	−S9	+S9
0	141	140	9	14
1	158	191	14	31
3	161	192	8	57
10	172	246	6	107
30	8	86	1	7
Source: Nakamura et al. (1979)				

None of the bacterial reverse mutation assays used *Escherichia coli* WP2 uvrA or *E. coli* WP2 uvrA (PKM101), which should more likely identify oxidizing or alkylating mutagens than the *Salmonella* strains used in the majority of TCEP studies. However, Follmann and Wober (2006) did test TCEP using *S. typhimurium* TA102, which can also identify such mutagens, and found that TCEP did not induce reverse mutations with this strain.

Beth-Hubner (1999) also reported negative results in a reverse gene mutation assay using *Saccharomyces cerevisiae D4* and in two mouse lymphoma assays (using the thymidine kinase locus).

M.1.3 Other Genotoxicity Assays

Table_Apx M-3 summarizes two sister chromatid exchange (SCE) assays (Galloway et al., 1987; Sala et al., 1982), *in vitro* comet assays measuring DNA damage and repair (Bukowski et al., 2019; Follmann and Wober, 2006), two cell transformation assays (Sala et al., 1982), and a DNA binding assay using TCEP (Lown et al., 1980). Beth-Hubner (1999) also summarized an eye mosaic test (somatic mutation and recombination) using *Drosophila melanogaster*.

These assays test for potentially harmful effects on genetic material such as DNA damage, cell transformation, DNA alkylation and chromosomal damage. However, unlike gene mutation and chromosomal aberrations studies, the changes measured in these assays may not be persistent and transmissible.

Two studies of TCEP induction of SCEs identified equivocal results in Chinese hamster ovary cells (positive in one of two trials with S9, negative without S9) and positive results without a dose-response in Chinese hamster lung fibroblasts (Galloway et al., 1987; Sala et al., 1982), suggesting some genetic damage, but without an understanding of the mechanism of action for this damage. The OECD test guideline related to evaluation of SCEs (OECD TG 479) was deleted in 2014 because the mechanism for this effect is not known (OECD, 2017).

TCEP was not considered to be an alkylating agent in an *in vitro* DNA binding assay (Lown et al., 1980).

Bukowski et al. (2019) conducted *in vitro* comet assays (alkaline and neutral) in peripheral mononuclear blood cells (PMBCs) and identified DNA damage at the highest concentration of TCEP tested (1 mM). Cell toxicity was not evaluated in the study, but previous results identified viability of PMBCs to be 92 percent of controls at 1 mM TCEP. DNA damage to the PMBCs was repaired within 2 hours (Bukowski et al., 2019). Another comet assay did not identify DNA damage in Chinese hamster fibroblasts at TCEP concentrations up to 1 mM with or without metabolic activation (Follmann and Wober, 2006).

Sala et al. (1982) identified a high level of cell transformation in Syrian hamster embryo (SHE) cells but a lower level with metabolic activation when using C3H10T1/2 cells. On page 24, OECD (2007) states that "cell transformation has been related to structural alterations and changes in the expression of genes involved in cell cycle control, proliferation and differentiation." The genomic changes may result from direct or indirect genetic interactions or non-genotoxic mechanisms. Tamokou and Kuete (2014) notes that the SHE assay is believed to detect early steps in the process of carcinogenesis, and that C3H10 cell assays related to later changes.

Taniai et al. (2012a) found no statistically significant increase in immunoreactive cells associated with repair of double-strand DNA double-strand breaks or regulation of cell cycle checkpoints after such DNA damage in kidneys of male rats dosed with 350 mg/kg-day TCEP for 28 days.

Table_Apx M-3. TCEP Genotoxicity Studies

Test Type	Exposure		Metabolic Activation	Positive Controls	Outcome	Reference(s)
	Species (Sex)/ Route	Concentration/Dose/ Duration				
Chromosomal aberrations – *in vivo*						
Micronucleus	Chinese hamsters (M+F)/ intraperitoneal	0, 62.5, 125, 250 mg/kg Single administration	NA	Yes	Equivocal, weakly positive for micronuclei	Sala et al. (1982)
Chromosomal aberrations – *in vitro*						
Chromosomal aberrations	Chinese hamster ovary cells	0, 160, 500, 1600 µg/mL 12 hr without activation 2 hr with activation	± S9 from rat livers induced with Aroclor 1254	Yes	Negative for chromosomal aberrations	Galloway et al. (1987) and NTP (1991b)
Gene mutations – *in vitro*						
Mammalian cell forward mutation assay (6-thioguanine-resistant mutants)	Chinese hamster lung fibroblasts (V79 cells)	500, 1,000, 2,000 µg/mL; no mention of cytotoxicity	± S9 from rat livers (not induced)		Negative for mutagenicity (both +/- S9); full results shown only for – S9	Sala et al. (1982)
Bacterial reverse mutation assay (pre-incubation assay)	*Salmonella typhimurium* strains TA97a, TA98, TA100, TA102, TA104, TA1535, TA1537, TA1538	100 nM to 1 mM	± S9	Yes	Negative for mutagenicity	Follmann and Wober (2006)
Bacterial reverse mutation assay (pre-incubation assay)	*Salmonella typhimurium* strains TA98, TA100, TA1535, TA1537	0, 10, 33, 100, 333, 1,000, 3,333 µg/plate	± S9 from rat and hamster livers induced by Aroclor 1254	Yes, dependent on bacterial strain	Not mutagenic up to toxic doses; trials showed toxicity/slight toxicity at the highest dose	Haworth et al. (1983) and NTP (1991b)
Bacterial reverse mutation assay	*Salmonella typhimurium*	0, 1, 3, 10, 30 µM/plate	± S9 from Kanechlor 500 (PCB)	Not identified	Positive in TA100 and TA1535.	Nakamura et al. (1979)

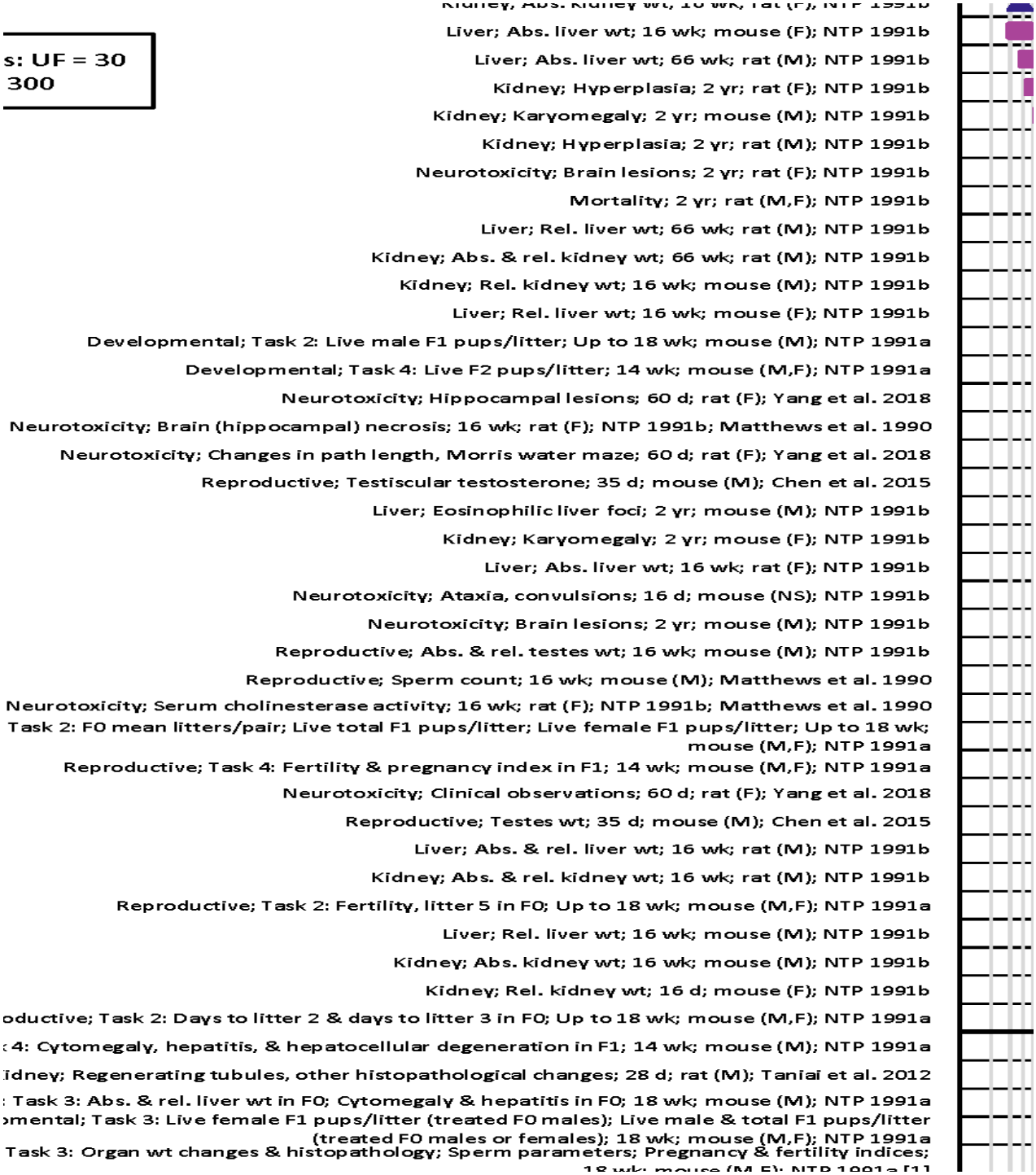
Test Type	Exposure		Metabolic Activation	Positive Controls	Outcome	Reference(s)
	Species (Sex)/ Route	Concentration/Dose/ Duration				
	strains TA98, TA100, TA1535, TA1537, TA1538	[= 286.65, 859.95, 2,866.5, 8,599.5 µg/plate]			The highest concentration showed cytotoxicity.	
In vitro bacterial reverse mutation assay	*Salmonella typhimurium* strains TA100, TA1535, TA1538	1,390 and 13,900 µg/plate [a]	± S9 from normal Sprague-Dawley rats and from rats induced by Aroclor 1254	None stated	Negative for mutagenicity [No statistical methods cited; visual inspection showed lack of dose response]	Prival et al. (1977)
In vitro bacterial reverse mutation assay	*Salmonella typhimurium* strains TA98, TA100, TA1535, TA1537, TA1538	Compounds were tested up to 5 mg/plate or toxic dose, whichever was lower	+ S9 from rats induced by Aroclor 1254 [unclear whether TCEP was tested without S9]		Negative for mutagenicity	Simmon et al. (1977)
In vitro bacterial reverse mutation assay	*Salmonella typhimurium* strains TA 98, TA100, TA1535, TA1537, TA1538	0, 0.1, 10, 100, 500, 2000 µg/plate; No cytotoxicity observed	± S9 from rats induced by Aroclor 1254		Negative for mutagenicity	BIBRA (1977)
Other genotoxicity assays						
In vitro Sister chromatid exchange	Chinese hamster ovary cells	Without S9: One trial, 26 hr incubation 5,16,50, 160 µg/mL; ***With S9***: Two trials, 2 hr incubation; Trial 1: 160, 500, 1,600 µg/mL; Trial 2: 1200, 1400, 1600 µg/mL	+/- S9 from rats	Yes	Equivocal overall Without activation – negative; With activation – Trial 1 had significant responses at the two highest doses; Trial 2 was negative at all doses; lowest concentration with stat	Galloway et al. (1987) and NTP (1991b)

Test Type	Exposure		Metabolic Activation	Positive Controls	Outcome	Reference(s)
	Species (Sex)/ Route	Concentration/Dose/ Duration				
					significant increase was 500 ug/mL; Trial 1 reached a 20% increase in SCEs [No mention whether cytotoxicity was observed.]	
In vitro Sister chromatid exchanges	V79 cells Chinese hamster lung fibroblasts	343, 490, 700, 1,000 µg/ml (experiment I); 2,000, 3,000 µg/mL (experiment II)			SCEs induced with no clear dose response (toxic observed at 3000 µg/mL, with mitosis partially inhibited)	Sala et al. (1982)
In vitro comet assay: DNA damage	Human: peripheral blood mononuclear cells	1 to 1,000 µM (alkaline version) 10 to 1,000 µM (neutral version)		Yes – H202 (alkaline version); 9 Gy (neutral version)	DNA damage observed at 1 mM in both assays (single and double strand breaks in alkaline version; double strand breaks in the neutral version). Cell viability was not assessed in the current assay but Mokra et al. (2018) identified viability as slightly decreased at 1 mM TCEP (92% of controls)	Bukowski et al. (2019)
In vitro comet assay: DNA repair	Human: peripheral blood mononuclear cells	100, 500, 1,000 µM (alkaline) 500, 1,000 µM (neutral) for 24 hr to induce damage; 60-120 min for repair assay			Single and double strand breaks and alkali-labile sites occurred observed at 1,000 µM were repaired after 2 hr (alkaline) Double strand breaks at 1,000 µM were repaired after 2 hr (neutral)	Bukowski et al. (2019)
In vitro comet assay	V79 Chinese hamster fibroblast cells	1 to 1,000 µM for 24 hr	+/- S9	Yes – potassium dichromate	No DNA strand breaks observed with or without S9	Follmann and Wober (2006)
In vitro cell transformation	Syrian hamster embryo cells	400, 500, 600, 800 µg/mL			High level of transformation	Sala et al. (1982)

Test Type	Exposure		Metabolic Activation	Positive Controls	Outcome	Reference(s)
	Species (Sex)/ Route	Concentration/Dose/ Duration				
In vitro cell transformation	C3H10T1/2 cells	900 and 1,500 µg/mL	Yes		Low incidence of transformed foci with metabolic activation (S9)	Sala et al. (1982)
DNA binding	*In vitro* PM2-CCC-DNA	5 mM in 180 min			No alkylation observed	Lown et al. (1980)

Appendix N EXPOSURE RESPONSE ARRAY FOR HUMAN HEALTH HAZARDS

The following exposure response array (Figure_Apx N-1) presents HEDs for all studies and hazard endpoints that yielded *likely* or *suggestive* evidence integration conclusions. The information is arrayed by lowest to highest HED for NOAELs and BMDLs; all PODs based on LOAELs are listed separately.



Figure_Apx N-1. Exposure Response Array for Likely and Suggestive Human Health Hazard Outcomes

Appendix O OCCUPATIONAL EXPOSURE VALUE DERIVATION AND ANALYTICAL METHODS USED

EPA has calculated an 8-hour time-weighted average (TWA) existing chemical occupational exposure value to summarize the occupational exposure scenario (OES) and sensitive health endpoints into a single value. This calculated value may be used to support risk management efforts for TCEP under TSCA section 6(a), 15 U.S.C. §2605. EPA calculated the value rounded to 0.008 ppm (0.09 mg/m^3) for inhalation exposures to TCEP as an 8-hour TWA and for consideration in workplace settings (see Appendix O.1) based on the lifetime cancer inhalation unit risk (IUR) for kidney cancer.

TSCA requires risk evaluations to be conducted without consideration of costs and other non-risk factors, and thus this most sensitive occupational exposure value represents a risk-only number. If risk management for TCEP is implemented following the risk evaluation, EPA may consider costs and other non-risk factors, such as technological feasibility, the availability of alternatives, and the potential for critical or essential uses. Any existing chemical exposure limit (ECEL) used for occupational safety risk management purposes could differ from the occupational exposure value presented in this appendix based on additional consideration of exposures and non-risk factors consistent with TSCA section 6(c).

This calculated value for TCEP represents the exposure concentration below which exposed workers and occupational non-users are not expected to exhibit any appreciable risk of adverse toxicological outcomes. This value accounts for potentially exposed and susceptible populations (PESS). The value is derived based on the most sensitive human health effect (*i.e.*, cancer) supported by the weight of scientific evidence. This value is expressed relative to benchmarks and standard occupational scenario assumptions of 8 hours per day, 5 days per week exposures for a total of 250 days exposure per year, and a 40-year working life.

EPA expects that at the most sensitive occupational exposure value of 0.008 ppm (0.09 mg/m^3) for lifetime exposure, workers and occupational non-users (ONUs) also would be protected against non-cancer effects from acute, intermediate, and chronic durations. EPA has not separately calculated a short-term occupational exposure value (STEV) because EPA did not identify hazards for TCEP associated with this very short exposure duration.

EPA did not identify a government-validated method for analyzing TCEP in air, but Appendix O.2 presents a method described by La Guardia and Hale (2015) and Grimes et al. (2019). The identified limit of detection (LOD) and limit of quantification (LOQ) using the method and the resulting monitoring data from Grimes et al. (2019) are below the lowest calculated occupational exposure value, indicating that monitoring below these levels may be achievable and that some workplaces may already be achieving the occupational exposure value.

The Occupational Safety and Health Administration (OSHA) has not set a permissible exposure limit (PEL) as an 8-hour TWA for TCEP (https://www.osha.gov/laws-regs/regulations/standardnumber/1910/1910.1000TABLEZ2). EPA did not locate other exposure limits for TCEP.

O.1 Occupational Exposure Value Calculations

This section presents the calculations used to estimate the occupational exposure values using inputs derived in this risk evaluation. Multiple values are presented below for hazard endpoints based on different exposure durations (described further in section 5.2.5). For TCEP, the most sensitive

occupational exposure value is based on cancer following lifetime exposure and the resulting 8-hour TWA is rounded to 0.008 ppm (0.09 mg/m^3).

Most Sensitive Occupational Exposure Value (Lifetime Cancer)

The EV_{cancer} is the concentration at which the extra cancer risk is equivalent to the benchmark cancer risk of 1×10^{-4}:

$$EV_{cancer} = \frac{Benchmark_{cancer}}{IUR} * \frac{AT_{IUR}}{ED * EF * WY} * \frac{\mathrm{IR_{resting}}}{\mathrm{IR_{workers}}}$$

$$= \frac{1\times10^{-4}}{5.26\times10^{-2}\ per\ ppm} * \frac{24\frac{h}{d}*\frac{365d}{y}*78y}{8\frac{h}{d}*\frac{250d}{y}*40y} * \frac{0.6125\frac{\mathrm{m^3}}{hr}}{1.25\frac{\mathrm{m^3}}{hr}} = 7.96\times10^{-3}\ \mathrm{ppm}$$

$$EV_{cancer}\ \left(\frac{\mathrm{mg}}{\mathrm{m^3}}\right) = \frac{EV\ ppm\ * MW}{Molar\ Volume} = \frac{0.00796\ \mathrm{ppm} * 285\frac{g}{mol}}{24.45\ \frac{L}{mol}} = 0.0928\ \frac{\mathrm{mg}}{\mathrm{m^3}}$$

Acute Non-cancer Occupational Exposure Value

The acute occupational exposure value (EV_{acute}) was calculated as the concentration at which the acute margin of exposure (MOE) would equal the benchmark MOE for acute occupational exposures using the following equation:

$$\mathrm{EV_{acute}} = \frac{\mathrm{HEC_{acute}}}{Benchmark\ MOE_{acute}} * \frac{\mathrm{AT_{HECacute}}}{ED} * \frac{\mathrm{IR_{resting}}}{\mathrm{IR_{workers}}} =$$

$$\frac{4.41\ \mathrm{ppm}}{30} * \frac{\frac{24h}{d}}{\frac{8h}{d}} * \frac{0.6125\frac{\mathrm{m^3}}{hr}}{1.25\frac{\mathrm{m^3}}{hr}} = 0.216\ \mathrm{ppm} = 2.51\ \frac{\mathrm{mg}}{\mathrm{m^3}}$$

Intermediate Non-cancer Occupational Exposure Value

The intermediate occupational exposure value ($EV_{intermediate}$) was calculated as the concentration at which the intermediate MOE would equal the benchmark MOE for intermediate occupational exposures using the following equation:

$$\mathrm{EV_{intermediate}} = \frac{\mathrm{HEC_{intermediate}}}{Benchmark\ MOE_{\mathrm{intermediate}}} * \frac{\mathrm{AT_{HEC\ intermediate}}}{ED*EF} * \frac{\mathrm{IR_{resting}}}{\mathrm{IR_{workers}}}$$

$$= \frac{1.27\ \mathrm{ppm}}{30} * \frac{\frac{24h}{d}*30d}{\frac{8h}{d}*22d} * \frac{0.6125\frac{\mathrm{m^3}}{hr}}{1.25\frac{\mathrm{m^3}}{hr}} = 0.0849\ \mathrm{ppm} = 0.990\ \frac{\mathrm{mg}}{\mathrm{m^3}}$$

The hazard value is the same for the intermediate and chronic OESs. The chronic occupational exposure value ($EV_{chronic}$) can be calculated as the concentration at which the chronic MOE would equal the benchmark MOE for chronic occupational exposures. However, EPA has determined that because the same key health effect applies to both intermediate and chronic exposure contexts, the relevant averaging time should be considered equivalent across both exposure scenarios. Therefore, the resulting $EV_{chronic}$ would be the same or higher than the $EV_{intermediate}$ based on exposures and EPA is presenting only the $EV_{intermediate}$.

Where:

AT_{IUR}	=	Averaging time for the cancer IUR, based on study conditions and adjustments (24 hr/day for 365 days/yr) and averaged over a lifetime (78 yrs) (see *Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Environmental Release and Occupational Exposure Assessment* (U.S. EPA, 2024n) and Section 5.2.5).
$AT_{HECacute}$	=	Averaging time for the POD/HEC used for evaluating non-cancer acute occupational risk based on study conditions and HEC adjustments (24 hr/day) (see Section 5.2.5).
$AT_{HECintermediate}$	=	Averaging time for the POD/HEC used for evaluating non-cancer intermediate occupational risk based on study conditions and HEC adjustments (24 hr/day for 30 days) (see Section 5.2.5).
$AT_{HECchronic}$	=	Averaging time for the POD/HEC used for evaluating non-cancer chronic occupational risk based on study conditions and HEC adjustments (24 hr/day for 365 days/yr) (see Section 5.2.5) and assuming the same number of years as the high-end working years (WY, 40 years) for a worker.
$Benchmark_{cancer}$	=	Benchmark for excess lifetime cancer risk, based on 1×10^{-4} extra risk
Benchmark MOE_{acute}	=	Acute non-cancer benchmark margin of exposure, based on the total uncertainty factor of 30 (see Section 5.2.5.1.1)
Benchmark $MOE_{intermediate}$	=	Intermediate non-cancer benchmark margin of exposure, based on the total uncertainty factor of 30 (see Section 5.2.5.1.2)
Benchmark $MOE_{chronic}$	=	Chronic non-cancer benchmark margin of exposure, based on the total uncertainty factor of 30 (see Section 5.2.5.1.2)
EV_{cancer}	=	Occupational exposure value (mg/m^3 and ppm) based on lifetime cancer risk at 1×10^{-4}
EV_{acute}	=	Occupational exposure value based on neurotoxicity from acute exposure
$EV_{intermediate}$	=	Occupational exposure value based on reproductive toxicity from intermediate exposure
$EV_{chronic}$	=	Occupational exposure value based on reproductive toxicity from chronic exposure
ED	=	Exposure duration (8 hr/day) (see Table 5-5)
EF	=	Exposure frequency (1 day for acute, 22 days for intermediate, and 250 days/yr for chronic and lifetime) (see Section 5.1.2.1)

HEC	=	Human equivalent concentration for acute, intermediate, or chronic non-cancer occupational exposure scenarios (see Table 5-50, Table 5-51, and Table 5-52)
IUR	=	Inhalation unit risk (per mg/m^3 and per ppm) (see Table 5-53)
IR	=	Inhalation rate (default is 1.25 m^3/hr for workers and 0.6125 m^3/hr assumed from "resting" animals from toxicity studies)
Molar Volume	=	24.45 L/mol, the volume of a mole of gas at 1 atm and 25 °C
MW	=	Molecular weight of TCEP (285 g/mole)
WY	=	Working years per lifetime at the 95th percentile (40 years) (*Risk Evaluation for Tris(2-chloroethyl) Phosphate (TCEP) – Supplemental Information File: Supplemental Information on Environmental Release and Occupational Exposure Assessment* (U.S. EPA, 2024n))

Unit conversion:

1 ppm = 11.7 mg/m^3 (see equation associated with the EV_{cancer} calculation)

O.2 Summary of Air Sampling Analytical Methods Identified

EPA conducted a search to identify relevant NIOSH, OSHA, and EPA analytical methods used to monitor for the presence of TCEP in air. The following sources were included for the search:

1. NIOSH Manual of Analytical Methods (NMAM); 5th Edition
   - URL: https://www.cdc.gov/niosh/nmam/default.html
2. NIOSH NMAM 4th Edition
   - URL: https://www.cdc.gov/niosh/docs/2003-154/default.html
3. OSHA Index of Sampling and Analytical Methods
   - URL: https://www.osha.gov/dts/sltc/methods/
4. EPA Environmental Test Method and Monitoring Information
   - URL: https://www.epa.gov/measurements-modeling/index-epa-test-methods

EPA did not identify any government-validated air sampling methods for TCEP. However, a method was described and used by La Guardia and Hale (2015) and Grimes et al. (2019). The method and associated LOD/LOQ are summarized in Table_Apx O-1.

Table_Apx O-1. Limit of Detection (LOD) and Limit of Quantification (LOQ) Summary for Identified Air Sampling Analytical Methods for TCEP

Air Sampling Analytical Methods	Year Published	LOD	LOQ	Notes	Source
Full-shift personal sampling	2019	16 ng/m^3 (1.6×10^{-5} mg/m^3; 1.37×10^{-5} ppm)	16 ng/m^3 (1.6×10^{-5} mg/m^3; 1.37×10^{-5} ppm)	Method reports LOD/LOQ of the overall procedure as 16 ng/m^3 (1.6×10^{-5} mg/m^3; 1.37×10^{-5} ppm) using an Institute of Medicine (IOM) sampler with a glass fiber filter at a flow rate of 2 L/min for the inhalable fraction of particulates and custom OVS-2 tubes at 1 L/ per min for vapor. Samples were sent to lab for analysis and quantification.	Methods described in La Guardia and Hale (2015) and Grimes et al. (2019)
ppm = parts per million; ppb = parts per billion; ppt = parts per trillion					