

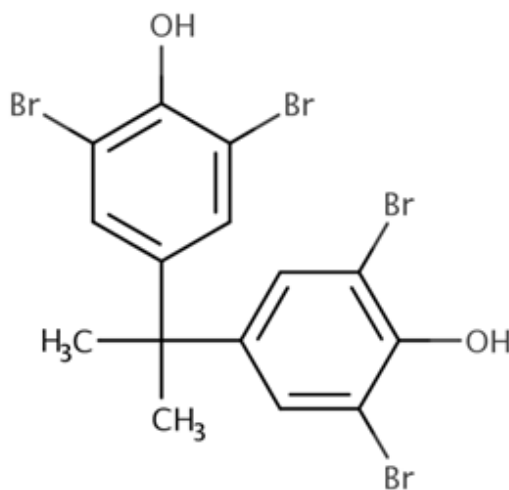


United States
Environmental Protection Agency

**Draft Environmental Media, General Population Exposure, and
Environmental Exposure for
4,4'-(1-Methylethylidene)bis[2,6-dibromophenol]
(TBBPA)**

Technical Support Document for the Draft Risk Evaluation

CASRN 79-94-7



June 2026

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153 KEY ABBREVIATIONS AND ACRONYMS

154	7Q10	Lowest 7-day average flow that occurs (on average) once every 10 years
155	30Q5	30 consecutive days of lowest flow over a 5-year period

156	ADD	Average daily dose
157	ADR	Acute dose rate
158	BAF	Bioaccumulation factor
159	BCF	Bioconcentration factor
160	BSAF	Biota-sediment accumulation factor
161	CADD	Chronic average daily dose
162	CASRN	Chemical Abstracts Service Registry Number
163	CDR	Chemical Data Reporting
164	COU	Condition of use
165	dw	Dry weight
166	DMR	Discharge Monitoring Report
167	ECHO	EPA's Enforcement and Compliance History Online database
168	EPA	Environmental Protection Agency (U.S.)
169	IIOAC	Integrated indoor-outdoor air calculator (model)
170	K _{OC}	Organic carbon/water partition coefficient
171	K _p	Dermal permeability coefficient
172	LOQ	Limit of quantitation/quantification
173	lw	Lipid weight
174	MOE	Margin of exposure
175	NAICS	North American Industry Classification System
176	NEI	National Emissions Inventory
177	NPDES	National Pollutant Discharge Elimination System
178	OES	Occupational exposure scenario
179	POD	Point of departure
180	POTW	Publicly Operated Treatment Work
181	RCRA	Resource Conservation and Recovery Act
182	TBBPA	4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] or Tetrabromobisphenol A
183	TMF	Trophic magnification factor
184	TRI	Toxics Release Inventory
185	TRV	Toxicity reference value
186	TSCA	Toxic Substances Control Act
187	TSD	Technical support document
188	U.S.	United States
189	ww	Wet weight

SUMMARY

The U.S. Environmental Protection Agency (EPA or the Agency) evaluated the reasonably available information for various environmental media concentrations and estimated exposure for the general population and environment using previously peer reviewed screening-level approaches. When needed, more refined quantitative analyses selected based on the evaluated population and the relevant exposure routes and pathways were conducted. Screening-level approaches enable timely assessments and conserve resources by focusing efforts on chemicals, uses, locations, populations, or exposure pathways and routes with the highest potential exposures. This technical support document (TSD) details the exposure analysis supporting the risk evaluation conclusions for 4,4'-(1-methylethylidene)bis[2,6-dibromophenol] (TBBPA) Toxic Substance Control Act (TSCA) conditions of use (COUs), focusing on scenarios with the upper-bound exposure estimates in a screening-level assessment. Available monitoring data as well as modeled exposure estimates are considered in this preliminary assessment. The key points are summarized below:

- EPA assessed environmental concentrations of TBBPA in air, water, and land (soil and biosolids) for use in the general population and environmental exposure assessments.
- For the general population, no acute exposure scenarios were quantified or summarized in this assessment because acute toxicity is not expected for humans ([U.S. EPA, 2026c](#)). Because EPA conducted a dose-response assessment using uterine tumor effects in its derivation of chronic human hazard values through a non-genotoxic, non-linear biological threshold of effects, lifetime average daily doses to estimate cancer risks were not calculated. It is therefore expected to be protective of cancer and non-cancer health outcomes associated with chronic human exposure ([U.S. EPA, 2026c](#)).
- The following general population exposure pathways were evaluated either qualitatively or quantitatively:
 - For the land pathway, EPA determined that TBBPA will have low leaching potential and mobility in groundwater because of its high binding affinity to particulate media and organic carbon. TBBPA is not expected to persist in soil despite the affinity for particulates and organic carbon because it is readily degradable in aerobic and anaerobic soils, reducing its overall mobility and persistence in soil. Furthermore, current Toxics Release Inventory (TRI) release data suggest TBBPA is not currently actively and knowingly applied to agricultural, residential, or public lands as biosolids or compost. If applied, current sludge stabilization technologies required for the land application of biosolids also substantially reduce the mass of TBBPA.
 - For the water pathway, TBBPA released to water is expected to predominantly partition into sediment and suspended particles in the water column. The screening analysis applied conservative model inputs (*e.g.*, high-end daily releases) and upper-bound modeled surface water concentrations to assess intermediate and chronic human exposure. EPA also used acute drinking water intake rates to estimate intermediate exposure for the general population because they were not available for intermediate durations. This is expected to overestimate exposure but provides EPA robust confidence that they represent the upper bound and are thus health protective.
 - For the ambient air pathway, TBBPA released to air is expected to predominantly be deposited into the soil compartment where it is not expected to persist. Persistence in soil is unlikely because TBBPA is readily degradable in both aerobic and anaerobic soils. EPA's screening analysis used conservative model inputs (*e.g.*, meteorological data, combining air releases from different facilities) and upper-bound modeled outdoor air

and soil concentrations from air deposition to soil to assess chronic human exposure via inhalation and soil ingestion. Although referred to as soil ingestion, this pathway includes ingestion of outdoor settled dust plus inhalation and subsequent swallowing of suspended soil and dust particles. An incidental soil ingestion rate by children was used for the screening analysis because childhood incidental soil ingestion is the highest among all lifestages. Overall, this approach and the results provide EPA robust confidence that estimated exposures represent the upper bound and are thus health protective.

- For environmental biomonitoring and trophic transfer, the following pathways were evaluated either qualitatively or quantitatively:
 - A screening-level trophic transfer analysis was conducted for a worm-eating mammal consuming earthworms as a prey item and an aquatic-dependent terrestrial species consuming fish that prey on sediment invertebrate midges (chironomids).
 - Dietary exposure estimates were calculated based on water and releases to soil (via air deposition) from the COU with the highest modeled environmental releases as reported in Sections 3 and 4.
 - Maximum TBBPA concentrations from modeled and monitored data presented no exposure of TBBPA greater than the calculated toxicity reference value (TRV) from the screening-level trophic transfer analysis. Similarly, the screening-level trophic transfer analysis for terrestrial mammals based on the highest modeled and monitored releases of TBBPA from air deposition to soil also resulted in dietary exposure concentrations below the TRV (Table 10-6).
 - The screening-level trophic transfer assessments for terrestrial invertebrates, mammals, and aquatic-dependent mammals have moderate confidence that TBBPA exposures from the highest release exposure scenarios are well below the hazard thresholds that would present risk.

1 ENVIRONMENTAL MEDIA CONCENTRATION OVERVIEW

This TSD is part of the *Draft Risk Evaluation for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026f](#)). 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol], also called tetrabromobisphenol A (CASRN 79-94-7). TBBPA is a brominated flame retardant that consists of a central methylene bridge connecting two 2,6-dibromophenol rings. TBBPA is primarily used as a reactive flame retardant in epoxy resin printed circuit boards by covalently binding to epoxy and polycarbonate resins. TBBPA is also used as an additive flame retardant in acrylonitrile-butadiene-styrene (ABS) resins for electronic enclosures and electronic consumer products with an outer plastic casing. In addition, TBBPA is used to synthesize other brominated flame retardants and TBBPA derivatives.

This TSD describes the use of reasonably available information to estimate environmental concentrations of TBBPA in different media and the use of the estimated concentrations to evaluate exposure to the general population and the environment from releases associated with COUs under TSCA. TBBPA is a TRI-reportable substance included on the TSCA Inventory, making it reportable under the Chemical Data Reporting (CDR) rule. No facilities reported point source water discharges during 2019 to 2023 within EPA's Discharge Monitoring Reports (DMRs) or air emissions within the National Emissions Inventory (NEI) ([U.S. EPA, 2026e](#)). TBBPA releases for all COUs are based on facility location data reported in TRI, except for the Use of laboratory chemicals COU and Electrical and electronic waste recycling COU. EPA evaluated the reasonably available information for releases of TBBPA from facilities that use, manufacture, or process TBBPA under industrial and/or commercial COUs. General population and environmental exposures occur when TBBPA is released into the environment and the environmental media are subsequent pathways for exposure. As described in the *Draft Release Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026e](#)), releases of TBBPA are expected to air, surface water, and disposal to landfills. Table 1-1 provides a list of the assessed COUs for TBBPA.

Table 1-1. Conditions of Use (COUs) Assessed

Life Cycle Stage	Category	Subcategory
Manufacture	Domestic manufacture	N/A
	Import	N/A
Processing	As a reactant	Flame retardant in plastic material and resin manufacturing
		Intermediate in all other chemical product and preparation manufacturing
	Incorporating into formulation, mixture, or reaction product	Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing
		Adhesive manufacturing
	Incorporation into article	Reactive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., printed circuit boards and semiconductor packages)
		Reactive flame retardant in transportation equipment manufacturing (e.g., interior material for automotive and aviation articles)
		Additive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and

Life Cycle Stage	Category	Subcategory
Processing		component manufacturing; Plastic product manufacturing (e.g., plastic electronic enclosures)
	Repackaging	N/A
	Recycling	N/A
Distribution in commerce	N/A	N/A
Industrial use	Construction, paint, electrical, and metal products	Powder coatings
Industrial and commercial use	Construction, paint, electrical, and metal products	Reactive flame retardant in electrical and electronic articles
		Additive flame retardant in electrical and electronic articles (i.e., plastic enclosures)
		Reactive flame retardant in automotive and aviation articles (i.e., laminate and prepreg material)
		Building/construction materials
	Packaging, paper, plastic, hobby products	Plastic and rubber products
	Other	Laboratory chemical (e.g., reagent, chemical standard for research, equipment calibration, sample preparation, reference material for instrument calibration)
Disposal	Disposal	Disposal

EPA relied on reported environmental releases in TRI, environmental monitoring data, and the fate and transport of TBBPA to determine which environmental pathways to consider. Details on the environmental partitioning and media assessment can be found in the *Draft Physical Chemistry and Fate Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026d](#)). Briefly, based on TBBPA's fate properties (e.g., Henry's Law constant, log K_{OC}, water solubility) and environmental transport assessment, EPA anticipates TBBPA to be predominantly in soil and sediment. Exposure from groundwater and landfills were thus not quantified because TBBPA is expected to strongly adsorb to organic materials and is readily degradable. Exposure from land application of biosolids was not quantified either because 10 years of TRI data showed no land or agricultural application of TBBPA. Sludge stabilization technologies required for land application of biosolids also substantially reduce TBBPA levels, and TBBPA is expected to be degradable in aerobic and anaerobic soils. Therefore, EPA performed a qualitative assessment for all these land exposure scenarios (Section 3).

Because TBBPA is also released to the ambient air and surface water from industrial facilities and processes based on TRI data, inhalation of ambient air and ingestion of water are possible exposure pathways for the general population. General population exposures via drinking water ingestion, fish ingestion, ambient air inhalation, and soil ingestion affected by air deposition were quantified. Modeled concentrations were compared to environmental monitoring data when possible. EPA did not assess incidental oral and dermal exposures to TBBPA through swimming because acute toxicity is not expected ([U.S. EPA, 2026c](#)). For chronic exposures, the Agency acknowledges that some populations may swim in or access open bodies of water (e.g., rivers and lakes) regularly in warmer climates or intermittently during the warm seasons. However, EPA expects only limited exposure through incidental exposure to surface water as further described in Section 6. Lastly, EPA conducted a quantitative analysis of the human milk pathway using biomonitoring data, which are not TSCA-COU specific.

Further descriptions of each exposure pathway can be found in the sections linked in Table 2-1.

Additionally, EPA reviewed biomonitoring studies (see Sections 10.2 and 10.4) and provided qualitative descriptions of the potential dietary exposures to aquatic and terrestrial organisms via feeding (trophic) relationships. Aquatic food web studies of TBBPA indicate no trophic magnification; it is not bioaccumulative under typical environmental conditions due to rapid metabolism and elimination observed in test organisms and based on the physical and chemistry and environmental fate properties. Therefore, trophic transfer screening-level analysis was not conducted for aquatic organisms. Although TBBPA is not expected to biomagnify within the terrestrial environment, relevant environmental exposures are possible through dietary exposure via the land pathway for soil invertebrates and terrestrial mammals as well as for aquatic-dependent mammals via the water pathway. EPA conducted screening-level approaches for terrestrial risk estimation based on exposure via trophic transfer for terrestrial and aquatic-dependent mammals.

2 SCREENING LEVEL HUMAN HEALTH ASSESSMENT OVERVIEW

TBBPA is a TRI-reportable substance and is on the TSCA Inventory, making it reportable under the CDR rule. No facilities reported point source water discharges between 2019 to 2023 within EPA's Discharge Monitoring Reports or air emissions within NEI (U.S. EPA, 2026e). TBBPA releases for all COUs are based on facility location data reported in TRI, except for the Use of laboratory chemicals COU and Electrical and electronic waste recycling COU. Because neither COU had TRI data, environmental releases were mathematically modeled. For this draft assessment/TSD, EPA used a screening-level approach that relies on conservative assumptions including default input parameters for modeling environmental media concentrations to characterize exposure resulting from the upper bound of the expected distribution. Details on the use of screening-level analyses in exposure assessment can be found in EPA's *Guidelines for Human Exposure Assessment* (U.S. EPA, 2019a). Upper-bound exposure estimates were defined as individuals with the greatest intake rate of TBBPA per body weight. These exposure estimates are also protective of individuals having less exposure either due to a lower intake rate or exposure to lower environmental media concentrations. This is explained further in Section 2.1.

2.1 Estimating Upper Bound Exposure

As described in the *Draft Release Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol]* (TBBPA) (U.S. EPA, 2026e), releases of TBBPA are expected to occur to air, surface water, and land. Figure 2-1 provides a graphic representation of where and in which media-specific TBBPA is expected to be found due to environmental releases and the corresponding route of exposure.

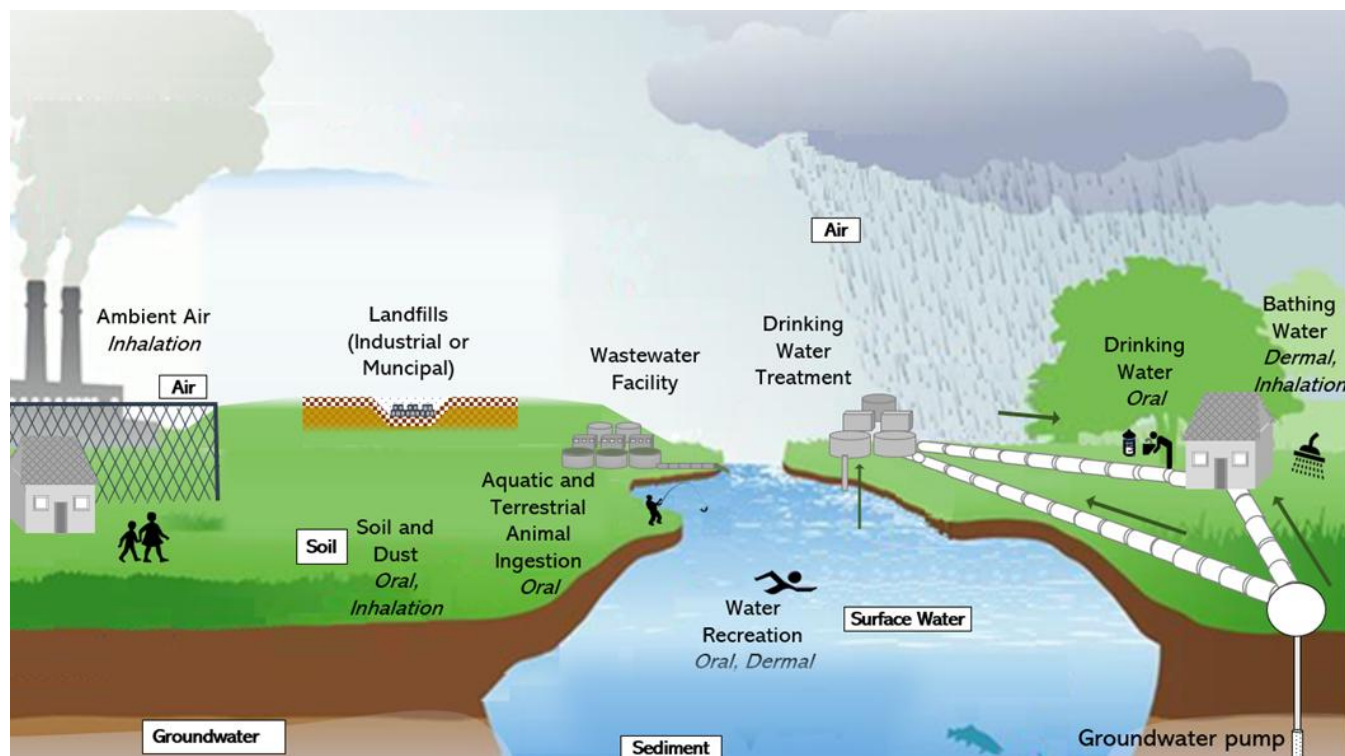


Figure 2-1. Potential Human Exposure Pathways for the General Population

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 are depicted with arrows.

For a screening-level analysis, upper-bound exposures were estimated for each assessed exposure pathway. EPA's *Guidelines for Human Exposure Assessment* defines upper-bound exposure estimates as a "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" ([U.S. EPA, 2019a](#)). EPA also considered fenceline populations in proximity to releasing facilities as part of the ambient air exposure assessment by using pre-screening methodology described in EPA's *Draft TSCA Screening-level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities (Version 1.0)* ([U.S. EPA, 2022](#)). For other exposure pathways, EPA's screening method assessing high-end exposure scenarios used release data that reflect exposures expected to occur in proximity to releasing facilities, which would include fenceline populations.

Table 2-1 summarizes the upper-bound human health exposure scenarios that were considered in the screening-level analysis including the lifestages assessed as the highest exposed populations based on intake rate and body weight. Exposure scenarios were assessed quantitatively only when environmental media concentrations were quantified for the appropriate exposure scenario. Because TBBPA environmental releases from landfills and landfill leachate (and therefore, resulting groundwater concentrations) were not quantified, oral exposure from groundwater resulting from TBBPA release to the environment via landfill leachate was not quantitatively assessed. However, the scenarios were assessed qualitatively for exposures potentially resulting from landfills. Additionally, the assessment focused solely on scenarios leading to intermediate and chronic exposure because acute toxicity of TBBPA is not expected for human receptors ([U.S. EPA, 2026c](#)).

377 **Table 2-1. Human Health Exposure Scenarios Assessed in the Screening Analysis for TBBPA**

Life Cycle/Category/Subcategory	Exposure Pathway	Exposure Route	Exposure Scenario	Lifestage	Analysis (Quantitative or Qualitative)
All COUs	Biosolids	No specific exposure scenarios were assessed for qualitative assessments			Qualitative, Section 3.2
All COUs	Landfills	No specific exposure scenarios were assessed for qualitative assessments			Qualitative, Section 3.3
All COUs	Surface water	No specific exposure scenarios were assessed for qualitative assessments			Qualitative, Section 6
- Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing - Disposal	Drinking water	Oral	Ingestion of drinking water	Adult, youth, and children	Quantitative, Section 5.1.1
	Fish ingestion	Oral	Ingestion of fish for tribal populations	Adult (fish ingestion rates were similar to if not higher than younger lifestages)	Quantitative, Section 7.1
- Manufacture – Domestic manufacture - Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – As a reactant – Intermediate in all other chemical product and preparation manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Adhesive manufacturing - Processing – Incorporation into article – Reactive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product	Ambient air	Inhalation	Inhalation of ambient air concentration for general population	Adult, youth, and children	Quantitative, Section 8.2.1
		Oral	Soil ingestion from air deposition for general population	Children	Quantitative, Section 8.2.2

Life Cycle/Category/Subcategory	Exposure Pathway	Exposure Route	Exposure Scenario	Lifestage	Analysis (Quantitative or Qualitative)
manufacturing (<i>e.g.</i> , printed circuit boards and semiconductor packages) • Processing – Incorporation into article – Reactive flame retardant in transportation equipment manufacturing (<i>e.g.</i> , interior material for automotive and aviation articles) • Processing – Incorporation into article – Additive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (<i>e.g.</i> , plastic electronic enclosures) • Industrial and commercial use – Construction, paint, electrical, and metal products – Reactive flame retardant in automotive and aviation articles (<i>i.e.</i> , laminate and prepreg material) • Disposal • Electrical and electronics recycling					
All COUs	Human milk	Oral	Human milk ingestion for general population	Infants (<1 year old)	Quantitative, Section 9

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3 LAND PATHWAY

Land pathways encompass the fate, transport, and exposure of TBBPA as it relates to the terrestrial environment (*e.g.*, soil, landfills, agricultural land) and its inhabitants. TBBPA may be directly released to the terrestrial environment through several key pathways: (1) land application of biosolids, (2) disposal in landfills (Section 3.3), (3) surface impoundments, and (4) deep well injection. Deep well injection was not further evaluated because of the absence of data. TBBPA may also be released to terrestrial media from the conveyance and deposition of TBBPA found in other environmental media (*i.e.*, atmosphere, surface water, and groundwater). Persistence in terrestrial pathways may lead to exposure through several key human health exposure pathways: (1) dermal absorption from soil/biosolids, (2) ingestion of soil/biosolids, (3) ingestion of plants/animals, (4) and inhalation of airborne particulates. This exposure assessment evaluates terrestrial exposure of TBBPA resulting from land application of biosolids (Section 3.2) and landfills (Section 3.3).

Discussion regarding TBBPA in biosolids and the land application of TBBPA is provided in Section 3.2. Because there was no direct reporting regarding TBBPA in biosolids, estimates for biosolids rely on existing literature regarding TBBPA presence in wastewater sludges and TRI-submitted releases to POTWs, which ranged from 0 to 7.7 kg per year TBBPA (Table 3-1).

Discussion of the impact TBBPA in landfills is reported in Section 3.3. TRI data submissions report annual TBBPA deposited landfills ranging from 417 to 48,844 kg per year for Resource Conservation and Recovery Act (RCRA) Subtitle C Landfills and 0 to 82,231 kg per year in non-RCRA Subtitle R Landfills (*i.e.*, Other Landfills) from 2014 through 2023 ([U.S. EPA, 2026e](#)).

3.1 Terrestrial Release Data

Facilities which manufacture greater than or equal to 25,000 lb per year or process or otherwise use 10,000 lb or more per year of TBBPA while meeting the facility TRI reporting requirements (*i.e.*, ≥ 10 employees operating in a qualifying industry) are required to submit annual release information to the TRI database. Ten years of data were collected from the TRI database from 2014 to 2023 (Table 3-1) ([U.S. EPA, 2026e](#)). The following six on-site TRI release reporting categories were identified as receiving pathways that could result in direct TBBPA releases to terrestrial pathways: (1) RCRA Subtitle C Landfills, (2) Other Landfills, (3) RCRA Subtitle C Surface Impoundments, (4) Other Surface Impoundments, (5) Land Treatment/Application, and (6) Other Land Disposal.

On-site release categories include transfers and disposals within the same facility or operating unit located in the same physical and geographic area. Landfills include the release or disposal of TBBPA containing materials to any facility in which the refuse is buried. Land impoundments include the release of TBBPA or TBBPA-containing materials to uncovered holding areas that allow for the volatilization or settling of waste materials (*e.g.*, settling lagoons, evaporation ponds). Both landfills and surface impoundments may fall under RCRA Subtitle C—depending on the accepted waste type, operation, and licensure of the facility (*i.e.*, RCRA Subtitle C Landfills or RCRA Subtitle C Surface Impoundments). Landfills and surface impoundments that do not fall under RCRA Subtitle C fall into the “other” categories (*i.e.*, Other Landfills, Other Surface Impoundments). TBBPA that is knowingly applied or injected into soil for either waste or land management is considered under “Land Treatment/Application” of biosolids (*e.g.*, biosolid fertilizers, sludge composting). Other Land Releases incorporates all other reported releases that do not clearly align with any of the established categories of terrestrial releases (*e.g.*, open pit landfills, waste piles).

Two off-site release categories were also identified in the TRI data: (1) Off-Site Transfer and (2) Discharge to Publicly Operated Treatment Works (POTWs). Off-Site Transfers include the transfer and disposal of any TBBPA or TBBPA-containing materials to a secondary, geographically or physically separate facility from the material origin. Off-site transfer and disposal of TBBPA and TBBPA-containing materials may include the transfer of materials to off-site landfills, surface impoundments, underground injections, land disposal/application, or other land disposal facilities. However, because the off-site transfer locations are physically and geographically distinct from the origin site and may not fall under the same ownership as the originating facilities, these locations are considered under a separate category under the TRI submissions guidelines. Off-site releases do not include the release to POTWs, which is its own distinct TRI reporting category. Liquid waste containing TBBPA may be transferred to POTWs via pipe or sewer, where it will then undergo treatment via wastewater treatment plant. The fate, removal, or transformation of TBBPA varies based on a plant's treatment system and technologies and may result in the release of TBBPA to receiving streams, land application sites, or atmospheric emissions. Treatment and removal of TBBPA in POTWs is discussed in the *Draft Physical Chemistry and Fate Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026d](#)).

Annual releases were calculated by aggregating all reported TRI releases for TBBPA. High and low daily releases are determined by dividing the total annual releases by the maximum (365) and minimum (2) number of assumed operating days in the year. More information regarding TRI release data can be found in *Draft Release Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026e](#)).

Table 3-1. TBBPA Terrestrial TRI Release Data from 2014–2023

Receiving Pathway	Annual Release (kg/yr)	Low Daily Release (kg/day)	High Daily Release (kg/day)	Section in this Draft TSD
Deep Well Injection	No submissions	N/A	N/A	—
RCRA Subtitle C Surface Impoundments	No submissions	N/A	N/A	—
Other Surface Impoundments	No submissions	N/A	N/A	—
RCRA Subtitle C Landfills	417–48,844	1.1–134	209–24,422	Section 3.3
Other Landfills	0–82,231	0–226	0–41,116	Section 3.3
Land/Agricultural Application	No submissions	N/A	N/A	Section 3.2
Other Land Disposal	0–45	0–0.12	0–23	Section 3.2
Transfer to Off-Site Location	1,332–56,065	3.7–154	666–28,032	N/A
Discharge to POTW	0–7.7	0–0.02	0–4	Section 3.2
POTW = public-owned treatment work; RCRA = Resource Conservation and Recovery Act; TSD = technical support document				

No facilities reported TRI submissions for the land disposal of TBBPA through “RCRA Subtitle C surface impoundments,” “Other Surface impoundments,” or “land treatment/application farming.” There were TRI facility submissions for the release of TBBPA to “RCRA Subtitle C landfills,” “Other Landfills,” “Transfer to off-site location,” and “Discharge to POTW” categories.

In addition to submitted TRI data, peer-reviewed literature, gray literature, and environmental monitoring databases were systematically reviewed to obtain concentrations of TBBPA in terrestrial land pathway media (*i.e.*, biosolids, wastewater sludge, agricultural soils, landfills, and landfill leachate) ([U.S. EPA, 2026g](#)). No reasonably available monitoring data were found from a review of government regulatory and reporting databases related to soil, landfills, or biosolids (*e.g.*, California Environmental

Data Exchange Network [CEDEN], Water Quality Portal [WQP]). The U.S. EPA Sewage Sludge Survey, a national survey of wastewater sewage sludge contaminants of concern, identified TBBPA in sewage sludge. Results from the sewage sludge survey are discussed further in Section 3.2. The impacts of TRI release and other environmental data are discussed for biosolids and landfills in Sections 3.2 and 3.3.1, respectively.

3.2 Biosolids

The term “biosolids” refers to treated sludge that meet the EPA pollutant and pathogen requirements for land application and surface disposal and can be beneficially recycled (40 CFR part 503) ([U.S. EPA, 1993a](#)). Biosolids generated during the treatment of industrial and municipal wastewater may lead to exposure to the public in several ways, including the application of biosolids to agricultural fields and pastures, residential lawns, and municipal or other public lands (*e.g.*, public parks, national and state forests) for nutrient supplementation. Both biosolids and sludges may also be incinerated or disposed of offsite through RCRA/municipal solid waste (MSW) landfills, sewage sludge mono-fills, RCRA deep well injection, or through sludge incineration. Landfill disposal will be discussed in further depth in Section 3.3.

TBBPA is introduced into biosolids through the sorption of TBBPA to organic and particulate media during wastewater treatment, where it is later physically separated into the solids sludge line and further dewatered and stabilized to form usable biosolids. TBBPA present in wastewater operations may result in biosolids that contain TBBPA. Sorption to primary and secondary sludges is the major mechanism for TBBPA removal in wastewater treatment, making up 90% of the overall removal rate. Chemical transformation (*i.e.*, hydrolysis, photolysis, biodegradation) is the minor removal mechanism, comprising 9% of the overall removal of TBBPA in wastewater treatment ([U.S. EPA, 2026d](#)).

Ten years of TRI data reported no on-site land or agricultural application of TBBPA nor any on-site surface impoundment disposal (Table 3-1) ([U.S. EPA, 2026e](#)). TRI reports releases to POTW ranging from 0 to 7.7 kg per day, which may result in sludges, and later biosolids, containing TBBPA if not adequately removed ([U.S. EPA, 2026e](#)). TBBPA may also be land applied under TRI’s “Transfer to Off-Site Locations” category, which range from 1,332 to 56,065 kg of TBBPA annually. However, the exact quantity of off-site transfers that result in TBBPA land application is unknown due to manner of TRI reporting requirements but is assumed to be negligible.

No U.S.-based studies were identified evaluating the presence of TBBPA in sludge. Several international studies identified TBBPA and its primary degradation products, tri-BBPA, di-BBPA, mono-BBPA, and brominated phenols ([Liu et al., 2016b](#)) in sewage sludges. Four literature reviews identified TBBPA in 12 countries (*i.e.*, Canada, England, Ireland, Germany, Netherlands, Denmark, Sweden, Spain, South Korea, China, and the Faroe Islands) in 18 unique studies. TBBPA concentrations in wastewater sludges ranged from up to 0.2 ng/g to 1,329 ng/g dry weight (dw) ([Abou-Elwafa Abdallah, 2016](#); [Liu et al., 2016b](#); [Malkoske et al., 2016](#); [Swedish Environmental Protection Agency, 2017](#)), with notable elevated concentrations in wastewater sludges from the treatment of electronics manufacturing and process facilities and e-waste recycling and disposal facilities’ waste streams ([Malkoske et al., 2016](#)).

Once treated to EPA standards for stabilization and vector reduction, sludges may be land applied in commercial or residential applications as biosolids. Common sludge stabilization techniques include chemical stabilization, anaerobic digestion, and composting. Anaerobic stabilization processes can reduce the concentration of TBBPA through anaerobic biodegradation. Observed half-lives of anaerobic digestion range from 5.1 to 19 days, exceeding 96% transformation of TBBPA over a 90-day treatment

period ([Wildlife International Ltd, 2006b](#); [Yang et al., 2019](#)). Aerobic composting may also reduce the concentration of TBBPA in sludges. Aerobic soils have demonstrated half-lives ranging from 5.3 to 99 days ([Springborn Laboratories, 1989a](#); [ECB, 2008](#); [Nyholm et al., 2010](#); [Li et al., 2014](#); [2015b](#); [Yang et al., 2016](#); [Wang et al., 2017](#)), yielding a 47 to up to 99% reduction in TBBPA over a 90-day treatment period. Nitrifying bioreactors have also shown promise in potential to degrade TBBPA in sludges, with one study demonstrating a half-life of 5.5 days and yielding approximately (up to) 99% reduction over a 90-day treatment period ([Li et al., 2015a](#)). Mineralization was far slower, with a half-life of 115 days and an estimated removal of 42% assuming a 90-day treatment period ([Li et al., 2015a](#)). At least one study identified TBBPA in anaerobically digested biosolids intended to be land applied at a concentration of 0.51 ng/g dw ([Brandli et al., 2007](#)).

Once stabilized, biosolids may be applied to public (e.g., national forests, public parks), agricultural (e.g., crop fields, range lands), industrial (e.g., reclamation sites), and residential (e.g., home gardens, lawns) land to promote the growth of plants and maintain a high nutrient density. Biosolids are typically applied to agricultural fields either as a slurry overtop the soil or as dry fertilizer, either as a surficial layer or incorporated into the topsoil. Incorporation of biosolids may cause significant dilutionary effects. The frequency and rate of application depend heavily on the application needs (e.g., crop fertilizer, forest land, reclamation site), applied soil chemistry, biosolids nutrient chemistry, and needs of the grower. Biosolids are typically applied at least once a year but may be applied more often depending on the soil and cover crop needs (Table 3-2) ([U.S. EPA, 2000a](#)).

Table 3-2. Typical Biosolids Application Frequencies

Application	Minimum Recommended Frequency (year ⁻¹)
Corn	1
Small grain	1–3
Soybeans	1
Hay	1–3
Forested land	0.2–0.5
Range land	0.5–1
Reclamation sites	1

Once applied, biosolids are available for uptake by plants, mechanical transport from wind and runoff, and degradation through abiotic (i.e., photolysis, hydrolysis) or biotic (i.e., aerobic biodegradation) means.

Abiotic degradation processes are not expected to be a major mechanism of removal for TBBPA in land-applied biosolids. TBBPA is susceptible to photolysis with a conservative half-life of 11.7 days, but only the uppermost exposed layer of TBBPA in the applied biosolids would be subject to direct sunlight. The remaining TBBPA, especially if incorporated or applied in a liquid slurry, would be shielded from sunlight by the overburden soil. TBBPA is not expected to undergo any significant hydrolysis because it lacks the necessary reactive functional groups for hydrolysis ([ECB, 2008](#); [EC/HC, 2013](#); [NCBI, 2020](#); [NICNAS, 2020](#)).

Biologic pathways are expected to be the dominant degradation pathway for TBBPA in land-applied biosolids. Two international studies have been identified which directly monitor TBBPA degradation in agricultural fields. Li and colleagues ([2015b](#)) conducted a 143-day study on the biodegradation of ¹⁴C-TBBPA in loamy sand soil under aerobic conditions with a half-life of 14.7 days ([Li et al., 2015b](#)).

TBBPA was largely converted to dibrominated byproducts (e.g., tri-BBPA, di-BBPA, mono-BBPA, bromo-phenols) but were largely immobilized through sorption to the soil. Relative TBBPA content (calculated as a percentage of applied TBBPA) following land application were estimated using the range of reported degradation rates ([Li et al., 2015b](#)) in aerobic soils and the recommended application rates from Table 3-2. If applied annually, TBBPA would not be expected to be appreciated significantly in the soil, with an annual estimated degradation of less than or equal to 99.9% over a 365-day period, assuming aerobic conditions are maintained in the topsoil. Biosolids may be applied as frequently as every 3 months without causing significant accumulation in soil, achieving 99% degradation in 97 days if appropriate soil conditions are maintained.

Nyholm et al. ([2010](#)) reported degradation rates of applied sludges in soil for both stabilized and unstabilized sludges. After incorporation, soils reached a sludge fraction of 0.05 (i.e., 0.5% by mass) with a nominal sludge concentration of 40 to 70 ng/g dw, within the range of previously reported sludge and biosolids concentrations. A half-life of 65 days was reported for TBBPA in untreated activated sludge and 93 days for TBBPA in sludge stabilized using anaerobic digestion ([Nyholm et al., 2010](#)). The decrease in degradation in digested sludge was attributed to the vector reduction of the final biosolids during treatment, as required for U.S. application (40 CFR 503.32), and shifting the microbial degradation from the acclimated activated sludge community to the unacclimated soil microbial community ([Nyholm et al., 2010](#)). Relative TBBPA content (calculated as a percentage of applied TBBPA) following land application were estimated using the range of reported degradation rates ([Nyholm et al., 2010](#)) in aerobic and partially aerobic soils and the recommended application rates from Table 3-2. If applied annually, approximately 2 to 7% of applied TBBPA would remain at re-application. If aerobic soil conditions are not maintained, degradation may slow. Degradation of TBBPA in activated sludge applied to anaerobic soils yielded a half-life of 430 days—an order of magnitude slower than in aerobic soils—resulting in 55% of TBBPA remaining at the time of reapplication (365 days).

TBBPA biodegradation in unamended soils may closely resemble that in land-applied soils, with similar environmental and matrix conditions. Aerobic biodegradation half-lives of TBBPA in aerobic soils ranged from 5.3 to 99 days ([Springborn Laboratories, 1989a](#); [ECB, 2008](#); [Nyholm et al., 2010](#); [Li et al., 2014](#); [2015b](#); [Yang et al., 2016](#); [Wang et al., 2017](#)), which would result in less than 8% residual TBBPA remaining if aerobic conditions were maintained at the time of re-application after 1 year. Biosolids could be reapplied between 10 times a year to once every 2 years without a significant buildup of TBBPA in the soil, depending on the system specific rate. Biodegradation half-lives of TBBPA in anaerobic soils varied widely from 16 to 449 days ([Springborn Laboratories, 1989b, c](#); [Ronen and Abeliovich, 2000](#); [Voordeckers et al., 2002](#); [Wildlife International Ltd, 2006a](#); [Nyholm et al., 2010](#); [Chang et al., 2012](#); [Wei et al., 2018](#); [Yang et al., 2019](#)). Relative TBBPA content (calculated as a percentage of applied TBBPA) following land application were estimated using the range of reported degradation rates ([Springborn Laboratories, 1989b, c](#); [Ronen and Abeliovich, 2000](#); [Voordeckers et al., 2002](#); [Wildlife International Ltd, 2006a](#); [Nyholm et al., 2010](#); [Chang et al., 2012](#); [Wei et al., 2018](#); [Yang et al., 2019](#)) in anaerobic soils using the recommended application rates from Table 3-2. Application of biosolids to anaerobic soils would result in as high as 57% of TBBPA remaining at the time of reapplication after 1 year. Biosolids containing TBBPA could be applied as frequently as three times a year to once every 8 years to prevent TBBPA accumulation, depending on the existing soil chemistry and application needs.

Land-applied biosolids will also be subject to mechanical weathering and fate processes. TBBPA is expected to be capable of medium- to long-range atmospheric transport (LRTP) when uplifted or volatilized with an expected persistence of up to 70 days and a maximum estimated range of 230 miles

from the emission source ([U.S. EPA, 2026d](#)). Elevated concentrations in nearby soil and agricultural land with concentrations as high as 2,900 ng/g dw due to dry deposition of TBBPA particulate have been reported surrounding active outdoor waste and landfill operations ([Malkoske et al., 2016](#)). Stormwater runoff is also expected to transport TBBPA bound to soil particulate. Although no studies have directly reported TBBPA in stormwater runoff originating from land application of biosolids, several studies have reported TBBPA contamination in surface waters surrounding outdoor operations, such as recycling facilities or active landfills (Section 3.3.2). TBBPA is not expected to readily leach from the soil into the groundwater or nearby surface water bodies due to its high sorption and high retardation resulting in low mobility in soil and sediment systems ([U.S. EPA, 2026d](#)).

TBBPA may be taken up by crops or plants following the application of biosolids. Several studies have reported the uptake of TBBPA in contaminated soils into several crops, including leafy plants, eucalyptus, peanuts, and various grains. Uptake has also been identified in various biota living in proximity to TBBPA-containing soil and surface water, including uptake and vertical transfer within the trophic chains ([He et al., 2010a](#); [He et al., 2010b](#); [He et al., 2013](#)). TBBPA has been identified in freshwater fish (e.g., crucian carp, mud carp, northern snakehead, Nile tilapia, suckermouth catfish), birds (e.g., chinese pond heron, white-breasted waterhen, slaty-breasted rail, common snipe, spotted dove, chinese francolin) ([He et al., 2010a](#); [He et al., 2010b](#); [He et al., 2013](#)). Although trophic magnification does not appear to be significant, TBBPA is capable of bioconcentration from soil and surface waters and undergo trophic transfer ([He et al., 2010a](#); [He et al., 2010b](#); [He et al., 2013](#)).

3.2.1 Weight of Scientific Evidence Conclusions

As noted in Section 3.1, smaller facilities (≤ 9 full-time employees) that use under the threshold quantity of TBBPA annually (*i.e.*, manufacturing < 25,000 lb, process/use < 10,000 lb annually) are not required to submit release data to the TRI database. TRI values are likely an underestimation of the true annual and daily release quantities when facilities are not required to report to TRI release to the environment, introducing uncertainty. There is also uncertainty in the relevancy of the biosolids monitoring data to the COUs considered in this draft assessment. Additionally, data regarding the treatment efficacy of sludge treatment processes (e.g., composting, anaerobic digestion, aerobic digestion, chemical stabilization) were limited.

Overall, TBBPA is not expected to have significant exposure potential from the land application of biosolids. Current TRI release data suggest TBBPA is not currently actively and purposely applied to agricultural, residential, or public lands as biosolids or compost. However, sludge data does show potential for unintentional application of TBBPA in some wastewater sludges. Although TBBPA may be present in wastewater sludge, current sludge stabilization technologies required for the land application of biosolids significantly reduce the mass of TBBPA incorporated into final biosolids. Applied TBBPA is expected to be readily degradable in aerobic and moderately-to-readily degradable anaerobic soils. TBBPA is not expected to accumulate in applied soils if aerobic conditions are maintained and reapplication is limited to the recommended guidelines. Anaerobic soils may pose a greater likelihood to TBBPA accumulation if applied in extreme cases. However, most of the available data suggest anaerobic biodegradation will occur at comparable rates to aerobic conditions and pose a low likelihood of accumulation with reapplication. Although some exposure may occur from the mechanical transport of TBBPA through atmospheric uplift or runoff conveyance, the extent of exposure is unknown with the limited data available and no reported TRI land applications of TBBPA. Finally, land application of TBBPA may result in an uptake of TBBPA into plants or crops during the growing process. However, the extent of exposure is unknown due to the limited reporting data and extent of surveyed literature. There is no evidence to suggest that TBBPA will accumulate in exposed organisms resulting from the direct application of TBBPA through biosolids applied as fertilizers.

3.3 Landfills

Landfills are a potential source of chemicals in the environment. TBBPA may be deposited into landfills through various waste streams such as consumer waste, residential waste, industrial waste, e-waste, and municipal waste, including dewatered wastewater biosolids or non-recyclable plastics from e-waste recycling operations.

Landfills are conceptually divided into three primary zones: (1) daily cell, (2) upper landfill, and (3) lower landfill. The daily cell includes the current, uncovered cell to which influent material is actively being deposited. It is assumed to be open to the elements and subject to sunlight, wind, rain, and other environmental conditions. Once a daily cover is placed on the active cell, it becomes part of the upper landfill.

The upper landfill comprises several upper lifts of the landfill, each typically 8 to 10 feet thick, which contain waste from the most recent several years. The upper landfill is covered using preapproved material, which immobilizes waste, shields it from the elements, and limits stormwater infiltration. Temporary or permanent plant cover material may also be applied to the upper landfill's surface, further stabilizing the soil and protecting it from further erosion. The upper landfill is expected to be exposed to normal environmental temperatures and pressures below the frost line (*i.e.*, 1 atm, 0–37 °C, aerobic conditions) ([2024a, b](#)), where biotic processes are the dominant route of degradation. In colder climates, soil may fall below freezing (0 °C) if the refuse falls above the natural frost line or exists in a permafrost environment, which may slow or halt biological degradation ([2024a, b](#)).

As time progresses, chemical and biological reactions use up the remaining oxygen in each landfill cell. Once the lift enters an anoxic or anaerobic state, it becomes part of the lower landfill. The lower landfill contains older material with fully anaerobic cells. The pressure and temperature are elevated due to the increased pressure, continuing biological processes, acid-base and oxidation reactions, hydration, and carbonation, which may generate additional trapped heat. While some anaerobic activity may continue in younger sections of the landfill, the accumulation of heat and pressure over time may cause temperatures to exceed 70 °C, exceeding the habitable range for most microorganisms, inhibiting or preventing biodegradation ([Huang, 2013](#)). Leachate collection systems and geomembranes are present at the lowest layer of the landfill to prevent leachate from infiltrating into the surrounding environment. Leachate collection systems have been required in all landfills since 1991. Landfills constructed prior to 1991 may lack adequate leachate collection and geomembrane systems (40 CFR 258.40).

3.3.1 Sources of TBBPA in Landfills

TBBPA may be deposited into municipal landfills through residential, municipal, and industrial waste streams. Ten years of TRI data report annual on-site landfill disposal ranging from 417 to 48,844 kg per year for RCRA Subtitle C compliant landfills and from 0 to 82,231 kg per year to non-RCRA Subtitle C compliant landfills (Table 3-1) ([U.S. EPA, 2026e](#)). TBBPA may also be disposed of into landfills under TRI's "Transfer to Off-Site Locations" category, which range from 1,332 to 56,065 kg of TBBPA annually. However, the exact quantity of off-site transfers that result in TBBPA disposal in off-site landfill is unknown due to manner of TRI reporting requirements.

There is limited information TBBPA's content in individual waste streams. However, TBBPA is found in many common products that may be disposed of in municipal or industrial landfills ([U.S. EPA, 2026e](#)). Electronics manufacturing and e-waste recycling are two significant sources of TBBPA. The TBBPA content of electronic waste (*i.e.*, e-waste) can have a wide range and are present in electronic plastic components and printed circuit boards, which have been reported to make up to 30 and 3% by mass of domestic waste streams, respectively ([Duan, 2016](#); [Malkoske et al., 2016](#)). TBBPA has also

been found in plastic electronics housings ($23,000 \pm 2,000$ ng/g), plastic covers ($7,300 \pm 3,000$ ng/g), wood television housings (80 ± 40 ng/g), printed circuit boards (43 ± 18 ng/g), and cable (5.0 ± 2.0 ng/g) (Morf et al., 2005). TBBPA in non-recyclable materials accounted for nearly 100% of the mass flow through e-waste recycling and disposal, with 57% of TBBPA by mass found in fine-grained plastic debris, 35% in plastic and wood castings, 5% in fine-grained metal debris, 3% in other fine particulate and dust, and less than 1% in-tact printed circuit board and copper cables (Morf et al., 2005). The non-recyclable material from e-waste is likely to be disposed of in landfills along with e-waste which has not been properly disposed, such as those discarded into residential curbside waste.

3.3.2 Daily Cell

Waste will first be deposited in the uppermost lift of the daily cell, where it will be exposed to direct sunlight. TBBPA has been reported to have a photolysis half-life of 11.7 days (U.S. EPA, 2026d). Exposure to sunlight in the daily cell is limited to less than 1 day of sunlight, approximately 14 to 16 hours during periods of maximum daylight (CDC, 2026). With a half-life of 11.7 days (≈ 281 hours), an estimated maximum of 3.9% of TBBPA will degrade by photolysis in the uppermost, exposed layer assuming continuous sunlight exposure and a 11.7-day half-life (U.S. EPA, 2026d). Since TBBPA is typically incorporated into plastic materials and thus shielded from ultraviolet (UV) light, photolysis will have a negligible effect on TBBPA concentrations in landfills. The actual losses of TBBPA to degradation are likely orders of magnitude lower than the maximum 3.9% and losses of TBBPA to photolysis are assumed to be negligible. Thus, abiotic degradation processes are not expected to have a significant role in the daily cell.

TBBPA may degrade in the daily cell via aerobic biodegradation. No literature has been identified evaluating the biodegradation of TBBPA in landfills. As such, aerobic soil biodegradation half-lives were used to estimate the extent of degradation possible, due to their similar microbial environmental conditions (e.g., 1 atm, 20–25 °C, aerobic, and water availability). Aerobic biodegradation half-lives of TBBPA in soils ranged from 5.3 to 99 days (Springborn Laboratories, 1989a; ECB, 2008; Nyholm et al., 2010; Li et al., 2014; 2015b; Yang et al., 2016; Wang et al., 2017), which would result in a maximum 12% degradation of bioavailable TBBPA over a single day. However, because TBBPA is incorporated into plastics and other materials, only a small fraction of TBBPA would be readily bioavailable during the 1-day period, and actual reductions in concentration from aerobic biodegradation are likely minimal to negligible. Aerobic biodegradation will continue in the upper landfill and will further degrade TBBPA as it becomes available from the degradation of waste.

Overall, abiotic and biotic degradation processes in the daily cell are limited to free and reactive TBBPA. The nature of TBBPA in consumer products suggests that the majority of TBBPA will not be free for reaction. Therefore, TBBPA in consumer products is unlikely to undergo chemical transformation through abiotic or biotic processes and assumed to be persistent.

Exposed daily cells are open to the immediate environment and subject to mechanical transport processes. Windy conditions may uplift dust and fine particulate material into the atmosphere. Once in the atmosphere, TBBPA may be subject to medium- to long-range transport with an expected persistence of up to 70 days and a maximum estimated range of 230 miles from the emission source (U.S. EPA, 2026d). Up to 3% by mass of plastics containing TBBPA were converted into fine particulate and dust with TBBPA concentrations as high as 29,000 ng/g dw in e-waste recycling facilities, TBBPA-based product manufacturing, and flame-retardant product manufacturing facilities (Malkoske et al., 2016). In outdoor operations, such as outdoor e-waste or receiving landfills, dust and particulate contributed to elevated concentrations in nearby soil and agricultural land with concentrations as high as 2,900 ng/g dw in the most severe instances (Malkoske et al., 2016). Off-gas

filters typically used in indoor e-waste recycling facilities capture up to 99.998% of fine dust and particulate, preventing its emission into the atmosphere, redirecting it to alternative disposal streams ([Malkoske et al., 2016](#)). It is likely that a significant fraction of TBBPA containing fine particulate material may escape the daily cell and be transported offsite, where it will undergo dry deposition or settle onto surrounding environments. As such, TBBPA present in daily cells may pose exposure potential to surrounding lands from the uplift and atmospheric transport of TBBPA-containing dust and fine particulate materials to surrounding areas through atmospheric conveyance and dry deposition.

Areas surrounding e-waste facilities are known to have elevated concentrations of TBBPA in surrounding soil, air, water, and sediment ([Malkoske et al., 2016](#)). TBBPA has also been reported in biota living adjacent to e-waste facilities, including in plants (*e.g.*, grains, peanuts, eucalyptus, leafy plants), freshwater fish (*e.g.*, crucian carp, mud carp, northern snakehead, Nile tilapia, suckermouth catfish), birds (*e.g.*, chinese pond heron, white-breasted waterhen, slaty-breasted rail, common snipe, spotted dove, chinese francolin) ([2010a](#); [He et al., 2010b](#); [2013](#)). TBBPA has been shown to accumulate in local biota, with bioaccumulation factors ranging from 10.3 to 20.1 L/g lipid weight (lw) and biota-sediment accumulation factors ranging from 1.2 to 2.3 (unitless) in freshwater bodies near e-waste recycling facilities with elevated sediment and water concentrations ([He et al., 2013](#)). When exposed, TBBPA accumulates mostly in the lipid centers of fish. The highest uptake is recorded in fat and liver tissue, with trace uptake recorded in muscle, heart, kidneys, or bladder tissue and no uptake recorded in the digestive tract, roe, or chyme ([Tang et al., 2015](#)). TBBPA was also present at elevated levels in tissues with direct exposure to the environment, including scales and gills ([Tang et al., 2015](#)). In lower trophic levels, there was a noted correlation between lipid content and tissue concentration, while the correlation diminished at higher levels, suggesting that there is likely little to no trophic magnification occurring between predator and prey species ([Tang et al., 2015](#)). The difference between trophic levels, however, suggests that uptake of TBBPA expands beyond environmental uptake (*e.g.*, dermal, gill), with ingestion and sediment exposure likely playing a role in TBBPA uptake ([Tang et al., 2015](#); [Malkoske et al., 2016](#)). While the addition of a daily cover would stabilize smaller particulate and dust susceptible to uplift, open pit landfills would be subject to increased removal of dust and fine particulates from the landfill, potentially to be deposited in surrounding environments.

3.3.3 Upper Landfill

Once the daily cover has been placed, TBBPA will be shielded from further photolysis or mechanical transport. TBBPA will, however, remain to be susceptible to hydrolysis and biological degradation. Hydrolysis is not a viable degradation mechanism because it lacks any hydrolytic reactive functional groups ([ECB, 2008](#); [EC/HC, 2013](#); [NCBI, 2020](#); [NICNAS, 2020](#)).

The lack of abiotic and mechanical removal pathways results in aerobic biodegradation as the primary mechanism for removal in the upper landfill. No studies were identified evaluating the biodegradation of TBBPA in landfills. Soil systems will be used as a surrogate due to their similar microbial conditions and abundance of associated literature. Aerobic biodegradation half-lives of TBBPA in soils ranged from 5.3 to 99 days ([Springborn Laboratories, 1989a](#); [ECB, 2008](#); [Nyholm et al., 2010](#); [Li et al., 2014](#); [2015b](#); [Yang et al., 2016](#); [Wang et al., 2017](#)), which would result in over 99% degradation in less than 2 years. Mineralization of TBBPA is orders of magnitude slower, with negligible mineralization in all but the most ideal laboratory conditions ($t_{1/2} \geq 3,728$ days) ([Wang et al., 2019](#); [Chen et al., 2019](#)). When mineralization does not occur, TBBPA forms several intermediates, including monomethyl ether TBBPA, dimethyl ether TBBPA, and other brominated products ([Wang et al., 2019](#)). Available TBBPA is not expected to be persistent in upper landfills with aerobic biodegradation being the major route of removal. The majority of TBBPA, however, is incorporated into plastics and is not likely to be readily bioavailable for biodegradation. Degradation of TBBPA in plastics is thus limited by the mechanical

degradation of the waste itself, not the metabolic capacity of the landfill's microbial community, significantly increasing the persistence of TBBPA in plastics and other weather-resistant materials.

3.3.4 Lower Landfill

Once the landfill cell becomes anaerobic, it is considered a part of the lower landfill. As with the upper landfill, TBBPA is not expected to undergo further abiotic degradation due to its lack of reactive functional groups and sunlight ([ECB, 2008](#); [EC/HC, 2013](#); [NCBI, 2020](#); [NICNAS, 2020](#)). Anaerobic biodegradation is the primary and sole degradation pathway in the lower landfill. As with the upper landfill, anaerobic soil and sediment biodegradation is used as a landfill surrogate due to its similar microbial environmental conditions and availability of data. As cells approach the anoxic and anaerobic state, TBBPA will continue to degrade at rates like aerobic degradation, with a half-life of 21 days at the aerobic-anaerobic interface ([Sun et al., 2014](#)). TBBPA mineralization rates in transitory environments are similarly slow to biodegradation in aerobic environments, with a half-life exceeding 374 days in ideal conditions ([Sun et al., 2014](#)). Once fully anaerobic, biodegradation half-lives of TBBPA varied widely from 16 to 449 days ([Springborn Laboratories, 1989b, c](#); [Ronen and Abeliovich, 2000](#); [Voordeckers et al., 2002](#); [Wildlife International Ltd, 2006a](#); [Nyholm et al., 2010](#); [Chang et al., 2012](#); [Wei et al., 2018](#); [Yang et al., 2019](#)). TBBPA anaerobically degraded following a debromination pathway creating tri-BBPA, di-BBPA, mon-BBPA, and finally BPA ([Ronen and Abeliovich, 2000](#); [Chang et al., 2012](#); [Sun et al., 2014](#); [Wei et al., 2018](#)). TBBPA also degraded via an O-methylation pathway to generate brominated phenols ([Sun et al., 2014](#)). As with the upper landfill, free TBBPA is not expected to be persistent in lower landfills with aerobic biodegradation being the major route of removal. The majority of TBBPA, however, is incorporated into plastics and is likely to not be readily bioavailable for biodegradation; therefore, biodegradation will be limited by the mechanical degradation of the waste itself, and not the metabolic capacity of the landfill's microbial community, significantly increasing the persistence of TBBPA in plastics and other weather-resistant materials.

Leaching is the only significant mechanical pathway for TBBPA removal in the lower landfill. TBBPA can leach with a water solubility of 4.15 mg/L at a pH of 7 ([Kuramochi et al., 2008](#)). Water solubility reduces under acidic conditions with a water solubility of 0.17 mg/L (pH = 3) in its anionic form ([Kuramochi et al., 2008](#)). Increased acidity in the lower landfill may reduce leaching potential. TBBPA has been observed in leachate produced in landfills from Japan (0 to 620 µg/L), Sweden (1–20 µg/L), the Netherlands (≤ 320 µg/L), and China (1–15.4 µg/L) ([Gong et al., 2016](#); [Morris et al., 2004](#); [Osako et al., 2004](#); [Zhou et al., 2013](#); [Stubbings and Harrad, 2014](#); [Daso et al., 2017](#)). Particulate material may also be conveyed along with leachate as it moves through the landfill and is collected and treated. TBBPA has been observed bound to leachate particulate with concentrations as high as 620 µg/kg in untreated and 11 µg/kg in treated leachate ([Osako et al., 2004](#)). Conventional leachate capture and treatment systems are capable of significantly reducing TBBPA concentrations in leachate with leachate treatment system removal rates exceeding 89% for TBBPA ([Osako et al., 2004](#); [Zhou et al., 2013](#)). Leaching is not considered a significant pathway for TBBPA exposure to the environment in post-1991 landfills with adequate geomembrane liners and leachate collection and treatment systems. Leachate that is captured but not adequately treated prior to release can lead to exposure in receiving streams and wildlife.

Landfills constructed prior to 1991 may lack adequate geomembrane liners and leachate collection systems to fully mitigate the potential of leachate release. No studies have been identified reporting TBBPA presence in groundwater around nearby landfills. However, with a water solubility of 4.15 mg/L and known detection in landfill leachate, infiltration of leachate into nearby soil and groundwater systems cannot be ruled out through omission. TBBPA would be unlikely to be mobile in groundwater and soil systems. TBBPA is expected to bind to organic media and particulate ([U.S. EPA, 2026d](#)). If

released into the soil through leachate, TBBPA is expected to bind to the surrounding soil and be highly retarded. Any TBBPA released from the lower landfill in leachate would likely experience a reduction in temperature outside of the landfill bounds, allowing it to degrade via aerobic and anaerobic degradation, depending on the individual system. It is likely that released TBBPA, both bound to particulate or present in leachate, would degrade before reaching any surrounding aquifers. As such, TBBPA in leachate infiltration is not expected to be a viable exposure route for TBBPA in landfills.

3.3.5 Weight of Scientific Evidence Conclusions

There is uncertainty in the relevancy of the landfill leachate monitoring data to the COUs considered in this evaluation. As noted in Section 3.1, smaller facilities with nine or less full-time employees or use under the threshold quantity of TBBPA annually (*i.e.*, manufacturing < 25,000 lb, process/use < 10,000 lb annually) are not required to submit release data to the TRI database. TRI values are likely an underestimation of the true annual and daily release quantities when considering non-qualifying facility releases and thus increase uncertainty.

Based on the biodegradation data for conditions relevant to landfills, there is high confidence that TBBPA will not be persistent in landfills. TBBPA in exposed sections of the landfill may pose some exposure potential due to mechanical transport processes, which may result in TBBPA's conveyance to surrounding areas through atmospheric and stormwater runoff conveyance. Overall, due to high-quality physical and chemical property data, there is robust confidence that TBBPA is likely to readily degrade in the landfill and not be persistent. TBBPA may be present in landfill leachate but is not likely to be persistent with appropriate leachate capture and treatment systems with high removal rates in treated leachate. Any uncaptured leachate is not expected to mobilize TBBPA due to its physical and chemical characteristics causing it to bind strongly to the soil particulate matter.

4 SURFACE WATER AND SEDIMENT CONCENTRATION

EPA queried TRI for environmental releases to water from 2019 to 2023 and identified water release data for 11 of the COUs, covering 86 facilities. Of those 86 facilities, only 5 reported either direct or indirect releases to surface water, as described in the *Draft Release Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026e](#)). Among those five facilities, two of them directly discharge to water and represent two COUs ([1] Processing – As a reactant – Flame retardant in plastic material and resin manufacturing and [2] Processing – Incorporating into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing). The remaining facilities are expected to be released to a POTW. EPA applied the releases to water as reported in TRI to model the resulting environmental media concentrations from TSCA COUs presented in Table 1-1.

The Agency also searched peer-reviewed literature, gray literature, and databases of environmental monitoring data to obtain concentrations of TBBPA in ambient surface water and aquatic sediments. Although the available monitoring data were limited, TBBPA was found in detectable concentrations in ambient surface waters and aquatic sediments. Section 4.1 presents EPA-modeled surface water concentrations and modeled sediment concentrations; Section 4.2.1 includes a summary of monitoring concentrations for ambient surface water; and Section 4.2.2 includes monitoring concentrations for sediment found from the systematic review process.

4.1 Modeling Approach for Estimating Concentrations in Surface Water and Sediment

EPA conducted modeling with EPA's Variable Volume Water Model (VVWM) in Point Source Calculator tool (PSC) ([U.S. EPA, 2019c](#)) to estimate surface water and sediment concentrations of TBBPA resulting from COU releases under TSCA. PSC inputs include physical and chemical properties of TBBPA (*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 surface water, sediment, and sediment pore-water concentrations.

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. TBBPA has a log K_{oc} of 5.57 (average of ECB ([2008](#)), Islam ([2015](#)), and Yang et al. ([2015](#))) indicating a high potential to sorb to suspended particles in the water column and settled sediment in the benthic environment.

Physical, chemical, and environmental fate properties selected by EPA for this assessment were applied as inputs to the PSC Model (Table 4-1). Selected values are described in detail in the *Draft Physical Chemistry and Fate for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026d](#)). The PSC model relies on the Heat of Henry parameter, which was estimated from temperature variation of the Henry's Law constant calculated by HENRYWIN™ in EPI Suite™ ([U.S. EPA, 2017a](#)).

Table 4-1. PSC Model Inputs (Chemical Parameters)

Parameter	Value ^a
K _{oc}	374,074 mL/g
Water Column Half-life	44 days at 25 °C
Photolysis Half-life	0.031 days at 45° latitude
Hydrolysis Half-life	Blank – TBBPA is not expected to be susceptible to hydrolysis because it lacks functional groups that hydrolyze under environmental conditions.
Benthic Half-life	16 days at 23 °C
Molecular Weight	543.88 g/mol
Vapor Pressure	4.68E-08 torr
Water Solubility	4.15 mg/L
Henry's Law Constant	1.45E-10 atm·m ³ /mol at 21.7 °C
Heat of Henry	54,041 J/mol
Reference Temperature	25 °C
^a For details on selected values, see <i>Draft Physical Chemistry and Fate Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)</i> (U.S. EPA, 2026d).	

A common setup for the model environment and media parameters was applied consistently across all PSC runs. The standard EPA water body characteristics were used to parameterize the water column and sediment parameters (Table 4-2), which is applied consistently in the screening scenarios. Standardized water body geometry was also applied consistently across runs, with a standardized width of 5 m, length of 40 m, and depth of 1 m, representing a small section of the receiving stream. Only the release parameters (daily release amount and days of release) and the hydrologic flow rate were changed between model runs for this chemical.

Table 4-2. Standard EPA Waterbody Characteristics for PSC Model Input

Parameter	Value
DFAC (represents the ratio of vertical path lengths to depth as defined in EPA's Exposure Analysis Modeling System [EXAMS]) (U.S. EPA, 2019c)	1.19
Water column suspended sediment	30 mg/L
Chlorophyll	0.005 mg/L
Water column f_{oc} (fraction of organic carbon associated with suspended sediment)	0.04
Water column dissolved organic carbon (DOC)	5.0 mg/L
Benthic depth	0.05 m
Water column biomass	0.4 mg/L
Benthic porosity	0.50
Benthic bulk density	1.35 g/cm ³
Benthic f_{oc}	0.04
Benthic DOC	5.0 mg/L
Benthic biomass	0.006 g/m ²
Mass transfer coefficient	0.00000001 m/s

A required input for the PSC Model is the hydrologic flow rate of receiving water body. Because TRI has reported releases to water, relevant flow data from the associated receiving water body were collected, as described further in this section. Databases that were queried to estimate a flow rate include

EPA's Enforcement and Compliance History Online (ECHO) contains facilities with a National Pollutant Discharge Elimination System (NPDES) permit, National Hydrography Dataset Plus (NHDPlus), and NHDPlus V2.1 Flowline Network Enhanced Runoff Method (EROM) Flow. For facilities that do not report to TRI, a generic distribution of hydrologic flows was developed by collecting flow data for facilities across a North American Industry Classification System (NAICS) code associated with each COU for a TBBPA-releasing facility. Flow information was extracted from the same databases. The 50th, 75th, and 90th percentile (P50, P75, P90, respectively) flows from the distribution were applied to represent variation in the potential receiving water bodies.

Different hydrological flow rates were used for different exposure scenarios. The 30Q5 flows (*i.e.*, the lowest 30-day flow that occurs [on average] once in a 5-year period) are used to estimate acute, incidental human exposure through swimming or recreational contact. Although acute toxicity of TBBPA is not expected ([U.S. EPA, 2026c](#)), acute exposure was calculated to estimate intermediate exposure as detailed in Section 6. The annual average flow represents long-term flow rates, but a harmonic mean provides a protective estimate for screening and is preferred for assessing potential chronic human exposure via drinking water. The harmonic mean is also used for estimating human exposure through fish ingestion because it takes time for chemical concentrations to accumulate in fish. Lastly, for aquatic or ecological exposure, a 7Q10 flow (*i.e.*, the lowest 7-day flow that occurs [on average] once in a 10-year period) is used to estimate exceedances of concentrations of concerns for aquatic life ([U.S. EPA, 2007](#)). The regression equations for deriving the harmonic mean and 7Q10 flows are provided in Appendix B.

Receiving water body TBBPA concentrations were estimated at the point of release (*i.e.*, in the immediate water body receiving the effluent). Concentrations were modeled for the two COUs that reported direct releases to water in TRI (*i.e.*, after any onsite treatment and without any transfers to off-site publicly owned treatment works [POTWs]), as well as for one POTW that indirectly receives industrial waste containing TBBPA according to TRI. For the POTW indirectly receiving industrial waste, TRI reports only the amount of TBBPA being transferred from the industrial facility to the receiving POTW. Because treatment is expected at the receiving POTW prior to discharge to surface water, EPA applied a mean removal efficiency of 88.8% in its modeling. See the *Draft Chemistry, Fate, Release, and Exposure Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* for more details on the derivation of the removal efficiency based on systematic review ([U.S. EPA, 2026d](#)). The number of operating days is a range for one COU to be consistent with test order data submitted by the American Chemistry Council (ACC) ([Integral Consulting, 2025](#)). The ACC TBBPA Consortium comprised 21 companies that operate 26 TBBPA sites across the United States. For some sites that release to water, a range of operating days was provided. As such, EPA modeled surface water and sediment concentrations using the lower and upper bounds of the range. Additional details on the test order data can be found in Section 2.2.2 of the *Draft Release Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026e](#)). While some information was provided on release days, the test order primarily applies to inhalation monitoring data for workers and therefore not relevant to this general population and environmental exposure assessment.

Three other POTWs reported indirectly receiving industrial waste containing TBBPA. However, for screening purposes, the resultant media concentrations from these WWTPs were not used in the screening assessment. That is because lower daily releases combined with higher flow rates of the receiving water body yielded media concentrations that were several orders of magnitude lower than the POTW considered.

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Table 4-3 and Table 4-4 show the surface water concentrations modeled using the 7Q10 flow and harmonic mean and 30Q5 flows, respectively.

Table 4-3. TBBPA Concentrations in Water and Sediment in the Receiving Waterbody at 7Q10 Flow

COU	Number of Operating Days Per Year	Daily Release (kg/day) ^a	7Q10 Total Water Column Concentration (µg/L)	7Q10 Total Benthic Pore Water Concentration (µg/L)	7Q10 Benthic Sediment Concentration (µg/kg)
Processing – As a reactant – Flame retardant in plastic material and resin manufacturing ^b	350	4.15E-02	4.47E-04	1.37E-04	2.05
	4	3.63	3.91E-02	1.32E-03	19.8
Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing ^b	12	3.78E-02	12.8	1.16	1.74E04
Disposal ^c	12 ^d	1.02	8.52	7.51E-01	1.12E04

7Q10 = lowest 7-day flow in a 10-year period
^a Details on operating days and daily releases are provided in the *Draft Release Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026e](#)).
^b No wastewater treatment removal efficiency was applied to these COUs because the facilities reported all as direct releases to water.
^c A mean wastewater removal efficiency of 88.8% was applied for this COU in estimating the facilities' releases to water. TRI reported only the amount of TBBPA-containing wastewater transferred from an industrial facility to a POTW for treatment. It is therefore reasonable to assume that the amount transferred is not the amount discharged, but rather after treatment at the POTW.
^d The number of operating days per year corresponds to the number of days, as reported in TRI, a facility discharges TBBPA-containing wastewater to this POTW. It is not the number of days that the POTW discharges to surface water.

Table 4-4. PSC Modeling Results for Total Water Column Using Harmonic Mean Flow and 30Q5 Flow

Scenario	Number of Operating Days	Daily Release (kg/day) ^a	30Q5 Flow (m ³ /d)	Harmonic Mean Flow (m ³ /d)	30Q5 Concentration (µg/L)	Harmonic Mean Concentration (µg/L)
Processing – As a reactant – Flame retardant in plastic material and resin manufacturing ^b	350	4.15E-02	115,692,482	265,609,908	3.59E-04	1.56E-04
	4	3.63	115,692,482	265,609,908	3.14E-02	N/A ^c
Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing ^b	12	3.78E-02	2,367	3,877	9.39	N/A ^c
Disposal ^d	12 ^e	0.1	3,120	4,955	8.52	N/A ^c

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Scenario	Number of Operating Days	Daily Release (kg/day) ^a	30Q5 Flow (m ³ /d)	Harmonic Mean Flow (m ³ /d)	30Q5 Concentration (µg/L)	Harmonic Mean Concentration (µg/L)
30Q5 = 30 consecutive days of lowest flow over a 5-year period ^a Details on operating days and daily releases are provided in the <i>Draft Release Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)</i> (U.S. EPA, 2026e). ^b No wastewater treatment removal efficiency was applied to these COUs because the facilities reported them as direct releases to water. ^c A harmonic mean was not estimated because 12 release days do not correspond to long-term flow conditions. ^d A mean wastewater removal efficiency of 88.8% was applied for this COU in estimating the facilities' releases to water. TRI reported only the amount of TBBPA-containing wastewater transferred from an industrial facility to a POTW for treatment. It is therefore reasonable to assume that the amount transferred is not the amount discharged, but rather after treatment at the POTW. ^e The number of operating days per year corresponds to the number of days, as reported in TRI, a facility discharges TBBPA-containing wastewater to this POTW. It is not the number of days that the POTW discharges to surface water.						

4.2 Measured Concentrations

EPA identified monitoring studies through systematic review to provide context to modelling results. The monitoring studies presented here were not used as part of the analysis for quantifying exposure estimates. EPA also searched the WQP, 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. Measured concentrations of TBBPA in surface water and sediment are presented in Sections 4.2.1 and 4.2.2, respectively.

4.2.1 Measured Concentrations in Surface Water

U.S. EPA ([1978](#)) collected surface water samples near an organo-bromide production facility in El Dorado, Arkansas in 1977. TBBPA was not detected (<50 µg/L) in the two surface water samples. Measured surface water concentrations from the WQP retrieved in 2022 resulted in 98 sample results. All samples were collected in 2010 from the effluent of multiple POTWs in Oregon and were below the detection limit of 100 µg/L ([EPA et al., 2022](#)). The detection limit is two orders of magnitude above the highest modeled concentrations. However, no further information is provided in the WQP to determine possible errors in reporting.

Systematic review identified an additional 18 studies from nine non-U.S. countries. See Appendix C for a summary. Detection limits, if provided, were less than 10 µg/L across all studies. He et al. ([2013](#)) reported the maximum concentration (2,830 µg/L) from samples collected in a river flowing through a highly industrialized area in China. No other measured concentrations from the 17 other studies exceeded 10 µg/L. It is important to note that uncertainties associated with international monitoring data limit their applicability to this assessment. Whether these data do or do not reflect conditions in the United States or COUs under TSCA is unknown. Information needed to link the monitoring data to foreign industrial processes and crosswalk those to TSCA COUs is not available. Several studies, particularly the ones in China, also investigated regions with a heavy presence of industrial or e-waste recycling activities. Furthermore, regulation on industry emissions and/or discharges to water often vary between the United States and foreign countries.

4.2.2 Measured Concentrations in Sediment

This section discusses the four sediment monitoring studies from the United States or Canada identified through systematic review. The monitoring studies presented here provide context to the modeling

results and were not used as part of the analysis for quantifying exposure estimates. In addition, Appendix D summarizes 38 international studies that were excluded from discussion in this section.

U.S. EPA (1978) collected sediment samples from the vicinity of multiple organo-bromide production facilities in El Dorado, Arkansas, in 1976 and 1977. TBBPA concentrations ranged from non-detect (<100 ng/g) to 330,000 ng/g (wet or dry weight not reported) in the eight samples that were tested. The detection frequency was not provided.

Quade (2003) analyzed bottom sediment collected from eight stations on Lake Ontario in 2002. Because the chemical analysis did not include partitioning or fractionating steps to separate TBBPA from any dimethyl TBBPA (MeTBBPA) that might have been originally present in the sediment, and since the samples were methylated during analysis, the reported TBBPA concentrations are the sum of both TBBPA and MeTBBPA. TBBPA was quantified in three of the eight samples (0.029–0.063 ng/g dw) and was below the detection limit of 0.002 ng/g in the other five samples. Quade (2003) noted that laboratory recoveries were small in some samples (34–38%), possibly due to organic content of the samples. However, the reported concentrations accounted for the losses during preparation and clean-up because calculations were based on the response to the labeled internal standard.

Lu et al. (2015) analyzed surficial (0–10 cm) and deeper (20–30 cm) sediment samples collected from Lake Erie in May and June 2004. TBBPA was below the limit of detection (0.05 ng/g dw) in all samples (n = 56)—except one surficial sediment sample from a site in the western basin near the Detroit River where TBBPA concentration was 0.5 ng/g dw.

Table 4-5. Summary of Measured TBBPA Concentrations in Sediment in the United States and Canada

Reference	Sampling Location	TBBPA Concentration (ng/g)
U.S. EPA (1978)	United States	FOD: 75% (n = 8) ND to 330,000 (ND was not reported)
Quade (2003)	Lake Ontario	FOD: 37.5% (n = 8) <0.002 to 0.064 dw
Lu et al. (2015)	Lake Erie	FOD: 1.8% (n = 56) <0.05 to 0.5 dw
dw = dry weight; FOD = frequency of detection; ND = non-detect		

4.3 Evidence Integration for Surface Water and Sediment

4.3.1 Strengths, Limitations, and Sources of Uncertainty for Modeled and Monitored Surface Water Concentration

EPA conducted modeling with PSC to estimate concentrations of TBBPA within surface water, pore-water, and sediment. PSC considers model inputs of physical and chemical properties of TBBPA (*i.e.*, K_{ow}, K_{OC}, water column half-life, photolysis half-life, hydrolysis half-life, and benthic half-life) and allows EPA to estimate surface water, sediment, and pore-water concentrations. The use of vetted physical and chemical properties of TBBPA increases confidence in the application of the PSC model. Only the chemical release amount, days of chemical release, and the receiving water body hydrologic flow were changed for each COU/occupational exposure scenario (OES). A standard EPA water body was used to represent a consistent receiving water body scenario.

Flow data collected via the EPA ECHO API (Application Programming Interface) and the NHDPlus V2.1 EROM flow database include self-reported hydrologic reach codes on NPDES permits and the best available flow estimations from the EROM flow data. The EROM database provides modeled monthly average flows for each month of the year, and the lowest of the monthly average flows was selected as a substitute for the 30Q5 flow to model general population exposure. The harmonic mean flow was also used for the general population assessment. The 7Q10 was applied for only the environmental exposure analysis as a screening approach where a potential contaminant may be predicted to be elevated due to period low flow conditions. Confidence in the flow values is considered robust because they are based on receiving waterbodies reported in TRI and standard regression equations from EPA's Exposure and Fate Assessment Screening Tool ([U.S. EPA, 2007](#)). Programming Interface) and the NHDPlus V2.1 EROM flow database include self-reported hydrologic reach codes on NPDES permits and the best available flow estimations from the EROM flow data.

Direct releases to surface water reported via TRI were applied as the actual loading to surface water. Resulting surface water concentrations from indirect discharges to off-site POTWs from TRI were also modeled. However, a removal efficiency was applied prior to modeling because discharges to POTWs are not directly released into the environment but to treatment facilities. The modeled data represent estimated surface water concentrations at the point of discharge for facilities that are actively releasing TBBPA to surface water. Confidence in the modeled surface water concentrations is robust because they are based on high-quality physical and chemical property data, as well as TRI reporting.

4.3.2 Weight of Scientific Evidence Conclusions

EPA used daily release information as reported in TRI to estimate surface water concentrations for use in the general population and environmental exposure assessment. As discussed in further detail in the *Draft Release Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026e](#)), the Agency was not able to quantitatively estimate site-specific releases for the final use of products or articles COUs. Disposal sites handling post-consumer, end-use TBBPA were not quantified due to the wide and dispersed use of TBBPA in electronics and electrical articles. Pre-consumer waste handling, treatment, and disposal are assumed to be captured in upstream COUs. All COUs had releases estimated using data from reporting facilities in TRI. The TRI database received a high-data quality rating through EPA's systematic review process. The overall weight of scientific evidence conclusion for this assessment is moderate across water releases for the various COUs. Factors that decreased the overall confidence for this estimate include the uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites.

For reported releases to TRI, relevant flow data from the associated receiving water body were collected by querying multiple EPA databases and permit IDs under NPDESs. The flow data include self-reported hydrologic reach codes on NPDES permits and the best available flow estimates from EPA and USGS databases. Other model inputs were derived from reasonably available literature collected and evaluated through EPA's systematic review process for TSCA risk evaluations. All monitoring and experimental data included in this analysis were from articles rated "medium" or "high" quality from this process.

Release concentrations were estimated at the point of release in the receiving water body to evaluate the upper end of potential exposure concentrations for a given release. Modeled surface water concentrations were based on TRI reporting of releases to water, as well as high-quality physical and chemical property data derived from reasonably available literature and evaluated through EPA's systematic review process for TSCA risk evaluations. Overall, EPA has robust confidence that the high-end and protective estimated surface water concentrations modeled using the Processing as a reactant in plastic material and resin manufacturing COU; Processing: Additive flame retardant in plastic, resin, and

1088 paints and coating manufacturing COU; and Disposal COU are appropriate to use in its screening-level
1089 assessment of general population and environmental exposure.

5 DRINKING WATER EXPOSURE TO GENERAL POPULATION

Drinking water in the United States typically comes from surface water (*i.e.*, lakes, rivers, and reservoirs) and groundwater. The source water then flows to a treatment plant where it undergoes a series of water treatment steps before being distributed to homes and communities. Public drinking water systems often use a combination of treatment processes that include coagulation, flocculation, sedimentation, filtration, and disinfection to meet drinking water quality standards. The exact treatment processes used to meet drinking water quality standards differ between public water systems. As described in Section 3.3, TBBPA is not expected to significantly migrate from landfills through groundwater infiltration because high hydrophobicity and high affinity for soil sorption is expected to retard or immobilize TBBPA in the surrounding soil. Therefore, drinking water exposure in this assessment is focused on drinking water sourced from surface water.

5.1 Modeling Approach for Estimating Concentrations in Drinking Water

5.1.1 Drinking Water Ingestion

Modeled surface water concentrations estimated in Section 4.1 were used to estimate potential TBBPA drinking water exposures. For this screening analysis, no further removal from drinking water treatment was applied. Acute toxicity of TBBPA pertaining to an exposure duration of one day is not expected ([U.S. EPA, 2026c](#)) and thus not calculated. Intermediate toxicity with an exposure duration of up to 30 days is expected. Equation 5-1 was used for estimating the non-cancer intermediate dose. A key input for this equation is the drinking water intake rate for various age groups. Since intermediate duration ingestion rates are not available, this scenario relies on ingestion rates consistent with an acute duration. Using these acute exposure factors are likely to overestimate exposure for the intermediate scenarios. However, in the absence of more specific exposure factors, EPA selected the acute intake rates to be health protective. Moving forward, drinking water exposure estimates using an acute dose rate will be referred to as the intermediate dose to align with human hazard values even though an acute drinking water intake rate was used. Equation 5-2 calculates the non-cancer potential average daily dose, which corresponds to the chronic drinking water exposure.

Equation 5-1. Intermediate Drinking Water Ingestion Calculation

$$\text{Intermediate Dose} = \frac{SWC \times \left(1 - \frac{DWT}{100}\right) \times IR_{dw} \times RD \times CF1}{BW \times AT}$$

Where:

SWC	=	Surface water concentration (ppb or $\mu\text{g/L}$; 30Q5 concentration for intermediate corresponding to the lower number of release days)
DWT	=	Removal during drinking water treatment (assume 0%)
IR_{dw}	=	Drinking water intake rate (L/day)
RD	=	Release days (days/year for average daily dose [ADD]; 30 days for intermediate)
$CF1$	=	Conversion factor ($1.0 \times 10^{-3} \text{ mg}/\mu\text{g}$)
BW	=	Body weight (kg)
AT	=	Exposure duration (years for ADD; 30 days for intermediate)

Equation 5-2. Average Daily Drinking Water Ingestion Calculation

$$ADD_{POT} = \frac{SWC \times \left(1 - \frac{DWT}{100}\right) \times IR_{dw} \times ED \times RD \times CF1}{BW \times AT \times CF2}$$

Where:

ADD_{POT}	=	Potential average daily dose (mg/kg-day)
SWC	=	Surface water concentration (ppb or µg/L; harmonic mean for ADD corresponding to the higher number of release days)
DWT	=	Removal during drinking water treatment (assume 0%)
IR_{dw}	=	Drinking water intake rate (L/day)
ED	=	Exposure duration (years for ADD; 1 day for acute dose rate [ADR])
RD	=	Release days (days/year for ADD; 1 day for ADR)
$CF1$	=	Conversion factor (1.0×10^{-3} mg/µg)
BW	=	Body weight (kg)
AT	=	Exposure duration (years for ADD; 1 day for ADR)
$CF2$	=	Conversion factor (365 days/year)

As previously mentioned, non-cancer intermediate exposure estimates from drinking water ingestion were calculated using an acute intake rate because no information was available on intermediate intake. The intermediate dose and ADD from drinking water for non-cancer effects were calculated using the acute 95th percentile ingestion rate for drinking water. This corresponds to the exposure estimates for an acute duration but will hereafter be referred to as an intermediate dose to align with human hazard values. EPA used the 30Q5 and harmonic mean modeled surface water concentrations to calculate the intermediate dose and average daily dose (ADD), respectively. Table 5-1 summarizes the drinking water doses for adults, infants, and toddlers. These estimates do not incorporate additional dilution or degradation beyond the point of discharge and in this case, it is assumed that the surface water outfall is located very close (within a few km) to the drinking water intake location. Applying dilution factors would account for the decreased concentration from the point of discharge to a drinking water intake location and decrease the dose for all scenarios.

1161 **Table 5-1. Drinking Water Doses Across Lifestages in mg/kg-day**

Scenario	Release Days	Water Column Concentrations		Adult (21+ years)		Infant (Birth to <1 year)		Toddler (1–5 years)	
		30Q5 Conc. (µg/L)	HM Conc. (µg/L)	Int. (mg/kg-day)	ADD (mg/kg-day)	Int. (mg/kg-day)	ADD (mg/kg-day)	Int. (mg/kg-day)	ADD (mg/kg-day)
Processing – As a reactant – Flame retardant in plastic material and resin manufacturing	350	3.59E-04	1.56E-04	1.44E-08	1.64E-09	5.07E-08	4.20E-09	1.80E-08	1.80E-09
	4	3.14E-02	NA ^a	1.26E-06	N/A ^a	4.43E-06	NA ^a	1.58E-06	NA ^a
Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing	12	9.39	NA ^a	3.78E-04	N/A ^a	1.33E-03	NA ^a	4.71E-04	NA ^a
Disposal	12	8.52	NA ^a	3.43E-04	NA ^a	1.20E-03	NA ^a	4.28E-04	NA ^a
30Q5 = 30 consecutive days of lowest flow over a 5-year period; ADD = average daily dose; HM = harmonic mean; Int. = intermediate dose; N/A = not applicable ^a The HM was not calculated using these release days because the HM represents long-term average flow conditions. Therefore, the average daily doses for chronic exposure durations that are estimated using a HM were not calculated either.									

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5.2 Measured Concentrations in Drinking Water

EPA searched peer-reviewed literature, gray literature, and databases of environmental monitoring data to obtain concentrations of TBBPA in drinking water. No studies were identified that reported concentrations of TBBPA in drinking water.

5.3 Evidence Integration for Drinking Water

EPA estimated low (no greater than 1.40×10^{-4} mg/kg-day for the highly exposed subgroup) exposure to TBBPA via drinking water when considering expected treatment removal efficiencies, even under high-end release scenarios. This assessment assumes that concentrations at the point of intake for the drinking water system are equal to the concentrations in the receiving water body at the point of release, where treated effluent is being discharged from a facility. In reality, some distance between the point of release and a drinking water intake would be expected, providing space and time for additional reductions in water column concentrations via degradation, partitioning, and dilution. Some form of additional treatment would typically be expected for surface water at a drinking water treatment plant, including coagulation, flocculation and sedimentation, and/or filtration. This treatment would likely result in even greater reductions in TBBPA concentrations prior to releasing finished drinking water to customers.

Additional qualitative considerations suggest that actual measured concentrations in raw and finished water would decrease further. EPA also expects drinking water treatment to further remove TBBPA.

5.4 Weight of Scientific Evidence Conclusions

EPA has robust confidence that assuming the same TBBPA concentrations in drinking water as in surface water was appropriate for a screening-level analysis. The confidence assigned was based on the use of conservative assumptions for the screening approach resulting in an upper bound of exposure. As described in Section 3.3, EPA did not assess drinking water estimates as a result of leaching from landfills to groundwater and subsequent migration to drinking water wells due to TBBPA's expected preferential partitioning to organic matter, limited mobility in organic matter/soil, and low water solubility.

6 SURFACE WATER EXPOSURE TO GENERAL POPULATION

Incidental oral or dermal exposures due to swimming is not quantitatively evaluated because acute toxicity of TBBPA pertaining to an exposure duration of one day is not expected ([U.S. EPA, 2026c](#)). For chronic exposures, EPA acknowledges that some populations may swim in or access open bodies of water (*e.g.*, rivers and lakes) regularly in warmer climates or intermittently during the warm seasons. However, EPA expects only limited exposure through incidental exposure to surface water (*i.e.*, both the 30Q5 and harmonic mean concentrations of TBBPA in surface water are very low to no more than 5 µg/L). The same 30Q5 and harmonic mean concentrations were used to evaluate exposure from drinking water ingestion. The primary difference in calculating incidental ingestion during swimming and the oral exposure from drinking water was the water intake rate. The intake rate for incidental ingestion during swimming is up to two orders of magnitude lower than for drinking water. Exposure estimates will also be proportionally lower for incidental ingestion during swimming.

Dermal exposure is also expected to be lower than drinking water because of a low permeability coefficient (K_p). The permeability quantifies how easily a chemical penetrates the skin, which is crucial for calculating an absorbed dose for evaluating systemic effects. The K_p for TBBPA was determined to be a low value of 3.72×10^{-7} cm/h. This originates from a dermal TBBPA absorption analysis conducted by Charles River Laboratories ([2024](#)) in response to an EPA test order issued under TSCA section 4(a)(2). Details on the experimental design and conditions can be found in U.S. EPA ([2026c](#)).

7 FISH INGESTION EXPOSURE TO GENERAL POPULATION

TBBPA is unlikely to bioconcentrate and bioaccumulate in fish but is considered a lipophilic organic compound (log K_{ow} 4.75 at pH 7.53 and 25 °C) ([U.S. EPA, 2026d](#)). Because of the latter chemical property, EPA quantitatively estimated human exposure to TBBPA through fish ingestion. Calculating TBBPA concentrations in fish tissue is determined by the surface water concentrations of TBBPA and a bioaccumulation factor (BAF). The surface water concentrations were modeled using reported releases to water from TRI for three COUs (Table 7-1). The harmonic mean flow is typically used to estimate fish tissue concentrations because it represents long-term average flow conditions, and it takes time for chemical concentrations to accumulate in fish. However, the harmonic mean is not appropriate for some COUs with few release days (*i.e.*, <1 month). In those instances, EPA used the 30Q5 as a conservative screening-level analysis even though the 30Q5 is expected to occur less frequently and unlikely to be sustained long-term.

The BAF for TBBPA using the Arnot-Gobas method for upper trophic organisms is 153 L/kg wet weight (ww) (see *Draft Chemistry, Fate, Release, and Exposure Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol]* (TBBPA) ([U.S. EPA, 2026d](#))). That BAF indicates TBBPA's low potential for bioconcentration and bioaccumulation. The computed concentrations of TBBPA in fish tissue are on a wet weight equivalent. Table 7-1 compares the fish tissue concentration (ww) calculated using an estimated BAF with the measured fish tissue concentrations obtained from systematic review. The monitoring studies ($n = 25$) were only used to provide context to modeling results and not to quantify exposure estimates. Of the 25, 7 were excluded because of insufficient information to normalize lipid weight to wet weight. Overall, fish tissue concentrations computed with modeled surface water concentrations are within the same order of magnitude as empirical fish tissue data.

Table 7-1. Fish Tissue Concentrations Calculated from Modeled Surface Water Concentrations and Monitoring Data

Data Description and Source	Surface Water Concentration	Fish Tissue Concentration
Modeled surface water concentration	1.56E-07 mg/L (HM) for Processing – As a reactant – Flame retardant in plastic material and resin manufacturing COU	2.39E-05 mg/kg ww
	9.39E-03 mg/L (30Q5 ^a) for Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing COU	1.44 mg/kg ww
	8.52E-03 mg/L (30Q5 ^a) for Disposal	1.30 mg/kg ww
Fish tissue monitoring data ($n = 19$ from 10 countries) (Gebbink et al., 2019 ; Poma et al., 2018 ; Chessa et al., 2019 ; Netherlands Institute for Fisheries Research, 2002 ; Frederiksen et al., 2007 ; van Leeuwen and de Boer, 2008 ; Evenset et al., 2009 ; McHugh et al., 2010 ; Yang et al., 2012 ; Hloušková et al., 2013 ; Svihlikova et al., 2015 ; Tang et al., 2015 ; Zhang et al., 2015 ; Tao et al., 2016 ; Kotthoff et al., 2017 ; Zhang and Kelly, 2018 ; Norwegian Environment Agency, 2019b ; Green et al., 2021)	N/A	ND (<3.0E-05) to 6.06 mg/kg ww No U.S. studies

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Data Description and Source	Surface Water Concentration	Fish Tissue Concentration
30Q5 = 30 consecutive days of lowest flow over a 5-year period; HM = harmonic mean; ND = non-detect; ww = wet weight ^a The harmonic mean flow is typically used—estimate fish tissue concentrations because it represents long-term average flow conditions, and it takes time for chemical concentrations—accumulate in fish. However, the HM is not appropriate this COU with only 12 release days. EPA used the 30Q5 as a conservative screening-level analysis even though the 30Q5 is expected—occur less frequently and unlikely—be sustained long-term.		

7.1 Fish Ingestion Exposure Estimates

Acute toxicity of TBBPA pertaining to an exposure duration of 1 day is not expected ([U.S. EPA, 2026c](#)) and thus not calculated. Intermediate toxicity with an exposure duration of up to 30 days is expected, but no intermediate fish IR is available. In the absence of more specific exposure factors, EPA selected an acute fish IR for estimating intermediate exposure to be health protective. Equation 7-1 shows how exposure from fish ingestion was calculated.

Equation 7-1. Fish Ingestion Calculation

$$ADR \text{ or } ADD = \frac{(SWC \times BAF \times IR \times CF1 \times CF2 \times ED)}{AT}$$

Where:

<i>ADR</i>	=	Acute dose rate (mg/kg-day). Referred to as an intermediate dose hereafter because acute toxicity is not expected.
<i>ADD</i>	=	Average daily dose (mg/kg-day)
<i>SWC</i>	=	Surface water (dissolved) concentration (µg/L)
<i>BAF</i>	=	Bioaccumulation factor (L/kg ww)
<i>IR</i>	=	Fish ingestion rate (g/kg-day)
<i>CF1</i>	=	Conversion factor (1.0×10 ⁻³ mg/µg)
<i>CF2</i>	=	Conversion factor (1.0×10 ⁻³ kg/g)
<i>ED</i>	=	Exposure duration (years)
<i>AT</i>	=	Averaging time (year)

EPA estimated exposure from fish consumption for only tribal populations to characterize the upper bound of exposure. Tribal populations have the highest fish ingestion rates among the general population even when including subsistence fishers. U.S. EPA ([2011b](#)) (Chapter 10, Table 10-6) summarizes relevant studies on current tribal-specific fish ingestion rates that covered 11 tribes and 94 Alaskan communities. The highest central tendency value (a mean) ingestion rate per kg of body weight was reported in a 1997 survey of adult members (16+ years) of the Suquamish Tribe in Washington. Adults from the Suquamish Tribe reported a mean ingestion rate of 2.7 g/kg-day, or 216 g/day, assuming an adult body weight of 80 kg. In comparison, the ingestion rates for the adult subsistence fishers and general population are 142.2 and 22.2 g/day, respectively. A total of 92 adults responded to this 1997 survey that was funded by the Agency for Toxic Substances and Disease Registry (ATSDR) through a grant to the Washington State Department of Health.

In addition to the current mean fish ingestion rate, EPA reviewed literature and surveys to identify a high-end (*i.e.*, 90th or 95th percentile) current fish ingestion rate for tribal populations. The surveys asked participants to estimate their daily fish consumption over the course of a year by meal size and meal frequency. The highest 95th percentile fish and shellfish ingestion rate was 874 g/day, or 10.9 g/kg-day assuming a body weight of 80 kg, for male adults (18+ years) of the Shoshone-Bannock Tribes

in Idaho ([Polissar, 2016](#)). The 95th percentile ingestion rate for males and females combined was not much lower at 10.1 g/kg-day ([Polissar, 2016](#)). The Suquamish Tribe also reported similar high-end (90th percentile) current ingestion rates for adults ranging from 8.56 to 9.73 g/kg-day ([Duncan, 2000](#)). Estimated high-end fish ingestion rates were lower for other tribes in Alaska, the Pacific Northwest, Great Lakes region, and northeastern North America. To evaluate a current high-end exposure scenario, EPA used the highest 95th percentile ingestion rate of 10.9 g/kg-day.

EPA proceeded with using two current ingestion rates: 216 g/day (2.7 g/kg-day) for a central tendency and 874 g/day (10.9 g/kg-day) as a high-end or 95th percentile. The same ingestion rates were used to estimate both the intermediate and ADD for tribal populations. U.S. EPA ([2011b](#)) provides fish ingestion rates specific to younger lifestages for select tribes, but adults still consume higher amounts of fish per kilogram of body weight at the 50th and 95th percentiles. As a result, exposure estimates focused on adults. See *Draft Fish Ingestion Risk Calculator for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026b](#)) for all exposure estimates.

Table 7-2 presents the exposure estimates that were calculated with modeled surface water concentrations based on TRI reported releases and an estimated BAF.

Table 7-2. Adult Tribal Fish Ingestion Doses by Surface Water Concentration

Surface Water Concentration and Scenario	Intermediate/ADD (mg/kg-day)	
	Current Mean IR	Current Tribal IR, 95th Percentile
Processing – As a reactant – Flame retardant in plastic material and resin manufacturing	6.44E-08	2.60E-07
Processing – Incorporation in – formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing	3.88E-03	1.57E-02
Disposal	2.55E-03	1.03E-02
ADD = average daily dose; Int. = intermediate exposure estimate; IR = ingestion rate		

7.2 Weight of Scientific Evidence Conclusions

7.2.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty

The modeled surface water concentrations are based on reported releases and receiving water body and hydrologic flow information. For two COUs, surface water concentrations based on the 30Q5 instead of a harmonic mean flow conditions were used because of the low release days (*i.e.*, <1 month). A 30Q5 does not represent long-term flow conditions that are preferable to estimating chemical concentrations in fish tissue. However, it is appropriate to use in a screening-level analysis. Overall, TBBPA concentrations in fish tissue calculated from the modeled surface water concentrations were within the same order of magnitude as measured fish tissue values in studies from countries outside of the U.S. Despite the use of a water concentration modeled using the 30Q5 flow rate, this comparison gives EPA moderate confidence that these calculated fish tissue concentrations are not grossly overestimated in the absence of U.S. biomonitoring studies in fish.

Lastly, TBBPA is not likely to bioconcentrate and bioaccumulate in fish because of its low water solubility and preferential sorption to organic matter, which limits its bioavailability. This is supported

1308 by the estimated BAF value of 153 L/kg ww ([U.S. EPA, 2026d](#)). Overall, TBBPA's environmental
1309 behavior is governed by its aqueous mobility and rapid metabolic transformation rather than strong lipid
1310 partitioning despite the chemical's lipophilicity.

8 AMBIENT AIR CONCENTRATION

EPA modeled ambient air concentrations to assess inhalation exposure to TBBPA. EPA's modeling estimates both short-term and long-term concentrations in ambient air as well as dry, wet, and total deposition rates.

8.1 Approach for Estimating Concentrations in Ambient Air

EPA used previously peer-reviewed methodology for fenceline communities ([U.S. EPA, 2022](#)) to evaluate exposures and deposition via the ambient air pathway for this assessment. This methodology uses the Integrated Indoor/Outdoor Air Calculator (IIOAC) Model. IIOAC is a spreadsheet-based tool that estimates outdoor air concentrations and deposition rates using pre-run results from a suite of dispersion scenarios in a variety of meteorological and land-use settings within EPA's American Meteorological Society/EPA Regulatory Model. Additional information on IIOAC can be found in the user guide ([U.S. EPA, 2019b](#)). IIOAC models the daily-average and annual-average concentrations of TBBPA in the ambient air at three distances (*e.g.*, 100; 100–1,000, and 1,000 m) from a releasing facility. It also estimates dry, wet, and total deposition rates of TBBPA from the ambient air to other media (*e.g.*, water and land) at those same distances.

For air releases, EPA queried both TRI and estimated releases for one COU that does not have TRI data ([U.S. EPA, 2026e](#)). For TRI facilities, EPA obtained annual releases of TBBPA across all COUs that manufacture, process, use, or dispose of TBBPA between 2019 and 2023 (Table 8-1). The ambient air releases were separated into releases by Albemarle Corporation South Plant, Magnolia, Arkansas (single facility) and by all other facilities (*n* = 29) combined as part of a screening assessment. The highest stack release was from Albemarle in 2020 (2,370 kg/site-year), and the stack release from all other facilities combined were lower. On the other hand, the highest fugitive release was from all other facilities in 2020 (410 kg/site-year) and were used in the screening assessment. The annual releases for the TRI facilities were divided by 365 and 2 days to calculate daily releases. These represent average daily releases per day, where 365 days per year represent best case scenario, and 2 days per year represent worst case scenario ([U.S. EPA, 2026e](#)). Two release days came from test order data submitted by ACC ([Integral Consulting, 2025](#)). The ACC consisted of 21 companies that operate 26 TBBPA sites across the United States. Those 26 sites are associated with 8 COUs ([U.S. EPA, 2026e](#)).

For the Electrical and electronic waste recycling COU, environmental releases were estimated using generic scenarios ([U.S. EPA, 2026e](#)). The media types that this COU releases to are air, landfill, incineration, or some combination of the above. EPA is uncertain as to the extent of releases to each media type. Therefore, for the ambient air screening analysis, EPA assumed all is released to air. EPA also assumed that an e-waste recycling facility is operated continuously with 2 weeks of annual downtime for maintenance, for a total of 350 days per year.

Daily releases is a direct input into IIOAC. Although the maximum stack or fugitive releases can vary between facilities and COUs, for this assessment, EPA assumed the releases occurred from the same location at the same time to determine a "total exposure" to TBBPA from both release types. Because these releases do not align spatially or temporally, the modeled concentrations used in this assessment give plausible high-end bounding model concentrations.

1353 **Table 8-1. COUs with Releases to Air, the Source of Air Release Data, and the Type of Release**

COU	Source of Air Releases	Type of Air Release
Manufacturing (Albemarle Corporation)	TRI	Stack
- Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – As a reactant – Intermediate in all other chemical product and preparation manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Adhesive manufacturing - Processing – Incorporation into article – Reactive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., printed circuit boards and semiconductor packages) - Processing – Incorporation into article – Reactive flame retardant in transportation equipment manufacturing (e.g., interior material for automotive and aviation articles) - Processing – Incorporation into article – Additive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., plastic electronic enclosures) - Industrial and commercial use – Construction, paint, electrical, and metal products – Reactive flame retardant in automotive and aviation articles (i.e., laminate and prepreg material) - Disposal	TRI	Fugitive
Electrical and electronic waste recycling	Estimated	Stack and fugitive

8.1.1 Release and Exposure Scenarios Evaluated

The release and exposure scenarios evaluated for this analysis are summarized below.

- Release: Annual release (kg/site-year)
 - Maximum daily release (kg/site-day)
- Release dataset: TRI and generic scenario
- Release type:
 - TRI facilities: Stack for Manufacturing COU and Fugitive for nine COUs as listed in Table 8-1
 - Estimated: Electrical and electronic waste recycling COU
- Distances evaluated: 100 m, 100 to 1,000 m, and 1,000 m
- Meteorological station
 - South (coastal): integrates meteorological data that result in highest ambient air concentrations (more conservative)
 - Northeast (inland): integrates meteorological data that result in highest air to soil deposition estimates
- Operating scenario:
 - For TRI facilities

- 365 days per year represents 24 hours a day, 7 days per week, and 52 weeks per year
 - 2 days per year represents 24 hours a day
- For estimated releases
 - 350 days per year represents 24 hours a day, 7 days per week, 50 weeks per year
- Topography: Urban and rural
- Particle Size:
 - Coarse (PM₁₀): Particulate matter with an aerodynamic diameter of 10 microns
 - Fine (PM_{2.5}): Particulate matter with an aerodynamic diameter of 2.5 microns

EPA used default release input parameters integrated within the IIOAC Model for both stack and fugitive releases along with a user-defined length and width for fugitive releases as listed in Table 8-2.

Table 8-2. IIOAC Input Parameters for Stack and Fugitive Air Releases

Stack Release Parameters	Value
Stack height (m)	10
Stack diameter (m)	2
Exit velocity (m/sec)	5
Exit temperature (K)	300
Fugitive Release Parameters	Value
Length (m)	10
Width (m)	10
Angle (degrees)	0
Release height (m)	3.05

8.1.2 IIOAC Model Output Values

The IIOAC Model provides multiple output values (see *Draft Ambient Air IIOAC Exposure Results and Risk Calculations for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026a](#)). A description of select outputs relied upon in this assessment are provided below. These outputs were selected because they represent an exposure scenario where modeled concentrations are expected to be higher, thus more protective of exposed populations.

- Fenceline average represents the daily-average and annual-average concentrations at 100 m distance from a releasing facility.
- High-end, daily-average represents the 95th percentile daily average of all modeled hourly concentrations across the entire distribution of modeled concentrations at 100 m.
- High-end, annual-average is the 95th percentile annual-average concentration across the entire distribution over a year of modeled concentrations at 100 m.
- High-end, annual average deposition rate is the 95th percentile annual-average deposition rate across the entire distribution of modeled deposition rates at 100 m (total, dry, and wet)

For this assessment, the high-end fenceline (100 m) average concentrations were used to evaluate exposures (and associated risks) to determine if there were any exposures of concern for the general population.

8.1.3 Modeled Results from IIOAC

All results for each scenario are available in the *Draft Ambient Air IIOAC Exposure Results and Risk Calculations for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026a](#)). EPA

used the highest estimated concentrations across all modeled scenarios to evaluate general population exposures near a releasing facility. This exposure scenario represents a national level exposure estimate inclusive of sensitive and locally impacted populations who live next to a releasing facility.

The IIOAC model provides source apportioned concentrations (fugitive and stack) based on the respective releases. To evaluate exposures for this ambient air assessment, EPA assumes the fugitive and stack releases occur simultaneously throughout the day and year. Therefore, the total concentration used to evaluate exposures and derive risk estimates in this ambient air assessment is the sum of the separately modeled fugitive and stack concentrations at 100 m from a releasing facility. The highest daily and annual fugitive and stack concentrations are provided in Table 8-3.

Table 8-3. Highest Daily-Average and Annual-Average IIOAC-Modeled Concentrations at 100 m from Releasing Facility

COU	Data Source	Source Type	Daily-Average Concentration (µg/m³)	Annual-Average Concentration (µg/m³)
Manufacturing ^a	TRI	Fugitive	150	3.40
- Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – As a reactant – Intermediate in all other chemical product and preparation manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Adhesive manufacturing - Processing – Incorporation into article – Reactive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., printed circuit boards and semiconductor packages) - Processing – Incorporation into article – Reactive flame retardant in transportation equipment manufacturing (e.g., interior material for automotive and aviation articles) - Processing – Incorporation into article – Additive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., plastic electronic enclosures) - Industrial and commercial use – Construction, paint, electrical, and metal products – Reactive flame retardant in automotive and aviation articles (i.e., laminate and prepreg material) - Disposal ^a		Stack	150	1.28
		Total	300	4.68
Electrical and electronic waste recycling ^b		EPA estimated	Fugitive	26.53
	Stack		1.10	1.06
	Total		27.63	26.50

^a While releases were reported in TRI for this COU, the combination of facilities to estimate fugitive, stack, and total releases does not result in spatially or temporally aligned releases. However, it is appropriate for a screening-level assessment.

^b Releases are estimated for a hypothetical facility because no TRI data were available for this COU.

The source appointed wet and dry deposition rates and the total deposition rates for the scenario are provided in Table 8-4.

Table 8-4. Highest Annual-Average HIOAC-Modeled Wet, Dry, and Total Deposition Rates at 100 m from Releasing Facility

COU	Data Source	Source Type	Total Annual-Average Deposition (g/m ²)		
			Total	Wet	Dry
Manufacturing ^a	TRI	Fugitive	8.28E-05	8.26E-05	2.48E-06
- Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – As a reactant – Intermediate in all other chemical product and preparation manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Adhesive manufacturing - Processing – Incorporation into article – Reactive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., printed circuit boards and semiconductor packages) - Processing – Incorporation into article – Reactive flame retardant in transportation equipment manufacturing (e.g., interior material for automotive and aviation articles) - Processing – Incorporation into article – Additive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., plastic electronic enclosures) - Industrial and commercial use – Construction, paint, electrical, and metal products – Reactive flame retardant in automotive and aviation articles (i.e., laminate and prepreg material) - Disposal^a		Stack	4.30E-04	4.28E-04	1.09E-05
		Total	5.12E-04	5.11E-04	1.33E-05
Electrical and electronic waste recycling ^b		EPA estimated	Fugitive	7.95E-04	7.87E-04
		Stack	5.29E-04	5.23E-04	6.47E-06
		Total	1.32E-03	1.31E-03	1.54E-05

^a While releases were reported in TRI for this COU, the combination of facilities to estimate fugitive, stack, and total releases does not result in spatially or temporally aligned releases. However, it is appropriate for a screening-level assessment.

^b Releases are estimated for a hypothetical facility because no TRI data were available for this COU.

8.2 Ambient Air Exposure and Air Deposition

8.2.1 Inhalation of Ambient Air

Because TBBPA is not expected to be acutely toxic, only the chronic risk estimates were calculated for both the inhalation and oral pathways. Refer to the *Draft Human Health and Environmental Hazard*

Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA) ([U.S. EPA, 2026c](#)) for more information on the hazard endpoints. To assess chronic non-cancer exposures to the general population, the total annual average modeled ambient air concentrations from IIOAC (Table 8-3) were used to derive exposure estimates. EPA assumed continuous exposure to the TBBPA concentrations modeled in ambient air (*i.e.*, 24 hours per day, 7 days per week, and 365 days per year). Risk characterization for this pathway can be found in the *Draft Risk Evaluation for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026f](#)).

8.2.2 Ingestion of Soil from Air Deposition

TBBPA may be introduced to soils through a variety of mechanisms, such as dry deposition of TBBPA emitted into the atmosphere in its solid particulate form or bound to atmospheric particulates. TBBPA soil concentration resulting from deposition was calculated using Equation 8-1. The annual deposition to soil (Tot_{Dep}) was taken from Table 8-4.

Equation 8-1. TBBPA Concentration in Soil via Air Deposition

$$Soil_{conc} = \frac{Tot_{Dep} \times CF}{Mix \times Dens}$$

Where:

$Soil_{conc}$	=	Concentration in soil (mg/kg)
Tot_{Dep}	=	Total annual deposition to soil (g/m ²)
CF	=	Conversion of g to mg
Mix	=	Mixing depth (m); default = 0.1 m from the European Commission Technical Guidance Document (ECB, 2003)
$Dens$	=	Density of soil; default = 1,700 kg/m ³ from the European Commission Technical Guidance Document (ECB, 2003)

To estimate chronic exposure from soil ingestion, CADD was calculated according to Equation 8-2 using the soil concentration from the previous equation.

Equation 8-2.

$$CADD = \frac{Soil_{conc} \times IR \times EF \times ED \times CF}{BW \times AT}$$

Where:

$CADD$	=	Chronic average daily dose (mg/kg-day)
$Soil_{conc}$	=	Concentration in soil (mg/kg)
IR	=	Intake rate of contaminated soil (mg/day)
EF	=	Exposure frequency (days/year)
ED	=	Exposure duration (years)
CF	=	Conversion factor (1×10 ⁻⁶ kg/mg)
BW	=	Body weight (kg)
AT	=	Averaging time (day)

For the soil intake rate, EPA's selected value includes ingestion of outdoor settled dust, or particles that have settled onto outdoor objects and surfaces from air deposition. Outdoor settled dust is generally present on the uppermost surface layer of soil. U.S. EPA ([2017c](#)) recommends a soil intake rate of 90 mg/day for children under 12 years of age. This intake rate also accounts for inhalation and subsequent swallowing of soil or dust particles. It decreases to 50 mg/day for children 12 years or older. The

exposure duration was assumed to be 12 years. To derive an intake rate that is protective of all lifestyles, EPA used an intake rate of 90 mg/day and a body weight of 11.8 kg for children aged one to less than 2 years ([U.S. EPA, 2011a](#)). CADD was only calculated for children aged 1 to less than 2 years because that lifestage had the greatest intake rate per kg of body weight, thus resulting in the highest exposure (Table 8-5). The full calculations for general population exposure to soil ingestion from air to soil deposition can be found in the *Draft Ambient Air IIOAC Exposure Results and Risk Calculations for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026a](#)).

Table 8-5. Exposure Estimates for Ingestion of Soil from Air Deposition

COU	Data Source	Assessed Population	Soil Concentration (mg/kg)	Chronic Doses (mg/kg-day)
Manufacturing	TRI	Children 1 to <2 years of age	3.10E-03	2.42E-08
- Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – As a reactant – Intermediate in all other chemical product and preparation manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Adhesive manufacturing - Processing – Incorporation into article – Reactive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., printed circuit boards and semiconductor packages) - Processing – Incorporation into article – Reactive flame retardant in transportation equipment manufacturing (e.g., interior material for automotive and aviation articles) - Processing – Incorporation into article – Additive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., plastic electronic enclosures) - Industrial and commercial use – Construction, paint, electrical, and metal products – Reactive flame retardant in automotive and aviation articles (i.e., laminate and prepreg material) - Disposal				
Electrical and electronic waste recycling	EPA estimated		7.89E-03	6.15E-08

8.3 Measured Concentrations in Ambient Air

EPA reviewed published literature as described in the *Draft Systematic Review Protocol for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026g](#)) to identify studies that present measured ambient air concentrations of TBBPA. EPA filtered the studies based on locations in the

United States with a publication date later than 1991 and did not find any studies that met these criteria. Although EPA acknowledges that there are studies outside these filters that measured TBBPA ambient air concentrations, for this assessment, the Agency only focused on recent studies conducted in the United States as these may be more readily associated with current operating scenarios and COUs under TSCA.

8.4 Weight of Scientific Evidence Conclusions

EPA has moderate confidence in the exposure estimates for this pathway. The release assessment handles all facilities except Albemarle as one site because the combined stack releases from all the facilities are still an order of magnitude less than Albemarle's stack releases alone. This approach results in modeled air concentrations that are neither spatially nor temporally aligned. However, it allows EPA to estimate a "total exposure" from all facilities. Furthermore, the Agency used conservative inputs in its IIOAC modeling and considered a series of exposure scenarios under varying operating scenarios and multiple particle sizes. This likely leads to overestimating the ambient air concentrations and deposition rates. The IIOAC modeling is also based on previously peer reviewed methodology and incorporates recommendations received during previous peer review and public comment. Although EPA has moderate confidence in the precision of the exposure estimates, the Agency has robust confidence in the values being representative of an upper bound of potential exposure from these releases and are health protective.

9 HUMAN MILK EXPOSURE

Infants are potentially susceptible because of their higher ingestion per body weight, immature metabolic systems, and the potential for chemical toxicants to disrupt sensitive developmental processes. EPA considered whether infants may be exposed to TBBPA through human milk by examining TBBPA's physical and chemical properties (*i.e.*, lipophilicity, molecular weight), exposure timing and life-stage considerations of point of departure (POD) derivation (*i.e.*, identified key hazard outcomes with considerations for additional potential infant susceptibility), and presence of biomonitoring data. EPA determined the most scientifically supportable approach to evaluate this pathway is to draw on available hazard and exposure information presented in Sections 9.1 and 9.2, respectively.

9.1 Potential for Hazards Unique to Nursing Infants

EPA considered the available hazard data to determine the extent to which human milk exposure is considered a significant route of exposure and if there is evidence of increased susceptibility associated with infant exposure. Across available gestational one- and two-generation exposure studies, the hazard POD for intermediate exposure durations are based upon increased incidence of polycystic kidney lesions in young male rats. It was derived from a study accounting for potential early lifestage sensitivity (*i.e.*, neonatal exposure). Comparing the POD for this effect to other health outcomes (*i.e.*, neurological, liver, and other kidney effects) from studies examining gestational and/or lactational exposure, this POD was determined to be the most sensitive and robust health endpoint. In other words, it is expected to be protective of all other health outcomes associated with intermediate exposures. For chronic scenarios, EPA's selected POD is based upon uterine tumor formation in adult female rats and was determined to follow a threshold for biological effects. Available evidence suggests that TBBPA is not acutely toxic. For further details on the human health studies and how they were used to derive a POD, see *Draft Human Health and Environmental Hazard Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026c](#)).

Overall, in considering the available evidence, hazard values designed to be protective of adult (maternal) exposure are also expected to be protective of nursing infants. EPA considers the proposed oral PODs for use in characterizing risk from exposure to TBBPA to be protective of potentially sensitive subpopulations and lifestages, including infants.

9.2 Potential for Exposure through Milk

9.2.1 Biomonitoring Information

TBBPA has the potential to accumulate in human milk because of its lipophilicity ($\log K_{OW} = 4.75$). EPA identified 18 biomonitoring studies from reasonably available information that investigated TBBPA or its metabolites in human milk. Milk samples were collected from various timepoints ranging from days to months after childbirth in North America (1 study), Europe (9 studies), and Asia (8 studies). The subsequent paragraphs summarize the available studies, and Appendix E provides additional details on each one.

Across all 18 studies, over 2,200 human milk samples were collected from 8 countries. TBBPA concentrations showed a large variability, ranging from less than 0.034 to 2,209 ng/g lw. The highest concentration was observed in a cohort of 63 Japanese mothers by Fujii et al. (2018), where TBBPA was detected in 97% of the samples. Fujii et al. (2018) also reported the minimum concentration of less than 0.034 ng/g and an arithmetic mean, median, and geometric mean of 48, 3.2, and 3.0 ng/g, respectively. Lankova et al. (2013) reported the next highest concentration at 688 ng/g lw among 50 women from the Czech Republic, with a detection frequency of 30%. Among the remaining studies, 15 did not measure

any single concentrations above 100 ng/g and central tendency (mean or median) above 10 ng/g. The only U.S. study by Carignan et al. (2012) detected low concentrations of TBBPA in milk. The chemical was observed in 15 of the 43 samples collected 2 to 8 weeks postpartum from 43 first-time mothers in 2004 to 2005. Concentrations ranged from below the limit of quantitation/quantification (LOQ) (<0.030) to 0.55 ng/g lw. The participants were 18 years and older, lived in the Greater Boston area for at least 3 years, spoke Spanish or English, and had pregnancies that were healthy and singlet. The geometric mean was not reported due to a low detection frequency (35%).

Four of the 18 studies measured TBBPA concentrations in milk on a wet weight (ww) basis and must be compared separately from those reporting based on a lw. These four had a total sample size of 115 (Martínez et al., 2019; Lankova et al., 2013; Akiyama et al., 2015). Martínez et al. (2019) did not provide a range, but the mean concentration from a cohort of 53 mothers was 580 ng/mL ww. In comparison, the maximum concentration from the other two studies was an order of magnitude lower at 16.2 ng/mL.

These biomonitoring studies provide empirical evidence of TBBPA's presence in human milk. Because only one U.S. study was identified and samples were collected 20 years ago, the degree to which these milk samples represent current TBBPA exposures in the United States is not known. Instead, to calculate a nursing infant's exposure, the maximum measured concentrations from Fujii et al. (2018) was used to calculate a nursing infant's exposure and for the purpose of a screening analysis. Equation 9-1 and Equation 9-2 show the calculation methods using TBBPA concentrations based on a lipid and wet weight, respectively. Milk ingestion rates (IR) for multiple age groups within the first year of life are available in EPA's *Exposure Factors Handbook* (U.S. EPA, 2011b) and in Table_Apx A-4. For this screening analysis, EPA used only IR for infants under a month old because it is the highest across the age groups. The milk IR also factors in an exposure duration, body weight, and averaging time; these parameters were thus excluded from the following equations.

Equation 9-1. Milk Ingestion Calculation Lipid Weight

$$\text{Exposure Estimate} = MC \times IR \times CF1$$

Where:

MC	=	Milk concentration (2,209 ng/g lipid weight (Fujii et al., 2018))
IR	=	Milk ingestion rate (9.0 g/kg-day)
CF1	=	Conversion factor (1×10^{-6} mg/ng)

Equation 9-2. Milk Ingestion Calculation Wet Weight

$$\text{Exposure Estimate} = MC \times IR \times CF1$$

Where:

MC	=	Milk concentration (580 ng/mL ww (Martínez et al., 2019))
IR	=	Milk ingestion rate (220 mL/kg-day)
CF1	=	Conversion factor (1×10^{-6} mg/ng)

EPA estimated intermediate and chronic non-cancer risks for a nursing infant using biomonitoring data. Acute exposure was not considered because TBBPA is not expected to be acutely toxic across all exposure routes (U.S. EPA, 2026c) and the absence of milk ingestion rates to characterize a peak daily

exposure. While chronic exposure represents repeated exposures covering at least 10% of lifetime in adults, EPA typically estimates chronic risk to a nursing infant when exposure throughout the first year of life may have sensitive effect levels comparable to or greater than identified hazards. Furthermore, TBBPA's intermediate duration POD suggested an early lifestage sensitivity. Therefore, chronic non-cancer risks were considered for infant doses in the first year of life. For cancer risks, TBBPA's potential cancer effects are not mediated through direct DNA damages (*i.e.*, no genotoxic mode of action) where any exposure is associated with some positive risk of developing cancer (see *Draft Human Health and Environmental Hazard Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol]* (TBBPA) ([U.S. EPA, 2026c](#))). In other words, TBBPA is not considered genotoxic or mutagenic.

Screening-level exposure estimates using the maximum measured TBBPA concentrations in human milk are presented in Table 9-1.

Table 9-1. Screening-Level Exposure from Human Milk Ingestion Based on Biomonitoring Data

Age Group	Exposure Estimates	
	Lipid Weight (mg/kg-day)	Wet Weight (mg/kg-day)
Birth to <1 month	1.98E-02	3.56E-03
Birth to <1 year	1.41E-02	2.47E-03
^a Intermediate risks are only estimated for the infant's first month of life, which had the highest doses because of the highest milk ingestion rate per kilogram of body weight. It is thus most protective for estimating shorter exposures. Furthermore, chronic risks cover much longer exposure durations (<i>i.e.</i> , at least 10% of lifetime in adults) than the 1 month represented by this exposure estimate.		
^b The exposure duration for an intermediate exposure is up to 30 days. The exposure duration for this age group exceeds 30 days.		

9.2.2 Potential to Model TBBPA Concentrations in Milk

EPA considered the potential to model TBBPA concentrations in human milk resulting from specific sources of maternal exposures to provide quantitative estimates of COU-specific milk exposures and risks. The Agency identified a pharmacokinetic model described in Kapraun et al. ([2022](#)) as the best available model to estimate milk concentrations resulting from the transfer of lipophilic chemicals from mothers to infants during gestation and lactation. The only chemical-specific parameter required by the Kapraun model is the elimination half-life in the animal species of interest.

Although such modeling may be possible for TBBPA, it is not needed to adequately assess exposures and risks to infants exposed through milk. As described in Section 9.1, EPA did not identify an infant-specific hazard POD that would be directly comparable to infant exposure estimates derived from modeling milk concentrations. Furthermore, EPA concluded that the PODs applied to assess risks to workers, consumers, and the general population in this assessment are expected to be protective of nursing infants.

9.3 Weight of Scientific Evidence Conclusions

Human milk is a relevant exposure pathway for TBBPA because available information on physical, chemistry and fate properties and biomonitoring data suggest that TBBPA can be and is present in human milk.

While available biomonitoring data were used to characterize a nursing infant's exposure to TBBPA, it is important to note that biomonitoring data represent an aggregate exposure from all TBBPA sources

and pathways, including both TSCA and non-TSCA uses. Biomonitoring data reflect total infant exposure through human milk ingestion, and the contribution of specific TSCA COUs to overall exposure cannot be determined. However, because biomonitoring data reflect total exposure and application of the highest measured concentrations in a screening assessment did not indicate risks, EPA does not expect the milk pathway to be a concern. A similar conclusion was reached by other regulatory and authoritative agencies: Australia's National Industrial Chemicals Notification and Assessment (NICNAS) ([2020](#)), European Chemical Bureau (ECB) ([2006](#)), Health Canada's Environment Canada (EC/HC) ([2013](#)), and the European Food Safety Authority (EFSA) ([2024](#)).

Overall, EPA expects the human health hazard values based on early-life (neonatal) and adult exposure scenarios considered throughout this assessment to be protective of nursing infants. These values are based upon studies that identify the most sensitive and robust effect levels for respective exposure duration scenarios. Furthermore, exposure estimates using the maximum measured concentrations in human milk from available biomonitoring data were low. EPA has therefore concluded that further modeling to quantify milk concentrations and infant exposures is not needed.

10 ENVIRONMENTAL BIOMONITORING AND TROPHIC TRANSFER

10.1 Approach and Methodology

EPA assessed the available studies related to the biomonitoring of TBBPA collected in accordance with the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances* ([U.S. EPA, 2021](#)) and *Draft Systematic Review Protocol for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol]* (TBBPA) ([U.S. EPA, 2026g](#)). Chemicals can be transferred from contaminated media and diet to biological tissue and accumulate throughout an organism's 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 reviewed biomonitoring studies and provided qualitative descriptions of the potential dietary exposures to aquatic and terrestrial organisms via feeding (trophic) relationships.

10.2 Measured Concentrations in Aquatic Species

Examination of each trophic level of biomonitoring information in aquatic species are presented here. In U.S. estuarine waters, bull shark (*Carcharhinus leucas*) (n = 13) averaged 13.2 µg/kg lw, while Atlantic sharpnose shark (*Rhizoprionodon terraenovae*) (n = 3) averaged 0.872 µg/kg lw ([Johnson-Restrepo et al., 2008](#)). In bottlenose dolphins from the west coast of Florida, its blubber contained had a mean TBBPA concentrations of 0.451 (n = 4), 1.86 (n = 5), and 1.18 (n = 6) µg/kg lw, and a range of 0.056 to 8.48 µg/kg lw ([Johnson-Restrepo et al., 2008](#)).

In other locations, TBBPA concentrations were reported for nine species of primary consumers (e.g., invertebrates). In the Netherlands, TBBPA concentrations in sea stars (*Asterias rubens*) (n = 3) from estuarine environments were reported to range from below one to 2.0 µg/kg lw ([Morris et al., 2004](#)). In the United Kingdom, hermit crabs (*Pagurus bernhardus*) (n = 9) and common welk (*Buccinum undatum*) (n = 3) from the North Sea exhibited TBBPA concentrations from below 1 to 35 µg/kg lw and 5 to 96 µg/kg lw, respectively ([Morris et al., 2004](#)). Mysid shrimp (*Neomysis integer*) (n = 3796) from the Scheldt estuary showed TBBPA concentrations of 0.8 to 0.9 µg/kg lw ([Verslycke et al., 2005](#)). In South Korea, mollusks (n = 10–20 of each) including *Ruditapes philippinarum*, *Nordotis discus*, and the *Octopus minor* exhibited mean TBBPA concentrations of 2.84, 4.43, and 8.11 µg/kg ww, respectively ([Choo et al., 2019](#)). In China, the Chinese mystery snail (*Cipangopaludina chinensis*) (n = 43) and the prawn (*Macrobrachium nipponense*) (n = 7) had whole body mean wet weight TBBPA concentrations of 1.97 and 1.57 µg/kg, respectively ([Tao et al., 2016](#)).

Fish are secondary and tertiary consumers with TBBPA concentrations reported for 15 species. The concentrations of TBBPA in fish vary significantly by location, reflecting differences in environmental exposure and contamination levels. In North Sea estuaries, whiting (*Merlangius merlangus*) muscle (n = 3) had a lipid-normalized mean of 136 µg/kg, whereas cod (*Gadus morhua*) liver (n = 2) were much lower from less than 0.3 to 1.8 µg/kg lw, indicating species and tissue differences within the same region ([Morris et al., 2004](#)). On Ireland's south coast, hake (liver) (n = 1) TBBPA concentrations were reported at less than 0.2 µg/kg lw ([Morris et al., 2004](#)). In Czechia, an overall mean of 1290 µg/kg wet weight was recorded across freshwater species (*Abramis brama*, *Squalius cephalus*, *Rutilus rutilus*, *Carassius carassius*, *Perca fluviatilis*, *Gobio gobio*, *Thymallus thymallus*, *Cyprinus carpio*, *Oncorhynchus*) (n = 48) ([Hloušková et al., 2013](#)). Along Poland's Southern Baltic coast, Baltic herring (*Clupea harengus*) exhibited lipid-normalized means of 2.3 µg/kg lw in

whole fish (n = 9), 3.27 µg/kg lw in liver (n = 6), and 1.6 µg/kg lw in muscle (n = 6) ([Reindl and Falkowska, 2014](#)).

In the Netherlands, between 2008 and 2013, *Solea solea* exhibited TBBPA concentrations in muscle tissue (n = 15) of 0.73 µg/kg wet weight, with lw measurements of 60.8 µg/kg ([Kotthoff et al., 2017](#)). In another study from the Netherlands, TBBPA was not detected (LOD not specified) in any of 23 pooled fish muscle samples from marine and freshwater locations ([van Leeuwen and de Boer, 2008](#)). In France, from 2007 to 2013, bream (*Abramis brama*) (n = 15) had concentrations from below the LOQ (0.45 µg/kg) to 0.76 µg/kg wet weight, and up to 36.2 µg/kg lw. The United Kingdom reported similar findings in the same period, with concentrations up to 1.19 µg/kg wet weight and up to 62.6 µg/kg lw in bream (n = 15). In Sweden, the concentrations were below LOQ (0.45 µg/kg) in muscle tissue of bream (n = 15) ([Kotthoff et al., 2017](#)). In the Czech Republic, fish concentrations (pooled *Squalius cephalus*, *Abramis brama*, *Aspius aspius*, *Rutilus rutilus*, *Carassius carassius*, *Cyprinus carpio*, *Chondrostoma nasus*; n = 59) ranged from 4.99 to 203 µg/kg lw ([Svihlikova et al., 2015](#)). In Germany, concentrations were below LOQ (0.45 µg/kg) in muscle tissue of bream (*Abramis brama*) (n = 15) ([Kotthoff et al., 2017](#)). In yellow eel from Belgium (n = 19), tissue concentrations ranged from less than 0.1 to 13 µg/kg lw ([Morris et al., 2004](#)). In Italy, concentrations of TBBPA in liver from carp (*Cyprinus carpio*) (n = 4) ranged from below LOD (0.011 µg/kg) to 0.5 µg/kg lw, in bream (*Abramis brama*) (n = 2) from 6.9 µg/kg to 15.4 µg/kg lw, in Sanders (*Sander lucioperca*) (n = 1) a concentration of 0.7 µg/kg lw was measured, and lastly in sheatfish (*Silurus glanis*) (n = 4), concentrations ranged from below LOD (0.011 µg/kg) to 1.6 µg/kg lw ([Luigi et al., 2015](#)). In England, studies conducted in 2008 reported TBBPA concentrations in an individual crucian carp (*Carassius carassius*) and an individual European perch (*Perca fluviatilis*) of 1.7 µg/kg lw and 1.3 µg/kg lw in muscle tissue, respectively. All other samples were below LOQ (0.29 µg/kg) ([Harrad et al., 2009](#)).

In a South China's e-waste pond, whole-body wet-weight means were 0.837 µg/kg ww in mud carp (*Cirrhinus molitorella*) (n = 5) and 0.384 µg/kg wet weight in northern snakehead (*Ophicephalus argus*) (n = 5). Tissue-specific wet-weight means in mud carp (n = 5) ranged from 0.03 µg/kg wet weight in muscle to 2.85 µg/kg wet weight in liver, with 2.60 µg/kg wet weight in fat and 1.42 µg/kg wet weight in gills; in snakehead (n = 5), tissue wet-weight means ranged from 0.04 µg/kg wet weight in muscle/heart to 1.33 µg/kg wet weight in scales. Lipid-normalized muscle mean concentrations were 4.3 µg/kg lw in mud carp and 9.7 µg/kg lw in northern snakehead, underscoring appreciable burdens in lipid-rich tissues ([Tang et al., 2015](#)). At a similar location, whole-body wet-weight means were comparable: mud carp (n = 12) 0.730 µg/kg wet weight, crucian carp (*Carassius auratus*) (n = 18) 0.590 µg/kg wet weight, and northern snakehead (n = 6) 0.350 µg/kg wet weight, while a background pond showed lower mud carp (n = 5) levels at 0.145 µg/kg ww ([Tao et al., 2016](#)). In a South China river catchment, lipid-normalized muscle means were 35.2 µg/kg lw in mud carp (n = 9), 21.2 µg/kg lw in plecostomus (*Hypostomus plecostomus*) (n = 10), and 18.1 µg/kg lw in Nile tilapia (*Tilapia nilotica*) (n = 15) ([He et al., 2013](#)). In the Pearl River Delta's e-waste region, crucian carp muscle showed a lipid-normalized median of µg/kg lw (range 0.23 to 1.74 µg/kg lw) (n = 7) ([He et al., 2010a](#)). Beyond e-waste and industrial sites, Lake Chaohu (China's fifth-largest freshwater lake) exhibited notable dry-weight means across tissues and fish species: whole organism (*Culter alburnus*, *Cyprinus carpio*, *Carassius auratus*, and *Silurus asotus*) (sample size not reported) 28.5 to 39.4 µg/kg dw; liver 16.0 to 37.5 µg/kg dw; kidney 75.2 to 126.4 µg/kg dw; gill 2.08 to 10.21 µg/kg dw; muscle 6.3 to 46.0 µg/kg dw; and adipose 12.0 to 21.9 µg/kg dw ([Yang et al., 2012](#)).

TBBPA was also reported in three species of upper trophic level species. In harbor porpoise (n = 9) from the United Kingdom's North Sea, its blubber contained a TBBPA mean concentration of 83 µg/kg lw, and a range of 0.1 to 418 µg/kg lw ([Morris et al., 2004](#)).

10.3 Modeled Concentrations in Aquatic Species

EPA derived estimated TBBPA concentrations in representative fish from modeled surface water concentrations due to industrial release OESs using VVWM-PSC. EPA used the BCF value of 160 L/kg (U.S. EPA, 2026d) and the highest 7Q10 surface water concentration of 8.52 µg/L, which resulted from the Disposal COU as a first-tier exposure assessment to increase confidence that derived TBBPA concentrations are not underestimated in representative fish tissues. The equation below was used to solve for C_{Fish} as the predicted TBBPA concentration (µg/kg ww) in representative fish.

In this assessment, EPA used Equation 10-1 for BCF at steady state:

Equation 10-1. Calculating BCF at Steady State

$$BCF = \frac{C_{Fish}}{C_{Water}}$$

Where:

BCF	=	Bioconcentration factor (L/kg)
C_{Fish}	=	TBBPA concentration in fish (µg/kg ww)
C_{Water}	=	TBBPA concentration in surface water (µg/L)

Table 10-1. Estimated TBBPA Concentration in Fish Tissues from Industrial Releases

Condition of Use	Daily Release (kg/day)	Surface Water Concentration (µg/L) ^a	TBBPA in Fish (µg/kg ww)
Processing – Incorporating into formulation, mixture, or reaction product – Flame retardant in plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing	0.037	12.8	2,048
Highest wet weight monitored fish tissue (common carp, whole fish) ^b			2.85
ww = wet weight; WWT = wastewater treatment			
^a Estimated surface water concentration of TBBPA (Section 3.3).			
^b Tang et al. (2015) reported 2.85 µg/kg ww in mud carp liver.			

10.4 Measured Concentrations in Terrestrial Species

Across avian studies, TBBPA concentrations vary by species, tissue, and location. In Norway, eggs of white-tailed sea eagle (*Haliaeetus albicilla*) (n = 12), peregrine falcon (*Falco peregrinus*) (n = 9), golden eagle (*Aquila chrysaetos*) (n = 15), and osprey (*Pandoin haliaetus*) (n = 5) contained TBBPA from below the LOQ (0.003 µg/kg) to 0.013 µg/kg wet weight (Herzke et al., 2005). In remote Norwegian seabird colonies, TBBPA in common eider, European shag, and European herring gull eggs (n = 54) were uniformly below 0.4 µg/kg wet weight (Huber et al., 2015). In the United Kingdom, cormorant liver (n = 5) contained a TBBPA mean of 7.1 µg/kg lw, with a range of 2.5 to 14 µg/kg lw (Morris et al., 2004). In the Netherlands, common tern eggs (n = 10) from an estuary near textile and flame-retardant facilities were reported below 2.9 µg/kg lw (Morris et al., 2004). Barn owl tissues from Belgium (n = 15)—muscle, liver, preen gland, and adipose—were below the LOQ (0.4 µg/kg), while body feathers contained TBBPA at a mean of 0.72 µg/kg dw, ranging from 0.40 to 6.06 µg/kg dw (Eulaers et al., 2014). Barn owl tissues (n = 11) from France similarly showed muscle, liver, preen

gland, and adipose below the LOQ (0.4 µg/kg), with body feathers at a mean of 1.08 µg/kg dw, ranging from 0.36 to 7.07 µg/kg dw (Eulaers et al., 2014). In South China's e-waste region, Chinese pond heron (n = 5) muscle had a median of 17.3 µg/kg lw, ranging from 13.3 to 24.3 µg/kg lw, white-breasted waterhen (n = 11) muscle had a median of 17.0 µg/kg lw, ranging from 2.81 to 148 µg/kg lw, common snipe muscle had a median of 5.43 µg/kg lw, ranging from 1.06 to 14.9 µg/kg lw, slaty-breasted rail (n = 8) muscle had a median of 10.3 µg/kg lw, ranging from 0.9 to 13.9 µg/kg lw, spotted dove (n = 4) muscle had a median of 9.01 µg/kg lw, ranging from 3.92 to 50.3 µg/kg lw, and Chinese francolin (n = 9) muscle had a median of 2.82 µg/kg lw, ranging from 2.42 to 5.48 µg/kg lw (He et al., 2010a).

In South China's Shenzhen mangrove nature reserve, TBBPA was detected in various species. *Bruguiera sexangula* (n = 6) had a mean of 0.007 µg/kg dw in roots, 0.27 µg/kg in stems, and a mean of 0.004 ng/g in leaves. *Kandelia obovata* (n = 6) displayed higher burdens, with roots at 0.20 µg/kg dw on average, stems had a mean of 0.22 µg/kg, and leaves had a mean of 0.04 ng/g. *Avicennia marina* (n = 5) exhibited the lowest concentrations among the three species, with roots at 0.003 ng/g dw, stems at 0.004 µg/kg, and leaves at 0.01 µg/kg (Li et al., 2019a). At another site in South China, Green foxtail contained a mean of 42.26 µg/kg dw and peanuts contained (n = 20 total for both species) a mean of 17.17 µg/kg (Wang et al., 2015). Camphor tree bark (sample size not reported) a median range of 130 to 600 µg/kg lw (Han et al., 2017). Lastly, eucalyptus leaves (n = 3) contained a mean of 8.9 µg/kg (He et al., 2010a).

10.5 Trophic Transfer

Trophic transfer is the process by which chemical contaminants can be taken up by organisms through diet and media exposures and be transferred through dietary consumption of prey from one trophic level to another. The physical and chemical properties of TBBPA, including log K_{ow} (4.75), BCF (mean 160 L/kg ww whole body), and BAF (1,386 L/kg wet weight in muscle) (U.S. EPA, 2026d) indicate that TBBPA has low bioaccumulation biomagnification potential. Exposure of environmental organisms to chemicals, including TBBPA, through ingestion or trophic interactions, depends on the persistence and bioaccumulative properties of the chemical inside and outside of the organisms. The concentration of TBBPA in an animal at any one time is the net result of exposure pathway, duration, and within-animal metabolism and excretion. This dietary exposure to wildlife assessment supports the broader TBBPA Risk Evaluation that critically evaluates the potential risks to aquatic and terrestrial organisms, with risk depending on both environmental exposures and hazards. In lieu of wild mammal hazard data, EPA derived a dietary hazard threshold using laboratory rodent experiments and extrapolated to representative wild mammals. Therefore, this section includes an assessment of the water pathway that leads to TBBPA in fish tissues that represent exposure concentrations to fish-eating mammals. This assessment also includes an assessment of the air deposition to soil pathway that leads to TBBPA in earthworms that represent exposure concentrations to earthworm-eating shrews.

TBBPA shows low-to-moderate bioaccumulation based on multiple metrics including BCF, BAFs, BSAF, and TMF. In laboratory BCF tests, fathead minnow and Eastern oyster had BCFs of roughly 160 and 148 L/kg ww respectively, with rapid depuration (1 day in fish and 3–5 days in oysters) (Fackler, 1989). Field BCFs converted to wet weight were 480 L/kg ww for oysters and 920 L/kg ww for mussels, and quantitative structure-activity relationship (QSAR) estimates were in the same range (EPI Suite™ 632 L/kg; OPERA 269 L/kg) (U.S. EPA, 2026d). When dietary exposure is included, laboratory scallop tests resulted in a BAF range of 514 to 1,386 L/kg ww (Hu et al., 2015). Field data from an e-waste pond showed higher BAFs in benthic invertebrates (e.g., Chinese mystery snail 5,184 L/kg; prawn 4,132 L/kg) and generally lower BAFs in fish (921–1,921 L/kg) (Tao et al., 2016). Food web studies of TBBPA indicate no trophic magnification. Trophic magnification factors (TMFs) were 0.25 in a freshwater e-waste pond web (Tao et al., 2016) and 0.545 in a marine web (Choo et al., 2019), both

showing trophic dilution ($TMF < 1$). Biota-sediment accumulation was variable by species and site. The mean biota-sediment accumulation factors (BSAFs) reported were 1.07 ± 3.92 for oysters and 6.96 ± 19.3 for mussels, whereas pelagic fish had a median BSAF of 0.312 (Gu et al., 2017). The weight of evidence indicates TBBPA is not bioaccumulative under typical environmental conditions, a conclusion consistent with the rapid metabolism and elimination observed in test organisms and based on the physical and chemistry and environmental fate properties. Because TBBPA is not expected to biomagnify across trophic levels in aquatic food webs, trophic transfer screening-level analysis will not be conducted for aquatic organisms. See the *Draft Physical Chemistry and Fate Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* (U.S. EPA, 2026d).

Although TBBPA is not expected to biomagnify within the terrestrial environment, relevant environmental exposures are possible through dietary exposure to soil and prey species contaminated with TBBPA. The current trophic transfer screening-level assessment was used to determine if terrestrial uptake and trophic transfer from TBBPA contaminated soils are a concern relative to the derived TRV for mammals. The screening-level trophic transfer analysis is conservative in that it assumes no *in vivo* metabolism or excretion for any representative organism.

Representative mammal species are chosen to connect the TBBPA transport exposure pathway via terrestrial trophic transfer from earthworm uptake of TBBPA from contaminated soil to the representative worm-eating mammal, the short-tailed shrew (*Blarina brevicauda*). Short-tailed shrew primarily feed on invertebrates with earthworms comprising approximately 31% (stomach volume) to 42% (frequency of occurrence) of their diet (U.S. EPA, 1993b). The calculations for assessing TBBPA exposure from soil uptake by earthworms and the transfer of TBBPA through diet to higher trophic levels used maximum soil concentration (0.003014 mg/kg) from air to soil deposition in Section 8.

EPA conducted screening-level approaches for terrestrial risk estimation based on exposure via trophic transfer using conservative assumptions for factors such as: area use factor, fraction of TBBPA absorbed from diet, soil, sediment, and water. Concentration of TBBPA within biota were calculated using the media to biota accumulation factor of 1.0 and the VVWM-PSC-modeled concentrations of TBBPA within the sediment and surface water 7Q10 flow rates for the OES/COU that resulted in the highest TBBPA concentration in each environmental medium. The screening-level approach employs a combination of conservative assumptions (*i.e.*, conditions for several exposure factors included within Equation 10-2 and Equation 10-3) and utilization of the maximum values obtained from modeled and/or monitoring data from relevant environmental compartments. For example, accumulation factors of 1.0 indicate 100% efficiency of chemical uptake which is standard conservative approach for the current screen level analysis. Following the basic equations as reported in Chapter 4 of EPA's *Guidance for Developing Ecological Soil Screening-levels* (U.S. EPA, 2005), wildlife receptors may be exposed to contaminants in soil by two main pathways: (1) incidental ingestion of soil while feeding, and (2) ingestion of food items that have become contaminated due to uptake from soil.

The representative aquatic-dependent terrestrial species is the American mink (*Mustela vison*), whose diet is highly variable depending on habitat. In a riparian habitat, American mink derives 74 to 92% of their diet from aquatic organisms, which includes 61 to 85% fishes, 4 to 11% crustaceans, and 2 to 3% amphibians (Alexander, 1977). Sediment and surface water concentrations of TBBPA modeled using VVWM-PSC represent the high-end annual release per COU/OES and were used as an input for the TBBPA concentration found in the American mink's diet in the form of water intake, incidental sediment ingestion, and a diet of fish. The representative fish for the screening-level trophic transfer analysis is the blacktail redhorse (*Moxostoma poecilurum*) serving as a prey item for the American

mink. The representative prey item for the blacktail redhorse was chironomid larvae (*Chironomus riparius*). These fish have the potential to be exposed to TBBPA within sediment through ingestion of sediment containing TBBPA during feeding. The general equation used to estimate dietary exposure via these two pathways is provided below and has been adapted to include consumption of water contaminated with TBBPA, and, for aquatic-dependent mammals, ingestion of TBBPA within sediment instead of soil:

Equation 10-2. Terrestrial and Aquatic Mammals

$$E_j = \left([S_j \times P_s \times FIR \times AF_{sj}] + [W_j \times WIR \times AF_{wj}] + \left[\sum_{i=1}^N B_{ij} \times P_i \times FIR \times AF_{ij} \right] \right) \times AUF$$

Equation 10-3. Fish

$$E_j = \left([S_j \times P_s \times FIR \times AF_{sj}] + \left[\sum_{i=1}^N B_{ij} \times P_i \times FIR \times AF_{ij} \right] \right) \times AUF$$

Where:

E_j	=	Exposure rate for contaminant (j) (mg/kg-bw/day)
S_j	=	Concentration of contaminant (j) in soil or sediment (mg/kg dw)
P_s	=	Proportion of total food intake that is soil or sediment (kg soil/kg food; SIR/((FIR)(body weight [bw])))
SIR	=	Sediment intake rate (kg of sediment [dry weight] per day)
FIR	=	Food intake rate (kg of food [dry weight] per kg body weight per day)
AF_{sj}	=	Absorbed fraction of contaminant (j) from soil or sediment (s) (for screening purposes set equal to 1)
W_j	=	Concentration of contaminant (j) in water (mg/L) assumed to equal water solubility for the purposes of terrestrial trophic transfer
WIR	=	Water intake rate (kg of water per kg body weight per day)
AF_{wj}	=	Absorbed fraction of contaminant (j) from water (w) (for screening purposes set equal to 1)
N	=	Number of different biota types (i) in diet
B_{ij}	=	Concentration of contaminant (j) in biota type (i) (mg/kg dw)
P_i	=	Proportion of biota type (i) in diet
AF_{ij}	=	Absorbed fraction of contaminant (j) from biota type (i) (for screening purposes set equal to 1)
AUF	=	Area use factor (for screening purposes set equal to 1)

Table 10-2. Terms and Values Used to Assess Trophic Transfer of TBBPA in Terrestrial Ecosystems

Term	Earthworm (<i>Eisenia fetida</i>)	Short-Tailed Shrew (<i>Blarina brevicauda</i>)
P_s	1	0.03 ^a
FIR	1	0.555 ^b
AF_{sj}	1	1

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Term	Earthworm (<i>Eisenia fetida</i>)	Short-Tailed Shrew (<i>Blarina brevicauda</i>)
P_i	1	1
AF_{wj}	1	1
AF_{ij}	1	1
N	1	1
AUF	1	1
S_j	mg/kg TBBPA	mg/kg TBBPA
B_{ij}	mg/kg TBBPA (soil)	mg/kg TBBPA (worm)
^a Soil ingestion as proportion of diet represented at the 90th percentile sourced from EPA's <i>Guidance for Developing Ecological Soil Screening-levels</i> (U.S. EPA, 2005). ^b Exposure factors (<i>FIR</i> and <i>WIR</i>) sourced from EPA's <i>Wildlife Exposure Factors Handbook</i> (U.S. EPA, 1993b).		

Table 10-3. Terms and Values Used to Assess Trophic Transfer of TBBPA from Aquatic to Terrestrial Ecosystems

Term	Blacktail Redhorse (<i>Moxostoma poecilurum</i>)	American Mink (<i>Mustela vison</i>)
P_s	0.32 ^a	5.35E-04 ^b
FIR	0.02 ^c	0.22 ^d
AF_{sj}	1	1
P_i	1	1
WIR	N/A	0.105 ^d
AF_{wj}	1	1
AF_{ij}	1	1
SIR	9.5E-04 ^e	1.20E-04 ^f
B_w	0.148 kg ^g	1.0195 kg ^h
N	1	1
AUF	1	1
S_j	mg/kg ⁱ TBBPA	mg/kg ⁱ TBBPA
W_j	mg/L TBBPA	mg/L ^j TBBPA
B_{ij}	mg/kg ^k <i>C. riparius</i>	mg/kg ^l Fish
^a Sediment ingestion as proportion of diet, calculated from the geometric mean of sediment as a proportion of diet reported in published literature for catostomids (Sule, 1985 ; Dauble, 1986 ; Doyle, 2011). ^b Sediment ingestion as proportion of diet, calculated by dividing the <i>SIR</i> by kg food, where kg food equal to <i>FIR</i> multiplied by body weight (bw) of the mink. ^c Daily feed rate reported from apparent satiation in laboratory growth study for juvenile black buffalo (<i>Ictiobus niger</i>) (Guy et al., 2018). ^d Exposure factors (<i>FIR</i> and <i>WIR</i>) sourced from EPA's <i>Wildlife Exposure Factors Handbook</i> (U.S. EPA, 1993b) for mink. ^e <i>SIR</i> reported as kg of sediment in diet at a <i>FIR</i> of 0.02 based on a mean body		

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Term	Blacktail Redhorse (<i>Moxostoma poecilurum</i>)	American Mink (<i>Mustela vison</i>)
	weight of 148 g (Guy et al., 2018) and sediment ingestion rate of 0.32.	
	^f Exposure factor (<i>SIR</i>) for mink sourced from EPA's <i>Second Five Year Review Report Hudson River PCBs Superfund Site to Appendix 11: Human Health and Ecological Risks</i> (U.S. EPA, 2017b).	
	^g Fish body weight used to calculate <i>FIR</i> (Guy et al., 2018).	
	^h Mink body weight used to calculate <i>P_s</i> sourced from EPA's <i>Wildlife Exposure Factors Handbook</i> (U.S. EPA, 1993b).	
	ⁱ Sediment concentration of TBBPA obtained using VVWM-PSC modeling for each respective COU/OES presented in Table 1-1.	
	^j Surface water concentration of TBBPA obtained using VVWM-PSC modeling for each respective COU/OES	
	^k Chironomid TBBPA concentration (mg/kg) calculated from modeled sediment concentration of TBBPA (VVWM-PSC).	
	^l Fish TBBPA dietary exposure rate (mg/kg bw/day) represented from application of Equation 10-3.	

At the screening-level, the conservative assumption is that the invertebrate diet for the short-tailed shrew is comprised entirely of earthworms from contaminated soil. The screening-level analysis for trophic transfer of TBBPA to the short-tailed shrew used the highest calculated soil contaminant concentration to determine if a more detailed assessment is required. The calculated TBBPA concentration in soil from modeled air deposition to soil was 3×10^{-3} mg/kg. The highest sediment concentration modeled via VVWM-PSC was used to represent TBBPA concentrations in media for trophic transfer for fish consuming chironomids to an aquatic-dependant mammal (American mink). Additional assumptions for this analysis have been considered to represent conservative screening values ([U.S. EPA, 2005](#)). Within this model, incidental oral soil or sediment exposure is added to the dietary exposure resulting in total oral exposure to TBBPA. In addition, EPA assumes that 100% of the contaminant is absorbed from the soil or sediment (AF_{sj}), water (AF_{wj}) and biota representing prey (AF_{ij}). The proportional representation of time an animal spends occupying an exposed environment is known as the area use factor (*AUF*) and has been set at 1 for all organisms. Values for calculated dietary exposure are shown in Table 10-4 for trophic transfer to shrew from the air to soil deposition value. For trophic transfer from surface water release of TBBPA to fish consuming chironomids and mink consuming fish are shown in Table 10-5 and Table 10-6, respectively. Chironomid concentrations (mg/kg) were calculated using sediment concentrations of TBBPA.

Exposure factors for mammals included food intake rate (*FIR*) and water intake rate (*WIR*) and were sourced from the EPA's *Wildlife Exposure Factors Handbook* ([U.S. EPA, 1993b](#)). The exposure factor for sediment intake rate (*SIR*) for mammals was sourced from the EPA's *Second Five Year Review Report Hudson River PCBs Superfund Site to Appendix 11: Human Health and Ecological Risks* ([U.S. EPA, 2017b](#)). *FIR* for the blacktail redhorse is represented with daily feed rate reported from apparent satiation in a laboratory growth study for juvenile black buffalo (*Ictiobus niger*) ([Guy et al., 2018](#)). The proportion of total food intake that is soil (*P_s*) is represented at the 90th percentile for short-tailed shrew and was sourced from calculations and modeling in EPA's *Guidance for Developing Ecological Soil Screening-levels* ([U.S. EPA, 2005](#)). The proportion of total food intake that is sediment (*P_s*) for representative taxa (American mink) was calculated by dividing the *SIR* by food consumption which was derived by multiplying the *FIR* by the body weight of the mink (sourced from *Wildlife Exposure Factors Handbook* ([U.S. EPA, 1993b](#))). The *SIR* for American mink was sourced from calculations in EPA's *Second Five Year Review Report Hudson River PCBs Superfund Site to Appendix 11: Human*

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Health and Ecological Risks (U.S. EPA, 2017b). For the purposes of the current screening-level trophic transfer analysis using the blacktail redhorse, EPA used a geometric mean of 0.32 for P_s as the proportion of total food intake that is sediment (kg sediment/kg food) based on several studies (Sule, 1985; Dauble, 1986; Doyle, 2011). The proportion of total food intake that is sediment (P_s) is 5.35×10^{-4} and was calculated with SIR (1.2×10^{-4} kg of sediment per day) sourced from calculation within EPA's *Second Five Year Review Report Hudson River PCBs Superfund Site to Appendix 11: Human Health and Ecological Risks* (U.S. EPA, 2017b). As a conservative assumption, in this screening-level assessment the American mink's diet is comprised entirely of fish as a prey item while the fish diet is comprised entirely of chironomids as a prey item. Similarly, the short-tailed shrew was assumed to have a diet comprised entirely of earthworms.

Table 10-4. Dietary Exposure Estimates Using EPA's Wildlife Risk Model for Eco-SSLs for Screening-Level Trophic Transfer of TBBPA (Air Deposition to Soil) to Short-Tailed Shrew

Condition of Use (Life Cycle Stage/Category/Subcategory)	Earthworm TBBPA Concentration (mg/kg) ^a	Shrew TBBPA Dietary Exposure Rate (mg/kg bw/day) ^b	TBBPA Terrestrial Invertebrate Hazard Threshold (mg/kg) ^c
Combined fugitive and stack release from multiple COUs; see Table 8-4	0.003014	0.0017	0.91
^a Estimated TBBPA concentration in representative soil invertebrate, earthworm, assumed equal to soil. ^b Dietary exposure (Equation 10-2) to TBBPA includes consumption of biota (earthworm), incidental ingestion of soil, and ingestion of water. ^c U.S. EPA (2026c)			

Table 10-5. Dietary Exposure Estimates Using EPA's Wildlife Risk Model for Eco-SSLs for Screening-Level Trophic Transfer of TBBPA (Releases to Surface Water) to Fish

COU (Life Cycle Stage/Category/Subcategory)	7Q10 Flow (m ³ /day)	TBBPA in Sediment Ingestion Rate (mg/kg bw/day) ^a	TBBPA in Chironomids Ingestion Rate (mg/kg bw/day) ^b	Fish TBBPA Dietary Exposure Rate (mg/kg bw/day) ^c
Processing – Incorporating into formulation, mixture, or reaction product – Flame retardant in plastic material and resin manufacturing; plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing	1,230	0.1	0.401	1.22
^a Calculated from Equation 10-3 with factors representing: concentration of TBBPA in sediment, proportion of food intake that is sediment, food intake rate, and absorbed fraction of TBBPA from sediment. ^b Calculated from Equation 10-3 with factors representing: concentration of TBBPA in prey, proportion of prey in diet, feed intake rate, and absorbed fraction of TBBPA from prey. ^c Dietary exposure (Equation 10-3) to TBBPA includes consumption of biota (chironomids) and ingestion of sediment during feeding.				

Table 10-6. Dietary Exposure Estimates Using EPA's Wildlife Risk Model for Eco-SSLs for Screening-Level Trophic Transfer of TBBPA (Releases to Surface Water) to Mink-Eating Fish

COU (Life Cycle Stage/ Category/Subcategory)	7Q10 Flow (m ³ /day)	TBBPA in Sediment Ingestion Rate (mg/kg bw/day) ^a	TBBPA in Water Intake Rate (mg/kg bw/day) ^b	TBBPA in Fish Ingestion Rate (mg/kg bw/day) ^c	Mink TBBPA Dietary Exposure Rate (mg/kg bw/day) ^d	TBBPA TRV for Mammals (mg/kg bw/day) ^e
Processing – Incorporating into formulation, mixture, or reaction product – Flame retardant in plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing	1,230	0.002	0.0005	1.22	0.27	250
^a Calculated from Equation 10-2 with factors representing: concentration of TBBPA in sediment, proportion of food intake that is sediment, food intake rate, and absorbed fraction of TBBPA from sediment. ^b Calculated from Equation 10-2 with factors representing: water intake rate, concentration of TBBPA in surface water, and absorbed fraction of TBBPA from water. ^c Calculated from Equation 10-2 with factors representing: concentration of TBBPA in prey, proportion of prey in diet, feed intake rate, concentration of TBBPA in surface water, and absorbed fraction of TBBPA from prey. ^d Dietary exposure from Equation 10-2 to TBBPA includes consumption of biota (fish), incidental ingestion of sediment, and ingestion of water. ^e Toxicity Reference Value (TRV) for mammals calculated using empirical toxicity data for rats as detailed within the <i>Draft Human Health and Environmental Hazard Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)</i> (U.S. EPA, 2026c).						

10.6 Weight of Scientific Evidence Conclusions

EPA uses several considerations when weighing scientific evidence to determine confidence in the dietary exposure estimates. These considerations include the quality of the database, consistency, strength and precision, and relevance (see Appendix A in the *Draft Human Health and Environmental Hazard Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* (U.S. EPA, 2026c)). This approach is in agreement with the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances* (U.S. EPA, 2021). For trophic transfer, EPA considers the evidence for worm-eating terrestrial mammals moderate and the evidence for fish-consuming, aquatic-dependent mammals moderate.

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

Quality of the Database, and Strength and Precision

Measured concentrations within aquatic species and terrestrial species were represented with empirical biomonitoring data. Empirical biomonitoring data for aquatic organisms were reasonably available with biota concentrations represented within a variety of aquatic taxa across multiple trophic levels. Biomonitoring data were available for aquatic species with TBBPA concentrations including invertebrate species (mollusk, worm, insect, echinoderm, and crustacean), fish, and mammal species.

The confidence in quality of the database for the chronic mammalian assessment using aquatic-dependent terrestrial species consuming fishes that prey on the sediment invertebrate chironomid is robust. The use of species-specific exposure factors (*i.e.*, feed intake rate, water intake rate, the proportion of soil or sediment within the diet) from reliable resources assisted in obtaining dietary exposure estimates ([U.S. EPA, 1993b](#), [2017b](#)), thereby resulting in a robust confidence for strength and precision of the dietary exposure estimates in terrestrial trophic transfer. Exposure factors for the fish species were obtained to represent potential sediment uptake from feeding activity and included: diet composition ([Dauble, 1986](#); [Boschung, 2004](#)), feed intake rate ([Guy et al., 2018](#)), and the proportion of sediment in diet ([Sule, 1985](#); [Dauble, 1986](#); [Doyle, 2011](#)). The confidence in quality of the database for the chronic mammalian assessment using a worm-eating mammal consuming earthworms as a prey item is robust. Confidence in strength and precision is robust.

Consistency

The confidence in consistency for the chronic mammalian assessment using a worm-eating mammal consuming earthworms as a prey item is moderate. The modeled soil concentration was represented by air deposition to soil, which includes the applicable probable parameters yet included a combination from multiple TRI releases. As such, EPA has moderate confidence that the subsequent screening-levels based on soil concentration and exposure estimates are protective for terrestrial organisms. For aquatic ecosystems, lipid-normalized concentrations of TBBPA were consistent across species or trophic levels which supports the conclusion that TBBPA does not biomagnify through dietary exposure. The confidence in consistency for the chronic mammalian assessment using aquatic-dependent terrestrial species consuming fishes that prey on the sediment invertebrate chironomid is robust.

Relevance (Biological and Environmental)

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 for dietary exposure estimates ([U.S. EPA, 1993b](#)). The confidence in biological relevance for the chronic mammalian assessment using a worm-eating mammal consuming earthworms as a prey item is moderate. Selection of a benthic oriented fish species increases confidence with considerations made for sediment ingestion due to feeding behavior and further increases confidence in representing exposure pathways from sediment to aquatic species. The application of conservative assumptions at each trophic level ensures a conservative approach to determining potential risk, whereby none of the calculated values exceeded the hazard threshold derived for mammals. Conservative assumptions associated with a lack of metabolic transformation within prey items such as chironomids, earthworms and fish increase the confidence in biological relevance that the resulting screening-level assessment is protective. The overall confidence is robust for biological relevance for the screening-level chronic mammalian assessment using an aquatic-dependent terrestrial species.

The screening-level trophic transfer analysis investigated dietary exposure resulting from TBBPA in biota and environmentally relevant media such as soil, sediment, and water. The analysis used equation terms (*e.g.*, area use factor and the proportion of TBBPA absorbed from diet, and soil or sediment) all set to the most conservative values, emphasizing a conservative approach to estimating exposure of TBBPA. Assumptions within the trophic transfer equations (Equation 10-2 and Equation 10-3) represent conservative screening values and those assumptions were applied similarly for each trophic level and representative species. The AUF, defined as the home range size relative to the contaminated area (*i.e.*, $AUF = \text{site} \div \text{home range}$) was designated as 1 for all organisms, which assumes a potentially longer residence within an exposed area or a large exposure area. These conservative approaches likely

2035 overrepresent TBBPA ability to transfer among the trophic levels, however, this increases confidence
2036 that risks are not underestimated. As a result, there is an overall robust confidence for environmental
2037 relevance of the dietary exposure estimates. The confidence in relevance for the chronic mammalian
2038 assessment using a worm-eating mammal consuming earthworms as a prey item is robust. The
2039 confidence in relevance for the chronic mammalian assessment using an aquatic-dependent terrestrial
2040 species consuming fishes that prey on the sediment invertebrate chironomid is robust.

11 CONCLUSION OF ENVIRONMENTAL EXPOSURE AND GENERAL POPULATION SCREENING-LEVEL ANALYSIS

11.1 General Population Exposure Conclusions

The general population can be exposed to TBBPA from various exposure pathways. General population exposures via drinking water ingestion, fish ingestion, ambient air inhalation, ingestion of soil from air deposition were quantified whereas exposures via the land pathway (*i.e.*, landfills, biosolids) were qualitatively assessed. Based on the high-end estimates of environmental media concentrations summarized in Table 11-1, general population exposures were estimated for the lifestage that would be most exposed based on intake rate and body weight. Table 11-2 summarizes the exposure pathways and lifestages that were assessed quantitatively or qualitatively.

Table 11-1. Summary of High-End TBBPA Concentrations in Various Environmental Media from Environmental Releases

COU	Release Media	Environmental Media	TBBPA Concentration
Processing as a reactant in plastic material and resin manufacturing	Water	Surface water (7Q10, 30Q5, and harmonic mean if applicable)	350 release days 7Q10: 4.47E-04 µg/L 30Q5: 3.59E-04 µg/L HM: 1.56E-04 µg/L
Processing: additive flame retardant in plastic, resin, and paints and coating manufacturing			4 release days 7Q10: 3.91E-02 µg/L 30Q5: 3.14E-02 µg/L
Disposal			7Q10: 12.8 µg/L 30Q5: 9.39 µg/L
- Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – As a reactant – Intermediate in all other chemical product and preparation manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Adhesive manufacturing - Processing – Incorporation into article – Reactive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (<i>e.g.</i>, printed circuit boards and semiconductor packages) - Processing – Incorporation into article – Reactive flame retardant in transportation equipment manufacturing (<i>e.g.</i>, interior material for automotive and aviation articles) - Processing – Incorporation into article – Additive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (<i>e.g.</i>, plastic electronic enclosures)	Ambient air	Fugitive and stack, 100 m	Daily-averaged total: 300 µg/m ³ Annual-averaged total: 4.68 µg/m ³

COU	Release Media	Environmental Media	TBBPA Concentration
- Industrial and commercial use – Construction, paint, electrical, and metal products – Reactive flame retardant in automotive and aviation articles (<i>i.e.</i>, laminate and prepreg material) - Disposal	Ambient air	Fugitive and stack, 100 m	Daily-averaged total: 300 µg/m ³
Electrical and electronics recycling			Annual-averaged total: 4.68 µg/m ³
			Daily-averaged total: 27.63 µg/m ³
			Annual-averaged total: 26.50 µg/m ³

Table 11-2. Pathways Assessed for High-End Exposure Scenarios for Highest Exposed Populations

COU	Exposure Pathway	Exposure Route	Exposure Scenario	Lifestage
All	Biosolids (3.2)	All scenarios were assessed qualitatively		
All	Landfills (3.3)	All scenarios were assessed qualitatively		
All	Surface water	Oral	Qualitatively evaluated exposure to TBBPA in surface water during swimming (Section 6)	
		Dermal	Qualitatively evaluated incidental ingestion of TBBPA in surface water during swimming (Section 6)	
- Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; paint and coating manufacturing - Disposal	Drinking water	Oral	Ingestion of drinking water (Section 5.1.1)	Adult, youth, and children
	Fish ingestion	Oral	Ingestion of fish for tribal populations (7.1)	Adult (21+ years)
- Processing – As a reactant – Flame retardant in plastic material and resin manufacturing - Processing – As a reactant – Intermediate in all other chemical product and preparation manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Flame retardant in: Plastic material and resin manufacturing; Plastics product manufacturing; Custom compounding of purchased resins; Paint and coating manufacturing - Processing – Incorporation into formulation, mixture, or reaction product – Adhesive manufacturing - Processing – Incorporation into article – Reactive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component	Ambient air	Inhalation	Inhalation of ambient air concentration for general population (Section 8.2.1)	Adult, youth, and children
		Oral	Soil ingestion from air deposition for general population (Section 8.2.2)	Infant and children (6 months to 12 years)

COU	Exposure Pathway	Exposure Route	Exposure Scenario	Lifestage
manufacturing; Plastic product manufacturing (e.g., printed circuit boards and semiconductor packages) • Processing – Incorporation into article – Reactive flame retardant in transportation equipment manufacturing (e.g., interior material for automotive and aviation articles) • Processing – Incorporation into article – Additive flame retardant in: Computer and electronic product manufacturing; Electrical equipment, appliance, and component manufacturing; Plastic product manufacturing (e.g., plastic electronic enclosures) • Industrial and commercial use – Construction, paint, electrical, and metal products – Reactive flame retardant in automotive and aviation articles (i.e., laminate and prepreg material) • Disposal • Electrical and electronics recycling	Ambient air	Oral	Soil ingestion from air deposition for general population (Section 8.2.2)	Infant and children (6 months to 12 years)

11.2 Weight of Scientific Evidence Conclusions for General Population Exposure

The weight of scientific evidence supporting the exposure estimate is decided based on the strengths, limitations, and uncertainties associated with the exposure estimates, which are discussed in detail for biosolids (Section 3.2.1), landfills (Section 3.3.5), surface water (Section 4.3.1), drinking water (Section 5.4), fish ingestion (Section 7.2.1), ambient air (Section 8), and human milk (Section 9.3). EPA summarized its weight of scientific evidence using confidence descriptors: robust, moderate, slight, or indeterminate. 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 determined moderate to robust confidence in its qualitative assessment and conclusions pertaining to exposures from landfills (Section 3.3), biosolids (Section 3.2), and surface water (Section 4). For its quantitative assessment, EPA modeled exposure due to various exposure scenarios resulting from different pathways of exposure. Exposure estimates used high-end inputs for the purpose of risk screening. When available, monitoring data were compared to modeled estimates to evaluate overlap, magnitude, and trends. For its quantitative exposure assessment of drinking water (Section 6), fish ingestion (Section 7), ambient air (Sections 8), and human milk (Section 9), EPA has robust confidence that the screening-level analysis was appropriately conservative to ensure the upper-bound exposure estimates were captured. EPA is confident that the upper-bound estimates of exposure from the screening-level analysis provide the necessary information to evaluate risks to the general population from releases into the environment. Refinements were conducted if this screening assessment indicated risks in any of the scenarios, which are further discussed in the *Draft Risk Evaluation for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* (U.S. EPA, 2026f).

11.3 Environmental Biomonitoring and Trophic Transfer Conclusions

Dietary exposure estimates were calculated based on water and releases to soil (via air deposition) from the COU/OES with the highest modeled environmental releases as reported within the Section 3. The Disposal OES resulted in the highest environmental releases to surface water and sediment and air

deposition to soil was used to estimate an applicable soil value. Although terrestrial hazard data for TBBPA were not available for mammalian wildlife species, studies in laboratory animals were used to derive hazard values for mammalian species. Specifically, empirical toxicity data from human health rodent models were used to estimate a TRV for terrestrial mammals at 250 of mg/kg-bw/day ([U.S. EPA, 2026c](#)) based on the *Guidance for Developing Ecological Soil Screening-levels (Eco-SSLs)*. The weight of evidence indicates TBBPA is not bioaccumulative under typical environmental conditions, a conclusion consistent with the rapid metabolism and elimination observed in test organisms and based on the physical and chemistry and environmental fate properties.

Results for calculated dietary exposures of TBBPA to mammals from modeled concentrations within relevant pathways such as water, sediment, and soil indicated exposure concentrations below the TRV. The conclusion of screening-level trophic transfer analyses for aquatic-dependent mammals with exposure pathways for surface water and sediment are presented within Table 10-6. Maximum TBBPA concentrations from modeled and monitored data presented no exposure of TBBPA greater than the calculated TRV from the screening-level trophic transfer analysis. Similarly, the screening-level trophic transfer analysis for terrestrial mammals based on the highest modeled and monitored releases of TBBPA from air deposition to soil also resulted in dietary exposure concentrations below the TRV (Table 10-6). Exposure pathways with aquatic-dependant mammals and terrestrial mammals as receptors were not examined further since, even with conservative assumptions, dietary TBBPA exposure concentrations from this analysis are not equal to or greater than the TRV. These results align with previous studies indicating that TBBPA has limited biomagnification potential as summarized within the *Draft Physical Chemistry and Fate Assessment for 4,4'-(1-Methylethylidene)bis[2,6-dibromophenol] (TBBPA)* ([U.S. EPA, 2026d](#)). The screening-level trophic transfer assessments for terrestrial invertebrates, mammals, and aquatic dependent mammals has moderate confidence that TBBPA exposures from the highest release exposure scenarios are well below the hazard thresholds that would present risk.

11.4 Weight of Scientific Evidence Conclusions for Environmental Biomonitoring and Trophic Transfer Conclusion

EPA weighed quality of the database, consistency, strength and precision, and relevance to determine confidence in dietary exposure estimates, consistent with the Draft Systematic Review Protocol. For trophic transfer, the confidence is moderate for worm-eating terrestrial mammals and for fish-consuming aquatic-dependent mammals.

Empirical biomonitoring across aquatic taxa and trophic levels (invertebrates, fish, mammals) supported robust confidence in the quality of the database by providing species-specific exposure factors (*i.e.*, feed intake rate, water intake rate, the proportion of soil or sediment within the diet) from reliable resources assisted in obtaining dietary exposure estimates ([U.S. EPA, 1993b, 2017b](#)), thereby resulting in a robust confidence for strength and precision of the dietary exposure estimates in terrestrial trophic transfer. Consistency is moderate for the worm-eating mammal scenario because soil concentrations were modeled from air deposition and combined TRI releases, and EPA has moderate confidence the screening-levels and exposure estimates are protective. The Agency has robust confidence in the relevance because the short-tailed shrew and American mink were appropriate representative mammals, and using consistent exposure factors addressed species-specific difference. The screening-level analysis applied equation terms set to the most appropriate values, likely overrepresenting trophic transfer yet ensuring protectiveness. Overall, confidence is robust for the environmental and biological relevance of the dietary exposure estimates, including the screening-level chronic mammalian assessments for both the worm-eating and aquatic-dependent scenarios.

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APPENDICES

Appendix A EXPOSURE FACTORS

Table_Apx A-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
Adult (16+ years)	80.0
^a Age group-weighted average	
^b See Table 8-1 of U.S. EPA (2011b) .	

Table_Apx A-2. Fish Ingestion Rate 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+ years) ^c	0.063	0.277
Subsistence fisher (adult) ^d	1.78	

^a Age group-weighted average, using body weight from Table_Apx A-1.

^b See Table 20a of [U.S. EPA \(2014\)](#).

^c See Table 9a of [U.S. EPA \(2014\)](#).

^d [U.S. EPA \(2000b\)](#)

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Table Apx A-3. Recommended Default Values for Common Exposure Factors

Symbol	Definition	Recommended Default Value	Recommended Default Value	Source
		Occupational	Residential	
ED	Exposure duration (h/day)	8	24	
EF	Exposure frequency (days/year)	250	365	
EY	Exposure years (years)	40	Varies for adults chronic, non-cancer 78 (lifetime) 1 Infant (birth to <1 year) 5 Toddler (1–5 years) 5 Child (6–10 years) 5 Youth (11–15 years) 5 Youth (16–20 years)	Number of years in age group Note: These age bins may vary for different measurements and sources
AT	Averaging time non-cancer	Equal to total exposure duration or 365 days/year × EY; whichever is greater	Equal to total exposure duration or 365 days/year × EY; whichever is greater	See pg. 6 to 23 of Risk assessment guidance for superfund, volume I: Human health evaluation manual (Part A). (U.S. EPA, 1989)
	Averaging time cancer	78 years (28,470 days)	78 years (28,470 days)	See Table 18-1 of the <i>Exposure Factors Handbook</i> (U.S. EPA, 2011b)
BW	Body weight (kg)	80	80 Adult 7.83 Infant (birth to <1 year) 16.2 Toddler (1–5 years) 31.8 Child (6–10 years) 56.8 Youth (11–15 years) 71.6 Youth (16–20 years) 65.9 Adolescent woman of childbearing age (16 to <21) to apply to all developmental exposure scenarios	See Table 8-1 of the <i>Exposure Factors Handbook</i> (U.S. EPA, 2011b) (refer to Figure 31 for age-specific BW) Note: Age bins may vary for different measurements and sources See Table 8-5 of the <i>Exposure Factors Handbook</i> (U.S. EPA, 2011b)
IR _{dw-acute}	Drinking water ingestion rate (L/day) to acute	3.219 Adult	3.219 Adult 1.106 Infant (birth to <1 year) 0.813 Toddler (1–5 years) 1.258 Child (6–10 years) 1.761 Youth (11–15 years) 2.214 Youth (16–20 years)	See Tables 3-15 and 3-33; weighted average of 90th percentile consumer-only ingestion of drinking water (birth to <6 years) (U.S. EPA, 2011b)
IR _{dw-chronic}	Drinking water ingestion rate (L/day) to chronic	0.880 Adult	0.880 Adult 0.220 Infant (birth to <1 year) 0.195 Toddler (1–5 years) 0.294 Child (6–10 years) 0.315 Youth (11–15 years) 0.436 Youth (16–20 years)	Chapter 3 of the <i>Exposure Factors Handbook</i> (U.S. EPA, 2011b), Table 3-9 per capita mean values; weighted averages for adults (years 21 to 49 and 50+), for toddlers (years 1–2, 2–3, and 3 to <6).

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Symbol	Definition	Recommended Default Value	Recommended Default Value	Source
		Occupational	Residential	
IR _{inc}	Incidental water ingestion rate (L/hr)		0.025 Adult 0.05 Child (6 to < 16 years)	<i>Evaluation of Swimmer Exposures Using the SWIMODEL Algorithms and Assumptions</i> (U.S. EPA, 2015)
IR _{soil}	Soil ingestion rate (mg/day)	50 Indoor workers 100 Outdoor workers	100 Infant (<6 months) 200 Infant to Youth (6 months to <12 years) 100 Youth to Adult (12+ years) 1,000 Soil Pica Infant to Youth (1 to <12 years) 50,000 Geophagy (all ages)	U.S. EPA Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (1991) Chapter 5 of the <i>Exposure Factors Handbook</i> (U.S. EPA, 2011b), Table 5-1, Upper percentile daily soil and dust ingestion
SA _{water}	Skin surface area exposed (cm ²) used for incidental water dermal contact		19,500 Adult 7,600 Child (3 to < 6 years) 10,800 Child (6 to < 11 years) 15,900 Youth (11 to < 16 years)	Chapter 7 of the <i>Exposure Factors Handbook</i> (U.S. EPA, 2011b), Table 7-1, Recommended Mean Values for Total Body Surface Area, for Children (sexes combined) and Adults by Sex
K _p	Permeability constant (cm/h) used for incidental water dermal contact	Not applicable for TBBPA because of the availability of a PBPK to estimate internal dose.		
SA _{soil}	Skin surface area exposed (cm ²) used for soil dermal contact	3,300 Adult	5,800 Adult 2,700 Child	EPA Risk Assessment Guidance for Superfund RAGS Part E for Dermal Exposure (U.S. EPA, 2004)
AF _{soil}	Adherence factor (mg/cm ²) used for soil dermal contact	0.2 Adult	0.07 Adult 0.2 Child	EPA Risk Assessment Guidance for Superfund RAGS Part E for Dermal Exposure (U.S. EPA, 2004)

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2710 **Table Apx A-4. Mean and Upper Milk Ingestion Rates by Age**

Age Group	Breast Milk Lipids Ingestion (mL/kg day) ^a		Breast Milk Lipids Ingestion (g/kg day) ^b		Breast Milk Ingestion (mL/kg day) ^a	
	Mean	Upper	Mean	Upper	Mean	Upper
Birth to <1 month	6.0	8.7	6.2	9.0	150	220
1 to <3 month	5.5	8.0	5.7	8.2	140	190
3 to <6 month	4.2	6.1	4.3	6.3	110	150
6 to <12 month	3.3	5.2	3.4	5.4	83	130
Birth to <1 year	4.1	6.2	4.2	6.4	104.8	152.5

^a EPA's *Exposure Factors Handbook* ([U.S. EPA, 2011b](#)), Table 15-1^b Converted using 1.03 g/mL as density of human milk.

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Appendix B ESTIMATING HYDROLOGICAL FLOW DATA FOR SURFACE WATER MODELING

For facilities reporting releases to TRI, relevant flow data from the associated receiving water body were collected. Databases that were queried to estimate a flow rate include EPA's ECHO that contains facilities with a NPDES permit, National Hydrography Dataset Plus (NHDPlus), and NHDPlus V2.1 Flowline Network Enhanced Runoff Method (EROM) Flow. Flows for facilities without TRI or DMR data were generated by collecting flow rates for facilities across NAICS code associated with TBBPA-releasing facilities. The same databases within ECHO were queried. The EROM database within ECHO provides modeled monthly average flows for each month of the year. While the EROM flow database represents averages across a 30-year time period, the lowest of the monthly average flows was selected as a substitute for the 30Q5 flow used in modeling, as both approximate the lowest observed monthly flow at a given location. The substitute 30Q5 flow was then plugged into the regression equation used by EPA's Exposure and Fate Assessment Screening Tool (E-FAST) ([U.S. EPA, 2007](#)) to convert between these flow metrics and solve for the 7Q10 using Equation_Apx B-1. In previous assessments, the EPA has selected the 7Q10 flow as a representative low flow scenario for biological impacts due to effluent in streams, while the harmonic mean represents a more average flow for assessing chronic drinking water exposure.

Equation_Apx B-1. Calculating the 7Q10 Flow

$$7Q10 = \frac{\left(0.409 \frac{cfs}{MLD} \times \frac{30Q5}{1.782}\right)^{1.0352}}{0.409 \frac{cfs}{MLD}}$$

Where:

7Q10 = Modeled 7Q10 flow (in million liters per day [MLD])
 30Q5 = Lowest monthly average flow from NHD (in MLD)

Furthermore, the harmonic mean flow was calculated using Equation_Apx B-2, derived from the relevant E-FAST regression ([U.S. EPA, 2007](#)).

Equation_Apx B-2. Calculating the Harmonic Mean Flow

$$HM = 1.194 \times \frac{\left(0.409 \frac{cfs}{MLD} \times AM\right)^{0.473} \times \left(0.409 \frac{cfs}{MLD} \times 7Q10\right)^{0.552}}{0.409 \frac{cfs}{MLD}}$$

Where:

HM = Modeled harmonic mean flow (in MLD)
 AM = Annual average flow from NHD (in MLD)
 7Q10 = Modeled 7Q10 flow from Equation_Apx B-1 (in MLD)

Quantified release estimates to surface water were evaluated with PSC modeling using the receiving water body flows. For each facility with surface water releases, the highest estimated release of TBBPA to surface water was used to estimate the corresponding TBBPA concentrations in the receiving water body. The total days of release associated with the highest COU release was applied as continuous days of release per year (e.g., a scenario with 250 days of release per year was modeled as 250 consecutive

2754 days of release, followed by 115 days of no release, per year). Raw daily concentration estimates from
2755 PSC were manually evaluated for the highest resulting concentrations in an averaging window equal to
2756 the total days of release (*e.g.*, a scenario with 250 days of release was evaluated for the highest 250-day
2757 average concentration).
2758
2759 The relevant NAICS codes for facilities associated with TBBPA releases are 541380, 541713, 541714,
2760 541715, and 621511. These correspond to the Use of laboratory chemicals COU.

Appendix C SURFACE WATER MONITORING LITERATURE

Table_Apx C-1. Measured TBBPA Concentrations in Water from Non-U.S. Studies

Reference	Country	Sampled Media and Region	TBBPA Concentration (µg/L)
Yang et al. (2012)	China	Freshwater lake	Max: 4.87
Xu et al. (2013)	China	Freshwater lake in a heavily developed and industrialized area	DF: 25% (n = 12) Range: 0.36 to 1.12 Mean: 0.64
He et al. (2013)	China	River flowing through highly industrialized area	DF: NR (n = 5) Range: 1,110 to 2,830 Mean: 1,750
Xiong et al. (2015)	China	Reservoir and river in electronic waste region	DF: NR (n = 14) Range: ND (NR) to 0.92 Mean: 0.23
Liu et al. (2016a)	China	Freshwater lake in a heavily developed and industrialized area	DF: 0.07 (n = 26) Range: ND (NR) to 0.012 Median: 0.04
Xiong et al. (2016)	China	River flowing through industrialized areas and downstream of 2 e-waste dismantling regions	DF: NR (n = 15) Range: 2.00E-05 to 2.70E-04 Mean: 1.97-04
Si et al. (2019)	China	Freshwater lake and rivers in industrial and agricultural regions	DF: 16% (n = 38) Range: 3.00E-04 to 6.10E-03 Median: 2.20E-03 Mean: 2.55E-03
Gong et al. (2017)	China	Seawater	DF: NR (n = 6) Range: ND (NR) to 1.80
Yin et al. (2011)	China	River	DF: NR (n = 3) Range: 2.39E-02 to 0.22
Harrad et al. (2009)	England	Freshwater lakes	DF: NR (n = 27) Mean: 1.40E-04 to 3.2E-03
Labadie et al. (2010)	France	River	DF: NR (n = 5) Range: 0.05 to 0.06
Kim et al. (2016)	Korea	Effluent combined from POTW treatment industrial and domestic waste	DF: NR (n = NR) Range: 1.09E-04 to 6.67E-03 Median: 2.50E-04 to 1.22E-03 Mean: 2.33E-04 to 1.85E-03
Choo et al. (2019)	Korea	Seawater	DF: 94% (n = 16) Range: ND (NR) to 2.79-03 Median: 3.56-04 Mean: 8.19-04
Peters et al. (2008)	Netherlands	Precipitation	DF: 36% (n = 50) Max: 2.6 75th, 90th percentile: 0.8, 1.1
Norwegian Environment Agency (2019a)	Norway	Stormwater Effluent	DF: NR (n = NR) Min. stormwater: <0.12 Min. effluent: <0.05
Chokwe et al. (2012)	South Africa	Influent	DF: NR (n = NR)

Reference	Country	Sampled Media and Region	TBBPA Concentration (µg/L)
		Effluent	Min. influent: 6.63 Min. effluent: 3.27
Gustavsson et al. (2018)	Sweden	Rivers	DF: 44% (n = 25) Range: <1.30E-03 to 6.20E-02 Median: 2.70E-02 Mean: 2.90E-02
DF = detection frequency; Min. = minimum; ND = non-detect; NR = not reported			

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Appendix D SEDIMENT MONITORING LITERATURE

Table_Apx D-1. Measured TBBPA Concentrations in Sediment from Non-U.S. and Canadian Studies (Dry Weight, ng/g)

Reference(s)	Region/ Country	Number of Studies	Range of TBBPA Concentration (ng/g)
Yang et al. (2012) ; Xu et al. (2013) ; He et al. (2013) ; Feng et al. (2012) ; Li et al. (2016) ; Liu et al. (2016a) ; Luo et al. (2010) ; Li et al. (2019b) ; SUNY (2019) ; Li et al. (2019a) ; Wang et al. (2016) ; Yang et al. (2014) ; Zhang et al. (2009) ; Xiong et al. (2015) ; Liu et al. (2014)	China	16	0.012 to 4.12E04
Sellström and Jansson (1995) ; Netherlands Institute for Fisheries Research (2002) ; Morris et al. (2004) ; Verslycke et al. (2005) ; Harrad et al. (2009) ; Evenset et al. (2009) ; Guerra et al. (2010) ; Labadie et al. (2010) ; Hloušková et al. (2014) ; Sühling et al. (2015) ; Luigi et al. (2015) ; Kotthoff et al. (2017) ; Ganci et al. (2019) ; Norwegian Environment Agency (2019a)	Europe	14	<0.10 to 9,750
Lee et al. (2015) ; Gu et al. (2017) ; Choo et al. (2019)	South Korea	3	0.023 to 150
Watanabe et al. (1983a) ; Suzuki and Hasegawa (2006) ; Watanabe et al. (1983b)	Japan	3	<0.2 to 140
Zhang et al. (2015)	Singapore	1	0.056 to 0.139
Chokwe et al. (2017)	South Africa	1	<0.513 to 20.99

Appendix E SUMMARY OF HUMAN MILK BIOMONITORING STUDIES

Table_Apx E-1. Measured TBBPA Concentrations in Human Milk

Location	Population	Sampling Year(s)	N (# of ND) ^a	TBBPA Concentration (ng/g Lipid Weight)		Comments	Reference
				Range ^b	Central Tendency ^c		
North America							
Boston, MA	43 primiparous women 18+ years of age	2004–2005	43 (28)	<0.03 to 0.55	NR	1 sample collected from each participant 2–8 weeks postpartum	Carignan et al. (2012)
Europe							
France	93 women; mean age of 32.5 years	2004–2006	77 (43)	<LOD to 37.34 (LOD not reported)	4.11 mean; 0.48 median	Samples collected 3 and 4 days after childbirth LOD not reported	Cariou et al. (2008)
	Demographics not reported for donors	2010	106 (62)	<0.5 to 15.1	3.5 and 3.2 mean upper and lower bound, respectively	Samples collected 2 weeks postpartum LOQ 2.5	Inthavong et al. (2017)
	26 women aged 23–46 years; mean age 34	2005	23 (NR)	0.034 to 9.4 LOD, LOQ, MDL not reported.	0.17 median		Antignac et al. (2008)
Czech Republic	50 women aged 20–43 years; median 30	2010	50 (35)	<2 to 688 <60 to 16.2 ng/mL ww	NR		Lankova et al. (2013)
	231 women aged 18–42 years; median 30	2019–2021	231 (231)	<0.1 to <0.1	<0.1	Samples collected 3–5 days after delivery Samples collected from 2 regions to 1 with a history of coal mining and heavy industry and another without major industrial burden	Parizek et al. (2023)
Ireland	109 primiparous	2010	11 (9)	<0.29 to 0.17	0.33 and 0.05 for		Pratt et al.

Location	Population	Sampling Year(s)	N (# of ND) ^a	TBBPA Concentration (ng/g Lipid Weight)		Comments	Reference
				Range ^b	Central Tendency ^c		
	women; mean age 32.7 years, BMI 23.3		(pooled samples)	(upper bound)	mean upper and lower bound		(2013)
United Kingdom	10 primiparous women; a mean age 26.3 years	2010 to 2011	120 (27)	<0.05 to 0.201	0.06 mean; 0.04 median; 5th and 95th percentile also reported (0.03 and 0.17)	12 milk samples collected monthly from each mother starting first month postpartum. LOD not reported LOQ 0.05 ng/g	Harrad and Abdallah (2015)
	34 women	NR	34 (12)	<0.04 to 0.65	0.06 mean; <0.04 median		Abdallah and Harrad (2011)
Spain	53 women aged 25 to 43 years old	NR	53 (50)	Not reported	18.7 mean lw 580 mean ng/mL ww		
Asia							
China	1,237 primiparous women from 12 provinces	2007	Urban: 12 (2) Rural: 12 (3) (pooled samples)	Urban: <LOD to 5.12 Rural: <LOD to 4.46 LOD not reported	Urban: 0.93 mean Rural: 0.96 mean	Samples 3 to 8 weeks after childbirth. Personal information such as occupation, diet and smoking habits was recorded; no occup. exposure reported LOD not reported	Shi et al. (2009)
China (urban and suburban)	103 women; mean age 30.1 years, BMI 22.7	2011	103 (48)	<0.06 to 12.46	0.41 mean; 0.10 median 5th, 25th, 75th and 95th percentiles were ND, ND, 0.38, and 1.25, respectively.	Samples collected 3 to 40 weeks after childbirth. Personal information such as dietary habit and socioeconomic and lifestyle factors was recorded. No occupational exposure. LOD 0.06 ng/g (unadjusted for lw).	Shi et al. (2013a)

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Location	Population	Sampling Year(s)	N (# of ND) ^a	TBBPA Concentration (ng/g Lipid Weight)		Comments	Reference
				Range ^b	Central Tendency ^c		
China	NR	2011	12 (8)	<0.06–1.85	0.28 mean		Shi et al. (2013b)
	Exact number not reported; 50–60 donors from rural provinces and 50 donors from urban provinces.	2011	29 (0) (pooled samples)	0.140–66.8 LOD, LOQ, MDLs not reported.	7.58 mean 1.21 median	Samples collected 3 to 8 weeks after delivery. Women from 16 provinces that cover 48% of China's land area and 56% of the Chinese population. No occupational exposure but possibility of residential exposure from local industrial sites.	Shi et al. (2017)
Japan	9 women aged 26–39 years; mean 31.4)	2005–2006	9 (0)	0.39–2.22 LOQ 0.2	1.04 mean; 0.72 median		Fujii et al. (2014)
	19 women aged 19–39 years	2012–2013	19 (1)	<0.018 to 8.7	1.9 mean	Three samples were collected 1 week, 1 month and 3 months after childbirth.	Nakao et al. (2015)
	64 women; mean 31.1	2008–2010	64	<0.034 to 2,209	48 mean; 3.2 median 3.0 geometric mean	Samples collected 4–8 weeks after childbirth. Personal information such as number of births, occupation (<i>i.e.</i> , housewife or office worker), smoking status, and alcohol usage was collected.	Fujii et al. (2018)
	12 women aged 23–37 years	2011–2013	12 (0)	0.00674 to 0.344 ng/mL ww LOQ 3.6E–04 ng/mL	0.064 mean	Samples collected 1 month after delivery	Akiyama et al. (2015)

LOD = limit of detection; LOQ = limit of quantification; lw = lipid weight; MDL = method detection limit; ND = non-detect; NR = not reported

^a N refers to the number of samples, unless otherwise noted. The number of non-detect values is reported in parenthesis. Values reported as “<X” are assumed to be non-detect.

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Location	Population	Sampling Year(s)	N (# of ND) ^a	TBBPA Concentration (ng/g Lipid Weight)		Comments	Reference
				Range ^b	Central Tendency ^c		
^b The range is the minimum and maximum values reported. Non-detect values are shown as less than the detection limit. ^c The central tendency values shown are as reported in the reference(s).							

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