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Draft Supplement to the Risk Evaluation for 1,4-Dioxane

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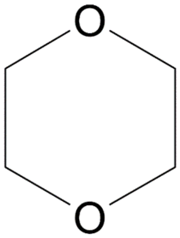

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Docket

Supporting information can be found in the public docket (Docket ID: EPA-HQ-OPPT-2022-0905).

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Internal Reviewers

This assessment was provided for review to scientists in EPA's Program and Region Offices. Comments were submitted by:

- Office of the Administrator/Office of Children's Health Protection
- Office of Air and Radiation
- Office of Chemical Safety and Pollution Prevention/Office of Pesticide Programs
- Office of General Council
- Office of Land and Emergency Management
- Office of Research and Development
- Office of Water

External Reviewers

This assessment was provided for review to other federal agencies and Executive Offices of the President, including:

- Consumer Product Safety Commission
- Department of Defense
- Department of Health and Human Services/National Institute of Environmental Health Sciences
- Department of Health and Human Services/National Institute for Occupational Safety and Health
- Department of Justice
- Department of Labor/Occupational Safety and Health Administration
- Executive Office of the President/Office of Management and Budget
- Food and Drug Administration
- National Aeronautics and Space Administration
- Small Business Administration Office of Advocacy

Summary of Risk Findings and Support for Risk Determination

1,4-Dioxane is a solvent used in a variety of commercial and industrial applications in the United States. It is also produced as a byproduct in several manufacturing processes and may remain present as a byproduct in consumer and commercial products, including soaps, detergents, and cleaning products. Health effects of concern for 1,4-dioxane include cancer and effects in liver and olfactory tissue. People may be exposed to 1,4-dioxane through occupational exposure, consumer products, or contact with water, land, or air where 1,4-dioxane has been released to the environment from industrial and commercial sources or from consumer and commercial products washed down the drain or disposed of in landfills.

The risk evaluation for 1,4-dioxane published in 2020 evaluated risks from a range of occupational and consumer uses of 1,4-dioxane, risks to aquatic species, and risks to the general population resulting from incidental recreational contact with water. It did not evaluate general population exposures to 1,4-dioxane in drinking water or air and did not evaluate the full range of exposure that may result from 1,4-dioxane produced as a byproduct.

This draft supplement completes the Toxics Substances Control Act (TSCA) risk evaluation for 1,4-dioxane by (1) more comprehensively evaluating risks from 1,4-dioxane present as a byproduct; and (2) evaluating risks from general population exposures to 1,4-dioxane released to water, air, and land. This analysis identified cancer risk estimates higher than 1 in 10,000 (1×10^{-4}) for a range of typical and high-end occupational exposures to 1,4-dioxane produced as a byproduct. It also identified cancer risk estimates higher than 1 in 1 million (1×10^{-6}) for a range of general population exposure scenarios associated with 1,4-dioxane in drinking water sourced downstream of release sites and in air within 1 km of releasing facilities. Although these risk estimates include inherent uncertainties and the overall confidence in specific risk estimates varies, the analysis provides support for the Agency to make a determination about whether 1,4-dioxane poses an unreasonable risk and to identify drivers of unreasonable risk among exposures for people (1) with occupational exposure to 1,4-dioxane under some conditions of use, (2) who rely on sources of drinking water downstream of release sites, and (3) breathing air near release sites.

Subsequent to this draft supplement, EPA is releasing a draft revised risk determination for 1,4-dioxane. The updated risk determination considers the results presented in this draft supplement as well as those published in the 2020 Risk Evaluation for 1,4-Dioxane.

EXECUTIVE SUMMARY

This draft document is a supplement to the *Final Risk Evaluation for 1,4-Dioxane*, published December 2020 (2020 RE). EPA conducted this supplemental analysis because contrary to the law's requirement for TSCA risk evaluations to be carried out on the "chemical substance" under the conditions of use (COUs), the 2020 RE excluded certain known human exposure pathways that are important to understanding the health implications of exposure to 1,4-dioxane. This supplement effectively completes EPA's risk evaluation on the chemical substance and positions the Agency to comprehensively address identified unreasonable risks.

1,4-Dioxane is primarily used as a solvent in commercial and industrial applications. It can also be produced as a byproduct of several common manufacturing processes, including but not limited to ethoxylation processes used in the production of surfactants used in soaps and detergents and production of polyethylene terephthalate (PET) plastics. Even though it is not intentionally added, 1,4-dioxane produced as a byproduct may remain present in consumer and commercial products, including soaps and detergents, cleaning products, antifreeze, textile dyes, and paints/lacquers. 1,4-Dioxane is released to the environment from industrial and commercial releases and from consumer and commercial products that are washed down the drain or disposed of in landfills. People may be exposed to 1,4-dioxane through occupational exposure, consumer products, or contact with water, land, or air where 1,4-dioxane has been released to the environment.

The 2020 RE did not evaluate risks from two critical areas: (1) general population exposures to 1,4-dioxane in drinking water or air, and (2) the full range of exposure that may result from 1,4-dioxane produced as a byproduct. During review of the draft risk evaluation, peer reviewers and public commenters raised concerns that failure to consider these exposure pathways could leave portions of the population at risk. These concerns include the fact that 1,4-dioxane has been detected in drinking water and is not readily removed through traditional treatment. In addition, 1,4-dioxane produced as a byproduct results in occupational exposures that were not evaluated in the 2020 RE. Finally, 1,4-dioxane produced as a byproduct also contributes to 1,4-dioxane in drinking water through industrial releases and down-the-drain (DTD) disposal of consumer and commercial products.

This draft supplement expands on the analysis of COUs in which 1,4-dioxane is present as a byproduct to include additional COUs for which information is reasonably available and consider associated occupational exposures. This draft supplement also evaluates risks to the general population—including potentially exposed or susceptible subpopulations (PESS)—from exposure to 1,4-dioxane through drinking water or air resulting from all industrial releases (including those resulting from 1,4-dioxane produced as a byproduct) as well as DTD releases of consumer and commercial products.

EPA's evaluation of these additional human exposure pathways included new methods and novel applications of existing methods that will be subject to peer review at a Science Advisory Committee on Chemicals (SACC) meeting in September 2023. Following review by the public and the SACC, the Agency will finalize this supplemental risk evaluation. At that time, EPA will initiate steps to address unreasonable risks identified through its complete evaluation of 1,4-dioxane.

Approach

For this draft supplemental risk evaluation, EPA is relying on the physical and chemical properties, chemical lifecycle information, environmental fate and transport information, and the hazard identification and dose-response analysis presented in the 2020 RE. EPA evaluated cancer and non-cancer risks from occupational and general population exposure scenarios using available modeling and/or monitoring information. The Agency also considered site-specific exposures, including combined or additive releases from multiple releasing facilities within a single air or water exposure pathway. This draft supplement considers PESS throughout the human health exposure assessment and risk characterization.

Exposure

Occupational Exposure: EPA estimated both high-end and central tendency occupational exposures through inhalation and dermal absorption. High-end exposure estimates were used because they attempt to capture potential variability in exposure across facilities and individuals and may be representative of PESS and "sentinel" exposures. In some cases, high-end estimates reflect uncertainty around the extent of this variability. EPA estimated occupational exposure for most COUs based on available monitoring data. For COUs without occupational monitoring data, EPA applied Monte Carlo methods to estimate exposures using generic scenarios and emission scenario documents. These methods are generally consistent with Monte Carlo approaches used in previous TSCA risk evaluations.

General Population Exposure: EPA evaluated general population exposures to 1,4-dioxane through drinking water and air that could result from releases to surface water, groundwater, land, and air. The Agency evaluated a range of exposure scenarios for each pathway, including (but not limited to) high-end exposure scenarios. To be protective of PESS and sentinel exposures, EPA developed risk estimates on the scenarios, populations, and life stages with the highest levels of exposure. For drinking water, EPA evaluated life stage-specific exposures for adults, formula-fed infants, and children. For air exposures, because the impacts of lifestage differences could not be quantified adequately, air concentrations are used for all lifestages. To address exposure to fenceline communities, EPA considered air exposures within 10 km of a release site. For water releases, EPA considered exposures to communities relying on drinking water sourced near release sites.

Hazard

All hazard values used in this draft supplement were derived from the points of departure (PODs) previously peer-reviewed by the SACC and published in the 2020 RE. Some of the exposure scenarios evaluated in this analysis required duration adjustments to the previously established hazard values; however, the underlying hazard endpoints and PODs remain the same. Health effects of concern for 1,4-dioxane include cancer and adverse, non-cancer effects in liver and olfactory tissue.

Risk Characterization

EPA evaluated both cancer and non-cancer risks for each exposure pathway. Because cancer is the primary risk driver for 1,4-dioxane, results presented here are cancer risk estimates. Overall confidence in the risk estimates varies across exposure pathways and COUs, depending on the data, models, and assumptions used. For risk estimates, EPA has medium to high confidence in the underlying hazard PODs used as the basis for risk characterization. Therefore, exposure-related considerations drive differences in confidence among risk estimates. Differences in central tendency and high-end risk estimates may reflect variability in exposure across the population (*e.g.*, due to differences in the frequency or intensity of occupational exposures) and/or uncertainty in the exposure assessment (*e.g.*, due to incomplete information on release amounts or variations in flow rates of receiving water bodies and/or drinking water intake locations).

Estimates of Occupational Risks: EPA estimated cancer and non-cancer risks for a set of new occupational COUs where 1,4-dioxane is present as a byproduct.

- Dermal Exposure: Dermal occupational exposure is expected to occur as a result of worker activities such as transfer operations, application of 1,4-dioxane containing formulations, and the cleaning of equipment. COU-specific weight fractions and evaporation drive the variability in results. Cancer risk estimates for dermal exposures range from 8.1×10^{-7} to 8.6×10^{-4} for central tendency exposure and from 5.0×10^{-6} to 1.5×10^{-2} for high-end exposures across COUs. Overall confidence in risk estimates for occupational dermal exposures is medium for all occupational exposure scenarios.
- Inhalation Exposure: Inhalation exposure to 1,4-dioxane is expected to occur as a result of the scenario-specific considerations described in the bullets below. Cancer risk estimates for inhalation exposure range from 8.3×10^{-12} to 1.8×10^{-3} for central tendency exposures and from 5.4×10^{-11} to 2.3×10^{-2} for high-end exposures across COUs. Occupational exposure scenarios with the highest estimates of risk from inhalation exposure are summarized below:

- **Polyethylene Terephthalate (PET) Manufacturing.** Workers may inhale 1,4-dioxane generated as a byproduct of PET plastic manufacturing. Cancer risk estimates for inhalation exposure range from 1.8×10^{-3} for central tendency exposures to 2.3×10^{-2} for high-end exposures. There is uncertainty regarding the risk estimates because the analysis relied on decades-old monitoring data and the extent to which the monitoring data reflect current practices is unknown. Overall confidence in risk estimates for PET plastic manufacturing is medium.

- **Hydraulic Fracturing Operations.** 1,4-Dioxane inhalation exposures may occur during hydraulic fracturing operations due to its documented presence in scale inhibitors, additives, friction reducers, and surfactants used in fracturing fluid formulations. Cancer risk estimates for inhalation exposure range from 7.0×10^{-5} for central tendency exposures to 9.5×10^{-3} for high-end exposures. There is uncertainty regarding the model inputs used to estimate exposures and the extent to which they reflect the actual distribution of hydraulic fracturing occupational exposures and workplace practices. Overall confidence in risk estimates for hydraulic fracturing operations is medium to high.

- **Ethoxylation Processes.** 1,4-Dioxane may be generated as a byproduct in ethoxylation reactions during the manufacture of common surfactants that result in worker inhalation exposure. Cancer risk estimates for inhalation exposure range from 4.6×10^{-4} for central tendency exposures to 5.9×10^{-4} for high-end exposures. There are numerous uncertainties due to the limited monitoring data, unknown concentrations, and the mass of 1,4-dioxane generated as a byproduct during ethoxylation. There is also uncertainty in the worker activities covered by the monitoring data and whether all foreseeable activities, corresponding exposures, and workplace operations are represented. Overall confidence in risk estimates for ethoxylation processes is low to medium.

- **Industrial/Commercial Use of Dish Soap and Dishwasher Detergent.** 1,4-Dioxane inhalation exposures are expected during the use of dish soap and dishwashing detergent from unloading and transferring detergent formulation, transport container cleaning, and washing operations due to the presence of 1,4-dioxane as a surfactant byproduct. Cancer risk estimates for inhalation exposure range from 4.0×10^{-4} for central tendency exposures to 1.0×10^{-3} for high-end exposures. There is uncertainty as to the representativeness of these estimates due to the age of the monitoring data, number of non-detects, and the limited sample size. Overall confidence in risk estimates for dish soap and dishwasher detergent use is low to medium.

Estimates of Risk to the General Population Exposure through Drinking Water Sourced from Surface Water: EPA estimated cancer and non-cancer risks for a range of general population exposures to surface water used as drinking water. 1,4-Dioxane is not readily removed through typical wastewater or drinking water treatment processes. Sources of 1,4-dioxane in surface water include direct and indirect industrial releases from COUs where 1,4-dioxane is manufactured, processed, or used; industrial COUs where 1,4-dioxane is present due to production as a byproduct (including PET manufacturing, ethoxylation processes and hydraulic fracturing operations); and DTD releases of 1,4-dioxane present in consumer and commercial products.

Monitoring data demonstrates that 1,4-dioxane is present in some source water and finished drinking water samples. Measured concentrations in finished drinking water samples resulted in cancer risk estimates greater than 1 in 1 million at the high end of the distribution of monitoring samples. However, available surface water monitoring datasets are not designed to reflect source water impacts of direct and indirect releases into water bodies. Therefore, EPA estimated concentrations modeled for a range of specific release scenarios. The Agency evaluated the performance of the models against monitoring data from site-specific locations serving as cases studies. This evaluation demonstrated strong concordance between modeled concentrations and monitoring data, thereby increasing confidence in risk estimates based on modeled concentrations.

EPA used modeled water concentrations to evaluate risks from a range of sources individually and in aggregate (*i.e.*, by evaluating risks from water concentrations resulting from multiple sources of 1,4-dioxane releasing to the same water bodies). For each of the sources assessed, risk estimates from modeled concentrations in receiving water bodies at the point of release may be greater than 1 in 1,000,000 or 1 in 100,000 under some conditions.

- <u>Industrial Releases to Surface Water</u>: Risk from individual facilities vary substantially within and across COUs, with cancer risk estimates ranging from 5.4×10^{-13} to 0.025. Overall confidence in risk estimates for specific facilities depends on confidence in facility-specific release data, but confidence in the overall analysis is medium-high
- <u>Down-the-drain Releases to Surface Water</u>: EPA evaluated the conditions under which down-the-drain releases contribute to different levels of risk and identified plausible scenarios in which risks from down-the-drain releases result in risks greater than 1 in 1 million. Risk estimates from modeled down-the-drain releases are highest in locations where large populations contribute to these releases and where they are discharged to streams with low flow. Overall confidence in this analysis is medium.
- <u>Hydraulic Fracturing Releases to Surface Water</u>: Cancer risk estimates from modeled hydraulic fracturing waste releases to surface water are 3.57×10^{-8} for median modeled releases and 1.45×10^{-6} for 95th percentile modeled releases. Overall confidence in this analysis is medium.
- <u>Aggregate Releases to Surface Water</u>: Probabilistic modeling provides a distribution of risk estimates reflecting a range of drinking water scenarios that account for aggregate sources of 1,4-dioxane in water. Overall confidence in risk estimates for specific facilities depends on confidence in facility-specific release data used as model inputs.

The relative contribution from different sources varies under different conditions and is likely to be driven by site-specific factors including the amounts released from each source, flow rates of receiving water bodies, and proximity of releases to drinking water intakes. In locations where industrial releases are particularly high, industrial releases are likely to drive risk. In locations where large populations contribute to down-the-drain releases and receiving water bodies have relatively low flow rates, down-the-drain releases on their own may drive risks. Each of the sources evaluated may contribute to drinking water risks under some conditions. Furthermore, this analysis illustrates the fact that sources of 1,4-dioxane produced as a byproduct, including those from industrial releases and DTD releases, can contribute to risks from 1,4-dioxane in water. The analyses in this draft supplement describe the conditions under which different levels of risk may occur.

Estimates of Risk to the General Population Exposure through Drinking Water Sourced from Groundwater: EPA estimated cancer and non-cancer risks for a range of general population exposures to groundwater used as drinking water. Sources of 1,4-dioxane in groundwater include leachate from landfills and disposal of hydraulic fracturing waste. Down the drain releases to septic fields from consumer and commercial products containing 1,4-dioxane as well as historical disposals of 1,4-dioxane are other potential sources of groundwater contamination but were not considered in this assessment. Overall confidence in these risk estimates is low to medium.

- 1,4-Dioxane in Groundwater from Hydraulic Fracturing: Cancer risk estimates for modeled groundwater concentrations are 4.0×10^{-7} for median modeled releases and 8.6×10^{-6} for 95th percentile modeled releases.
- 1,4-Dioxane in Groundwater Resulting from Landfill Leachate: Cancer risk estimates increase under scenarios with higher leachate concentrations and loading rates. Monitoring data for groundwater contamination surrounding landfills was not readily available for comparison.

Estimates of Risk to the General Population through Air: EPA estimated cancer and non-cancer risks for a range of general population exposures to 1,4-dioxane in air. 1,4-Dioxane concentrations in air depend on the facility-specific release amount, stack height(s), topography, and meteorological conditions—not on specific COUs.

Potential sources of 1,4-dioxane in air include industrial releases reported to Toxics Release Inventory (TRI), fugitive emissions from hydraulic fracturing, and emissions from institutional and industrial laundries. The highest estimated risks occurred within 1,000 m of industrial release sites. EPA also estimated risk from the aggregate exposures from multiple facilities releasing 1,4-dioxane in proximity to fenceline communities. This aggregate analysis did not identify locations with aggregate cancer risk greater than 1×10^{-6} that did not already have cancer risk above that level from an individual facility and therefore did not have a substantial impact on the overall findings.

1,4-Dioxane concentrations in air depend on the facility-specific release amount, stack height(s), topography, and meteorological conditions.

- Air Releases Reported to TRI: Cancer risk estimates for 95th percentile modeled air concentrations with 1,000 m (approximately 0.6 mile) of the highest risk facilities in each COU range from 1.0×10^{-10} to 1.1×10^{-4}. Cancer risk estimates based on 50th percentile modeled exposure concentrations within 1,000 m of the highest risk facilities range from 2.5×10^{-11} to 8.3×10^{-5}. Although individual risk estimates for specific locations should be interpreted with caution, most estimates are informed by moderate to robust modeling approaches and input data. Overall confidence in risk estimates for inhalation exposures resulting for air concentrations modeled for industrial releases ranges from low to high, depending on the level of confidence in release information underlying risk estimates for specific facilities and COUs.
- Fugitive Air Emissions from Hydraulic Fracturing Operations: Cancer risk estimates within 1,000 m of hydraulic fracturing operations range from 2.2×10^{-8} to 7.1×10^{-5} for a range of model air model scenarios across a range of high-end (95th percentile) and central tendency release scenarios. Overall confidence in risk estimates for inhalation exposures resulting for air concentrations modeled based on releases from hydraulic fracturing operations is medium.
- Emissions from Industrial and Institutional Laundries: Cancer risk estimates within 1,000 m of industrial and institutional laundries range from 1.5×10^{-11} to 3.8×10^{-8} across a range of high-end exposure scenarios. Overall confidence in risk estimates from inhalation exposures resulting from industrial and institutional laundries is medium.

1 INTRODUCTION

1,4-Dioxane is one of the first 10 chemicals undergoing the Toxics Substances Control Act (TSCA) risk evaluation process after passage of the Frank R. Lautenberg Chemical Safety for the 21st Century Act, which amended TSCA. 1,4-Dioxane is primarily used as a solvent in a variety of commercial and industrial applications such as the manufacture of other chemicals (*e.g.*, adhesives, sealants) or as a processing aid or laboratory chemical. It is produced as a byproduct in several manufacturing processes, including ethoxylation, sulfonation, sulfation, and esterification. Although there are no direct consumer uses, 1,4-dioxane produced as a byproduct in the aforementioned processes can be present in commercial and consumer products, including soaps, detergents, and cleaning products. Use of these products may result in direct occupational and consumer exposures. Disposal of these products down-the-drain (DTD) may contribute to general population exposure to 1,4-dioxane present in some surface water.

1.1 Regulatory Context

In the 2019 draft 1,4-dioxane risk evaluation, EPA reviewed the exposures, hazards, and risks of 1,4-dioxane from occupational exposures and surface water exposures to environmental organisms. It also included the physical and chemical properties, lifecycle information, environmental fate and transport information, and hazard identification and dose-response analysis. However, the 2019 draft risk evaluation excluded general population exposures through drinking water and air and conditions of use (COUs) in which 1,4-dioxane is present as a byproduct. These exclusions were based in part on an interpretation that EPA had broad discretionary authority under TSCA to categorically exclude conditions of use from the scope of its evaluations, and, as described in the 2018 *1,4-Dioxane Problem Formulation*,[1] that certain exposure pathways need not be considered if they were under the jurisdiction of other EPA regulatory programs or analytical processes.

These analyses were reviewed by the Science Advisory Committee on Chemicals (SACC) in 2019. The SACC raised a number of concerns regarding the evaluation and approach, but particularly noted its concerns about the Problem Formulation straying from "basic principles of risk assessment," the omission of well-known exposure routes, and that general lack of comprehensiveness undermining EPA's ability to protect against risks to human health and the environment. As stated in the meeting minutes and final report[2] from the July 2019 SACC meeting, "there was general dissatisfaction in the Committee that the human health risk characterization did not extend to the general population since there was no indication in the Evaluation that other offices in the EPA had plans to conduct such a characterization." Furthermore, "several committee members also observed that failure to assess 1,4-dioxane exposure in the general population may leave substantial portions of the population at risk. This is particularly concerning for drinking water." The SACC also raised concerns about potential risks from 1,4-dioxane produced as a byproduct, recommending that "EPA should provide a detailed discussion of the scientific basis for the exclusion of impurity or byproduct formation of 1,4-dioxane."

Public stakeholders also raised concerns about water monitoring data demonstrating the presence of 1,4-dioxane in drinking water. Commenters also identified additional sources of 1,4-dioxane that had not been assessed, including COUs in which 1,4-dioxane is produced as a byproduct, and raised concerns that some of these COUs may be an important source of exposure to 1,4-dioxane in water.

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[1] The 1,4-Dioxane Problem Formulation is available at https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/14-dioxane-problem-formulation.

[2] The SACC July 2019 meeting minutes and final report (Document ID EPA-HQ-OPPT-2019-0237-0064) are available in the docket at https://www.regulations.gov/document/EPA-HQ-OPPT-2019-0237-0064.

In November of 2020, EPA released a supplement to the draft 1,4-dioxane risk evaluation for public comment. The November 2020 supplement assessed eight additional COUs of 1,4-dioxane as a byproduct in consumer products and general population exposure from incidental contact with surface water. The EPA determined that the additional analysis did not warrant SACC review as no novel science was utilized.

In December 2020, the Agency published the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c) (referred hereafter as the "2020 RE") (U.S. EPA, 2020c). The 2019 draft and 2020 supplement were both incorporated into the 2020 RE, which assessed risks for

- worker and occupational non-user (ONU) exposures to 1,4-dioxane through 16 industrial and commercial COUs;
- consumer and bystander exposures to 1,4-dioxane present as a byproduct[3] in eight consumer product categories;
- general population exposure via incidental/recreational contact with 1,4-dioxane present in surface water from industrial releases; and
- aquatic species' exposures to 1,4-dioxane present in surface water.

In January 2021, the White House issued Executive Order 13990 instructing that the federal government be guided by the best science and be protected by processes that ensure the integrity of Federal decision-making, and established the Administration's policy of, among other concerns, following the science, improving public health and protecting the environment, limiting exposure to dangerous chemicals, and prioritizing environmental justice when delivering on these concerns. Executive Order 13990 also instructs agencies to (1) review actions issued between January 20, 2017, and January 20, 2021, which may be inconsistent with or present obstacles to implementing the policy established in the order and; (2) consider suspending, revising, or rescinding such actions.

Upon further review, EPA determined that the approach taken in the 2020 RE (*i.e.*, the exclusion of reasonably foreseeable exposures to workers, as well as exposures to the general population from air, water, and disposal) was inconsistent with the plain language of TSCA section 6 and left potential risks—including risks to potentially exposed and susceptible subpopulations (PESS)—unaccounted for. The law's requirement that EPA conduct risk evaluation on a "chemical substance" under the COUs requires the Agency to determine the chemical's COUs and to not otherwise exclude those COUs from the scope of the risk evaluation.

In June of 2021, EPA announced that additional analysis was needed to consider critical exposure pathways not assessed in the final risk evaluations for the first 10 chemicals (including, but not limited to, ambient air, ambient water, and drinking water). For many of the first 10 risk evaluations, EPA applied the *Draft Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities Version 1.0*,[4] published in January 2022 to determine whether further analysis was needed. For 1,4-dioxane, however, EPA determined that a more in-depth analysis was needed to address concerns about known drinking water contamination (described in Section 1.3.1.3) and to more fully evaluate COUs in which 1,4-dioxane is present as a byproduct (described in Section 1.3.1.1), and signaled its intention to re-open and formally supplement the 1,4-dioxane risk evaluation.

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[3] Byproduct means a chemical substance produced without a separate commercial intent during the manufacture, processing, use, or disposal of another chemical substance(s) or mixture(s).

[4] The draft *Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities Version 1.0* is available at https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/tsca-screening-level-approach-assessing-ambient-air-and.

This draft supplement to the 2020 RE is intended to complete the risk evaluation on the chemical substance 1,4-dioxane as required under TSCA by (1) expanding the analysis of COUs in which 1,4-dioxane is present as a byproduct to include additional COUs and consider associated occupational exposures; and (2) evaluating risks from general population exposures to 1,4-dioxane released to surface and groundwater, air, and land.

EPA's evaluation of these additional human exposure pathways included new methods and novel applications of existing methods. This draft supplement is the first under amended TSCA to evaluate: exposures and risks from a chemical produced as a byproduct, aggregate risks for communities relying on drinking water sourced from surface water receiving a chemical from multiple sources, risks for communities relying on drinking water sourced from groundwater, aggregate risks for communities exposed through air near multiple release sites, and consideration of multiple years of environmental release data.

1.2 Scope

This draft supplement is intended to evaluate risks from exposure pathways and COUs for 1,4-dioxane that were not assessed in the 2020 RE. Additional exposure pathways and new COUs included in this draft supplement were identified based on information submitted in previous public comments and other reasonably available information. For the current analysis, EPA is relying on the physical and chemical properties, as well as lifecycle information, environmental fate and transport information, and hazard identification and dose-response analysis presented in the 2020 RE (Sections 1.1, 1.4, 2.1, and 3.2 of the 2020 RE, respectively). Furthermore, this supplement does not reevaluate the occupational, consumer, or ecological exposure pathways and risks that were previously assessed in the 2020 RE.

This draft supplement to the 2020 RE evaluates risks for the following exposure pathways:

- Occupational exposure to
    - 1,4-dioxane present as a byproduct in commercial products (corresponding to consumer products considered in the 2020 RE); and
    - 1,4-dioxane produced or present as a byproduct in additional industrial COUs for which information on the presence of 1,4-dioxane is reasonably available, including ethoxylation processing, polyethylene terephthalate (PET) manufacturing, and hydraulic fracturing (Sections 3.1, 5.2.1).
- General population exposures to
    - 1,4-dioxane present in drinking water sourced from surface water as a result of all direct and indirect industrial releases and DTD releases of consumer and commercial products (Sections 2.3.1, 3.2.2 and 5.2.2.1);
    - 1,4-dioxane present in drinking water sourced from groundwater contaminated as a result of disposals (Sections 2.3.2, 3.2.2.2 and 5.2.2.1.6); and,
    - 1,4-dioxane released to air from industrial and commercial sources (Sections 2.3.3, 3.2.3, and 5.2.2.3).

Many of the COUs assessed in this draft supplement contribute to more than one exposure pathway. For example, 1,4-dioxane present as a byproduct of PET manufacturing may contribute to occupational exposures during manufacturing as well as general population exposures through releases to water and air. In addition, for many of the exposure pathways assessed, multiple COUs contribute to 1,4-dioxane exposure. For example, many COUs can contribute to general population exposures to 1,4-dioxane in surface water, including industrial releases from a range of COUs and DTD releases of consumer and commercial products. In this draft supplement, EPA evaluated general population exposures resulting from all known releases, including releases associated with COUs evaluated in the 2020 RE and releases associated with new COUs introduced in this draft supplement due to the presence of 1,4-dioxane produced as a byproduct.

1.3 Use Characterization

1.3.1 Conceptual Models

The life cycle diagram for 1,4-dioxane in Figure 1-1 summarizes the conditions of use that are within the combined scope of the 2020 RE and the current draft supplement. The life cycle diagram has been updated from the 2020 RE to highlight additional sources of 1,4-dioxane produced as a byproduct, including commercial products and industrial uses, releases, and disposals (*e.g.*, PET manufacturing, ethoxylation byproducts, disposal of hydraulic fracturing produced waters).

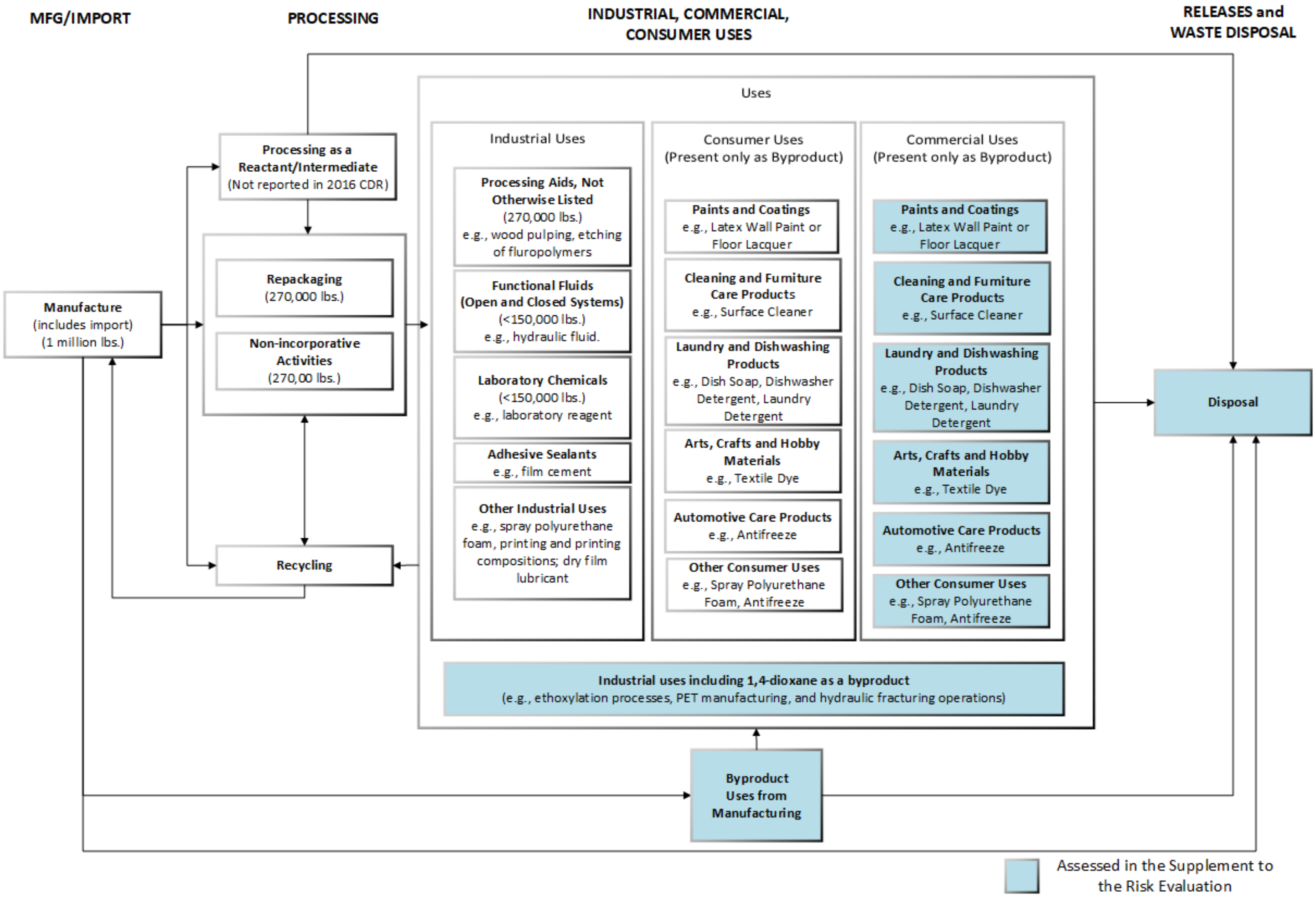

Figure 1-1. 1,4-Dioxane Life Cycle Diagram

Note: This life cycle diagram has been expanded from what was published in the 2020 RE to include additional sources of 1,4-dioxane produced as a byproduct (indicated in blue boxes). See Appendix D for a complete table of COUs considered in the 2020 RE and the current supplement.

1.3.1.1 1,4-Dioxane as a Byproduct

1,4-Dioxane produced as a byproduct may be a source of exposure for several of the occupational and general population exposure pathways evaluated in this draft supplement. Byproduct refers to a chemical substance produced without a separate commercial intent during the manufacture, processing, use, or disposal of another chemical substance(s) or mixture(s). In this assessment, the term byproduct is used to refer to 1,4-dioxane produced during manufacturing or industrial processes, including 1,4-dioxane that remains present in downstream processes or in consumer and commercial products.

In the 2020 RE, EPA evaluated risks to consumers and bystanders from 1,4-dioxane present as a byproduct in consumer products. In this draft supplement, EPA expanded on the previous evaluation to consider risks from all other pathways of exposure to 1,4-dioxane produced as a byproduct for which information is reasonably available. Figure 1-2 summarizes both what is known about the processes that may result in 1,4-dioxane production and how it may contribute to human exposures through a range of exposure pathways.

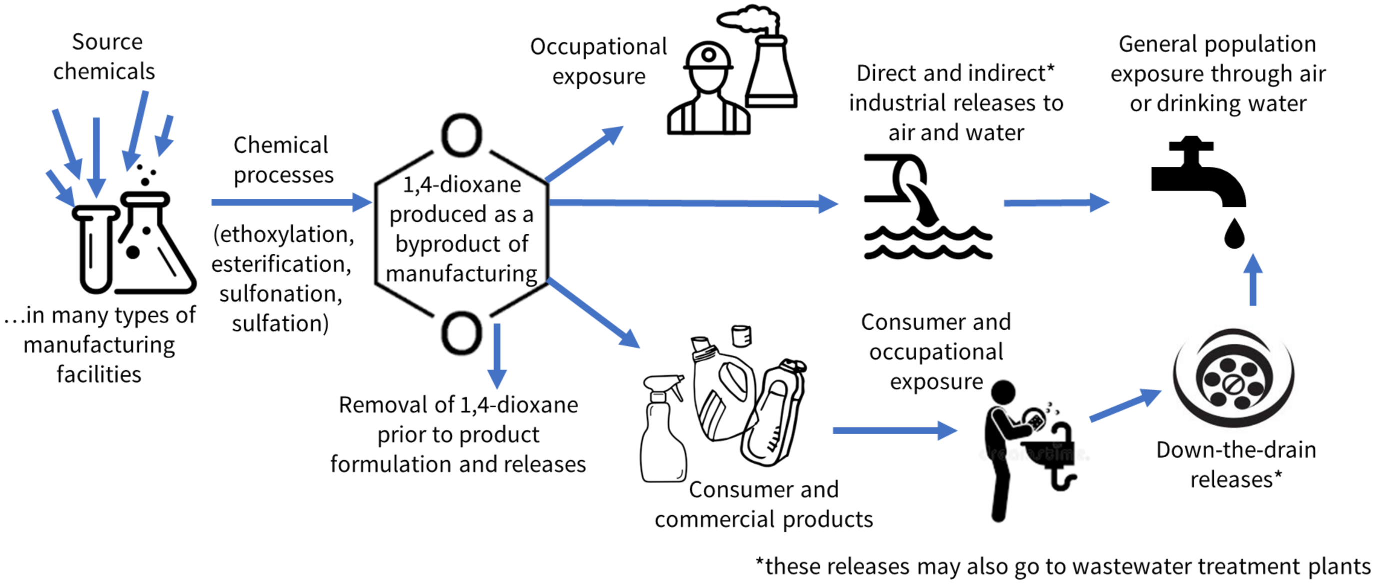

Figure 1-2. Production of 1,4-Dioxane as a Byproduct and Potential Exposure Pathways

1,4-Dioxane is produced as a byproduct in several common manufacturing reactions, including in manufacturing of PET plastics and in ethoxylation reactions during the manufacture of common surfactants. In some facilities, additional processing steps may remove 1,4-dioxane produced as a byproduct prior to product formulation and environmental releases, but the full extent of this practice across industries is not known. Occupational exposure to 1,4-dioxane produced as a byproduct may occur at manufacturing facilities and hydraulic fracturing operations. Releases of 1,4-dioxane from manufacturing and industrial sites may also contribute to general population exposures through drinking water and air.

1,4-Dioxane produced as a byproduct has also been detected in consumer and commercial products, resulting in potential exposure to consumers and bystanders (evaluated in the 2020 RE) or workers and ONUs (evaluated as described in Section 3.1.2). For example, 1,4-dioxane exposures are expected during the use of dish soap and dishwashing detergent from unloading and transferring detergent formulation, transport container cleaning, and washing operations due to the presence of 1,4-dioxane as a surfactant byproduct. In addition, consumer and commercial products containing 1,4-dioxane may contribute to general population exposures when released down the drain.

In this draft supplement, EPA evaluated pathways of exposure to 1,4-dioxane produced as a byproduct that were not previously assessed. Specifically, EPA considered 1,4-dioxane present as a byproduct in commercial products corresponding to the consumer products considered in the 2020 RE. EPA considered the direct occupational exposures that result from use of these commercial products as well as the DTD releases of consumer and commercial products, which contribute to general population exposures through surface water. EPA also identified a new set of COUs where 1,4-dioxane is produced or present as a byproduct, based on information submitted by public commenters and other reasonably available information. For each of these new COUs, EPA evaluated occupational exposure as well as industrial releases that contribute to general population exposures. The available information supporting inclusion of each of the new COUs is described below. A more detailed list of the new COUs and COU subcategories considered in this draft supplement is presented in Section 2.1.1. The following COUs are known to produce 1,4-dioxane as a byproduct based on reasonably available information, but 1,4-dioxane produced as a byproduct may also be present in other industries that have not yet been identified:

- **Industrial/Commercial Use of Products Containing 1,4-Dioxane as a Byproduct.** 1,4-Dioxane is present in a range of commercial products (including textile dyes, antifreeze, surface cleaners, dish soaps, laundry detergents, and paint and floor lacquer) because it is produced as a byproduct during the manufacture of ingredients such as ethoxylated surfactants. While 1,4-dioxane present as a byproduct in consumer products was previously assessed, evaluation of 1,4-dioxane in these corresponding commercial products is new in this draft supplement. 1,4-Dioxane present in commercial products can result in occupational exposure in commercial settings as well as DTD releases that contribute to general population exposures.
- **PET Manufacturing.** 1,4-Dioxane has been identified as a byproduct in the manufacture of PET (U.S. EPA, 2017c). EPA does not have information on the byproduct concentration of 1,4-dioxane in PET. PET is produced by the esterification of terephthalic acid to form bishydroxyethyl terephthalate (BHET) (Forkner et al., 2004). BHET polymerizes in a transesterification reaction catalyzed by antimony oxide to form PET (Forkner et al., 2004). In 2014, 20.6 million metric tons of PET were used in the United States (McDaniel and DesLauriers, 2015). 1,4-Dioxane produced during PET manufacturing may result in occupational exposures and may contribute to general population exposures via releases to water and air.
- **Ethoxylation Processing.** 1,4-Dioxane may be formed as a byproduct of reactions based on condensing ethylene oxide or ethylene glycol during manufacture of detergents, shampoos, surfactants, some food additives, and certain pharmaceuticals (HHS, 2016). In cosmetic ethoxylated raw materials and ethoxylated alkyl sulfates, 1,4-dioxane has been detected at concentrations of 0.48 to 1,410 ppm (U.S. EPA, 2020c; Saraji and Shirvani, 2017; Davarani et al., 2012; Black et al., 2001). Polyethoxylated raw materials are widely used in cosmetic products as emulsifiers, foaming agents, and dispersants (Black et al., 2001). They are produced by polymerizing ethylene oxide, usually with a fatty alcohol, to form polyethoxylated alcohols which may be used to synthesize other products such as sulfated surface-active agent. During the ethoxylation process, 1,4-dioxane can be formed as a byproduct by the dimerization of ethylene oxide (Black et al., 2001). The volume of 1,4-dioxane produced as a byproduct of ethoxylation reactions is unknown. 1,4-Dioxane produced during ethoxylation processing may result in occupational exposures and may contribute to general population exposures via releases to water and air. It also contributes to the presence of 1,4-dioxane in consumer and commercial products.

- **Hydraulic Fracturing.** Hydraulic fracturing stimulates an existing oil or gas well by injecting a pressurized fluid containing chemical additives into the well (U.S. EPA, 2022d).1,4-Dioxane is measured in fracturing fluid, a water-based fluid that contains several chemical additives and in waste fluid (produced waters). FracFocus 3.0 contains self-reported information indicating that 1,4-dioxane is present in hydraulic fracturing fluid additives, as scale inhibitors, additives, biocides, friction reducers, and surfactants (GWPC and IOGCC, 2022). According to the FracFocus 3.0 database, 1,4 dioxane is present in weight fractions ranging from 2.3×10^{-11} to 0.05 within hydraulic fracturing additives and 1.00×10^{-12} to 4.30×10^{-6} in hydraulic fracturing fluids (GWPC and IOGCC, 2022).1,4-Dioxane has been documented to have a concentration of 60 µg/L in hydraulic fracturing produced waters (Lester et al., 2015). 1,4-Dioxane present in fracturing fluid may result in occupational exposures. It may also contribute to general population exposures via discharge to surface water, groundwater, or fugitive air emissions from fracturing operations.

1.3.1.2 Occupational Exposures

The conceptual model Figure 1-3 Figure 1-3presents the exposure pathways, exposure routes, and hazards to people from industrial and commercial releases and uses of 1,4-dioxane. Blue shading highlights the exposures evaluated in this draft supplement. Workers and ONUs may have acute (8 hr) or chronic (annual to lifetime) exposures to 1,4-dioxane produced as a byproduct during PET manufacturing, ethoxylation processes, or hydraulic fracturing operations. Workers and ONUs may also have acute or chronic exposures to 1,4-dioxane present as a byproduct in commercial products, including detergents, cleaners, and lacquers.

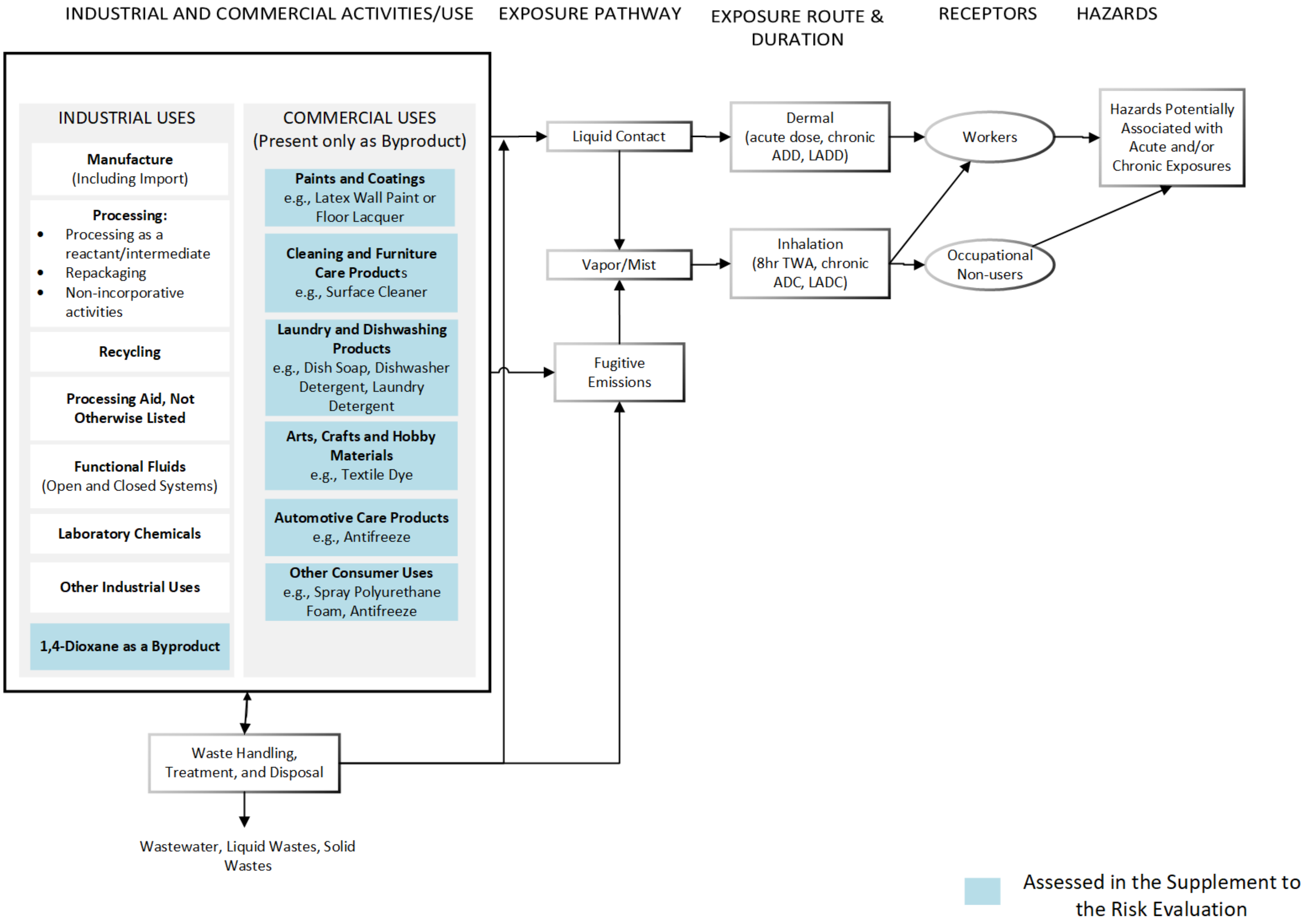

Figure 1-3. Conceptual Model for Occupational Exposures from Industrial and Commercial Activities

In this draft supplement, EPA evaluated acute and chronic cancer and non-cancer risks from occupational inhalation and dermal exposures to 1,4-dioxane produced as a byproduct during PET manufacturing and ethoxylation processes, hydraulic fracturing fluids and waste containing 1,4-dioxane, and commercial products containing 1,4-dioxane.

1.3.1.3 General Population Exposures

The conceptual model in Figure 1-4 presents general population exposure pathways and hazards from environmental releases and wastes associated with COUs (red, blue, and purple shading for each source in the figure corresponds to the environmental media to which they release). The disposal and release scenarios illustrated in Figure 1-1, Figure 1-2, and Figure 1-3 all contribute to the releases to air, water and land that may result in the general population exposures illustrated in Figure 1-4. The general population may be exposed to 1,4-dioxane released to surface water (blue shading), groundwater (red shading), and air (purple shading). Drinking water exposures are evaluated based on releases to both surface water and groundwater and these are each described in the drinking water subsection below.

EPA's evaluation of general population exposures considers potentially exposed or susceptible subpopulations (PESS). Exposures to 1,4-dioxane through air and water could result in risk to fenceline communities. As defined in the *Draft Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities Version 1.0*,[5] fenceline communities are members of the general population that are in proximity to air emitting facilities or a receiving water body, and who therefore may be disproportionately exposed to a chemical undergoing risk evaluation under TSCA section 6. For the air pathway, proximity goes out to 10,000 m from an air emitting source. For the water pathway, proximity does not refer to a specific distance measured from a receiving water body, but rather to those members of the general population that may interact with the receiving water body and thus may be exposed.

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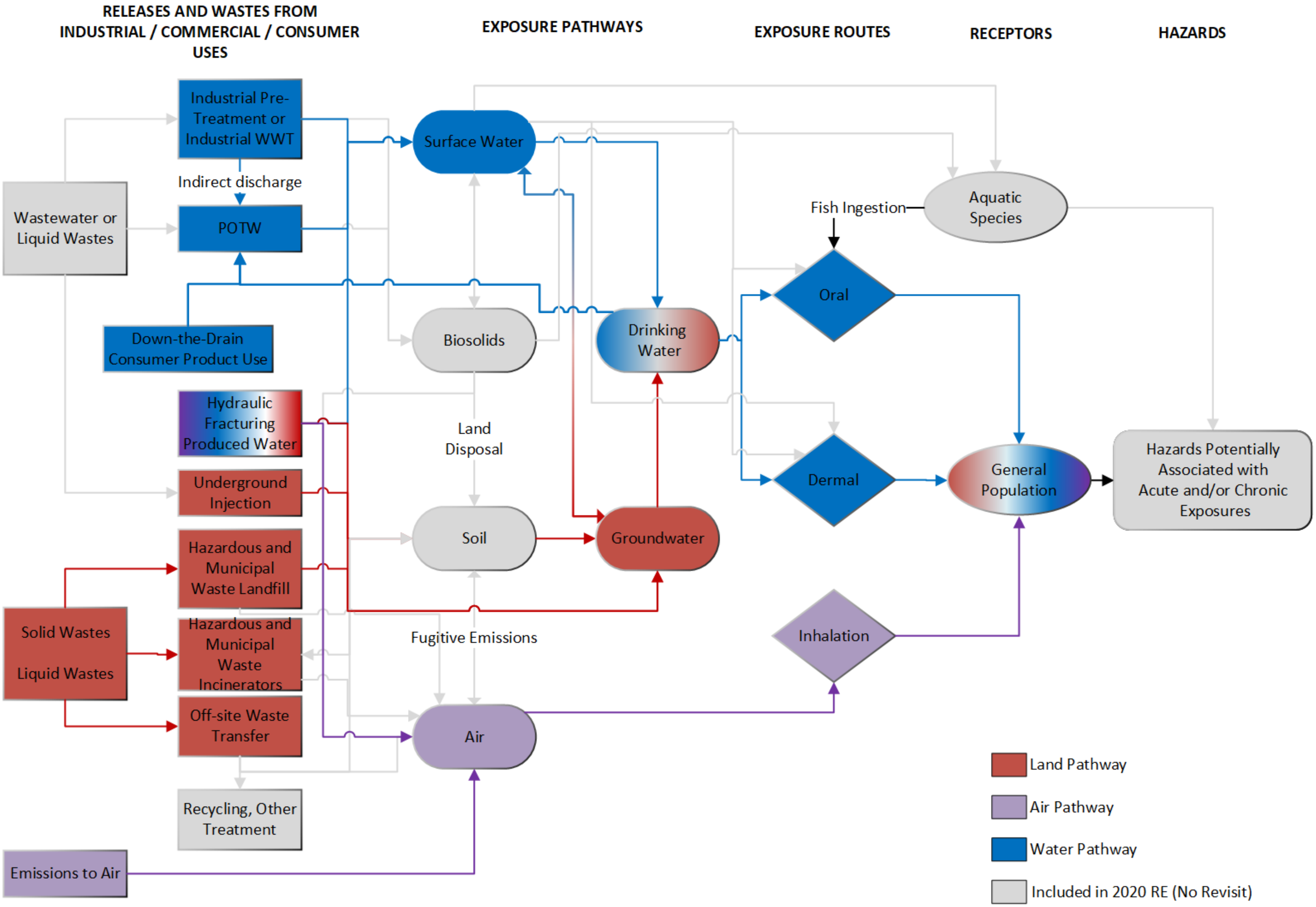
[5] The draft *Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities Version 1.0* is available at https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/tsca-screening-level-approach-assessing-ambient-air-and.



1292
1293 **Figure 1-4. Conceptual Model for Environmental Releases and General Population Exposures**

The 2020 RE included an assessment of acute incidental/recreational general population exposure to 1,4-dioxane in surface water resulting from industrial releases. It did not evaluate risks from other general population exposure pathways such as drinking water or air.

In this draft supplement, EPA evaluated additional general population exposure via air and drinking water. EPA evaluated acute (24-hour) and chronic (annual to lifetime) cancer and non-cancer risks from these exposure pathways. Where data were reasonably available, EPA incorporated releases of 1,4-dioxane produced as a byproduct into these pathways. The Agency also considered aggregate 1,4-dioxane exposures and risks from multiple releasing facilities or COUs for each pathway.

1.3.1.3.1 Drinking Water

1,4-Dioxane may enter surface water through direct and indirect industrial releases, DTD releases from consumer and commercial products via wastewater treatment facilities, and releases of wastewater from hydraulic fracturing sites. Similarly, 1,4-dioxane released or disposed of through various land pathways may reach groundwater under some conditions. There is potential for general population exposures to 1,4-dioxane if contaminated surface water or groundwater are used as drinking water. 1,4-Dioxane is mobile in water and does not readily degrade in water. Available data indicate that typical wastewater treatment and drinking water treatment methods are not effective at removing 1,4-dioxane. The subsections below explain how general population exposures through surface and groundwater were considered.

Surface Water Pathway

1,4-Dioxane was included in the third unregulated contaminant monitoring rule (UCMR3) (U.S. EPA, 2017d) published in May of 2012 requiring community water systems to monitor for 1,4-dioxane between 2013 and 2015. National and state water monitoring programs have detected 1,4-dioxane in drinking water and drinking water sources (as described in Section 2.3.1.10). In the absence of monitoring data, estimating 1,4-dioxane surface water concentrations can be complex because in addition to direct and indirect industrial and commercial releases, upstream sources from releasing facilities and DTD releases of consumer and commercial products contribute to surface water contamination.

EPA evaluated surface water concentrations (Section 2.3.1) and drinking water exposures (Section 3.2.2.1) that could result from direct and indirect industrial releases, DTD releases of consumer and commercial products, and disposal of wastewater from hydraulic fracturing sites. EPA used a novel aggregate model to predict water concentrations of 1,4-dioxane that could result from multiple sources that release 1,4-dioxane to the same receiving water bodies. Surface water concentrations estimated for each source in isolation and from multiple sources in aggregation were used to evaluate potential exposures (Section 3.2.2.1) and risks (Section 5.2.2.1) from general population oral exposure to 1,4-dioxane in drinking water.

Land Pathway to Groundwater

Groundwater contamination with 1,4-dioxane presents a potential risk when the chemical substance is released to landfills, underground injection wells, or surface impoundments. Due to its physical-chemical properties (*e.g.*, water solubility, Henry's law constant) and fate characteristics (*e.g.*, biodegradability, half-life in groundwater), 1,4-dioxane is anticipated to persist in groundwater for months to years. This persistence has resulted in higher 1,4-dioxane concentrations in groundwater relative to other media (ATSDR, 2012). EPA considered potential for groundwater contamination following disposal of waste containing 1,4-dioxane to landfills, underground injection of 1,4-dioxane waste, and disposals of hydraulic fracturing waste containing 1,4-dioxane. Groundwater concentrations estimated for each source (Section 2.3.2) were used to evaluate exposure (Section 3.2.2.2) and risks (Section 5.2.2.1.6) for communities who rely on groundwater as a source of drinking water.

1.3.1.3.2 Air

Industrial releases to air include those from sites where 1,4-dioxane is manufactured intentionally as well as those where it is produced or present as a byproduct. In this draft supplement, EPA evaluated exposures and risks for communities located near release sites (fenceline communities) because they are the members of the general population that are expected to be PESS due to their greater exposure. EPA applied the methodology presented in the *Draft TSCA Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities*[6] to evaluate risks from industrial air releases to fenceline communities. In addition to considering risks from individual facilities, EPA evaluated risks from aggregate exposures in cases where multiple facilities reporting 1,4-dioxane releases to air were in proximity. The Agency also evaluated potential risks to fenceline communities from air emissions of 1,4-dioxane modeled for hydraulic fracturing operations and industrial and commercial laundries.

1.3.1.3.3 Aggregate Exposure

EPA has defined aggregate exposure as “the combined exposures to an individual from a single chemical substance across multiple routes and across multiple pathways (40 CFR § 702.33).” In this draft supplement, EPA considered the combined 1,4-dioxane exposure an individual may experience due to releases to air or water from multiple sources. For general population drinking water exposure scenarios, EPA evaluated combined exposure and risks from multiple sources of 1,4-dioxane in surface water, including direct and indirect industrial releases, DTD releases, and upstream background contamination (Section 5.2.2.1). For general population air exposure scenarios, EPA evaluated combined exposure and risk across multiple facilities in proximity releasing to air (Section 5.2.2.3 and Appendix J.4). EPA qualitatively considered aggregate exposures across exposure routes (*i.e.*, across oral and inhalation) and across exposure pathways (*i.e.*, across air and water) but did not quantitatively aggregate these exposures due to uncertainties around the additivity of effects across routes. The rationale for the scope of aggregate analysis in this draft supplement and remaining sources of uncertainty are further discussed in Section 5.2.2.5.

1.3.2 Potentially Exposed or Susceptible Subpopulations

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

Considerations related to PESS can influence the selection of relevant exposure pathways, the sensitivity of derived hazard values, the inclusion of particular populations, and the discussion of uncertainties throughout the assessment. Factors that may contribute to increased exposure or biological susceptibility to a chemical include lifestage, pre-existing disease, lifestyle activities (*e.g.*, smoking, physical activity), occupational and consumer exposures (including workers and occupational non-users, consumers and

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[6] The draft *Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities Version 1.0* is available at https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/tsca-screening-level-approach-assessing-ambient-air-and.

bystanders), geographic factors (*e.g.*, fenceline communities), socio-demographic factors, nutrition, genetics, unique activities (*e.g.*, subsistence fishing), aggregate exposures, and other chemical and non-chemical stressors.

This draft supplement considers PESS throughout the human health exposure assessment and risk characterization. The hazard assessment and dose-response analysis used in this draft supplement incorporate all PESS considerations described previously in the 2020 RE. Section 5.2.2.4 provides a summary of how specific factors contributing to exposure and susceptibility were addressed in this assessment and identifies remaining sources of uncertainty for PESS.

1.4 Systematic Review

EPA used the TSCA systematic review process described in the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances* (U.S. EPA, 2021a) (hereinafter referred to as "2021 Draft Systematic Review Protocol") to identify information needed to evaluate additional COUs and exposure pathways considered in this draft supplement. Appendix C provides additional information on the literature search strategy, data screening, evaluation, extraction, and evidence integration steps performed in support of this assessment—including clarifications and updates made to the 2021 Draft Systematic Review Protocol to better address assessment needs for this draft supplement.

1.5 Document Outline

This draft supplement to the risk evaluation for 1,4-dioxane comprises the following sections and appendices:

- Section 1 presents information on the scope of the draft supplement. It also includes an overview of the systematic review process used in this analysis. Appendix A provides a list of abbreviations and acronyms used throughout this report while Appendix B provides the full name and links to all supplemental documents associated with this draft risk evaluation. A more detailed description of the systematic review protocol for this assessment is presented in Appendix C while Appendix D provides a crosswalk of COUs with occupational exposure scenarios.
- Section 2 presents an overview of releases and concentrations of 1,4-dioxane in the environment. A more detailed description of the industrial and commercial environmental release assessment is presented in Appendix E. Methods for estimating environmental concentrations of 1,4-dioxane are described in more detail in Appendix G (surface water), Appendix H (groundwater), and Appendix J (air).
- Section 3 presents the human exposure assessment for occupational and general population exposure pathways. Details of the occupational exposure assessment are presented in Appendix F and details of the general population exposure assessment are presented in Appendix I and Appendix J.
- Section 4 provides a summary of the human health hazard and dose-response assessment previously published in the 2020 RE and describes duration adjustments made for the current analysis.
- Section 5 presents risk characterization based on the conditions of use and exposure pathways evaluated in this draft supplement. Details of risk calculations and risk estimates are provided in the set of supplemental risk calculator files listed in Appendix B. Section 5 also includes a discussion of PESS based on both greater exposure and susceptibility, a description of aggregate and sentinel exposures, and a discussion of assumptions and uncertainties and the impact on the supplemental risk evaluation.

Figure 1-5 provides an overview of how the analyses presented in each section are integrated into risk characterization.

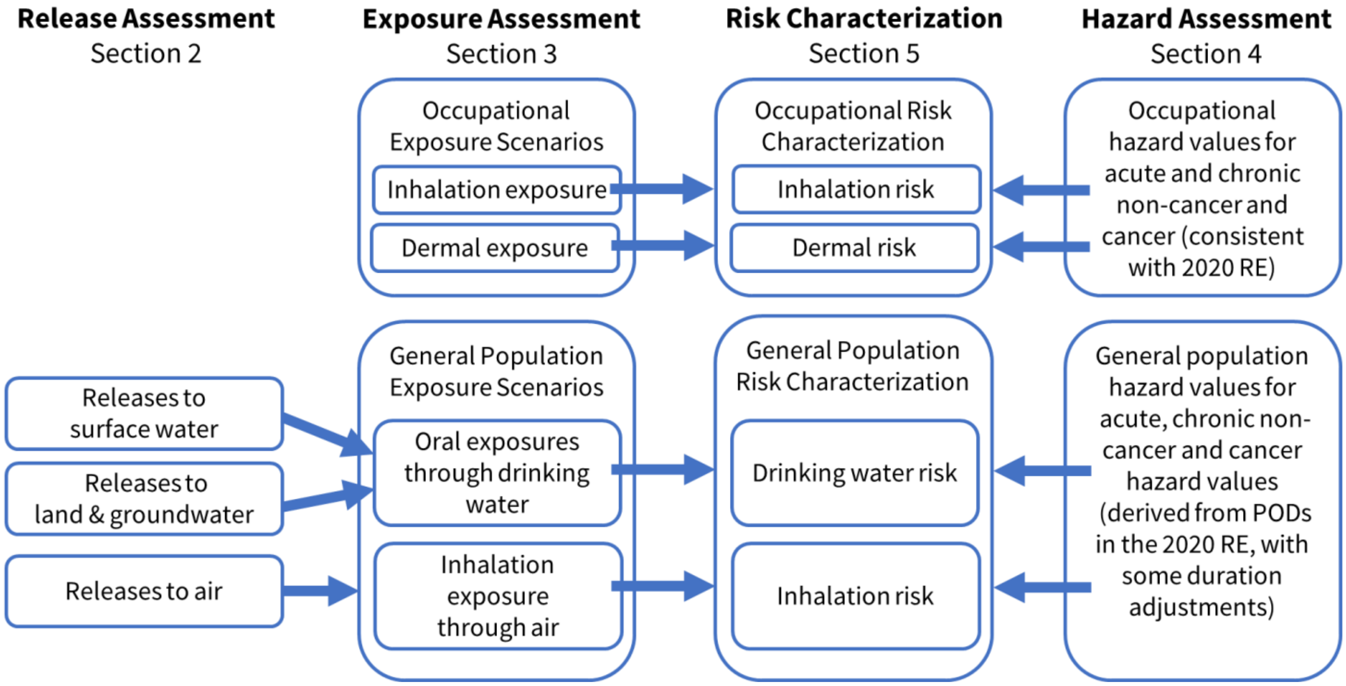

Figure 1-5. Overview of Analyses Included in this Draft Supplement to the Risk Evaluation for 1,4-Dioxane

2 RELEASES AND CONCENTRATIONS

2.1 Approach and Methodology

2.1.1 Industrial and Commercial Releases

Releases to the environment are one component of potential exposure and may be derived from reported data that are obtained through direct measurement via monitoring, calculations based on empirical data, and/or assumptions and models.

The original COUs for 1,4-dioxane are summarized in Table 1-4 of the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). Additional COUs included in this draft supplement due to 1,4-dioxane produced as a byproduct are presented in Table 2-1. For general population exposures, this draft supplement considers releases from all COUs (including the original COUs included in the 2020 RE and the additional COUs associated with 1,4-dioxane produced as a byproduct). For occupational exposures, this draft supplement focuses on the additional COUs associated with 1,4-dioxane produced as a byproduct. For additional information and context on the inclusion of these COUs in the draft supplement, refer to Sections 1.1 and 1.2. For the full table of COUs, including those previously assessed in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), see Appendix D.

A COU is a combination of life cycle stage, category, and subcategory, as shown in Table 2-1. The COU subcategory is the most granular description of the use. EPA mapped each COU to an occupational exposure scenario (OES). The purpose of an OES is to group, where appropriate, COUs based on similarity of the operations and data availability for each COU. For each OES, EPA estimated air, land, and water releases and occupational dermal and inhalation exposures. EPA mapped OESs to COUs using professional judgment based on reasonably available data and information that describe how releases and exposures take place within an occupational COU. EPA may group multiple COUs into an OES if the release and exposure potential is similar across the COUs and there is insufficient data to differentiate the COUs. This grouping minimized repetitive assessments. Alternatively, EPA may assign multiple OESs to one COU if there are several ways in which release and exposure takes place for the given COU and sufficient data exist to separately assess the OES. Appendix D.1 shows mapping between COUs and OESs. A crosswalk of the COUs with the OESs assessed is provided in Table 2-1.

As shown in Table 2-1, most COU life cycle stage, category, and subcategory combinations map to a single OES with a similar or identical name to the COU subcategory. However, for the COU subcategory of dish soap, dishwasher detergent, and laundry detergent, EPA assigned four OESs: (1) dish soap, (2) dishwasher detergent, (3) laundry detergent (industrial), and (4) laundry detergent (institutional). Institutional use of laundry detergent equates to commercial use.

EPA assessed environmental releases (air, water, and land) and occupational exposures (inhalation and dermal) to 1,4-dioxane for each of the OESs listed in Table_Apx D-1. EPA used the environmental release estimates for each OES for subsequent environmental concentrations and general population exposure calculations.

Table 2-1. Additional Categories and Subcategories of COUs and Associated OESs Included in the Scope of the Draft Supplement Due to the Presence of 1,4-Dioxane Produced as a Byproduct[a]

Condition of Use			OES Mapped to COU
Life Cycle Stage	**Category[b]**	**Subcategory[c]**	
Processing	Byproduct	Byproduct produced during the ethoxylation processes	Ethoxylation Process Byproduct
		Byproduct produced during the production of polyethylene terephthalate	Polyethylene Terephthalate (PET) Byproduct
Industrial use, commercial use	Other uses	Hydraulic fracturing	Hydraulic Fracturing
Consumer use, commercial use	Paints and coatings	Latex Wall Paint or Floor Lacquer	Paint and Floor Lacquer
	Cleaning and furniture care products	Surface Cleaner	Surface Cleaner
	Laundry and dishwashing products	Dish soap Dishwasher detergent Laundry detergent	Dish Soap Dishwasher Detergent Laundry Detergent (Industrial)[d] Laundry Detergent (Institutional)[d]
	Arts, crafts, and hobby materials	Textile dye	Textile Dye
Consumer use, commercial use	Automotive care products	Antifreeze	Antifreeze
Disposal	Disposal	Industrial pre-treatment	Disposal
		Industrial wastewater treatment	
		Publicly owned treatment works (POTW)	
		Underground injection	
		Municipal landfill	
		Hazardous landfill	
		Other land disposal	
		Municipal waste incinerator	
		Hazardous waste incinerator	
		Off-site waste transfer	

[a] New COUs and associated OESs where 1,4-dioxane is produced as a byproduct.
[b] These categories of COU reflect CDR codes and broadly represent conditions of use for 1,4-dioxane in industrial and/or commercial settings.
[c] These subcategories reflect more specific uses of 1,4-dioxane.
[d] Laundry detergent use may occur in industrial, commercial, or consumer settings. Sufficient information was available to separately assess each use setting. Thus, there are two OESs—one for industrial and one for institutional, which equates to commercial use.

2.1.1.1 General Approach and Methodology for Environmental Releases

Data reported to the Toxics Release Inventory (TRI)[7] and discharge monitoring reports (DMRs) are the primary sources of release data that EPA used for the release assessments. Under the Emergency Planning and Community Right-to-Know Act (EPCRA) section 313, 1,4-dioxane has been a TRI-reportable substance since 1987. The TRI database includes information on disposal and other releases of 1,4-dioxane to air, water, and land, in addition to how it is being managed through recycling, treatment, and burning for energy recovery. Under the Clean Water Act (CWA), surface water discharges reported in DMRs are based on required monitoring as part of a facility's National Pollutant Discharge Elimination System (NPDES) permit.

For each OES, a daily air, land, and water release was estimated based on annual releases, release days, and the number of facilities (Figure 2-1). The blue boxes represent primary sources of release data that are used to develop annual releases, release days, and number of facilities. The information in the green boxes is aggregated by OES to provide a daily release estimate.

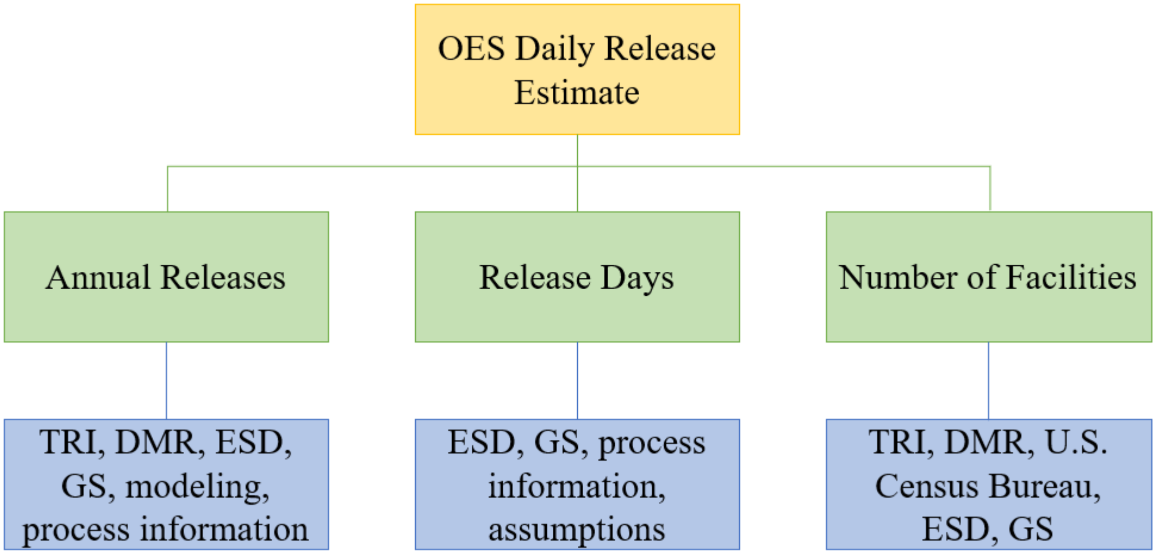

Figure 2-1. Overview of EPA's Approach to Estimate Daily Releases for Each OES

TRI = Toxics Release Inventory; DMR = discharge monitoring report; ESD = emission scenario document; GS = generic scenario

2.1.1.2 Water Release Estimates

EPA followed a similar approach for estimating industrial and commercial water releases as it did in the 2020 RE, with one key difference. Here, EPA evaluated multiple years of data using data from 2013 to 2019 TRI (U.S. EPA, 2022g) and 2013 to 2019 DMR (U.S. EPA, 2022c), as opposed to utilizing 1 year of data.

Where water releases are expected for an OES but TRI and DMR data were not available, EPA estimated industrial, and commercial water releases using two approaches. If available, Stochastic Human Exposure and Dose Simulation for High Throughput (SHEDS-HT) DTD modeling was used. SHEDS-HT predicts a per capita DTD loading of 1,4-dioxane, which is combined with an estimation of the population contributing to publicly owned treatment works (POTW) effluent on the modeled water

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[7] TRI page: https://www.epa.gov/toxics-releasE−inventory-tri-program.

body stream to produce an estimated DTD loading. If SHEDS-HT DTD modeling was not available for an OES without TRI or DMR data, EPA used data from literature, ESDs, and GSs in conjunction with Monte Carlo simulation where sufficient data were available to vary calculation input parameters to estimate industrial and commercial water releases (see Appendix E.3 for additional information).

For the following OESs, EPA either could not estimate water releases due to lack of reasonably available data or information or did not expect water releases based on volatility and use patterns:

- **Functional Fluids (Closed-Systems).** Water release data were not available for this OES. However, EPA expects that the sources of release for this OES to be similar to those for the Industrial Uses OES (per process information in the 2020 RE, Appendix G.6.4). Therefore, EPA grouped the water release assessment for Functional Fluids (Closed-Systems) into that for Industrial Uses. However, there is uncertainty in this assumption of similar release sources between these OESs.
- **Laboratory Chemical, Film Cement, and Dry Film Lubricant.** Wastewater discharges containing 1,4-dioxane were not expected for these OESs; releases from these OESs are expected to be to air from volatilizations and landfill/incineration from disposal of empty containers and other waste (2020 RE, Appendix G).
- **Antifreeze.** Wastewater discharges containing 1,4-dioxane were not expected for this OES; releases from this OES are expected to be to air from volatilizations during antifreeze changeouts and to landfill/incineration from disposal of empty antifreeze containers and spent antifreeze.
- **Paints and Floor Lacquer.** Wastewater discharges containing 1,4-dioxane were not expected for this OES; releases from this OES are expected to be to air from volatilizations during painting/drying and to landfill/incineration from disposal of empty paint containers, used paint brushes/rollers, or solvent washes of paint brushes/rollers.

2.1.1.3 Land Release Estimates

EPA used data from 2019 TRI (U.S. EPA, 2022g) to estimate industrial and commercial land releases that were mapped to each OES with the exception of the Disposal OES. For that OES, EPA performed a more detailed analysis using data from 2013 to 2019 TRI (U.S. EPA, 2022g). Where land releases are expected for an OES, but TRI data were not available, releases were estimated using reasonably available data from literature, ESDs, and GSs in conjunction with Monte Carlo simulation (Palisade, 2022a) to allow for variability in calculation input parameters where sufficient data were available to inform such variability.

EPA did not estimate daily land releases due to the high level of uncertainty in the number of release days. This uncertainty is because facility operating days does not correlate directly to releases. For example, a facility may wait until a dumpster or other waste receptacle is full before disposing of it. Because these releases may occur on a daily, weekly, and even monthly scale, EPA used the annual land releases reported in TRI data or modeled without estimating land releases for a different frequency (daily, weekly, monthly). See Appendix E.4 for additional information. Annual land release estimates were used to estimate potential groundwater contamination from landfill releases as described in Section 2.3.2.

For the following OESs, EPA was not able to estimate land releases due to lack of data or information or did not expect land releases due to physical form, use patterns, and lack of data:

- **Manufacturing, Import and Repackaging, and Functional Fluids (Open-Systems).** Data from 2019 TRI (U.S. EPA, 2022g) indicated that there were no releases of 1,4-dioxane to land from facilities that EPA mapped to these OESs. EPA did not have additional reasonably available information to estimate land releases from these OESs.

- **Functional Fluids (Closed-Systems).** See previous explanation in the "Water Release Estimates" section above.
- **3D Printing.** Industrial applications of this OES are expected to be accounted for in the Industrial Uses TRI data. Per Appendix G.6.8 of the 2020 RE, 3D printing ink containing 1,4-dioxane is used in research labs to print biomedical products. Because the 2019 TRI data for the Industrial Uses OES include medicinal and pharmaceutical manufacturing NAICS codes, medical research labs that conduct 3D printing with 1,4-dioxane inks may be captured in that OES. Therefore, EPA grouped the land release assessment for 3D Printing into that OES for Industrial Uses. However, the extent to which all potential 3D printing sites that use 1,4-dioxane are captured in the Industrial Uses TRI data is unknown.

EPA also notes that the Hydraulic Fracturing OES is associated with certain specific land releases that may not apply to other OESs, such as the releases of wastewater containing 1,4-dioxane to deep well injection or surface impoundments, which are considered land releases in this assessment. The Agency estimated these deep well injection and surface impoundment releases, which were used in addition to landfill releases, to estimate potential groundwater contamination from hydraulic fracturing described in Section 2.3.2.

2.1.1.4 Air Release Estimates

EPA applied the following tiered approach to developing air release, exposure, and risk estimates:

1. Pre-screening analysis,
2. Single-year fenceline analysis, and
3. Multi-year fenceline analysis.

2.1.1.4.1 Pre-screening Analysis

This analysis is described in the *Draft TSCA Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities* and consisted of extracting data for all facilities reporting 1,4-dioxane air releases to the 2019 TRI (U.S. EPA, 2022g). The extracted data were reviewed to identify the maximum single facility release reported across all reporting facilities. Additionally, the arithmetic average (mean) value of all reported releases was calculated. These two release values were used for further analysis to estimate exposure concentrations at select distances from the releasing facility.

2.1.1.4.2 Single-Year Fenceline Analysis

Where available, EPA used data from 2019 TRI to estimate industrial and commercial air releases in accordance with the *Draft TSCA Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities*. Facilities are only required to report to TRI if the facility has 10 or more full-time employees; is included in an applicable North American Industry Classification System (NAICS) code; and manufactures, processes, or uses the chemical in quantities greater than a certain threshold. Due to these limitations, some sites that manufacture, process, or use 1,4-dioxane may not report to TRI and are therefore not included in these datasets.

Where air releases are expected for an OES, but TRI data were not available, industrial, and commercial air releases were estimated using data from literature, ESDs, and GSs in conjunction with Monte Carlo simulation (Palisade, 2022a) to allow for variability in calculation input parameters where sufficient data were available to inform such variability. See Appendix E.5 for additional information.

For the following OESs, EPA was not able to estimate air releases due to lack of data or information:

- **Functional Fluids (Closed-Systems).** See previous explanation in the "Water Release Estimates" section above.
- **3D Printing.** Industrial applications of this OES are expected to be accounted for in the Industrial Uses TRI data. Per Appendix G.6.8 of the 2020 RE, 3D printing ink containing 1,4-dioxane is used in research labs to print biomedical products. Because the 2019 TRI data for the Industrial Uses OES include medicinal and pharmaceutical manufacturing NAICS codes, medical research labs that conduct 3D printing with 1,4-dioxane inks may be captured in that OES. Therefore, EPA grouped the air release assessment for 3D Printing into that OES for Industrial Uses. However, the extent to which all potential 3D printing sites that use 1,4-dioxane are captured in the Industrial Uses TRI data is unknown.
- **Textile Dyes, Dish Soap, and Dishwasher Detergent.** EPA did not find relevant reasonably available 1,4-dioxane or surrogate TRI data, literature sources, sufficient process information, nor ESD or GS with air release estimation approaches to estimate air releases for these OESs. Therefore, EPA was not able to estimate air releases for these OESs.

2.1.1.4.3 Multi-Year Analysis

The multi-year analysis incorporates Science Advisory Committee on Chemicals (SACC)[8] recommendations on the *Draft TSCA Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities* to evaluate multiple years of chemical release data to estimate exposures and associated risks to fenceline communities. This is achieved by conducting a facility-by-facility evaluation of all 1,4-dioxane releases reported to TRI from 2015 through 2020. Data for these 6 years were obtained from the TRI database (TRI basic plus files downloaded on August 5, 2022). Annual release data for 1,4-dioxane were extracted from the entire TRI data set for all facilities reporting air releases of 1,4-dioxane for one or more years between 2015 and 2020. Facilities were categorized into occupational exposure scenarios for exposure modeling purposes and later cross-walked to COUs for risk management purposes.

2.2 Environmental Releases

2.2.1 Industrial and Commercial Releases

This section summarizes the estimated air, water, and land releases for each OES; the weight of the scientific evidence conclusions for these estimates; and the strengths, limitations, assumptions, and key sources of uncertainty for these estimates.

2.2.1.1 Release Estimates Summary

EPA estimated air, water, and land releases of 1,4-dioxane using various methods and information sources—including TRI and DMR data, GSs and ESDs modeling with and without Monte Carlo, process information, and SHEDS-HT DTD Modeling. EPA combined its estimates for annual releases, release days, and number of facilities to estimate a range of daily releases for all OESs, including those presented in the December 2020 RE. The COUs associated with each OES are summarized in Table_Apx D-1. A summary of these industrial and commercial releases for air, water, and land are presented in Table_Apx E-3, Table_Apx E-5, and Table_Apx E-7, respectively. These release estimates are for total releases from a facility and may include multiple points of release, such as multiple outfalls for discharges to surface water or multiple point sources for air emissions. Note that for some release

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[8] Additional information about SACC is available at https://www.epa.gov/tsca-peer-review/science-advisory-committee-chemicals-basic-information.

estimates, there is uncertainty and variability in the potential media of release. In such cases, EPA did not have sufficient information to partition the release estimates between all potential media of release and they are replicated between the air, land, and water subsections if there is overlap in the potential media of release.

EPA mapped these releases by media, state, and tribal territory for the conterminous United States. Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not mapped since no estimated releases are known.

Surface water releases as reported by TRI and DMR are presented in Figure 2-2. For surface water releases, the data are divided based on the source of data (*i.e.*, DMR, TRI) and whether the release is from a direct (on-site) source, including on-site wastewater treatment systems, or indirect (offsite) source where the chemical substance was taken to a different location for potential release, such as a POTW. The largest releases have been from PET manufacturing in Tennessee (2,512,434 kg in 2019 and 15,168 kg in 2018), Alabama (170,526 kg in 2015; 125,903 kg in 2014; and 111,924 kg in 2017), and West Virginia (14,134 kg in 2016 and 12,229 kg in 2014).

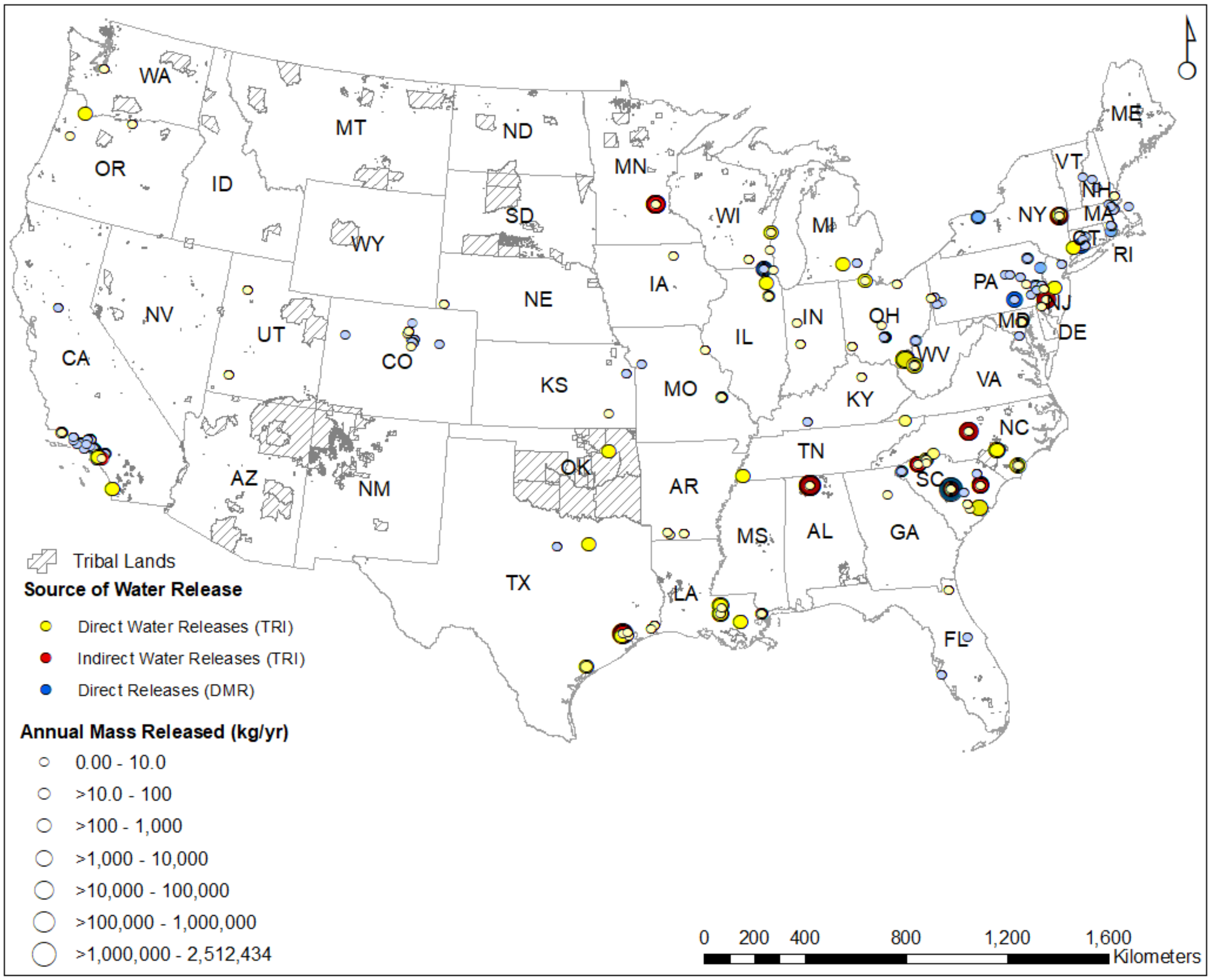

Figure 2-2. 1,4-Dioxane Annual Water Releases as Reported to TRI and DMR, 2013–2019

Note: Some symbols for individual years may overlap and obscure annual releases at each site.

Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown due to no known modeled or estimated releases.

Land disposals as reported by TRI are available in Figure 2-3. The largest disposals have been to on-site Class I Underground Injection Wells in Texas (169,035 kg in 2013; 42,865 kg in 2015; 10,729 kg in 2018), On-site Subtitle C Landfills in Oregon (7,321 kg in 2014; 7,000 kg in 2013; and 6,076 kg in 2015), and Offsite Other Landfills in Indiana (862 kg in 2019; 603 kg in 2018; and 354 kg in 2017). Air release as reported by TRI are available in Figure 2-4.

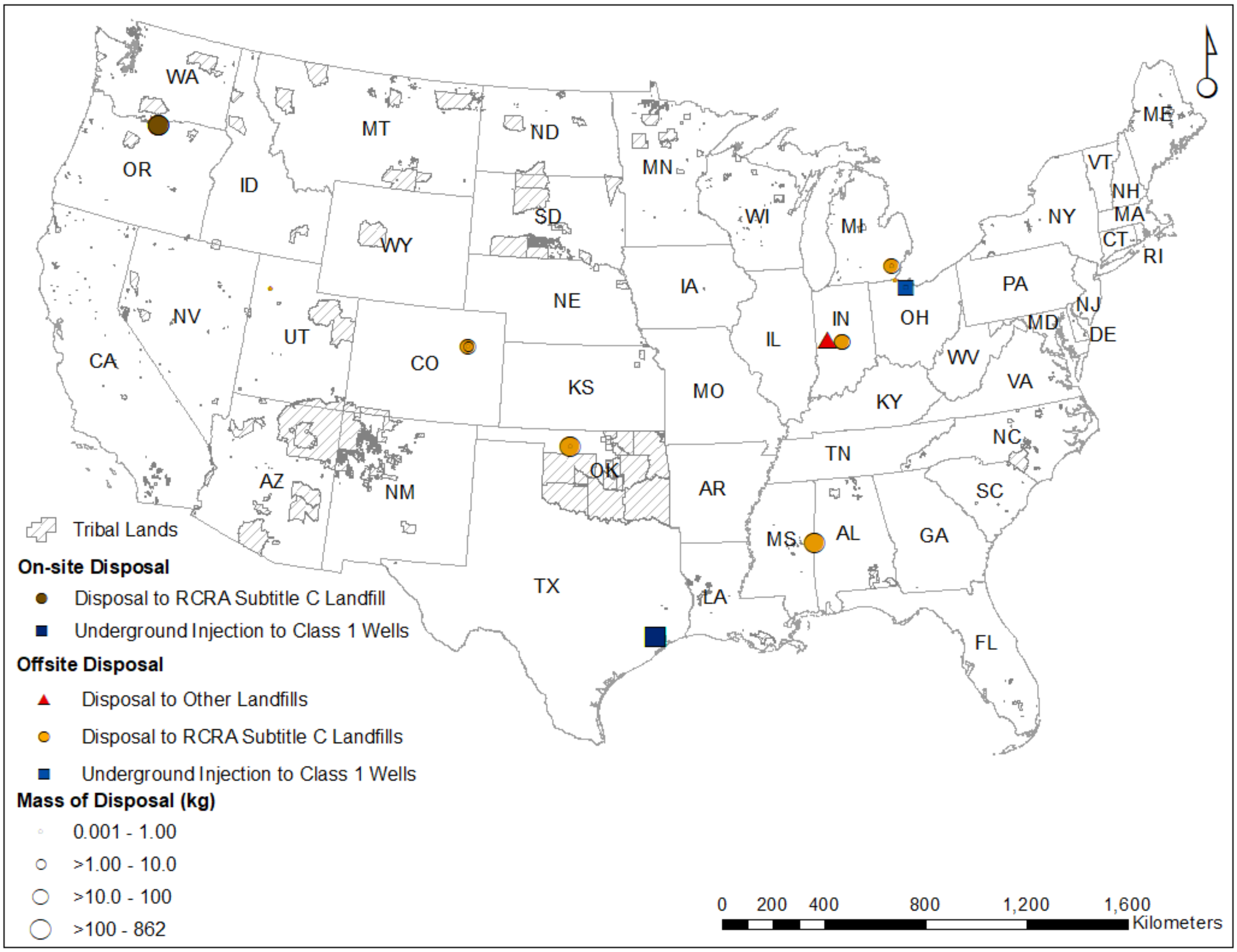

Figure 2-3. 1,4-Dioxane Annual Releases to Land as Reported to TRI, 2013–2019

Note: Some symbols for individual years may overlap and obscure annual releases at each site. Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown as there are no known modeled or estimated releases.

For air releases, the largest emissions have been in Illinois (9,943 kg/year), South Carolina (3,495 kg/year), and Texas (2,097 kg/year). Collectively, these figures give insight into the spatial distribution of releases and corresponding amount across the contiguous United States. A full summary of these estimates can be found in Appendix E.

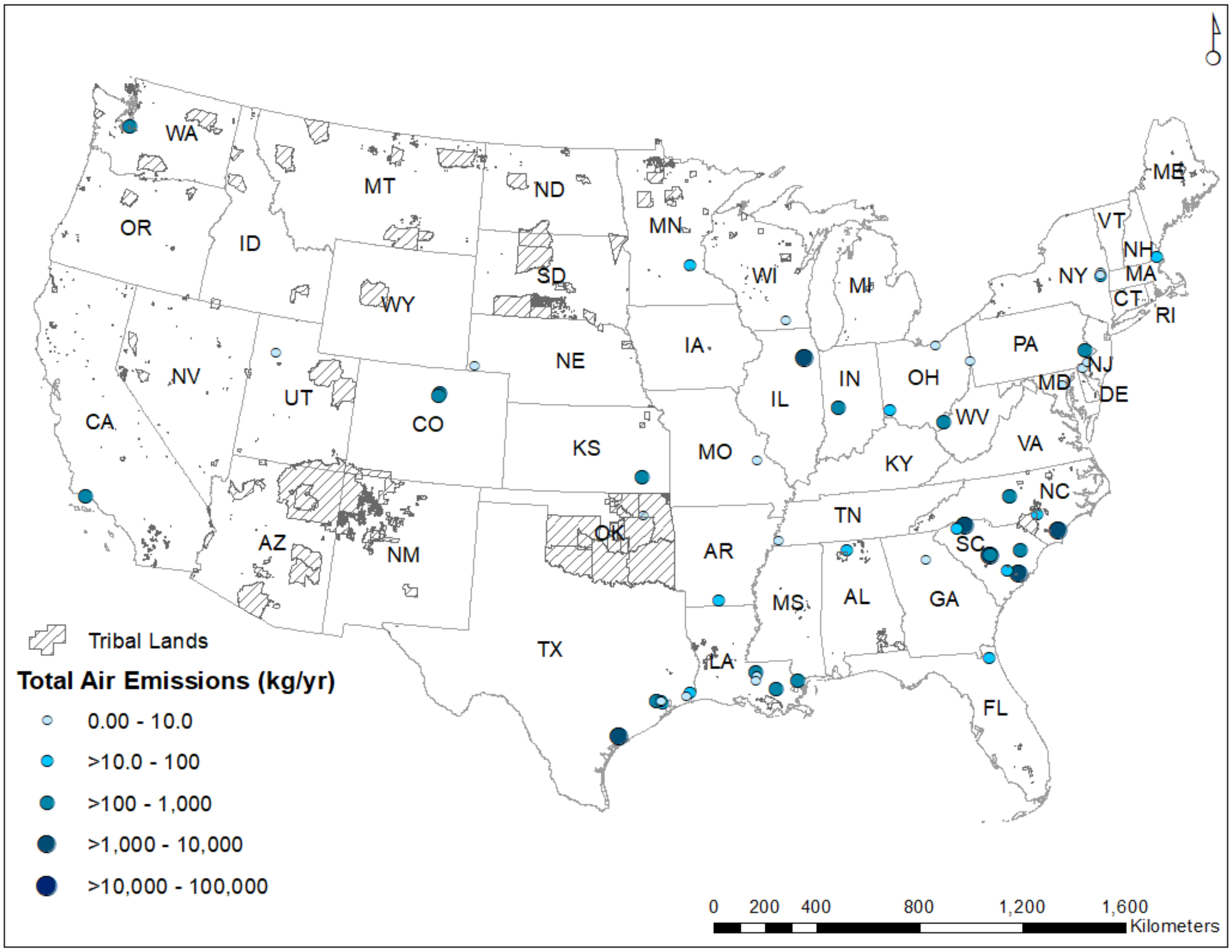

Figure 2-4. 1,4-Dioxane Annual Releases to Air as Reported by TRI, 2013–2019

Note: Some symbols for individual years may overlap and obscure annual releases at each site.
Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown as there are no known modeled or estimated releases.

In addition to mapping releases from TRI and DMR, EPA also mapped hydraulic fracturing sites reporting the presence of 1,4-dioxane in hydraulic fracturing operation fluids according to FracFocus 3.0 (GWPC and IOGCC, 2022). These operations are primary sited in a wide range of shale plays across the country (as indicated by the multi-colored plays mapped in Figure 2-5). The Delaware play in Texas has the largest number of operations (n = 158) followed by the Niobrara in Colorado (n = 86) and the Utica play that spreads across Pennsylvania and Ohio (n = 70).

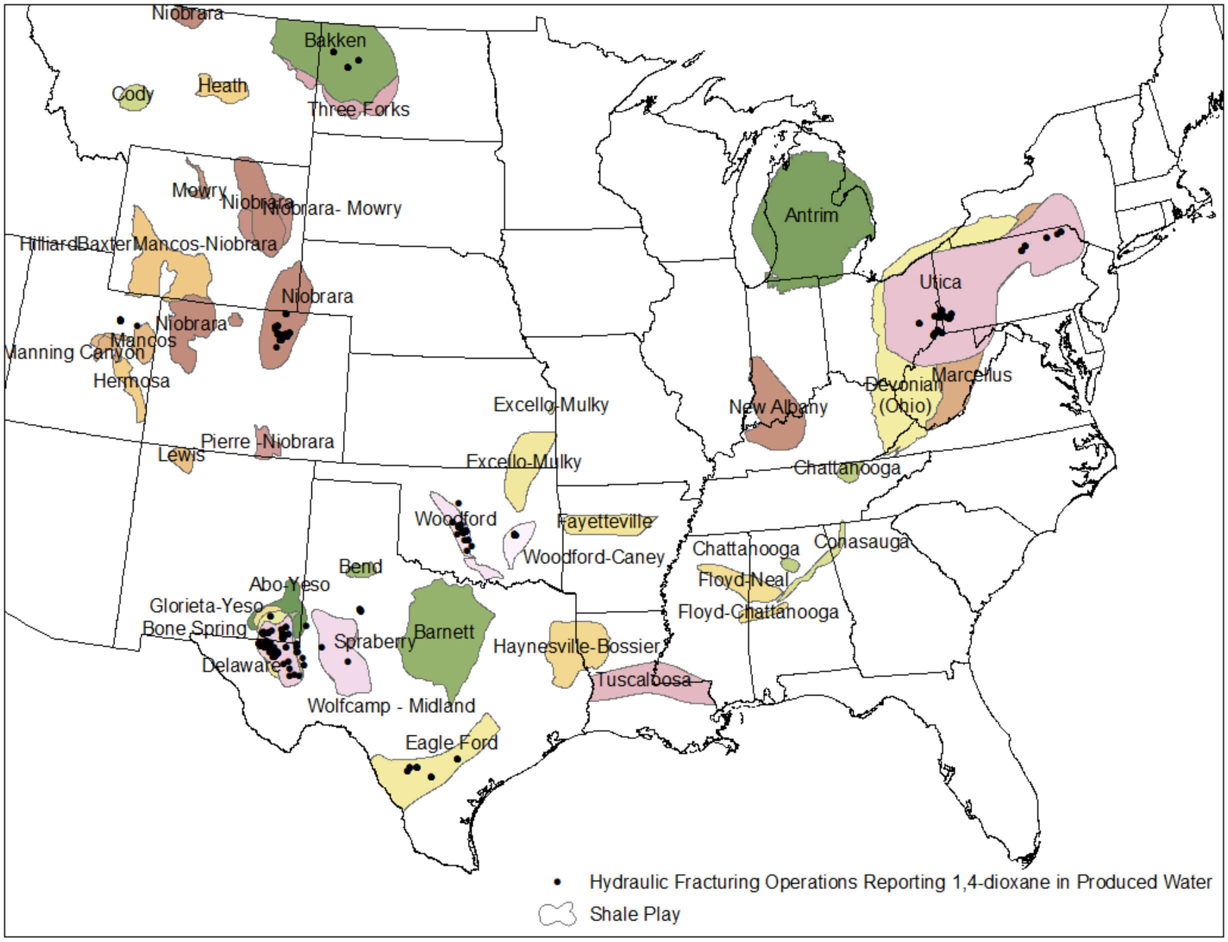

Figure 2-5. Locations of Hydraulic Fracturing Operations that Report 1,4-Dioxane in Produced Waters

Note: Some symbols for individual years may overlap and obscure annual releases at each site.
Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown as there are no known modeled or estimated releases.

The basis for overall data quality determinations is also included in the water, air, and land summary subsections below. Each source is evaluated on multiple metrics based on defined criteria. For air, water, and land releases, all monitoring data had data quality ratings of medium/high. Modeled data had data quality ratings of medium or high.

2.2.1.2 Weight of the Scientific Evidence Conclusions for Environmental Releases

EPA's judgment on the weight of the scientific evidence is based on the strengths, limitations, and uncertainties associated with the release estimates. The Agency considers factors that increase or decrease the strength of the evidence supporting the release estimate—including quality of the data/information, applicability of the release data to the COU (including considerations of temporal relevance, locational relevance) and the representativeness of the estimate for the whole industry. The best professional judgment is summarized using the descriptors of robust, moderate, slight, or indeterminant, according to EPA's *Application of Systematic Review in TSCA Risk Evaluations* (U.S. EPA, 2018c). For example, a conclusion of moderate weight of the scientific evidence is appropriate where there is measured release data from a limited number of sources such that there is a limited number of data points that may not cover most or all of the sites within the COU. A conclusion of slight weight of the scientific evidence is appropriate where there is limited information that does not sufficiently cover all sites within the COU, and the assumptions and uncertainties are not fully known or documented. See EPA's *Application of Systematic Review in TSCA Risk Evaluations* (U.S. EPA, 2018c) for additional information on weight of the scientific evidence conclusions.

For air, water, and land releases, all monitoring data had data quality ratings of medium/high. Modeled data had data quality ratings of medium or high. For supplemental releases modeled with TRI/DMR (PET Byproduct, Ethoxylation Byproduct, Disposal), the weight of the scientific evidence conclusion was moderate to robust because the reasonably available information relevant for the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited. For releases that used SHEDS-HT modeling (Surface Cleaner, Dish Soap, Dishwasher Detergent), the weight of the scientific conclusion was slight since there is uncertainty in the application of this modeling for a commercial setting, and this case study does not represent all sites in this OES. For supplemental releases that used GS/ESDs, the weight of the scientific conclusion was moderate when used in tandem with Monte Carlo modeling (Textile Dye, Laundries), and slight/moderate when used alone (Antifreeze, Paint and Floor Lacquer). For Hydraulic Fracturing, the weight of the scientific conclusion was moderate to robust since FracFocus 3.0, an ESD, and Monte Carlo modeling were used. See Appendix E.8 for a summary of EPA's overall weight of the scientific evidence conclusions for its release estimates for each of the assessed OESs.

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

EPA estimated air, water, and land releases of 1,4-dioxane using various methods and information sources, including TRI and DMR data, GSs and ESDs modeling with and without Monte Carlo, process information, and SHEDS-HT DTD Modeling.

TRI and DMR were determined to have the overall data quality determination of medium through EPA's systematic review process. Uncertainties for using TRI and DMR data are discussed in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). In summary, these uncertainties include underestimation of the number of sites for a given OES due to reporting thresholds in TRI, the accuracy of EPA's mapping of sites reporting to TRI and DMR to a specific OES, and quality of the data reported to TRI and DMR. There is additional uncertainty for sites that report to TRI with Form A because these sites do not report release quantities if the quantity did not exceed 500 lb for the total annual reportable release amount. For these sites, EPA assessed a "what-if" scenario, which assumes the entire 500 lb going to single media of release, noting that the 500 lb should not be added over all release media. Additional information on TRI uncertainties is provided in Appendix E.5. In addition, as discussed in Section 2.2.1, EPA used data from the 2019 reporting year to estimate air and land releases. A key source of uncertainty in the assessment of air and land releases is whether 2019 TRI data is representative of releases from other reporting years. This does not apply to the water release estimates because EPA used data from reporting years 2013 to 2019. A strength of using TRI is that it compiles the best readily available release data for all facilities that reported to EPA. However, not all facilities are required to report to TRI.

Some uncertainties of using DMR data include the accuracy of EPA's mapping of sites reporting to DMR to a specific OES, and quality of the data reported to DMR. Also, an uncertainty of using the ECHO Pollutant Loading Tool Advanced Search option is that average measurements may be reported as a quantity (kg/day) or a concentration (mg/L). Calculating annual loads from concentrations requires adding wastewater flow to the equation, which increases the uncertainty of the calculated annual load. In addition, for facilities that reported having zero pollutant loads to DMR, the EZ Search Load Module uses a combination of setting non-detects equal to zero and as one-half the detection limit to calculate the annual pollutant loadings. This method could cause overestimation or underestimation of annual and

Daily pollutant loads. A strength of using DMR data and the Pollutant Loading Tool is that the tool calculates an annual pollutant load by integrating monitoring period release reports provided to the EPA and extrapolating over the course of the year. However, this approach assumes average quantities, concentrations, and hydrologic flows for a given period are representative of other times of the year.

Where TRI and DMR data were not reasonably available, EPA used GS and ESDs. One uncertainty for this method is lack of specific 1,4-dioxane data. Because GS/ESDs are generic, assessed parameter values may not always be representative of applications specific to 1,4-dioxane use in each OES. Another uncertainty is lack of consideration for release controls. The GS/ESDs assume that all activities occur without any release controls, and in an open-system environment where vapor and particulates freely escape (U.S. EPA, 2022d; OECD, 2017, 2011a, b). Actual releases may be less than estimated if facilities utilize pollution control methods. Although 1,4-dioxane monitoring data are preferred to modeled data, EPA strengthened modeled estimates by using Monte Carlo modeling to allow for variation in environmental release calculation input parameters according to the GS/ESD and other literature sources. However, EPA did not utilize Monte Carlo modeling for all GS/ESD, which is a limitation of this assessment. Table_Apx E-8 includes information on which GS/ESDs were used in tandem with Monte Carlo modeling.

EPA used process information to quantify environmental releases for the film cement and dry film lubricant OESs. This process information is from the 2020 RE (U.S. EPA, 2020c) and the underlying sources were determined to have high overall data quality determinations through EPA’s systematic review process. To develop these release estimates, EPA made assumptions on the likely media of release for various releases sources and, in some cases, used standard EPA models in conjunction with process information to estimate the release quantity. A source of uncertainty in this approach is the representativeness of these estimates regarding all sites that use 1,4-dioxane for this OES.

Where no other data or information was reasonably available, EPA used SHEDS-HT down the drain (DTD) modeling to estimate commercial use environmental releases to surface water or land (Appendices E.3.2 and E.4.2). The results for this analysis are included in Table_Apx E-3. The main source of uncertainty is that the modeling EPA performed to estimate the total release amounts from each COU to surface water or land (via disposal to landfills) is based on information for SHEDS-HT in combination with information from a single case study location, Liverpool, OH. It is uncertain whether the release estimates generated from this case study are applicable to other areas of the country. EPA was unable to estimate the number of sites in Liverpool, OH, for the OES where this modeling approach was used; therefore, the release estimates were presented as totals for all sites as opposed to per-site estimates. To estimate land release, EPA used the modeled water releases from SHEDS-HT and back-calculating a 1,4-dioxane use rate based on the expected loss fraction to water for the OES. Finally, a land release loss fraction was applied to the back-calculated use rate. The uncertainty in this approach is due to the standard models and assumptions used to estimate loss fractions to water and land. Additionally, the same uncertainties listed above for the use of SHEDS-HT to estimate water releases are applicable to the approach for estimating land releases. EPA is unsure whether the use of SHEDS-HT results in a high-end or typical exposure scenario, so the use of this data may lead to over or underestimates of releases.

To assess daily air and water discharges, EPA divided annual release loads by the number of facility release days to estimate the daily release load for the facility. There is uncertainty if the assumed release duration is applicable to all sites for a given OES; therefore, the average daily releases may be higher if sites have fewer release days or lower if they have greater release days. Furthermore, 1,4-dioxane concentrations in air emissions and wastewater release to receiving water bodies at each facility may vary from day-to-day such that on any given day the actual daily releases may be higher or lower than the estimated average daily discharge. Thus, this approach minimizes variations in emissions and discharges from day to day. EPA did not estimate daily land releases due to the high level of uncertainty in the number of release days associated with land releases. The Agency expects that sites may not send waste to landfills every day and are more likely to accumulate waste for periodic shipments to landfills. However, sites that release to municipal landfills may have more frequent release days based on the frequency of shipments.

2.3 1,4-Dioxane Environmental Concentrations

2.3.1 Surface Water Pathway

Surface water contamination from 1,4-dioxane can occur from direct releases of wastewater from industrial operations, discharges from wastewater treatment plants containing DTD releases of 1,4-dioxane from consumer and commercial product usage (*i.e.*, dish soap, laundry detergent, etc.), and other activities where 1,4-dioxane may be present as a byproduct, such as in hydraulic fracturing operations. To understand possible exposure scenarios from these practices, EPA assessed exposures to the general population from ambient surface and drinking water. These exposures are due to 1,4-dioxane being directly or indirectly discharged to receiving water bodies.

The evaluation of these exposures considered both the review of reasonably available monitoring data to both ambient surface water and drinking water as well as the modeling of estimated exposures due to releases. Although EPA identified a robust set of surface and drinking water monitoring data (Section 2.3.1.1) indicating the presence of 1,4-dioxane in these pathways, it was collected independent of release data, and cannot be attributed to specific sources (Section 2.2). Therefore, EPA relied primarily on a series of modeling approaches to estimate concentrations of 1,4-dioxane in surface water near known release locations (Sections 2.3.1.2 and 2.3.1.3). For this assessment, EPA modeled concentrations resulting from industrial releases for all COUs releasing to surface water, including those assessed in the 2020 RE, as well as those producing 1,4-dioxane as a byproduct. To the degree possible, the relationship between monitoring and modeled data is further evaluated in Section 2.3.1.4.

2.3.1.1 Monitoring Data

Environmental concentration data for 1,4-dioxane in ambient surface water (*i.e.*, measured in rivers, streams, lakes, and ponds, rather than within industrial operations or drinking water systems) across the country, as well as routine monitoring conducted by public water systems (PWSs) of raw (untreated) source water and finished (treated) drinking water were collected from readily available public databases and publications. The methods for retrieving and processing ambient surface water and PWS data are described in detail in Appendix G.1.

Ambient Surface Water Monitoring

Data were retrieved from the Water Quality Portal (WQP) to characterize observed concentrations in ambient surface water (NWQMC, 2022). These monitored values may or may not represent locations used as a source for drinking water and are analyzed to characterize the observed ranges of 1,4-dioxane concentrations in ambient surface water—irrespective of the reasons for sample collection—and to provide context for the modeled surface water concentrations presented in Section 2.3.1.3. Data retrieved in July 2022 included sampling dates from 1997 to 2022 and resulted in 12,471 available sample results. Full details of the retrieval and processing of ambient surface water monitoring data from the WQP are presented in Appendix G. Table 2-2 shows the range of 1,4-dioxane concentrations detected in surface water samples. Most (*i.e.*, 92.3 percent) of the sample records available had no level of 1,4-dioxane detected above the reported detection limit for the analysis (referred to as “non-detects”), with limits of detection ranging from 0.001 to 28,000 µg/L across all samples. The 105 detected values ranged from 0.016 to 470 µg/L, with a median of 1.10 µg/L. Figure 2-6 and Figure 2-7 show the distribution of detected concentrations and reported detection limits of non-detect samples, respectively. The highest concentrations reported in this dataset are noted in the metadata to have been collected at the point of discharge from an industrial facility, while for most samples, the reason for sampling, or sample location in relation to expected releases is not included in the metadata. Figure 2-8 shows the spatial distribution of detected samples. For the entire dataset, including non-detects, approximately 70 percent of the samples were collected from the states of North Carolina, New Mexico, and New Jersey. Of the 105 detected values, 46 percent are located in Pennsylvania, 21 percent in North Carolina, and 14 percent in Illinois. In the absence of a national standardized study of 1,4-dioxane in ambient surface water (analogous to the UCMR monitoring in drinking water), and without more national coverage and metadata, it is difficult to characterize the national occurrence of 1,4-dioxane in surface water. It is apparent from the available monitoring data that certain areas may be more likely to have higher concentrations, while many others have little or no detected 1,4-dioxane. Over-representation of certain states or regions may reflect targeted sampling campaigns of specific locations expected to have higher concentrations, and conclusions about areas without monitoring data cannot be drawn without further exploration through modeling.

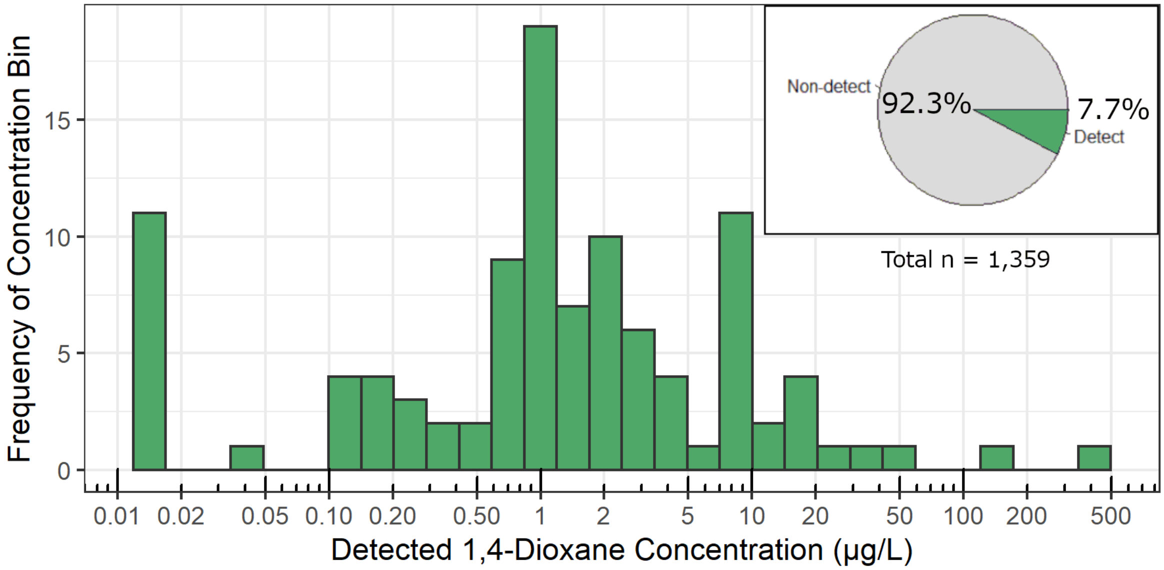

Figure 2-6. Frequency of Nationwide Measured 1,4-Dioxane Surface Water Concentrations Retrieved from the Water Quality Portal, 1997–2022

Note: Detectable levels of 1,4-dioxane may vary by location.

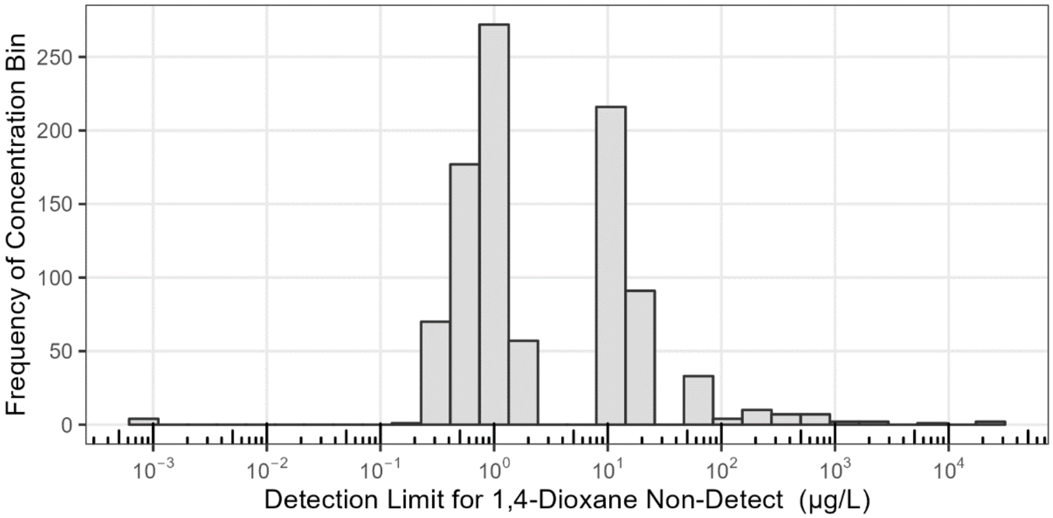

Figure 2-7. Frequency of Detection Limits for Nationwide Non-detect 1,4-Dioxane Surface Water Samples Retrieved from the Water Quality Portal, 1997–2022

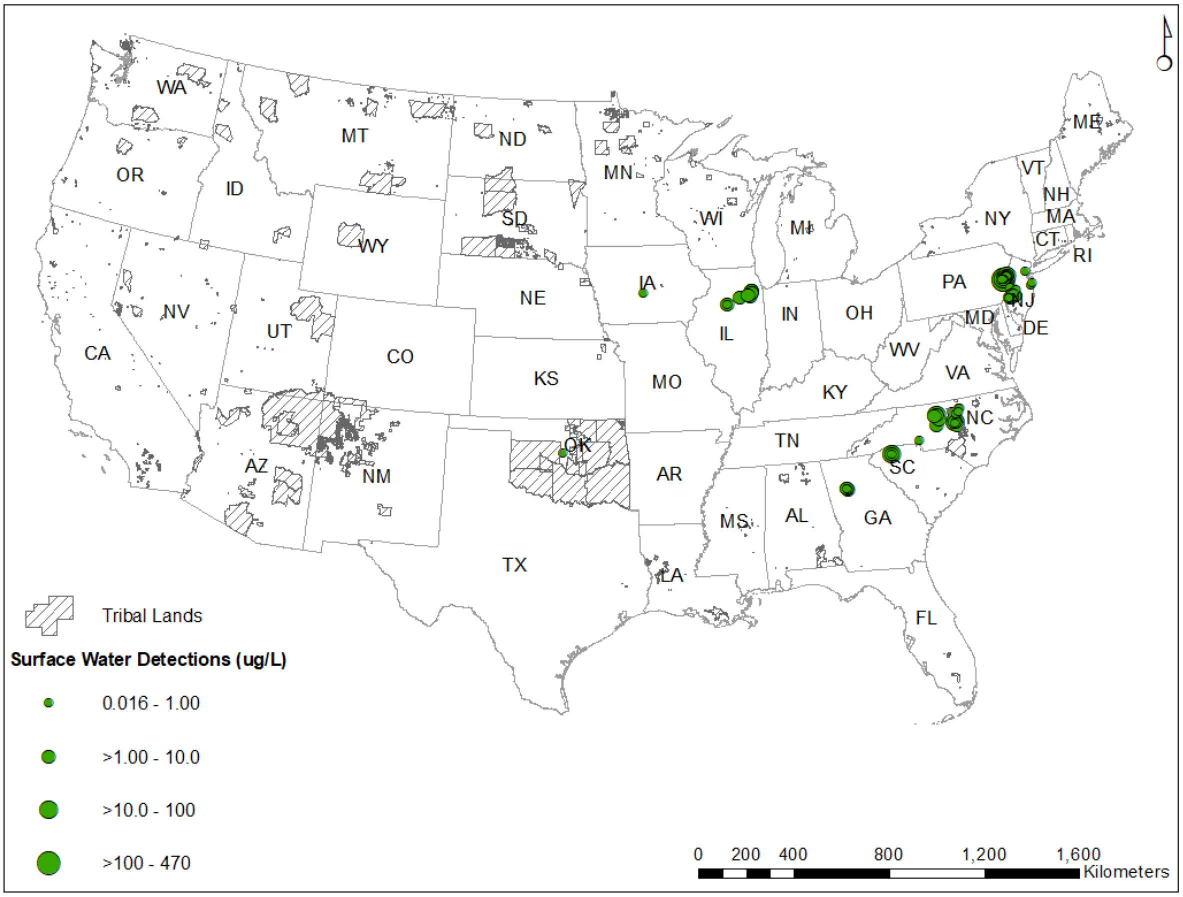

Figure 2-8. Detectable Concentrations of 1,4-Dioxane in Surface Water from the Water Quality Portal, 1997–2022

Note: Detectable levels of 1,4-dioxane may vary by sampling location.

Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown as there are no known monitoring data above detection limits.

Drinking Water Monitoring Data

The Safe Drinking Water Act (SDWA) authorizes the United States Environmental Protection Agency (US EPA) to set national health-based standards for drinking water to protect against both naturally occurring and man-made contaminants that may be found in drinking water. The National Primary Drinking Water Regulations (NPDWR) are legally enforceable primary standards and treatment techniques that apply to PWSs. Although states, tribes or territories that have been approved as the primary implementation authority for drinking water may require monitoring or impose limits for contaminants beyond those regulated under SDWA, there are not currently national requirements to routinely monitor or limit 1,4-dioxane in finished water from PWSs. In support of the SDWA, EPA often relies on data from the Unregulated Contaminant Monitoring Rule (UCMR) program as the best available occurrence information to support its regulatory determinations (*i.e.*, to judge whether a particular contaminant is known to occur or there is substantial likelihood the contaminant will occur in public water systems with a frequency and at levels of public health concern). UCMR monitoring is designed to produce a data set that is nationally representative of public water systems (PWSs) across the country, but its focus is on finished water (rather than source water) and it may not capture worst-case conditions. PWS monitoring data of finished drinking water were collected for 1,4-dioxane via EPA's published Third UCMR (UCMR3) dataset from 2013 to 2015, as well as raw and finished drinking water monitoring from additional individual state databases (CA, MA, and NY) from 2008 to 2022 (CA Water Board, 2022; NY DOH, 2022; Commonwealth of Massachusetts, 2018; U.S. EPA, 2017d). UCMR3 data were filtered to only include facilities flagged as using surface water, while the individual state data were filtered down to only those systems with surface water listed as the primary source in SDWIS. Datasets were processed to ensure that no samples were repeated in multiple datasets. These PWS monitoring data were collected to assess possible exposures to the general population through drinking water. Descriptions of the data retrieval and processing methods are presented in Appendix G.2.

The combined datasets resulted in 16,972 samples from 2,847 PWSs across 50 states (Table 2-2). Reported detection limits across the PWS datasets ranged from 0.0001 to 3 µg/L, with 81 percent of samples reporting a detection limit of 0.07 µg/L. To the extent that it could be determined from the database records, samples were separated into raw (untreated) water from the PWS intake or finished (treated) water being sent to the distribution system. The distribution of raw water monitoring concentrations is presented in Figure 2-9, and the distribution of finished drinking water concentrations is presented in Figure 2-10.

Table 2-2. Summary of PWS Monitoring Datasets of 1,4-Dioxane Monitoring in PWSs Using Surface Water as a Source

Dataset of Origin	Number of Samples	Minimum Concentration (µg/L)	Median Concentration (µg/L)	Maximum Concentration (µg/L)	Start Year	End Year
CA	1,797	0.25	0.5	1.5	2011	2022
MA	949	0.049	0.22	3.8	2008	2022
NY	615	7.20E−05	0.035	1	2015	2022
UCMR3	13,611	0.035	0.035	13.3	2013	2016

Note: for the summary presented in this table, results reported as below their respective detection limit were assigned a value of half of the detection limit.

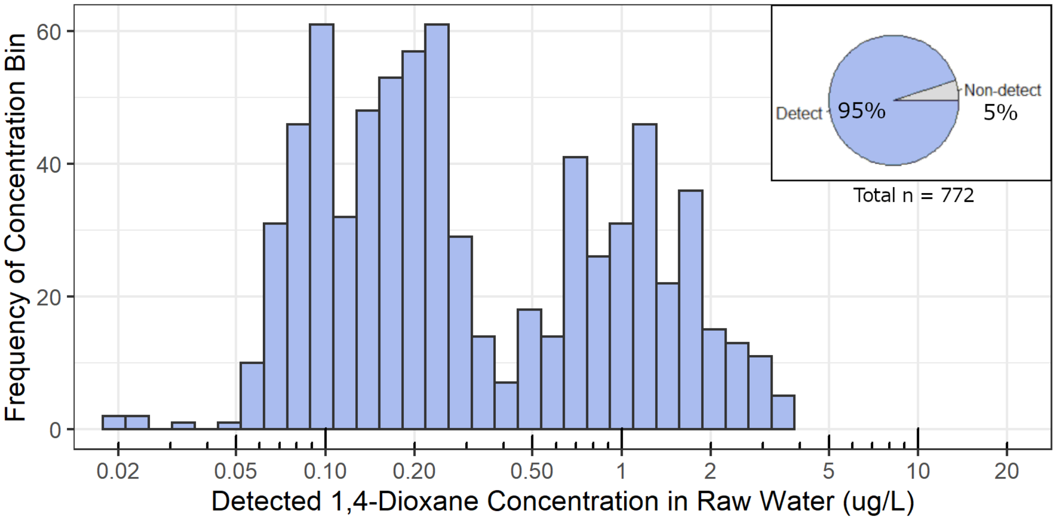

Figure 2-9. Frequency of 1,4-Dioxane Concentrations Monitored in Raw (Untreated) Drinking Water Derived from Surface Water

Data retrieved from state databases (CA, MA, and NY) between 2008–2022.

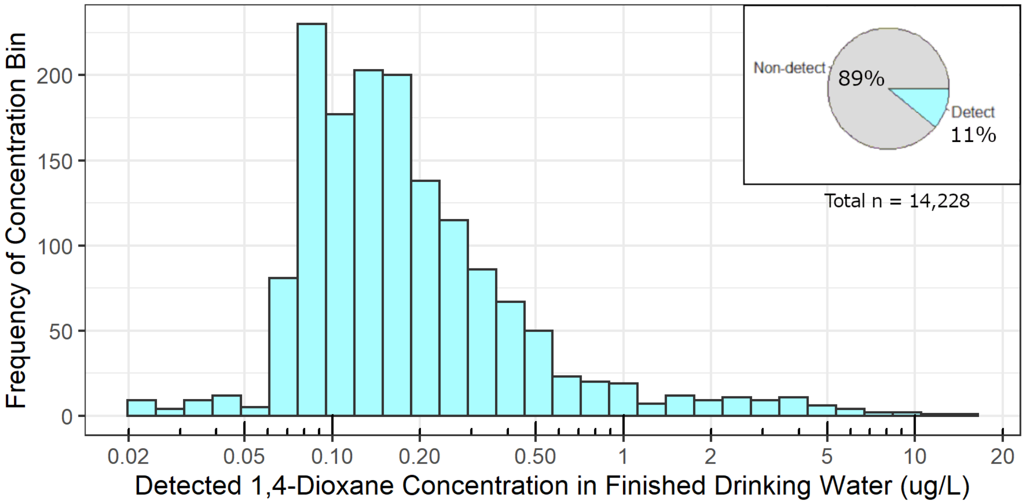

Figure 2-10. Frequency of 1,4-Dioxane Concentrations Monitored in Finished (Treated) Drinking Water Derived from Surface Water.

Data retrieved from the UCMR3 and state databases (CA, MA, and NY) between 2008–2022
Note: the detection limit for the method used in UCMR3, and the most common detection limit reported in state databases is 0.07 µg/L.

Water treatment systems may vary widely across the country based on available and utilized water treatment processes that depend on whether source water is groundwater or surface water. These processes typically include disinfection, coagulation/flocculation, sedimentation, and filtration (U.S. EPA, 2006a). In assessing drinking water exposures, the ability to treat and remove or transform chemicals in possible drinking water supplies should be considered. Typical treatment processes do not remove 1,4-dioxane from ambient surface water and groundwater prior to possible general population consumption as drinking water and treatment processes that do effectively remove 1,4-dioxane are uncommon. EPA therefore assumes zero removal in the following analyses to provide a conservative estimate of general population drinking water exposures. Even without treatment processes that remove

1,4-dioxane, multiple sources of water may be mixed within the same drinking water system which may result in finished water with lower concentrations than one of the higher contributing source water concentrations. EPA acknowledges that the surface water concentration at a single intake location may be higher than the finished drinking water once mixed with other sources. Further discussion of the prevalence of treatment processes across water systems, and the methodology for identifying raw and finished drinking water monitoring samples is presented in Appendix G.1.2.

Figure 2-11 shows the spatial distribution of UCMR3 samples at the county level, with 1,4-dioxane detected in 25 percent or 240 of 943 counties with participating water systems.

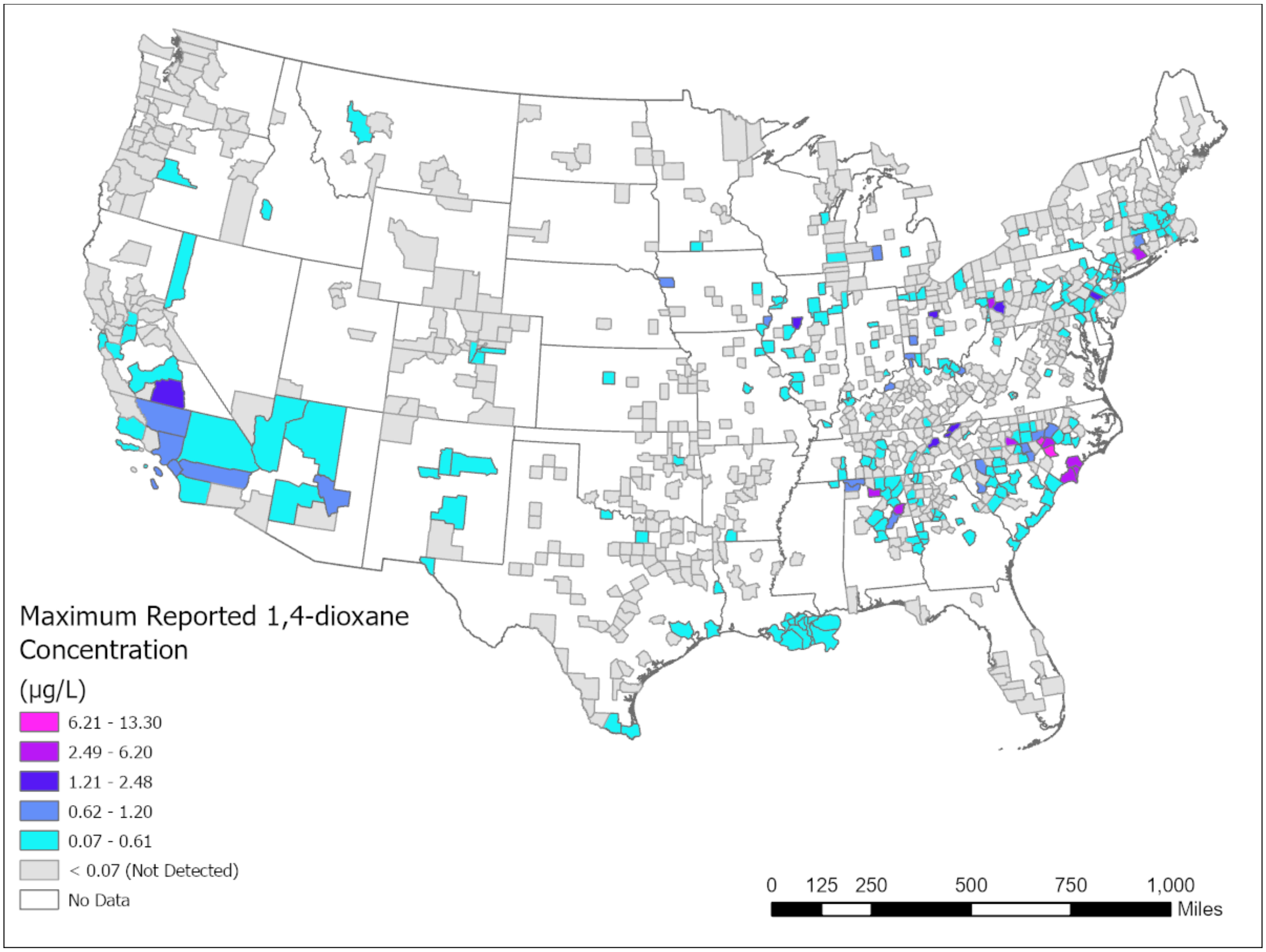

Figure 2-11. Map of Counties Containing PWSs that Reported Monitoring of Finished Drinking Water Drawn from Surface Water for 1,4-Dioxane under UCMR3

Note: UCMR3 monitoring of 1,4-dioxane required four sampling events, one for each season, to capture temporal variability. Each county highlighted may include one or multiple PWSs reporting data.

Monitored drinking water data were also included in exposure and risk estimates to assess the human health implications of drinking water concentrations in this range. Since the UCMR program and state monitoring datasets are not designed to reflect source water impacts of direct and indirect releases into water bodies, EPA's TSCA program relied on estimated concentrations modeled for a range of specific release scenarios to characterize risks from the water pathway. EPA evaluated the performance of the models used to estimate water concentrations with monitoring data from site-specific locations serving as cases studies. These case study comparisons demonstrated strong concordance between modeled concentrations and monitoring data, thereby increasing confidence in risk estimates based on modeled concentrations.

Although monitoring data confirm that 1,4-dioxane is present in drinking water in some locations, samples collected under the UCMR program are designed to be nationally representative of drinking water occurrence and not specifically associated with industrial releases of 1,4-dioxane. Since these monitoring data may not reflect the 1,4-dioxane concentrations that result from industrial releases, EPA relied on modeling to estimate 1,4-dioxane concentrations that occur near release sites.

2.3.1.2 Surface Water and Drinking Water Modeling

To assess possible general population exposures to 1,4-dioxane via industrial releases to surface water, concentrations of 1,4-dioxane in surface water were modeled using two separate approaches. First, a facility-specific approach aimed to quantify the maximum expected aqueous concentrations resulting from reported 1,4-dioxane discharges from individual facilities in isolation. Second, a probabilistic model was applied to assess the range of expected aqueous concentrations resulting from reported 1,4-dioxane discharges across a COU, with consideration of expected ranges of background concentrations of 1,4-dioxane from DTD loading and other unreported releases.

2.3.1.2.1 Modeling Methodology

A detailed description of modeling methods is presented in Appendix G.2.

As described in Section 2.2, annual releases of 1,4-dioxane to surface water from regulated dischargers were retrieved from TRI and DMR. To the extent possible, modeled hydrologic flow data (*i.e.*, stream flow) associated with the receiving water body to which each facility released was retrieved from the NHDPlus V2.1 dataset (U.S. EPA; U.S. GS, 2016). The receiving water body was identified either through NPDES permit information for the releasing facility, or the nearest identified NHDPlus V2.1 flowline. Detailed methods for the retrieval and processing of flow data are presented in Appendix G.2.1.

Facility-Specific Modeling

Facility-specific modeling was conducted to estimate concentrations in receiving water bodies resulting from the greatest facility-specific annual release reported between 2013 through 2019. This modeling approach employed the equations used to model releases from facilities in the E-FAST 2014 model (U.S. EPA, 2014) and is described in Appendix G.2.2. For each facility and annual release amount, three different scenarios for days of release per year were considered: 1 day, 30 days, and expected number of days of operation reported in Table_Apx E-2 (referred to as the "maximum" number of days and ranges from 250 to 365 days depending on OES). These additional scenarios with lower numbers of days of operation provide more conservative estimates of resulting surface water concentrations and are intended to evaluate the full range of possible facility release patterns based on the best available information. Two flow metrics were evaluated: the lowest monthly average flow from NHDPlus, and the harmonic mean flow derived from E-FAST 2014 methodology. The resulting concentrations from the facility-specific modeling are used in calculations of general population exposure and human health outcomes.

Probabilistic Modeling

The probabilistic modeling approach was conducted to consider multiple years of release data per facility and multiple modeled flow metrics from NHDPlus V2.1 (U.S. EPA; U.S. GS, 2016) per facility to generate a distribution of potential surface water concentrations resulting from releases across each COU. The underlying model for the probabilistic approach is a fit-for-purpose model developed by EPA in Microsoft Excel, the EWISRD-XL model (**E**stimating **W**ater **I**ndustrial **S**urface **R**elease and **D**own the Drain in Excel). The EWISRD-XL model was designed to model 1,4-dioxane inputs to a stream segment, including existing in-stream concentrations, DTD loading from consumer and commercial products, and industrial releases, as a steady-state snapshot of a single point in time (Figure 2-12). Outputs from the model include the resulting downstream concentration and the relative contribution from each input source to that concentration. Examples of the EWISRD-XL model applied to three specific case study locations (Brunswick County, NC, Columbia, TN, and Liverpool, OH) are included in three *Supplemental Information Files* (U.S. EPA, 2023o, p, q).

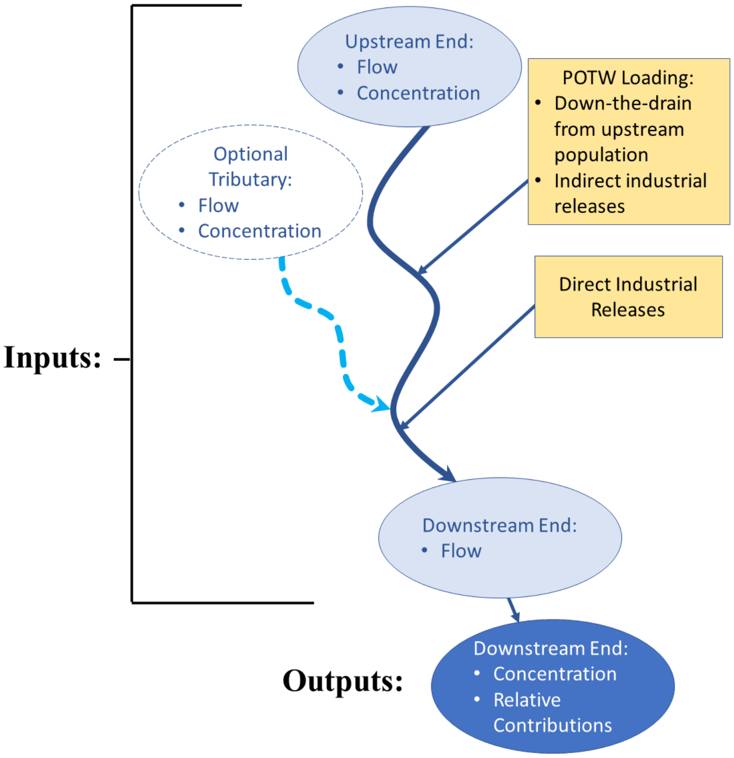

Figure 2-12. Schematic of the EWISRD-XL Model Inputs and Outputs

For the probabilistic 1,4-dioxane COU modeling, an R script (R Core Team, 2022) was developed to rapidly run multiple iterations of the EWISRD-XL model. In this configuration, called the EWISRD-XL-R model, the underlying calculations were performed by EWISRD-XL model, and an R script wrapper managed the processing of input and output data. For the probabilistic COU modeling, the EWISRD-XL-R model developed to calculate the receiving water body concentrations at the point-of-release by a facility. The EWISRD-XL-R results include the concentrations due only to releases from facilities, as well as an estimated background concentration of DTD and unmonitored releases. The full details of the underlying EWISRD-XL model and the probabilistic implementation are presented in Appendix G.2.3. Distributions of total concentrations resulting from combinations of facility releases and background concentrations were used for calculations of general population exposure and human health outcomes.

A series of case studies was developed with the EWISRD-XL model to evaluate its performance across various 1,4-dioxane release settings. These cases are presented in Appendix G.2.3.2.

2.3.1.2.2 Estimating Down-the-Drain Releases

To evaluate the anticipated ranges of DTD contributions of 1,4-dioxane to water bodies receiving POTW effluent, a range of combinations of hydrologic flows and populations served by a POTW were evaluated using the EWISRD-XL-R model. For this modeling exercise, only contributions from the DTD component were used to calculate resulting surface water contributions (*i.e.*, no facility releases or existing background concentrations were included). Hypothetical combinations of hydrologic flows and populations contributing to wastewater loading derived from the national distribution of hydrologic flows and populated places were selected to represent a range of results, which were then compared with concentrations expected from industrial releases and used to calculate ranges of human exposure and risk. More detailed methodology for this calculation is presented in Appendix G.2.3.4.

2.3.1.2.3 Hydraulic Fracturing

Hydraulic fracturing is a process used to extract oil and gas from shale plays. After hydraulic fracturing operations inject fluids to extract oil and gas, a substantial volume of water may be produced through flowback. The composition of these produced waters depends both on the geochemistry of the injected area and the injected fluids. 1,4-Dioxane has been reported to EPA as one of the chemicals present in these produced waters by 411 facilities via FracFocus 3.0 (GWPC and IOGCC, 2022). Estimated 1,4-dioxane loadings of produced water to surface water from hydraulic fracturing activities (described in Appendix E.9) were evaluated for expected ranges of resulting concentrations in receiving water bodies using the EWISRD-XL-R model. Hydraulic fracturing wells reporting 1,4-dioxane use by FracFocus 3.0 were mapped, and flow data from nearby water bodies were collected from NHDPlus V2.1. A Monte Carlo analysis was used to generate loadings to receiving water bodies from the distribution of modeled releases and to pair them with hydrologic flows, resulting in a distribution of possible surface water concentrations. Methodology for this analysis is presented in Appendix G.

2.3.1.2.4 Proximity to Drinking Water Sources

Drinking water exposures from facility-specific results assumed that the exposure occurs at the receiving water body to provide a conservative estimation of drinking water exposures. However, the evaluated water bodies may not be used as, or proximate to, actual drinking water sources and intakes. To give a more robust characterization of possible drinking water exposures, known facility-specific releases were mapped to drinking water sources using public water systems data stored in EPA's Safe Drinking Water Information System Federal Data Warehouse (U.S. EPA, 2022f). This dataset is updated quarterly, and the 2nd quarter 2022 version was used for this analysis. Following mapping, the colocation of and proximity of releases to drinking water sources were evaluated. Locations of raw water intakes for PWSs are considered sensitive by EPA Office of Water due to public safety concerns. Geospatial analysis and the NHDPlus V2.1 flowline network were used to assess whether any known drinking water intakes are located downstream of 1,4-dioxane releasing facilities. Methodology for this analysis is presented in Appendix G.2.4.

2.3.1.3 Modeling Results

2.3.1.3.1 Facility-Specific Results

The facility-specific results show the expected concentration at the point of release from the facility discharging 1,4-dioxane to receiving water bodies, without consideration of the contribution from other sources. The total number of modeled releases within a given OES may be greater than the number of 1,4-dioxane releasing facilities in cases where facilities indirectly dispose of 1,4-dioxane by transferring to another facility in addition to directly discharging 1,4-dioxane. Surface water concentrations resulting from facility-specific modeling for one day of release are summarized in Table 2-3 and represent the highest expected concentrations in receiving water bodies, due to the annual release amount being discharged in a single day. Surface water concentrations resulting from facility-specific modeling for maximum days of release are summarized in Table 2-4, and represent the lowest expected concentrations in receiving water bodies due to the annual release spread out over the most days. The single day release scenario allows consideration of a "worst-case scenario" given the available annual release information and can inform an upper limit of concentrations resulting from releases. The maximum days release scenario can inform a lower limit of expected concentrations from the available annual release data. Full discussion on the evaluation of multiple release days is given in Section 2.3.1.2.1, but the range of evaluated release days is intended to provide to full range of expected surface water concentrations resulting from possible facility release patterns and available information. As described in Section 5, these variations in concentration due to days of release do not affect chronic cancer risk estimates resulting from a particular releasing facility, due to annual averaging of exposure. Resulting concentrations varied widely, both across and within OESs, due to variability in facility release amounts as well as receiving water body flow magnitudes. Facility-specific releases are organized around their identified OES as fully described in Section 2.1 and Appendix D.

Table 2-3. Summary of Surface Water Concentration Results by OES from Facility-Specific Modeling of Annual Maximum Releases between 2013 and 2019 for 1 Operating Day per Year

OES	No. of Releases Modeled	Sum of Annual Releases Modeled (kg/year)	Annual Release by Facility (kg/site-year)			Surface Water Concentration (Lowest Monthly Flow) (µg/L)			Surface Water Concentration (Harmonic Mean Flow) (µg/L)		
			Min	**Mean**	**Max**	**Min**	**Mean**	**Max**	**Min**	**Mean**	**Max**
Disposal	25	16,997	1.36E−04	6.80E02	7.95E03	1.50E−02	6.45E05	9.52E06	1.50E−02	4.77E05	7.34E06
Ethoxylation Byproduct	8	112,076	4.54E−01	1.40E04	1.12E05	5.39E−03	2.58E06	2.07E07	3.01E−03	1.22E06	9.73E06
Functional Fluids (Open-System)	6	17,711	3.80E−01	2.95E03	1.75E04	1.39E01	1.57E03	4.78E03	6.07E00	7.40E02	2.21E03
Import and Repackaging	12	2,722	2.27E02	2.27E02	2.27E02	1.08E01	8.15E06	9.28E07	4.39E00	1.01E06	7.40E06
Industrial Uses	31	70,343	2.07E−01	2.27E03	2.62E04	1.33E−02	5.11E05	4.64E06	6.52E−03	4.53E05	5.15E06
Manufacture	2	7,034	1.67E03	3.52E03	5.36E03	8.31E04	1.63E06	3.18E06	8.31E04	1.63E06	3.18E06
PET Manufacturing	19	2,773,355	3.40E−01	1.46E05	2.51E06	2.77E00	1.07E06	1.66E07	1.28E00	1.05E06	1.66E07
Printing Inks	1	5	5.45E00	5.45E00	5.45E00	2.05E03	2.05E03	2.05E03	2.05E03	2.05E03	2.05E03
Remediation	16	46	3.40E−05	2.91E00	2.39E01	1.50E−03	1.83E03	1.79E04	3.54E−04	1.52E03	1.37E04
Overall	120	3,000,290	3.40E−05	2.50E04	2.51E06	1.50E−03	1.45E06	9.28E07	3.54E−04	5.92E05	1.66E07

Table 2-4. Summary of Surface Water Concentration Results by OES for Facility-Specific Modeling of Annual Maximum Releases between 2013 and 2019 for the Maximum Operating Days per Year

OES	No. of Releases Modeled	Sum of Annual Releases Modeled (kg/year)	Annual Release by Facility (kg/site-year)			Surface Water Concentration (Lowest Monthly Flow) (µg/L)			Surface Water Concentration (Harmonic Mean Flow) (µg/L)		
			Min	Mean	Max	Min	Mean	Max	Min	Mean	Max
Disposal	25	16,997	1.36E−04	6.80E02	7.95E03	5.99E−05	2.57E03	3.81E04	5.99E−05	1.90E03	2.94E04
Ethoxylation byproduct	8	112,076	4.54E−01	1.40E04	1.12E05	2.16E−05	1.03E04	8.26E04	1.20E−05	4.87E03	3.89E04
Functional Fluids (Open-System)	6	17,711	3.80E−01	2.95E03	1.75E04	5.63E−02	6.37E00	1.93E01	2.46E−02	3.00E00	8.95E00
Import and Repackaging	12	2,722	2.27E02	2.27E02	2.27E02	4.32E−02	3.26E04	3.71E05	1.76E−02	4.04E03	2.96E04
Industrial Uses	31	70,343	2.07E−01	2.27E03	2.62E04	5.31E−05	2.04E03	1.86E04	2.61E−05	1.81E03	2.06E04
Manufacture	2	7,034	1.67E03	3.52E03	5.36E03	3.32E02	6.52E03	1.27E04	3.32E02	6.52E03	1.27E04
PET Manufacturing	19	2,773,355	3.40E−01	1.46E05	2.51E06	1.11E−02	4.29E03	6.63E04	5.12E−03	4.20E03	6.63E04
Printing Inks	1	5	5.45E00	5.45E00	5.45E00	8.21E00	8.21E00	8.21E00	8.21E00	8.21E00	8.21E00
Remediation	16	46	3.40E−05	2.91E00	2.39E01	4.11E−06	5.01E00	4.90E01	9.69E−07	4.15E00	3.75E01
Overall	120	3,000,290	3.40E−05	2.50E04	2.51E06	4.11E−06	5.80E03	3.71E05	9.69E−07	2.37E03	6.63E04

Table 2-5 describes the crosswalk between identified OESs and relevant COUs under each for the identified facility releases to surface water. The full facility-specific analysis is included in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Release to Surface Water from Individual Facilities* (U.S. EPA, 2023h).

Table 2-5. OES-COU Crosswalk for Identified Facilities Releasing to Surface Water[a]

COU[b]			**OES[d]**
Life Cycle Stage	**Category**	**Subcategory[c]**	
Manufacturing	Domestic manufacture	Domestic manufacture	Manufacturing
	Import	Import Repackaging	Import and Repackaging
Processing	Processing as a reactant	Polymerization catalyst	Industrial Uses
	Non-incorporative	Basic organic chemical manufacturing (process solvent)	
	Byproduct	Byproduct produced during the ethoxylation process to make ethoxylated ingredients for personal care products	Ethoxylation Process Byproduct
		Byproduct produced during the production of polyethlene terephtalate	PET Byproduct
Industrial Use[a]	Intermediate use	Plasticizer intermediate Catalysts and reagents for anhydrous acid reactions, brominations, and sulfonations	Industrial Uses
	Processing aids, not otherwise listed	Wood pulping Extraction of animal and vegetable oils Wetting and dispersing agent in textile processing Polymerization catalyst Purification of process intermediates Etching of fluoropolymers	Industrial Uses
	Functional fluids (open and closed systems)	Polyalkylene glycol lubricant Synthetic metalworking fluid Cutting and tapping fluid	Functional Fluids (Open System)
Industrial Use, Commercial Use	Other Uses	Spray polyurethane foam Printing and printing compositions, including 3D printing Dry film lubricant Hydraulic fracturing	Printing Inks (3D)
Disposal	Disposal	Remediation	Remediation
Disposal	Disposal	Industrial pre-treatment Industrial wastewater treatment Publicly owned treatment works (POTW) Underground injection Municipal landfill	Disposal

COU[b]			OES[d]
Life Cycle Stage	**Category**	**Subcategory[c]**	
		Hazardous landfill Other land disposal Municipal waste incinerator Hazardous waste incinerator Off-site waste transfer	

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

[b] As mapped to COU Life Cycle Stage, Category, and Subcategory in Table_Apx D-1.

[c] Evaluated facilities within an OES may not encompass all listed COU subcategories.

[d] Note that identified OESs can encompass multiple COUs across different life cycle stages and categories.

To put the modeled releases in the context of the underlying data sources for release amounts and receiving water body flow, Table 2-6 presents the results of the process of assigning the receiving water body (by reach code in the NHDPlus 2.1 dataset) to each releasing facility. Those facilities with reach code information in their NPDES permit were regarded as the highest confidence in an accurate match to the actual discharging water body, followed by facilities matched geospatially to the nearest reach code within 1 km of the facility. Facilities matched to reaches beyond 1 km from the facility but within 2 km provided lower confidence, and those without reach code matches were substituted with the lowest non-zero flow within the OES as a conservative estimate. The full details of the flow matching process are presented in Appendix G.2.1. Due to the assumptions described in Section 2.2.1.3 required to model releases from facilities reporting only via TRI Form A, the percent of facilities within an OES using Form A is also reported.

Table 2-6. Summary by OES of Data Sources for Releases and Receiving Water Body Flow

		Method of Matching to Receiving Water Body				
OES	**Total Number of Releases**	**NPDES Permit Contains Reach Code**	**Nearest Reach (within 1 km)**	**Nearest Reach within 2 km**	**Lowest Non-zero Flow within OES Substituted**	**% of Releases Estimated from TRI Form A**
Disposal	25	22	0	1	1	8
Ethoxylation Byproduct	8	1	1	1	0	0
Functional Fluids (Open-System)	6	5	0	0	0	0
Import and Repackaging	12	1	2	2	7	100
Industrial Uses	31	11	3	5	7	45
Manufacture	2	1	0	0	0	0
PET Manufacturing	19	11	0	1	0	0
Printing Inks	1	1	0	0	0	0
Remediation	16	14	2	0	0	0
Total	120	67	8	10	15	23

A generic table of annual facility release and average flow rates for the receiving water body is presented in Table 2-7, which demonstrates the relationship between the facility and water body characteristics with regard to the resulting surface water concentrations. Table 2-8 shows the relative occurrence of each of the releases modeled for this assessment within binned ranges of releases and flows. Combined, these tables demonstrate that most facilities releasing 1,4-dioxane are initially discharging to smaller water bodies, even in some cases where large annual release amounts result in very high modeled concentrations.

Table 2-7. Hypothetical Mean Annual Concentrations (µg/L) for a Range of Annual Release and Flow Rate Combinations, for a Facility with 250 Days of Release per Year

		Annual Release Amount (kg)						
		1	10	100	1,000	10,000	100,000	1,000,000
Mean Annual Receiving Water Body Flow (cfs)	1	1.6	16	160	1.6E03	1.6E04	1.6E05	1.6E06
	10	0.16	1.6	16	160	1.6E03	1.6E04	1.6E05
	100	0.016	0.16	1.6	16	160	1.6E03	1.6E04
	1,000	1.6E−03	0.016	0.16	1.6	16	160	1.6E03
	10,000	1.6E−04	1.6E−03	0.016	0.16	1.6	16	160
	100,000	1.6E−05	1.6E−04	1.6E−03	0.016	0.16	1.6	16

Table 2-8. Occurrence of Facilities for Distributions of Maximum Annual 1,4-Dioxane Release Amounts and Receiving Water Body Flow

		Annual Release Amount (kg)					
		<10	10 to 100	100 to 1,000	1,000 to 10,000	10,000 to 100,000	>100,000
Mean Annual Receiving Water Body Flow (cfs)	<10	14%	8%	11%	6%	2%	<1%
	10 to 100	9%	7%	2%	<1%	1%	3%
	100 to 1,000	3%	6%	2%	1%	<1%	<1%
	1,000 to 10,000	1%	2%	2%	2%	2%	<1%
	10,000 to 100,000	3%	1%	3%	4%	2%	<1%

2.3.1.3.2 Concentrations from Down-the-Drain Loading

Water concentrations of 1,4-dioxane resulting from DTD releases depend on the population size (an indicator of the number of people using products and contributing to the releases) and the stream flows of the receiving water bodies. The representative per capita DTD loading developed from modeling results from SHEDS-HT was applied to a range of population sizes (100 to 1,000,000 people) and stream flows (300 to 30,000 cfs) to develop a distribution of potential surface water concentrations. Estimated surface water at the point of discharge by POTWs resulting from DTD releases ranged from less than 0.0001 to 110 µg/L (Table 2-9). The typical ranges of results from this analysis are comparable to the range of minimum to mean concentrations calculated from individual facility releases in Section 2.3.1.3.1.

Table 2-9. Estimated Surface Water Concentrations (µg/L) Due to DTD Loading for a Range of Populations and Hydrologic Flows

		Population Contributing to DTD Loading				
		100	**1,000**	**10,000**	**100,000**	**1,000,000**
Receiving Water Body Flow (cfs)	**100**	0.011	0.11	1.1	11	110
	300	3.6E−03	0.036	0.36	3.6	36
	1,000	1.1E−03	0.011	0.11	1.1	11
	3,000	3.6E−04	3.6E−03	0.036	0.36	3.6
	10,000	1.1E−04	1.1E−03	0.011	0.11	1.1

The occurrence of POTWs processing wastewater from various populations and the associated flows of the receiving water bodies were investigated using data from the ICIS-NPDES database (U.S. EPA, 2013), to inform the interpretation of the above ranges of DTD loading concentrations. For communities with a single POTW treating wastewater, most fell into the range of 100 to 10,000 people, with the annual average flow of the receiving water body less than 300 cfs (Table 2-10).

Table 2-10. Estimated Percent Occurrence of Combinations of Contributing Population to POTWs and Receiving Water Body Flow, from Combined ICIS-NPDES and 2020 Census Data

		Population Contributing to DTD Loading				
		<100	**100 to 1,000**	**1,000 to 10,000**	**10,000 to 100,000**	**100,000 to 1,000,000**
Mean Annual Receiving Water Body Flow (cfs)	**<100**	5%	44%	26%	4%	<1%
	100 to 300	<1%	3%	4%	1%	<1%
	300 to 1,000	<1%	2%	2%	1%	<1%
	1,000 to 3,000	<1%	1%	2%	<1%	<1%
	3,000 to 10,000	<1%	<1%	1%	<1%	<1%
	>10,000	<1%	1%	1%	1%	<1%

2.3.1.3.3 Concentrations from Hydraulic Fracturing

The Monte Carlo distribution of potential surface water concentrations resulting from hydraulic fracturing operations is presented in Table 2-11. Hydrologic flows in water bodies near hydraulic fracturing wells reporting 1,4-dioxane as a constituent of wastewater ranged from less than 10 to 44,300 cfs. Due to the very low flows in many nearby streams, resulting concentrations were sensitive to the receiving water body flow rate. The distribution of loading to surface water from hydraulic fracturing represents the loading from a single site of hydraulic fracturing operations (described in Appendix G.2.3.5) at the immediate point of discharge to the receiving water body. Concentrations estimated at the highest end are comparable to mean to high-end facility releases presented in Section 2.3.1.3.1. More than half of the modeled concentrations fell below the typical detection limit in drinking water of 0.07 µg/L.

Table 2-11. Distribution of Potential Concentrations in Surface Water Resulting from Hydraulic Fracturing Operations from a Single Site Reporting 1,4-Dioxane as an Ingredient

Monte Carlo Distribution	Concentration (µg/L)
Maximum	198
99th Percentile	7.79
95th Percentile	2.60
Median	0.064
5th Percentile	2.31E−04
Minimum	6.08E−12

2.3.1.3.4 Aggregate Probabilistic Results

The aggregate probabilistic model predicts surface water concentrations at the point of facility releases when incorporating potential contributions from DTD and other unmonitored sources. The model incorporates multiple years of release data and was run with 10,000 iterations for each OES using different combinations of direct and indirect facility releases, DTD releases, flows, and background concentrations. This results in a more descriptive distribution of the potential releases. At the highest end, the results of the aggregate probabilistic model are similar to those from the facility-specific modeling. This is due to both the facility-specific modeling and the highest end of the probabilistic modeling being based on the maximum reported releases from the modeled facilities. Additionally, the loading from facilities far outweighs the contribution from background sources at the higher end. The shape of the resulting distribution can be informative in its representation of the frequency of concentrations exceeding a certain threshold.

Resulting surface water concentrations ranged from 1.45×10^{-4} to 7.34×10^{3} µg/L. Summaries of the resulting concentrations by OES are presented in Table 2-12 and Figure 2-13. Overall, releases from facilities tended to result in greater 1,4-dioxane concentrations in surface water than the expected ranges of background concentrations. Background concentrations were derived from values of 1,4-dioxane measured by drinking water systems using surface water as a source that were not downstream of known 1,4-dioxane releases (Figure 2-11). The “% of Releases Greater than Background” column in Table 2-12 refers to the frequency of model runs (out of the 10,000 per OES) in which the resulting concentration from the facility release was greater than the generated background concentration resulting from DTD and other unregulated surface water loading. A low percentage for this metric may suggest that releases by a particular OES are typically outweighed by these other unreported releases with respect to their contribution to surface water concentrations.

Table 2-12. Aggregate Probabilistic Results Showing Distribution of Total 1,4-Dioxane Concentration in Surface Water (Release Plus Background)

OES	Min (µg/L)	5th Percentile (µg/L)	25th Percentile (µg/L)	Median (µg/L)	75th Percentile (µg/L)	95th Percentile (µg/L)	Max (µg/L)	% of Releases Greater than Background
Disposal	1.88E−03	1.51E−01	1.98E−01	3.50E−01	8.52E−01	1.94E00	2.02E01	81
Ethoxylation Process Byproduct	2.25E−03	9.81E−02	1.35E−01	2.74E−01	4.65E−01	2.55E00	1.46E01	72
Functional Fluids (Open-System)	1.78E−04	8.20E−02	1.11E−01	1.58E−01	2.79E−01	1.60E00	6.10E00	48
Import and Repackaging	5.70E−03	1.32E−01	2.83E−01	6.60E01	3.25E02	1.42E03	2.12E03	90
Industrial Uses	1.45E−04	5.15E−02	8.95E−02	1.22E−01	2.52E−01	1.33E01	2.26E02	44
Manufacture	1.10E02	3.32E02	3.32E02	7.19E02	2.32E03	5.48E03	7.34E03	100
PET Byproduct	5.44E−03	1.42E−01	2.13E−01	4.65E−01	4.36E00	1.30E02	2.23E03	84
Printing Inks (3D)	1.51E−01	3.17E−01	1.10E00	4.00E00	6.41E00	7.48E00	8.26E00	96
Remediation	5.34E−04	5.89E−02	9.04E−02	1.35E−01	2.73E−01	6.66E00	1.46E01	47

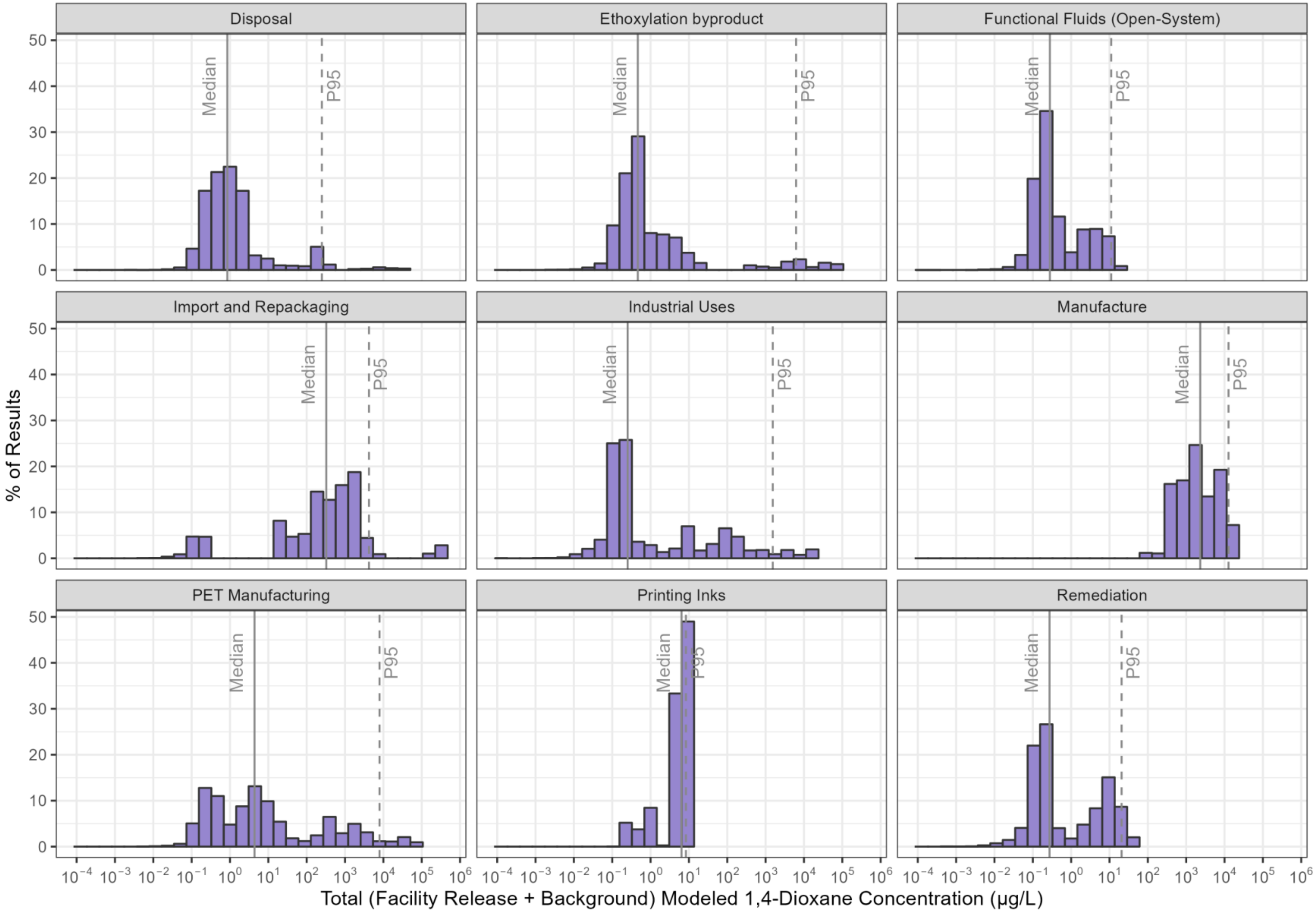

Figure 2-13. Distributions of Surface Water Concentrations Estimated by Aggregate Probabilistic Model for Each OES

Vertical lines indicate the median and 95th percentile (P95) surface water concentrations.

2.3.1.4 Comparison of Modeled and Monitored Surface Water Concentrations

At the higher end, the modeled concentrations from facility releases are several orders of magnitude greater than those observed in the 1,4-dioxane monitoring data (Figure 2-6 and Table 2-3). This difference in concentrations may be due to monitoring data being collected further downstream (allowing for additional dilution), or on reaches that are not impacted by releasing facilities. Many of the direct releasing facilities, and POTWs assessed for the DTD component, had a receiving water body specified on their NPDES permits that was associated with a very small stream or industrial canal. These small receiving water bodies, combined with larger loading values from the releases, resulted in high modeled concentrations in surface water at the point of release. As this water travels downstream, it is expected to eventually join with larger water bodies, where some decrease in concentration due to dilution would occur.

Because most of the reasonably available monitoring data were generally not co-located with 1,4-dioxane release sites, EPA relied primarily on modeling to estimate water concentrations that could result from releases. Where co-located monitoring data were available, EPA compared modeled concentrations to reasonably available monitoring data in the limited set of specific locations to evaluate the performance of the model. Comparisons of modeled vs. monitoring water concentrations for this limited set of “case study” locations demonstrate that modeled mean concentrations are generally consistent with mean concentrations reported in monitoring data. For example, the Cape Fear River upstream of the Brunswick County, NC drinking water intake was selected as a case study to test the model due to abundant monitoring data in the region. Water concentrations modeled based on upstream releases from an industrial facility in Fayetteville in combination with other upstream sources. As illustrated in Figure 2-14, modeled surface water concentrations generally fell within the ranges reported from monitored concentrations. Wide ranges of both monitored and modeled values were noted, indicating variability among inputs to the system. Details of the case study comparisons for Brunswick County and other locations are described in Appendix G.2.3.2. The concordance of monitoring and modeled concentrations increases confidence in the model used to estimate water concentrations from DTD releases and hydraulic fracturing, and to perform probabilistic modeling of aggregate concentrations from multiple sources.

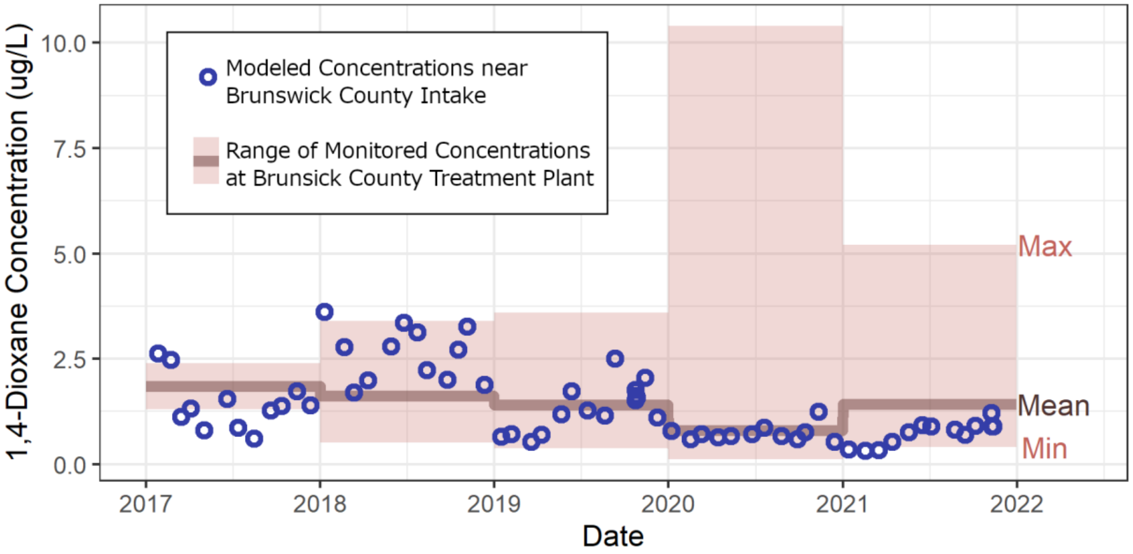

Figure 2-14. Case Study Comparison of Modeled and Monitored Concentrations in Brunswick County

2.3.1.5 Strengths, Limitations, and Sources of Uncertainty in Assessment Results for Monitored and Modeled Drinking Water and Surface Water Concentrations

The evaluation of general population drinking water exposure scenarios are impacted by uncertainties and assumptions surrounding inputs and the approaches used for modeling surface water concentrations and estimation of the drinking water doses. In Appendix E.8, EPA assesses the overall confidence of estimated releases for various OESs. For those OESs releasing to surface water, confidence is rated as medium to high depending on an individual OES.

The modeling used and the associated default and user-selected inputs have the ability to affect overall strength in evaluated general population exposures. The facility-specific releases methodology described in Section 2.3.1.2.1, and the results in 2.3.1.3.1, rely on a modeling framework that does not consider downstream fate or transport. However, the physical-chemical properties of 1,4-dioxane are expected to moderate this limitation due to its likelihood to stay in the water column, and due to the lack of removal during typical drinking water treatment process. To reduce uncertainties, EPA incorporated an updated flow network and flow data into this assessment that allowed a more site-specific consideration of release location and associated receiving water body flows. These facility-specific releases are also evaluated on a per facility basis that does not account for additional sources of 1,4-dioxane that may be present in the evaluated waterways. To help address these limitations in this risk evaluation, EPA conducted additional aggregate and probabilistic approaches, evaluated in Section 2.3.1.2.1 and Section 2.3.1.3.4, that give a more complete overall estimation of possible 1,4-dioxane concentrations. Finally, drinking water exposures from facility-specific results assume that the exposure occurs at the receiving water body. The water bodies evaluated may or may not be used as drinking water sources. To address this limitation, EPA evaluated the proximity of known 1,4-dioxane releases to known drinking water sources as well as known drinking water intakes as described in Section 2.3.1.2.4.

To evaluate the accuracy of the aggregate model, case studies described in Appendix G.2.3.2 compared modeled results to observed monitored concentrations. The three evaluated case studies give good general agreement between available monitoring with modeled values. Overall, this gives strength to the modeling assumptions, inputs and output calculations for areas that are lacking robust monitoring data. The model is able to effectively capture the general influences of both DTD loading, facility loading and upstream contributions to create an aggregation of possible ambient surface water concentrations of 1,4-dioxane. The monitored data encompassed both ambient surface water monitoring as well as drinking water system monitoring data. For the ambient surface water data, data is limited geographically and temporally with many states having no reported data and even those areas reporting measured values having limited samples over time. Monitored concentrations in close proximity to modeled releases were rare, often making direct comparisons of modeled results unavailable. In most cases, monitoring data represented water bodies without identified releases of 1,4-dioxane nearby.

The hydraulic fracturing analysis relies on a Monte Carlo distribution of loading values with some level of uncertainty and is itself a Monte Carlo simulation with potential receiving water body flows. The precision of such an analysis is lower at the most extreme (minimum and maximum) values.

2.3.2 Land Pathway (Groundwater)

Any activities where chemicals or wastes might be released to the environment has the potential to pollute groundwater. To understand possible exposure scenarios from these practices, EPA assessed drinking water exposure resulting from use of 1,4-dioxane contaminated groundwater due to chemical injection to Underground Class I Wells, leaching from landfills where 1,4-dioxane or products containing 1,4-dioxane have been disposed, and disposal of hydraulic fracturing produced water to surface impoundments. Sections 2.3.2.1 through 2.3.2.4 provide a description and an assessment of each disposal practice. Figure 2-15 and Figure 2-16 provide a visual summary of groundwater monitoring data available through the WQP (NWQMC, 2022).

2.3.2.1 Groundwater Monitoring Data

Measured, field-collected, data from environmental samples representing groundwater 1,4-dioxane concentrations across the country were collected as direct groundwater monitoring results. These results are collated by the National Water Quality Monitoring Council and stored in the WQP (NWQMC, 2022). Some monitoring results reported to the WQP included locations expected to be directly impacted by 1,4-dioxane releases. Data were available from 1997 to 2022, resulting in 8,110 available sample results. The distribution and detection percentages are presented in Figure 2-15 and mapped in Figure 2-16. The process for identifying this data is provided in Appendix H.1. This analysis is intended to characterize the observed ranges of 1,4-dioxane concentrations in groundwater, irrespective of the reasons for sample collection, and to provide context for the modeled groundwater concentrations presented in Sections 2.3.2.1 through 2.3.2.4. In order to better understand where highest groundwater concentrations are occuring, EPA arbitrarily portioned the data based on order of magnitude differences to best describe where and when data differences could be observed.

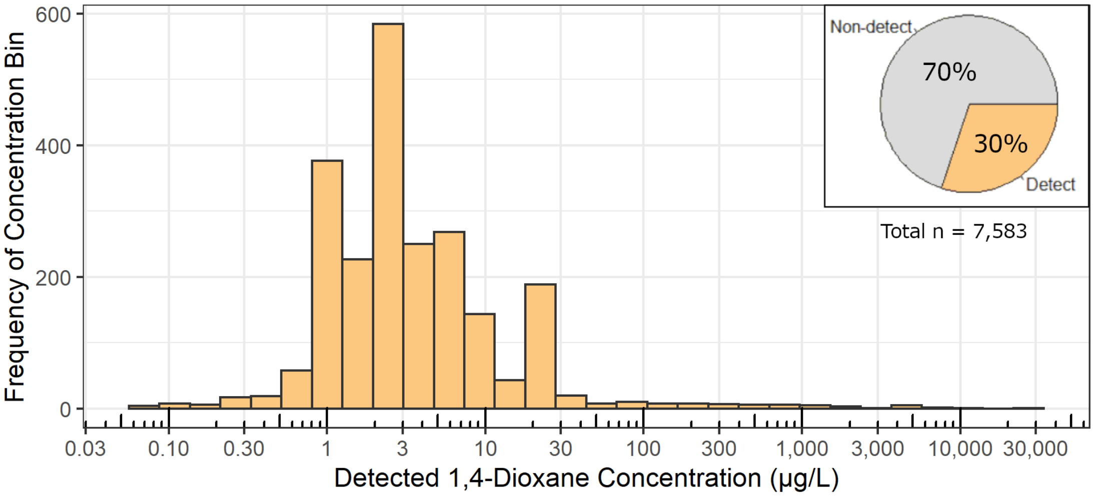

Figure 2-15. Frequency of Nationwide Detected 1,4-Dioxane Groundwater Concentrations (n = 2,284) Retrieved from the Water Quality Portal, 1997–2022

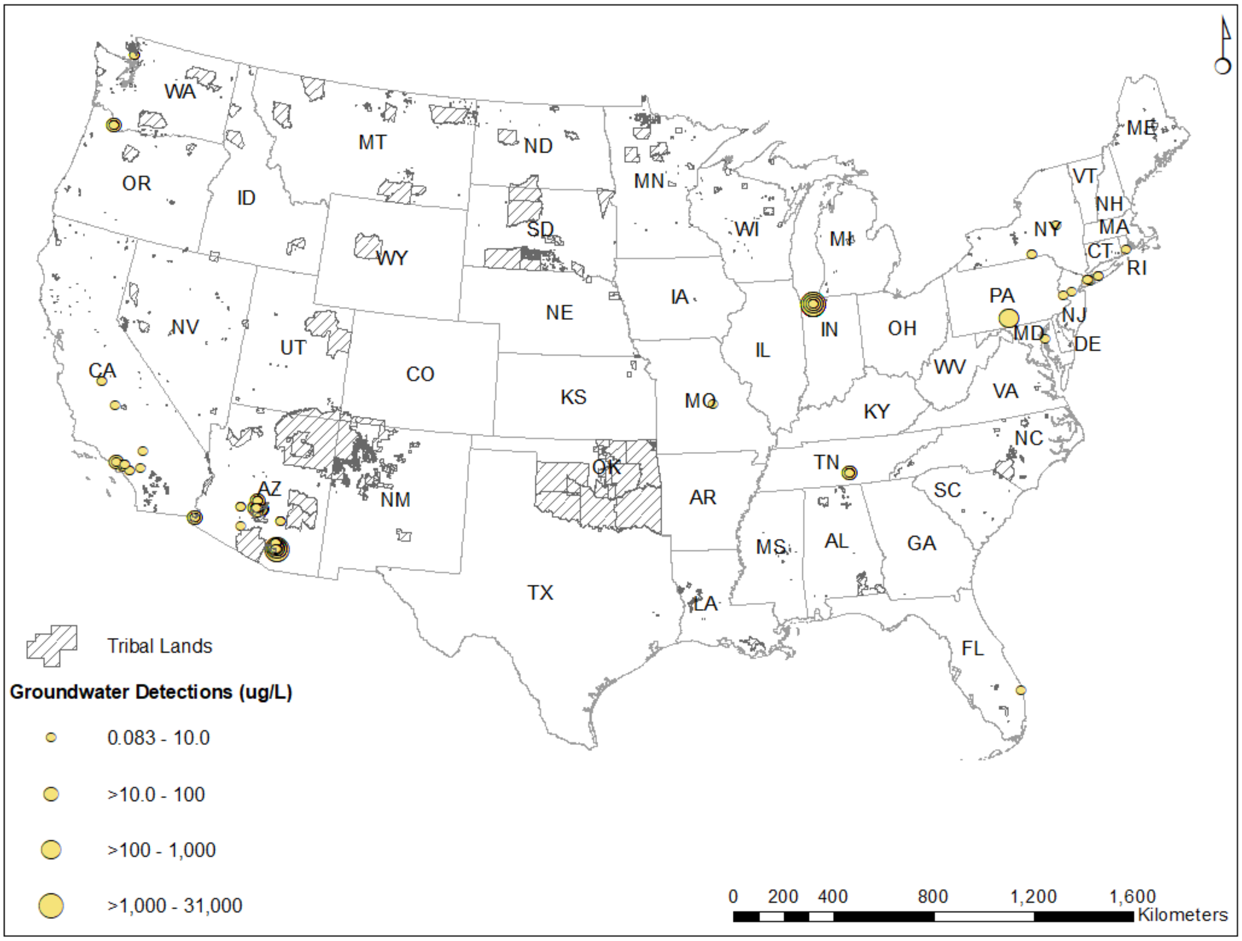

Figure 2-16. Detectable Concentrations of 1,4-Dioxane in Groundwater from the Water Quality Portal, 1997–2022

Note: Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown as there are no known monitoring data above detection limits.

Figure 2-15 shows the range of detected concentrations of 1,4-dioxane in groundwater from 1997 to 2022. During this period the detection limits ranged from 0.028 µg/L to 320 µg/L. The maximum detected concentration (31,000 µg/L) occurred in Westville, IN, in 1997 at a former waste-oil refinery. This site and many others identified in this monitoring data have ongoing remediation projects to address these contamination plumes.

Recent changes in industrial activities and disposal may have largely reduced groundwater contamination with 1,4-dioxane. As shown in Figure 2-17, samples collected prior to 2000 tended to be substantially higher in concentration relative to those collected after 2003. This finding may be an artifact of historical uses and industrial practices related to 1,4-dioxane. Although several samples are still above 10 µg/L, particularly in 2007, the bulk of data tend to fall between 1 and 10 µg/L. Without a thorough investigation of what practices have changed in industry, it is difficult to attribute this decline to a single event but indicates continued work to prevent groundwater contamination.

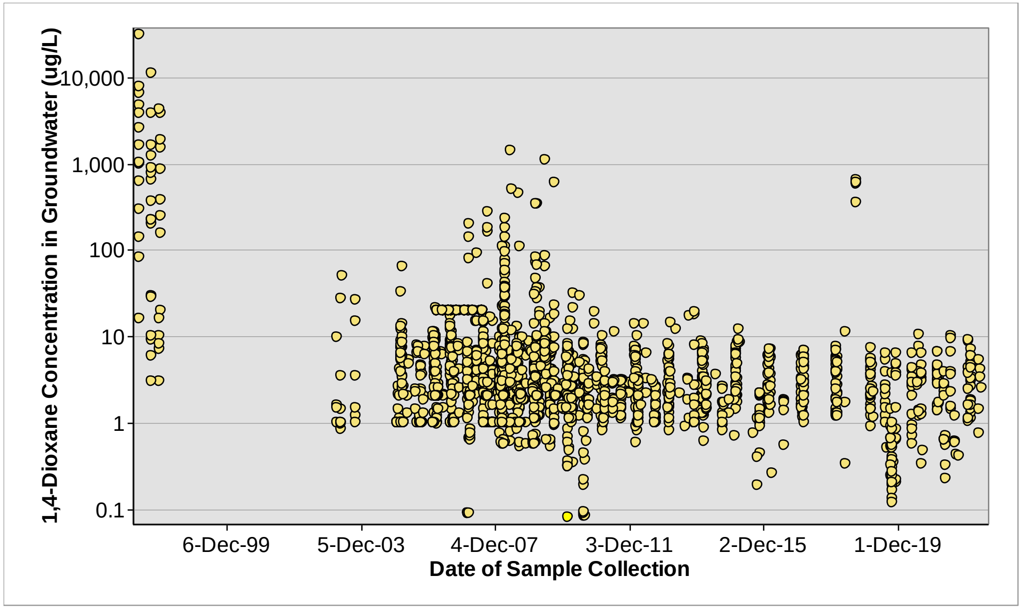

Figure 2-17. Groundwater Concentrations of 1,4-Dioxane vs. Sample Collection Date for Data Collected between 1997 and 2022

Figure 2-16 shows the spatial distribution of detected 1,4-dioxane concentration across the contiguous states. This map shows nine locations with concentrations of 1,4-dioxane greater than 10 µg/L. These tend to be attributed to past industrial activities causing extensive groundwater contamination. In addition to this monitoring data, groundwater contamination from disposing 1,4-dioxane to landfills has been documented in Alaska (Li et al., 2013), California (Li et al., 2015; Adamson et al., 2014), Michigan (Mohr and DiGuiseppi, 2010), New York (Lee et al., 2020), and recently in Ohio (https://cumulis.epa.gov/supercpad/cursites/csitinfo.cfm?id=0504014). EPA was not able to identify reasonably available information specific to groundwater concentrations near or around underground injection sites, landfills, or surface impoundments that received hydraulic fracturing produced water.

2.3.2.2 Disposal via Underground Injection

Underground injection is a method of disposal for hazardous wastes.[9] There are generally six different classes of underground wells, and only Class I Wells may be permitted to receive hazardous waste. Oversight of these wells requires that they are designed and constructed to prevent the movement of

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[9] Additional information about underground injection can be found at https://www.epa.gov/uic.

injected waste streams into drinking water systems. Wells typically consist of three or more concentric layers of pipe including surface casing, long string casing, and injection tubing. In addition, wells must be sited at locations with geologies that mitigate any movement of contaminants outside of a confined layer in case of a well failure. Extensive pre-siting geological tests confirm that the injection zone is of sufficient lateral extent and thickness and is sufficiently porous so that fluids injected through the well can enter the rock formation without extensive buildup of pressure or possible displacement of injected fluids outside of the intended zone.

Potential pathways through which injected fluids can migrate to underground sources of drinking water include failure of the well or improperly plugged or completed wells near the well. Well failures can be detected by continuous monitoring systems or mechanical integrity tests, at which point the wells would be shut-in until they are repaired. EPA's extensive technical requirements for Class I wells (40 CFR 148) are designed to prevent contamination of underground sources of drinking water through these pathways. Operators must conduct appropriate mechanical integrity tests yearly for hazardous wells and every 5 years for nonhazardous wells to ensure wells are fit for operation. Note that the loss or failure of mechanical integrity does not necessarily mean that wastewater will escape the injection zone. This added security can be attributed to redundant safety systems to protect against loss of waste confinement.

2.3.2.2.1 Summary of Assessment for Disposal to Underground Injection

According to EPA's TRI database, there are two locations where 1,4-dioxane has been disposed of via underground injection to Class I Wells. On-site disposals to Class I underground injection wells are provided in Table_Apx H-1. On-site Class I underground injection wells may be owned and operated by the producer of the waste. Off-site disposals to Class I underground injection wells are provided in Table_Apx H-2. Offsite Class I underground injection wells may be secondary entities that own and operate the well. Both on-site and offsite underground injection wells must be permitted and regularly inspected. Careful review of the permits and state databases corroborates that both sites are permitted and compliant. These sites have implemented groundwater migration controls and the Enforcement and Compliance History Online (ECHO) database (U.S. EPA, 2022e) indicates the site is currently in compliance.

In addition to reviewing these permits, EPA reviewed reasonably available groundwater monitoring data available via state databases as well as via the WQP (see Figure 2-15 and Figure 2-16) and found no evidence of groundwater contamination near the facilities. Because underground injection is not expected to result in groundwater contamination based on the reasonably available information, EPA did not quantitatively estimate groundwater concentrations, exposures, or risks from underground injection.

2.3.2.2.2 Strengths, Limitations, and Sources of Uncertainty in Assessment of Disposal to Underground Injection Wells

Because EPA did not quantitatively evaluate the potential exposure from disposing 1,4-dioxane via underground injection, the major source of uncertainty is limited to the accuracy of state databases providing monitoring data surrounding these wells. EPA believes these databases are reporting accurately where contaminations are known, but only explored states where the TRI database indicated there were disposals via underground injection. Disposals below the reporting requirement for TRI may not be captured.

2.3.2.3 Disposal to Landfills

Landfills may have various levels of engineering controls to prevent groundwater contamination. These can include industrial liners, leachate capturing systems, and routine integration of waste. However, groundwater contamination from disposal of consumer, commercial, and industrial waste streams continues to be a prominent issue for many landfills throughout the United States (Li et al., 2015; Li et al., 2013; Mohr and DiGuiseppi, 2010). These contaminations may be attributed to perforations in the liners, failure of the leachate capturing system, or improper management of the landfills. 1,4-Dioxane persists in groundwater and can migrate away from landfills into nearby communities at the same rate as hydraulic flow (Mohr and DiGuiseppi, 2010). When these communities rely on groundwater as their primary drinking water source, there is a potential for exposure via oral ingestion if that water is contaminated with 1,4-dioxane and does not undergo treatment. Depending on the distance between the landfill and a drinking water well, as well as the potential rate of release of landfill leachate into groundwater, the concentration of this exposure can vary substantially.

Landfills are regulated under the Resource Conservation and Recovery Act (RCRA) or TSCA. RCRA landfills can be classified as Subtitle C (hazardous waste landfills) or Subtitle D (municipal solid nonhazardous waste landfills). Subtitle C establishes a federal program to manage hazardous wastes from cradle to grave. The objective of the Subtitle C program is to ensure that hazardous waste is handled in a manner that protects human health and the environment. When waste generators produce greater than 100 kg per month of non-acutely hazardous waste, those hazardous wastes, including 1,4-dioxane, meeting the U108 waste code description in 40 CFR 261.33, must be treated to meet the land disposal restriction levels in 40 CFR part 268 and be disposed in RCRA subtitle C landfills. These disposals are captured partially through the Toxics Release Inventory and are reported for onsite facilities (Table_Apx H-3) and offsite facilities (Table_Apx H-4). Recent violations of permits are reported in the footnotes of each table.

Review of state databases does not suggest any readily available evidence of groundwater contamination near or coinciding with these operations that could affect a drinking water supply. Similar review of the data available via the WQP suggests that there are no known contaminations from RCRA Subtitle C Landfills as reported to the TRI program (see Figure 2 14 and Figure 2 15). The absence of groundwater contamination near RCRA Subtitle C Landfills may be attributed to many of the ongoing engineering controls built into these facilities as well as active monitoring of groundwater wells around facilities. As a result, EPA did not assess Subtitle C landfills further than understanding their permit violations.

Regulations established under Subtitle D ban open dumping of waste and set minimum federal criteria for the operation of municipal waste and industrial waste landfills, including design criteria, location restrictions, financial assurance, corrective action (clean up), and closure requirements. States play a lead role in implementing these regulations and may set more stringent requirements. National requirements for Subtitle D landfills are most specific for MSW landfills. MSW landfills built after 1990 must be constructed with composite liner systems and leachate collection systems in place. Composite landfill liners consist of a minimum of two feet of compacted soil covered by a flexible membrane liner, which work in concert to create a low hydraulic conductivity barrier and prevent leachate from being released from the landfill and infiltrating to groundwater. A leachate collection system typically consists of a layer of higher conductivity material above the composite liner that funnels leachate to centralized collection points where it is removed from the landfill for treatment and disposal. Despite these controls, releases may still occur due to imperfections introduced during construction or that form over time (Li et al., 2015; Li et al., 2013; Mohr and DiGuiseppi, 2010); thus, groundwater monitoring is required to identify and address any releases before there can be harm to human health and the environment. RCRA Subtitle D requirements for non-MSW landfills are less stringent. In particular, nonhazardous industrial landfills and C&D debris landfills do not have specified national requirements for construction and operation and certain landfills are entirely exempt from RCRA criteria. Under the Land Disposal Program Flexibility Act of 1996 (Pub.L. 104-119), some villages in Alaska that dispose of less than 20 tons of municipal solid waste daily (based on an annual average) may dispose of waste in unlined or clay-lined landfills or waste piles for open burning or incineration.

There are a several potential sources of 1,4-dioxane to Subtitle D landfills. Waste generators that produce less than 100 kg per month of non-acutely hazardous waste, including 1,4-dioxane meeting the U108 waste code, may dispose of this waste in these landfills. Nonhazardous industrial wastes also have the potential to contain 1,4-dioxane at variable concentrations. Consumer and commercial products may also contain 1,4-dioxane in relatively low amounts. The greatest potential for release of disposed 1,4-dioxane to groundwater is from landfills that do not have an adequate liner system. Thus, an objective of this assessment is to evaluate the potential for groundwater contamination in the absence of landfill controls.

This assessment was completed using the Hazardous Waste Delisting Risk Assessment Software (DRAS). DRAS was specifically designed to address the Criteria for Listing Hazardous Waste identified in Title 40 Code of Federal Regulations (40 CFR) Section 261.11(a)(3), a requirement for evaluating proposed hazardous waste delistings. In this assessment, DRAS is being utilized to determine potential groundwater concentrations of 1,4-dioxane after they have been disposed of into a non-hazardous waste landfill. The results of this assessment are found in Table 2-13. This assessment relied on the default waste loading rates for RCRA Subtitle C Landfills available in DRAS. Similarly, the assessment relied on the default values for 1,4-dioxane as the chemical of concern. Lastly, leachate concentrations were estimated for a range of possibilities until no risk could be identified at the lower end of those concentrations. Because DRAS calculates a weight adjusted dilution attenuation factor (DAF) rather than a groundwater concentration, a back of the envelop computation was used to convert the DAF to a potential concentration that people living within 1 mile of a landfill might be exposed if the release were not identified and remediated.

Table 2-13. Potential Groundwater Concentrations (µg/L) of 1,4-Dioxane Found in Wells within 1 Mile of a Disposal Facility Determined by Using the DRAS Model

Leachate Concentration (µg/L)	**Loading Rate (kg)**									
	4.55E−04	4.55E−03	4.55E−02	4.55E−01	4.55E00	4.55E01	4.55E02	4.55E03	4.55E04	4.55E05
1.00E−07	7.81E−13	7.46E−12	5.46E−11	5.21E−10	6.49E−09	6.17E−08	5.88E−07	5.62E−06	5.38E−05	5.13E−04
1.00E−06	7.81E−12	7.46E−11	5.46E−10	5.21E−09	6.49E−08	6.17E−07	5.88E−06	5.62E−05	5.38E−04	5.13E−03
1.00E−05	7.81E−11	7.46E−10	5.46E−09	5.21E−08	6.49E−07	6.17E−06	5.88E−05	5.62E−04	5.38E−03	5.13E−02
1.00E−04	7.81E−10	7.46E−09	5.46E−08	5.21E−07	6.49E−06	6.17E−05	5.88E−04	5.62E−03	5.38E−02	5.13E−01
1.00E−03	7.81E−09	7.46E−08	5.46E−07	5.21E−06	6.49E−05	6.17E−04	5.88E−03	5.62E−02	5.38E−01	5.13E00
1.00E−02	7.81E−08	7.46E−07	5.46E−06	5.21E−05	6.49E−04	6.17E−03	5.88E−02	5.62E−01	5.38E00	5.13E01
1.00E−01	7.81E−07	7.46E−06	5.46E−05	5.21E−04	6.49E−03	6.17E−02	5.88E−01	5.62E00	5.38E01	5.13E02
1.00E00	7.81E−06	7.46E−05	5.46E−04	5.21E−03	6.49E−02	6.17E−01	5.88E00	5.62E01	5.38E02	5.13E03
1.00E01	7.81E−05	7.46E−04	5.46E−03	5.21E−02	6.49E−01	6.17E00	5.88E01	5.62E02	5.38E03	5.13E04

Concentrations organized by potential loading rates (kg) and potential leachate concentrations (µg /L).

2.3.2.3.1 Summary of Assessment for Disposal to Landfills

EPA determined through modeling that groundwater concentration of 1,4-dioxane increased with increasing landfill load rate and increasing leachate concentration. With each progressive iteration of loading rate or leachate concentration, potential groundwater concentrations increase by an order of magnitude. When both loading rate and leachate increase by one order of magnitude, potential groundwater concentration increase by two orders of magnitude. These increases can largely be attributed to the increasing weight adjusted dilution attenuation factor and are what would be expected for a chemical substances with 1,4-dioxane's physical-chemical properties (water solubility, Henry's law constant) and fate characteristics (biodegradability, half-life in groundwater). 1,4-Dioxane migrates in groundwater at the rate of hydraulic flow and can persist for greater than 30 days in anaerobic environments (Adamson et al., 2014; Mohr and DiGuiseppi, 2010) as described in the 2020 RE. Thus, these concentrations are likely to represent the range of potential groundwater concentrations for sentinel PESS living within a 1-mile radius of a RCRA Subtitle D landfills and other non-Subtitle C landfills.

EPA also determined that the modeled concentrations are within the range of concentrations of 1,4-dioxane found in groundwater monitoring studies. A survey of monitoring studies in California has demonstrated that 1,4-dioxane concentrations in groundwater can range from 9 µg/L at 10th percentile to 13,460 µg/L at the 90th percentile (Adamson et al., 2014). Monitoring data from EPA's Third Unregulated Contaminant Monitoring Rule (UCMR3) reported 1,4-dioxane concentrations in groundwater ranging from 0.07 µg/L to 34 µg/L (Adamson et al., 2017; U.S. EPA, 2017d). Though many of the corresponding sites in these monitoring surveys are not specifically tied to the disposal of 1,4-dioxane to landfills, they provide context for what concentrations may be expected when contamination occurs. When focusing on groundwater concentrations of 1,4-dioxane surrounding landfills based on reasonably available information, EPA found concentrations of 1,4-dioxane ranging from 6.4 to 25 mg/L (Cordone et al., 2016). Leaching from unlined lagoons in Michigan resulted in groundwater concentrations highs ranging from 1,000 to 20,000 µg/L (Jackson and Lemke, 2019; Mohr and DiGuiseppi, 2010); four decades later concentrations are now reaching 2 mg/L or less after active treatment and natural attenuation. Mean concentrations of 1,4-dioxane in landfill leachate in the United States has ranged from 11.8 µg/L for municipal landfills to 44.6 µg/L for hazardous waste landfills (as described in (Mohr and DiGuiseppi, 2010)). These concentrations further support that the modeled concentrations are within the range of those reported in the literature.

2.3.2.3.2 Strengths, Limitations, and Sources of Uncertainty in Assessment Results for Disposal to Landfills

Uncertainties and limitations are inherent in the modeling of groundwater concentrations from disposing chemical substances into RCRA Subtitle D landfills and other non-Subtitle C landfills. These uncertainties include, but are not limited to, determining the total and leachable concentrations of waste constituents; estimating the release of pollutants from the waste management units to the environment; and, estimating transport of pollutants in a range of variable environments by process that often are not completely understood or are too complex to quantify accurately. To address some of these uncertainties and add strength to the assessment, EPA considered multiple loading rates and multiple leachate concentrations. These considerations add value to estimate exposure that falls at an unknown percentile of the full distribution of exposures.

A strength of the assessment is that the modeled data are within the range of monitoring data that have been evaluated at both the national scale (Adamson et al., 2017; U.S. EPA, 2017d) and local scales (Li et al., 2015; Li et al., 2013; Mohr and DiGuiseppi, 2010). However, the modeled results may not represent current conditions of waste management units in the United States. Both the DRAS model and EPACMTP are based on a survey of drinking water wells located downgradient from a waste management unit (U.S. EPA, 1988). Due to the age of the survey, it is unclear how the survey represents current conditions and proximity of drinking water wells to disposal units. Similarly, it is not clear if the surveyed waste management units are representative of current waste management practices.

2.3.2.4 Disposal of Hydraulic Fracturing Produced Water to Surface Impoundments

After hydraulic fracturing operations inject fluids to extract oil and gas, a substantial volume of water may be produced through flowback. Otherwise known as produced waters, the composition of this water depends both on the geochemistry of the injected area and the injected fluids (U.S. EPA, 2016a). 1,4-Dioxane has been reported to EPA as one of the chemicals present in produced waters by 411 facilities via FracFocus 3.0 (GWPC and IOGCC, 2022) (Table 2-14). A variety of options exist for these produced waters after use in hydraulic fracturing operations ranging from underground injection, treatment and subsequent use, treatment and discharge, or evaporation in surface impoundments. Each of these options are subject to state and federal regulations (U.S. EPA, 2016a). When produced waters are released to unlined surface impoundments, there is potential for groundwater contamination and subsequent human exposure via drinking water. Thus, EPA conducted an assessment to determine the range of groundwater concentrations within a 1-mile radius of surface impoundments receiving produced water from hydraulic fracturing operations.

Based on the results of Monte Carlo analysis presented in Table_Apx E-5, disposal to these surface impoundments could account for up to 3 percent of all produced waters. 1,4-Dioxane has been documented to have a concentration of 60 µg/L in these produced waters (Lester et al., 2015). Thus, EPA assessed the potential for disposing of hydraulic fracturing produced water at the 5th, 50th, 95th, and 99th percentiles as well as at the min, mean, and max to a managed surface impoundment assuming these loading rates and concentration using DRAS. The results are presented in Table 2-14.

2.3.2.4.1 Summary of Assessment for Disposal of Hydraulic Fracturing Produced Water

In general, EPA determined that groundwater concentrations of 1,4-dioxane would increase as more produced water was released to surface impoundments. The values presented in Table 2-14 represent the maximum 33-year receptor well concentration within a 1-mile radius of a hypothetical surface impoundment that leaches into groundwater. With each progressive iteration of summary statistic for loading rate, potential groundwater concentrations increase accordingly. This increase can be attributed to the decrease in the weight-adjusted dilution attenuation factor. As the mass of 1,4-dioxane entering an aquifer increases, it is less diluted and higher concentrations will be found downgradient. Due to its physical-chemical properties (*e.g.*, water solubility, Henry's law constant) and fate characteristics (*e.g.*, biodegradability, half-life in groundwater), 1,4-dioxane migrates in groundwater and can persist for greater than 30 days in anaerobic environments (Adamson et al., 2014; Mohr and DiGuiseppi, 2010) as described in the 2020 RE. Thus, these concentrations are likely to represent the range of potential groundwater concentrations for people living within a 1-mile radius of a surface impoundment, representing a sentinel PESS exposure.

Table 2-14. Total Annual Release Summary

Total Annual Release (kg/site-year)	Monte Carlo Analysis Summary Statistic	Weight Adjusted Dilution Attenuation Factor	Potential Groundwater Concentration (µg/L)
1.68E00	Max	3.18	1.89E−05
01.87E−01	99th Percentile	3.91	1.54E−05
6.52E−02	95th Percentile	3.91	1.54E−05
1.47E−02	Mean	84	7.10E−07
3.83E−03	50th Percentile	495	1.20E−07
3.24E−05	5th Percentile	495	1.20E−07
1.06E−11	Min	135,000	0.00E00

2.3.2.4.2 Strengths, Limitations, and Sources of Uncertainty in Assessment Results for Disposal from Hydraulic Fracturing Operations

Although it is well understood that 1,4-dioxane is present in produced waters from hydraulic fracturing as reported in FracFocus (GWPC and IOGCC, 2022), the number of studies reporting the concentration of the chemical substances in produced waters is limited (Lester et al., 2015). FracFocus is generally considered a moderately reliable source of information as it is based on data from thousands of fracking wells across the United States. Further, both the release assessment (as discussed in Section 2.2.1.2; see also Table_Apx E-7) and the groundwater concentration assessment (Table 2-14) are modeled using a Monte Carlo simulation. These conditions lower the confidence in the overall assessment.

2.3.3 Ambient Air Pathway

EPA developed and applied tiered methodologies and analyses to estimate ambient air concentrations and exposures to members of the general population. These methodologies and analyses focus on inhalation exposures to a sub-set of the general population referred to as fenceline communities. Fenceline communities are defined as a subset of the general population that are in proximity to air emitting facilities or a receiving water body, and who therefore may be disproportionately exposed to a chemical undergoing risk evaluation under TSCA section 6(b). For the air pathway, proximity goes out to 10,000 meters from an air emitting source. The methodology and analyses were first presented in the *Draft TSCA Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities* referred to here as the 2022 Fenceline Report.[10] The specific methodologies used in this assessment to evaluate general population exposures to 1,4-dioxane in air are briefly described in Figure 2-18. Additional details on the methodologies are provided in Appendix J.

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[10] The 2022 Fenceline Report is available at https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/tsca-screening-level-approach-assessing-ambient-air-and.

Ambient Air: Screening Methodology

- Methodology is independent of facility and use classifications. Analysis broadly estimates ambient air concentrations and associated exposures/risks based on maximum and mean releases at three pre-defined distances from a releasing facility. Designed to inform whether application of higher-tier analysis methodology is warranted.

Ambient Air: Single Year Methodology (AERMOD or IIOAC)

- Methodology is facility and scenario specific. Analysis evaluates ambient air concentrations and associated exposures/risks resulting from facility-specific releases across multiple distances from the source. Utilizes a single year of release data, but can be expanded to utilize multiple-years of release data from multiple data sets/sources.

Ambient Air: Multi-Year Analysis Methodology (IIOAC)

- Methodology is facility and scenario specific. Analysis evaluates ambient air concentrations and associated exposures/risks resulting from facility-specific releases at three pre-defined distances from a releasing facility. Utilizes multiple years of release data reported to TRI. Developed in response to SACC comments/recommendations on the 2022 Fenceline Report to consider multiple years of release data to estimate exposures and associated risks.

Figure 2-18. Brief Description of Methodologies and Analyses Used to Estimate Ambient Air Concentrations and Exposures

EPA used the air release estimates obtained using the methodology described in Section 2.1.1.3 as direct inputs for the models used to estimate exposure concentrations at various distances from a releasing facility. EPA expanded upon the methods described in the 2022 Fenceline Report in response to SACC comments/recommendations by evaluating potential aggregate concentrations from multiple facilities.

2.3.3.1 Measured Concentrations in Air

EPA did not identify quantitative outdoor air monitoring data for 1,4-dioxane.

2.3.3.2 Modeled Concentrations in Air

Because there is no air monitoring data for 1,4-dioxane, the Agency relied upon modeling to estimate exposure concentrations to fenceline communities at various distances from a releasing facility. Modeling was used for each analysis described in Figure 2-18 for 1,4-dioxane. For scenarios where the screening methodology indicated a need for further analysis, EPA performed a full analysis using the American Meteorological Society/Environmental Protection Agency Regulatory Model (AERMOD) and/or EOA's Integrated Indoor/Outdoor Air Calculator (IIOAC).[11] IIOAC analysis was performed for three COUs where no site specific data were available (Hydraulic Fracturing, Industrial Laundry Facilities, Institutional Laundry Facilities) and is briefly described in Section 2.3.3.2.4 with results presented and discussed in Sections 5.2.2.3.2 and 5.2.2.3.3. An expanded analysis to consider aggregate exposures was performed for 1,4-dioxane in response to SACC comments/recommendations on the 2022 Fenceline Report.

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[11] The IIOAC website is available at https://www.epa.gov/tsca-screening-tools/iioac-integrated-indoor-outdoor-air-calculator.

2.3.3.2.1 Ambient Air: Screening Methodology

The Ambient Air: Screening Methodology utilizes EPA's IIOAC model to estimate high-end and central tendency (mean) 1,4-dioxane exposure concentrations in ambient air at three distances from an emitting facility: 100, 100 to 1,000, and 1,000 m. EPA developed and evaluated a range of exposure scenarios for each of two categorical release amounts[12] designed to capture a variety of release types, topography, meteorological conditions, and release scenarios. A diagram of these exposure scenarios is provided in Appendix J. Findings from the Ambient Air: Screening Methodology were used to inform the need for a higher-tier analysis as well as provide insight into whether risk estimates above the benchmarks are or are not expected for 1,4-dioxane.

The Ambient Air: Screening Methodology design inherently includes both estimates of exposures as well as estimates of risks to inform the need, or potential need, for further analysis. If findings from the Ambient Air: Screening Methodology estimate risk (acute non-cancer, chronic non-cancer, or cancer) for a given chemical above (or below as applicable) typical Agency benchmarks, EPA generally will conduct a higher-tier analysis of exposures and associated risks for that chemical. If findings from the Ambient Air: Screening Methodology estimate risks that do not exceed (or fall below as applicable) benchmarks, EPA may still conduct a limited higher-tier analysis at distances very near a releasing facility (less than 100 m) to ensure potential risks are not missed.

A more detailed description of the Ambient Air: Screening Methodology for 1,4-dioxane is provided in Appendix J, along with summarized results. In general, for 1,4-dioxane, the results of this analysis identified risk estimates above screening benchmarks for cancer at multiple distances and for multiple releases (max and mean). In accordance with the tiered methodology presented to the SACC in the 2022 Fenceline Report, because risk estimates exceeded the benchmark for cancer based on the Ambient Air: Screening Methodology, EPA conducted multiple higher-tier analyses of all facilities reporting releases of 1,4-dioxane to TRI.

2.3.3.2.2 Ambient Air: Single Year Methodology (AERMOD)

The Ambient Air: Single Year Methodology (AERMOD) utilizes the EPA's American Meteorological Society/Environmental Protection Agency Regulatory Model (AERMOD)[13] to estimate 1,4-dioxane concentrations in ambient air at eight finite distances (5, 10, 30, 60, 100, 2,500, 5,000, and 10,000 m) and one area distance from an emitting facility.[14] EPA modeled two different types of release estimates, as applicable, for 1,4-dioxane: (1) facility-specific chemical releases with source attribution when 2019 TRI data was available, and (2) alternative release estimates representing a generic facility when 2019 TRI data was not available for an OES. Daily and period average outputs were obtained via modeling, and post-processing scripts were used to extract a variety of statistics from the modeled concentration distribution, including the 95th (high-end), 50th (central tendency), and 10th (low-end) percentile 1,4-dioxane concentrations at each distance modeled.

A summary of the concentration ranges estimated using the Ambient Air: Single Year Methodology (AERMOD) is provided in Table 2-15. The summary includes 11 OESs and select statistics (maximum,

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[12] The pre-screening methodology from the 2022 fenceline analysis evaluated two categorical release values across all facilities reporting releases to the 2019 TRI. The first is the maximum single facility release reported across all facilities; the second is the mean (arithmetic average) of all releases reported across all facilities reporting.

[13] See https://www.epa.gov/scram/air-quality-dispersion-modeling-preferred-and-recommended-models#aermod for more information.

[14] For the one "area distance" evaluated, receptors are placed in a cartesian grid between approximately 200 and 900 m, at 100 m spacing. This results in a total of 456 receptors. The exposure estimates for the area distance represent the arithmetic average (mean) exposure concentration across all 456 receptors within the "area distance" for each day.

mean, median, and minimum) calculated from the modeled concentration distributions within each OES at each distance modeled. The associated range of estimated concentrations is based on the maximum 95th percentile annual average exposure concentrations for each distance. Although the range of concentrations are provided, there are many instances where the range extends as many as 12 orders of magnitude from minimum to maximum concentration. This occurs because within each OES there are several individual facilities evaluated and, in most cases, the reported release values from each individual facility can vary widely (from 500 lb to several hundred thousand pounds), which in turn affects the range of estimated exposure concentrations at a given distance. Therefore, in trying to summarize the wide variety of releases into a single range, the variation in estimated concentrations will also appear extensive. This is not indicative of an inadequate analysis or methodology, but solely based on the variability of releases across facilities within a given OES.

Table 2-15. Summary of Select Statistics for the 95th Percentile Estimated Annual Average Concentrations from the "Full-Screening" Analysis for 1,4-Dioxane Releases Reported to TRI

OES	# Facilities Evaluated in OES	Statistic	Annual Average Concentration (ppm) Estimated within 5 to 10,000 m of Releasing Facilities								
			5	10	30	60	100	100 to 1,000	2,500	5,000	10,000
Disposal	15	Max	4.26E−03	5.05E−03	1.80E−03	6.90E−04	3.15E−04	2.95E−05	1.89E−06	6.28E−07	2.09E−07
		Mean	4.76E−04	6.92E−04	2.79E−04	1.12E−04	5.39E−05	6.00E−06	3.46E−07	1.18E−07	4.12E−08
		Median	8.44E−06	1.65E−05	9.35E−06	8.70E−06	5.81E−06	7.64E−07	4.53E−08	1.48E−08	4.81E−09
		Min	3.31E−15	9.85E−14	5.17E−11	9.72E−10	2.03E−09	1.10E−09	1.21E−10	3.81E−11	1.22E−11
Dry Film Lubricant	8	Max	1.61E−10	7.14E−09	5.10E−07	3.88E−06	6.29E−06	9.92E−07	2.79E−08	8.44E−09	3.68E−09
		Mean	2.06E−11	9.46E−10	1.90E−07	2.28E−06	4.05E−06	8.14E−07	1.95E−08	5.94E−09	2.45E−09
		Median	2.46E−13	3.58E−11	1.59E−07	2.21E−06	4.00E−06	7.75E−07	1.88E−08	6.02E−09	2.66E−09
		Min	4.05E−18	2.19E−13	5.64E−08	9.23E−07	2.39E−06	7.39E−07	1.36E−08	4.02E−09	1.40E−09
Ethoxylation Byproduct	6	Max	6.53E−03	1.36E−02	7.33E−03	3.09E−03	1.64E−03	3.81E−04	2.20E−05	9.00E−06	3.45E−06
		Mean	1.74E−03	3.05E−03	1.49E−03	6.18E−04	3.23E−04	7.03E−05	4.02E−06	1.62E−06	6.13E−07
		Median	2.44E−04	4.40E−04	2.08E−04	8.70E−05	4.40E−05	6.36E−06	3.09E−07	1.03E−07	3.40E−08
		Min	4.08E−14	6.32E−13	4.29E−10	5.22E−09	1.15E−08	4.99E−09	7.17E−10	3.39E−10	1.40E−10
Film Cement	1	Max	1.25E−04	1.31E−04	4.41E−05	2.28E−05	1.25E−05	2.29E−06	1.38E−07	4.60E−08	1.52E−08
		Mean	3.90E−05	4.87E−05	2.04E−05	9.36E−06	4.82E−06	7.95E−07	4.44E−08	1.48E−08	4.87E−09
		Median	2.02E−05	2.93E−05	1.74E−05	7.33E−06	3.57E−06	5.68E−07	2.48E−08	8.10E−09	2.65E−09
		Min	3.17E−06	6.36E−06	4.97E−06	1.95E−06	8.99E−07	1.32E−07	4.44E−09	1.42E−09	4.61E−10
Functional Fluids (Open-System)	2	Max	1.28E−05	2.36E−05	1.03E−05	1.08E−05	1.82E−05	7.42E−06	6.78E−07	2.47E−07	8.81E−08
		Mean	6.40E−06	1.18E−05	5.74E−06	7.71E−06	1.08E−05	4.24E−06	3.88E−07	1.45E−07	5.28E−08
		Median	6.40E−06	1.18E−05	5.74E−06	7.71E−06	1.08E−05	4.24E−06	3.88E−07	1.45E−07	5.28E−08
		Min	1.66E−11	1.93E−10	1.18E−06	4.61E−06	3.37E−06	1.06E−06	9.70E−08	4.30E−08	1.74E−08
Import and Repackaging	1	Single Facility	2.70E−11	5.57E−10	5.52E−08	4.17E−07	8.70E−07	3.21E−07	6.72E−08	4.12E−08	2.23E−08

OES	# Facilities Evaluated in OES	Statistic	Annual Average Concentration (ppm) Estimated within 5 to 10,000 m of Releasing Facilities								
			5	**10**	**30**	**60**	**100**	**100 to 1,000**	**2,500**	**5,000**	**10,000**
Industrial Uses	12	Max	4.19E−03	4.78E−03	1.54E−03	5.67E−04	2.80E−04	7.15E−05	8.79E−06	3.22E−06	1.13E−06
		Mean	8.76E−04	1.14E−03	4.08E−04	1.70E−04	9.63E−05	1.69E−05	1.48E−06	5.45E−07	1.94E−07
		Median	8.76E−05	1.14E−04	3.83E−05	1.65E−05	9.94E−06	2.23E−06	3.19E−07	1.17E−07	4.04E−08
		Min	7.75E−13	1.69E−12	2.40E−09	2.50E−08	1.23E−08	1.10E−09	6.36E−11	1.97E−11	6.14E−12
Laboratory Chemical Use	1	Max	2.06E−03	2.15E−03	7.26E−04	3.75E−04	2.06E−04	3.76E−05	2.27E−06	7.57E−07	2.50E−07
		Mean	6.84E−04	8.52E−04	3.58E−04	1.64E−04	8.46E−05	1.39E−05	7.77E−07	2.59E−07	8.55E−08
		Median	4.30E−04	5.65E−04	3.15E−04	1.36E−04	6.68E−05	1.08E−05	4.82E−07	1.59E−07	5.24E−08
		Min	7.39E−05	1.48E−04	1.16E−04	4.55E−05	2.09E−05	3.08E−06	1.03E−07	3.30E−08	1.07E−08
Manufacturing	1	Single Facility	8.73E−03	1.63E−02	7.69E−03	3.22E−03	1.59E−03	1.42E−04	8.21E−06	2.54E−06	7.92E−07
PET Manufacturing	13	Max	8.01E−03	9.57E−03	3.50E−03	1.40E−03	6.43E−04	1.07E−04	2.07E−05	1.24E−05	6.58E−06
		Mean	1.41E−03	1.89E−03	7.83E−04	3.36E−04	1.85E−04	3.31E−05	4.23E−06	2.08E−06	9.60E−07
		Median	8.00E−04	1.64E−03	5.21E−04	2.27E−04	1.42E−04	2.64E−05	2.48E−06	1.09E−06	3.94E−07
		Min	6.04E−12	8.54E−11	3.01E−08	2.43E−07	5.56E−07	3.02E−07	4.33E−08	2.07E−08	9.30E−09
Spray Foam Application	1	Max	7.79E−07	8.40E−07	2.85E−07	1.50E−07	8.55E−08	1.55E−08	1.72E−09	6.30E−10	2.45E−10
		Mean	2.68E−07	3.30E−07	1.34E−07	6.21E−08	3.28E−08	5.29E−09	4.78E−10	1.67E−10	5.97E−11
		Median	1.41E−07	1.95E−07	1.14E−07	4.88E−08	2.36E−08	3.64E−09	2.25E−10	7.40E−11	2.40E−11
		Min	2.51E−08	4.43E−08	3.45E−08	1.36E−08	6.07E−09	8.42E−10	3.26E−11	1.10E−11	3.74E−12

Based on the air concentrations estimated through the Ambient Air: Single Year Methodology, EPA also estimated potential aggregate air concentrations resulting from the combined releases of multiple facilities in proximity to each other. Details of the methods used to aggregate exposure and corresponding risk are presented in Appendix J.4.

2.3.3.2.3 Ambient Air: Multi-Year Analysis (IIOAC)

The multi-year analysis utilizes EPA's IIOAC model to estimate high-end and central tendency (mean) 1,4-dioxane concentrations in ambient air at three distances from an emitting facility: 100, 100 to 1,000, and 1,000 m. The multi-year analysis incorporates SACC recommendations on the 2022 Fenceline Report by evaluating multiple years of chemical release data to estimate exposures and associated risks to fenceline communities. This is achieved for 1,4-dioxane by expanding upon methodologies described in the 2022 Fenceline Report and conducting a facility-by-facility evaluation of all 1,4-dioxane releases reported to TRI (2015 through 2020). Data for these 6 years were obtained from the TRI database (TRI basic plus files downloaded on August 5, 2022). Annual release data for 1,4-dioxane were extracted from the entire TRI data set for all facilities reporting air releases of 1,4-dioxane for one or more years between 2015 and 2020. Facilities were categorized into occupational exposure scenarios for modeling purposes to inform the release scenarios evaluated.

The multi-year analysis provides highlights of the year-to-year variability that exists in the release data and illustrates the potential impact of considering multiple years of TRI data on exposure and risk estimates. The findings from the multi-year analysis can also be used in a comparative manner to determine how representative the single year of data used for the Ambient Air: Single Year Methodology (AERMOD) presented in the 2022 Fenceline Report is or to provide additional confidence in the findings from the Ambient Air: Single Year Methodology (AERMOD) described in the 2022 Fenceline Report for purposes of estimating exposures and associated risks to fenceline communities. In broader terms, the multi-year analysis provides both a broad analysis of multiple years of release data and enables a general comparison to the Ambient Air: Single Year Methodology (AERMOD) results described above and in the 2022 Fenceline Report.

2.3.3.2.4 Ambient Air: IIOAC Methodology for COUs without Site-Specific Data (Hydraulic Fracturing, Industrial, and Institutional Laundry Facilities)

For COUs without site-specific data, EPA's IIOAC model was used to estimate high-end and central tendency (mean) 1,4-dioxane concentrations in ambient air at three distances from an emitting facility (100, 100 to 1,000, and 1,000 m). This methodology was applied for three unique COUs (hydraulic fracturing, and industrial, and institutional laundry facilities) where there was no site-specific data available for modeling. Environmental releases (fugitive and stack) along with other data (like days of release) for these COUs were estimated using Monte Carlo modeling. As such, the Ambient Air: IIOAC Methodology for COUs without site-specific data was developed to allow modeling all possible iterations of releases provided across eight different exposure scenarios, including consideration of source attribution as well as actual days of release. Additionally, the product form for laundry detergent was provided, allowing for analysis of releases associated with detergent in vapor only form, as well as solid form (particulate) either coarse (PM10) or fine (PM2.5). A description of this methodology is provided in Appendix J along with a summary of the model inputs and exposure scenarios evaluated. A full list of the inputs, exposure scenarios, and results is provided in *1,4-Dioxane Supplemental Information File: Air Exposure and Risk Estimates for 1,4-Dioxane Emissions from Hydraulic Fracturing Operations* (U.S. EPA, 2023b) and *1,4-Dioxane Supplemental Information File: Air Exposures and Risk Estimates for Industrial Laundry* (U.S. EPA, 2023c).

2.3.3.3 Strengths, Limitations, and Sources of Uncertainty for Modeled Air Concentrations

EPA has medium to high confidence in the air concentrations estimated from TRI release data using IIOAC and AERMOD.

IIOAC

IIOAC estimates air concentrations at three pre-defined distances (100, 100 to 1,000, and 1,000 m). The inherent distance limitations of IIOAC do not allow estimation of exposures closer to a facility (less than 100 m from the facility) where we expect to see higher exposures from fugitive releases.

IIOAC uses meteorological data from 14 pre-defined meteorological stations representing large regions across the United States. This generalizes the meteorological data used to estimate exposure concentrations where competing conditions can influence the exposure concentrations modeled upwind and downwind of a releasing facility. To reduce the uncertainties associated with using regional meteorological data, EPA conducted a sensitivity analysis of all 14 pre-defined meteorological stations to identify which two within IIOAC tended to result in a high-end and central tendency estimate of exposure concentrations. This maintained a more conservative exposure concentration estimate which is then used in calculations to estimate risks. This approach adds confidence to the findings by ensuring, potential risks would be captured under a high-end exposure scenario, while also providing insight into potential risks under a less conservative exposure scenario (central tendency).

AERMOD

AERMOD is an EPA regulatory model and has been thoroughly peer reviewed; therefore, the general confidence in results from the model is high but relies on the integrity and quality of the inputs used and interpretation of the results. For the full analysis, EPA used releases reported to the 2019 TRI as direct inputs to AERMOD. Although there is some uncertainty around the representativeness of using only a single year of data, AERMOD successfully estimated exposure concentrations to fenceline communities. Furthermore, in response to SACC recommendations to use multiple years of data to estimate exposures and associated risks, EPA developed the IIOAC and conducted a multi-year analysis using 6 years of TRI data and compared the results to those of AERMOD and found exposure concentration estimates from the 2019 data is generally representative of other years.

AERMOD relied upon the latitude/longitude information reported by each facility to TRI as the location for the point of release. While this may generally be a close approximation of the release point for a small facility (for example a single building), it may not represent the release point within a much larger facility. Therefore, there is some uncertainty associated with the modeled distances from each release point and the associated exposure concentrations to which fenceline communities may be exposed. For small facilities where the latitude/longitude may closely approximate the release point, there is a less uncertainty that the estimated exposure at the associated distance is representative of exposure to fenceline communities at that distance. For larger facilities, where the latitude/longitude may be several hundred meters away from the actual release point, there is a higher uncertainty that the estimated exposure at the associated distance is representative of exposure to fenceline communities at that distance.

The TRI datasets used for both AERMOD and IIOAC do not include source specific stack parameters that can affect plume characteristics and associated dispersion of the plume. Therefore, EPA used pre-defined stack parameters within IIOAC to represent stack parameters of all facilities modeled using each of these methodologies. Those stack parameters include a stack height 10 m above ground with a 2-meter inside diameter, an exit gas temperature of 300° Kelvin, and an exit gas velocity of 5 m per second (see Table 6 of the IIOAC User Guide). These parameters were selected since they represent a slow-moving, low-to-the-ground plume with limited dispersion which results in a more conservative estimate of exposure concentrations at the distances evaluated. As such, these parameters may result in some overestimation of emissions for certain facilities modeled.

Additionally, the assumption of a 10×10 area source for fugitive releases may impact the exposure estimates very near a releasing facility (5 and 10 m from a fugitive release). This assumption places the receptor at 5 m directly on top of the release point which may result in an over or underestimation of exposure. This assumption places the 10-meter receptor just off the release point that may again result in either an over or underestimation of exposure depending on other factors like meteorological data, release heights, and plume characteristics.

For facilities reporting releases to TRI via a TRI Form A (which is allowed for use by those facilities releasing less than 500 lb of the chemical reported), EPA assumed the maximum release value of 500 lb for exposure modeling purposes. TRI Form A reporters do not provide source attribution (fugitive or stack releases) so EPA modeled each facility associated with a Form A submittal twice—once assuming all 500 lb of the reporting threshold was fugitive and once assuming all 500 lb of the reporting threshold was stack. There is no way to attribute a certain portion of the releases to each release type, so this modeling approach represents a conservative estimate, in terms of total release, but may overestimate exposure concentrations associated with each release type if a facility did not actually release all 500 lb via a single release type or even combined release type. To avoid the potential double counting of facility releases for TRI Form A reporters, when presenting potential exposures EPA presented only the highest (more conservative) exposure concentration estimated for either of the two release types for purposes of evaluating potential risks to fenceline communities. Given the exposure scenarios modeled, this tended to result from the exposure scenario which assumed all 500 lb of the release were fugitive releases.

3 HUMAN EXPOSURES

**1,4-Dioxane – Human Exposures (Section 3):
Key Points**

EPA evaluated reasonably available information for occupational exposures to 1,4-dioxane present as a byproduct, and general population, including PESS, exposures to 1,4-dioxane present in the environment.

- Occupational exposures for workers and ONUs to 1,4-dioxane present as a byproduct occur during manufacturing, through use of commercial products, or in hydraulic fracturing operations.
- General population exposures to 1,4-dioxane occur when 1,4-dioxane is present in potential drinking water sources or ambient air, particularly in fenceline communities.
- EPA considered the potential for increased exposures across PESS factors throughout the exposure assessment. PESS categories incorporated into this supplemental exposure assessment include
  - Lifestage (including formula-fed infant exposures),
  - Occupational exposures (including high-end exposure scenarios), and
  - Geography/site-specific factors (*i.e.*, fenceline community exposures)

3.1 Occupational Exposures

**1,4-Dioxane – Occupational Exposures (Section 3.1):
Key Points**

EPA considered the reasonably available information to evaluate occupational exposures.

- EPA estimated occupational exposures to 1,4-dioxane through air and skin. EPA estimated both high-end and central tendency exposures for occupational exposure scenarios associated with each COU.
- Exposure for most COUs was estimated based on monitoring data. For COUs without monitoring data, EPA applied Monte Carlo statistical modeling approaches to estimate exposures.

The following sections describe EPA's approach to assessing occupational exposures for OESs involving industrial and commercial products containing 1,4-dioxane as a byproduct. The assessed OESs include textile dye, antifreeze, surface cleaner, dish soap, dishwasher detergent, institutional and industrial laundries, paints and floor lacquer, PET byproducts, ethoxylation process byproducts, and hydraulic fracturing. For a crosswalk linking COUs to OESs, see Table 2-1. The remaining OESs have occupational exposure assessments in Section 2.4.1 of the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

EPA distinguishes between exposures to workers and exposures to ONUs. Normally, workers may handle 1,4-dioxane and have direct contact with the chemical, while ONUs are working in the general vicinity of workers but do not handle 1,4-dioxane and do not have direct contact with 1,4-dioxane being handled by the workers.

EPA evaluated acute and chronic inhalation exposures to workers and ONUs, and dermal exposures to workers. EPA did not assess dermal exposures to ONUs as EPA does not expect ONUs to have routine dermal exposures in the course of their work.

The occupational exposure assessment for each COU comprises the following components:

- **Process Description** of the COU, including the role of the chemical in the use; process vessels, equipment, and tools used during the COU; and descriptions of the worker activities, including an assessment for potential points of worker exposure.
- **Number of Sites** that use the chemical for the given COU.
- **Number of Workers and ONUs** potentially exposed to the chemical for the given COU. Unless mentioned otherwise in this report, the total number of workers and ONUs are number of personnel per site per day. The details on estimation of the number of workers and ONUs are discussed below for each COU.
- **Central Tendency and High-End Estimates of Inhalation Exposure** to workers and occupational non-users. See "General Approach and Methodology for Environmental Releases" for a discussion of EPA's statistical analysis approach for assessing inhalation exposure.
- **Dermal Exposure** estimates for multiple scenarios, accounting for simultaneous absorption and evaporation, and different protection factors of glove use.
- **Users** include adult workers (greater than 16 years old) exposed to 1,4-dioxane for 8-hour exposure.
- **ONUs** include female and male adult workers (greater than16 years old) exposed to 1,4-dioxane indirectly by being in the same work area of the building.

3.1.1 Approach and Methodology

EPA developed occupational exposure values representative of central tendency (50th percentile, mean) conditions and high-end (90th and 99.9th percentiles). Additional explanation of central tendency and high-end conditions are described in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

3.1.1.1 Process Description, Number of Sites, Number of Workers, and ONUs

EPA performed a literature search to find descriptions of processes involving 1,4-dioxane and worker activities that could potentially result in occupational exposures. This literature search was specific to the scope of this draft supplement and is described in Section 1.4. A summary of the data quality evaluation results for the 1,4-dioxane occupational exposure sources are presented in the attachment *Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure* (U.S. EPA, 2023t).

EPA used a variety of sources to supplement the data found through the Systematic Review process. The additional sources included relevant NIOSH Health Hazard Evaluations, Generic Scenarios, and ESDs. These sources were sometimes used to provide process descriptions of the COUs as well as estimates for the number of sites and number of workers. Because CDR data were not available for the COUs included in this occupational exposure assessment, EPA used data from the Bureau of Labor Statistics (BLS) and the U.S. Census' Statistics of US Businesses (SUSB) to estimate the number of sites, workers, and ONUs for each OES. This approach involved the identification of relevant Standard Occupational Classification (SOC) codes within the BLS data for the identified NAICS codes for each OES. First, EPA identified the affected NAICS codes. Then, EPA reviewed occupation descriptions to designate which SOC codes contained potentially exposed workers and ONUs. The Agency refined the estimates by using U.S. Census Bureau data. Next, EPA estimated the percentage of workers using 1,4-dioxane instead of other chemicals to calculate number of workers per site. Finally, this data was separated by COU. Additional details on this approach can be found in Appendix G.5 of the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). These estimates were utilized in Monte Carlo modeling of environmental releases and occupational exposures.

3.1.1.2 Inhalation Exposures Approach and Methodology

EPA used inhalation monitoring data from literature sources having high or medium data quality ratings during data evaluation. EPA used modeling approaches to estimate potential inhalation exposures where inhalation monitoring data were not available.

The Agency reviewed workplace inhalation monitoring data collected by government agencies such as OSHA and NIOSH, and monitoring data found in published literature (*i.e.*, personal exposure monitoring data and area monitoring data). Central tendency and high-end exposure values were calculated from the monitoring data provided in the sources depending on the size of the dataset (*i.e.*, number of data points). Where discrete sampling points were not provided in the source and EPA was unable to calculate central tendency and high-end values, the Agency used values of central tendency and high-end that were provided in the source. EPA's approach for evaluating central tendency and high-end estimates from inhalation monitoring data is further discussed in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). EPA used the following types of monitoring data of 1,4-dioxane from various sources to estimate occupational inhalation exposure:

- Personal sample monitoring data from directly applicable scenarios (*e.g.*, personal breathing zone [PBZ]). This type of monitoring data was used for the textile dye, surface cleaner, dish soap, paint and floor lacquer, PET byproduct, and the Ethoxylation Process Byproduct OESs.
- Personal sample monitoring data from potentially applicable or similar scenarios. Specifically, PBZ data from the dish soap OES was also used for the dishwasher detergent OES because these OESs are expected to be similar.

EPA used the following models and modeling approaches to estimate occupational inhalation exposure where no monitoring data were found:

- Monte-Carlo statistical modeling approaches, which was used for the antifreeze, laundry detergent, and hydraulic fracturing OES. EPA developed these models for the purposes of this assessment. The models and the associated sources of data used in the modeling are described in detail in Appendices F.7, F.8, and F.9, respectively.
- Additional modeling approaches, including the use of surrogate data and fundamental modeling approaches for the spray polyurethane foam OES in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). While this OES is included in the scope of this draft supplement, EPA evaluated occupational exposure estimates for this OES in the published risk evaluation and these estimates remain unchanged in this draft supplement.
- *EPA AP-42 Loading Model* estimates vapor releases that occur when vapor is displaced by liquid during container loading. It calculates a vapor generation rate (G) using the physio-chemical properties of the chemical.
- *EPA Mass Balance Inhalation Model* estimates occupational inhalation exposures assuming the air immediately around the source of exposure behaves as a well-mixed zone. EPA used the vapor generation rate (G), calculated using the EPA AP-42 Loading Model, in conjunction with this model to develop estimates of inhalation exposure.

- *Generic Model for Central Tendency and High-End Inhalation Exposure to Total and Respirable Particulates Not Otherwise Regulated (PNOR)* estimates occupational inhalation exposures to particulates containing the chemical using OSHA PNOR data.

EPA did not utilize occupational exposure limits to estimate occupational inhalation exposures in this assessment because sufficient monitoring data or modeling approaches were available for all OES.

EPA then used measured or modeled air concentrations to calculate exposure concentration metrics essential for risk assessment. These exposures are presented as 8-hour time weighted averages (TWAs) and used to calculate average daily concentrations (ADCs) and lifetime average daily concentrations (LADCs). The ADC is used to estimate chronic, non-cancer risks and the LADC is used to estimate chronic, cancer risks. These calculations required additional parameter inputs, such as years of exposure, exposure duration and frequency, and lifetime years. See Appendix F.1 for more information about parameters and equations used to calculate acute and chronic exposures.

3.1.1.3 Dermal Exposures Approach and Methodology

EPA modeled dermal doses using the *EPA Dermal Exposure to Volatile Liquids Model*. This model determines a dermal potential dose rate based on an assumed amount of liquid on skin during one contact event per day and the steady-state fractional absorption for 1,4-dioxane. The amount of liquid on the skin is adjusted by the weight fraction of 1,4-dioxane in the liquid to which the worker is exposed. This is the same approach that EPA used in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

3.1.1.4 Engineering Controls and Personal Protective Equipment

EPA's occupational exposure estimates do not assume the use of engineering controls or PPE. Additionally, as discussed in Section 5.2.1, the risk estimates are based on exposures to workers in the absence of PPE such as gloves or respirators. Reasonably available monitoring data or information on effectiveness of engineering control and PPE for reducing occupational exposures to 1,4-dioxane during the assessed OES were not available. This section presents a general discussion on engineering controls and PPE for informative purposes only.

OSHA recommends employers utilize the hierarchy of controls for reducing or removing hazardous exposures. The most effective controls are elimination, substitution, or engineering controls. Respirators, and any other personal protective equipment (PPE), are the last means of worker protection in the hierarchy of controls and should only be considered when process design and engineering controls cannot reduce workplace exposure to acceptable levels. OSHA’s Respiratory Protection Standard (29 CFR 1910.134) provides a summary of respirator types by their assigned protection factor (APF). OSHA defines the APF to mean the workplace level of respiratory protection that a respirator or class of respirators is expected to provide to employees when the employer implements a continuing, effective respiratory protection program according to the requirements of the OSHA Respiratory Protection Standard. Exposure limits, respirator requirements, worker respirator use rates, and a table of APFs for different types of respirators are provided in the 2020 RE (U.S. EPA, 2020c).

OSHA’s hand protection standard (29 CFR 1910.138) states that employers must select and require employees to use appropriate hand protection when employees are expected to be exposed to hazards such as those from skin absorption of harmful substances; severe cuts or lacerations; severe abrasions; punctures; chemical burns; thermal burns; and harmful temperature extremes. Dermal protection selection provisions are provided in § 1910.138(b) and require that appropriate hand protection is selected based on the performance characteristics of the hand protection relative to the task(s) to be performed, conditions present, duration of use, and the hazards to which employees will be exposed.

Unlike respiratory protection, OSHA standards do not provide protection factors (PFs) associated with various hand protections. Data regarding the frequency of effective glove use in industrial and commercial settings is limited. While there are no regulatory standards containing dermal protection factors, the European Center for Ecotoxicity and Toxicology of Chemicals (ECETOC) targeted risk assessment (TRA) tool includes fixed, assigned protection factors equal to 5, 10, or 20 for various dermal protection strategies. These are discussed in Appendix F.3 and further explained in the 2020 RE (U.S. EPA, 2020c).

3.1.2 Occupational Exposure Estimates

In this section, EPA provides a summary of the exposure estimates for each OES, including estimates for number of workers and ONUs, inhalation exposures, and dermal exposures. For the crosswalk linking COU to OES, see Table 2-1. Note that EPA assessed dermal exposures for all OESs with the same methodology, which is described at the end of this section.

3.1.2.1 Summary of Inhalation Exposure Assessment

EPA estimated central tendency and high-end occupational inhalation exposures using various methods and information sources—including OSHA data, NIOSH health hazard evaluation data, and GSs and ESDs with Monte Carlo modeling. EPA estimated inhalation exposures as 8-hour TWA values for the COUs included in this draft supplement per Table 2-1. Using the estimated central tendency and high-end inhalation exposures with the estimated exposure frequency, EPA then calculated the cancer and non-cancer exposures using the calculations described in Appendix F.1.

A summary of the occupational inhalation exposures is presented Table_Apx F-32. EPA used monitoring data to estimate occupational inhalation exposures to workers for the textile dye, surface cleaner, dish soap, dishwasher detergent, paint and floor lacquer, PET byproduct, and ethoxylation byproduct conditions of use. This monitoring data was found to be relevant to these scenarios and based on medium to high data quality. However, several of the scenarios had a low number of samples and may have preceded changes in current industry practices. Additionally, sufficient representation of the entire industry is uncertain due to the limited number of sites. For the remaining conditions of use included in this draft supplement, which are antifreeze, laundry detergent, and hydraulic fracturing, EPA did not find reasonably available monitoring data and estimated worker inhalation exposure using GSs and ESDs with Monte Carlo modeling. The applied models are directly relevant to these conditions of use, but the underlying distributions may not sufficiently capture variability across entire industry sectors. For both measured and modeled data, the degree of certainty to which these data represent the true distribution of exposure and the potential over- or underestimation of exposure is unknown.

Monitoring data and modeling approaches were not available to estimate occupational inhalation exposures for ONUs. The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical.

The PET byproduct and textile dyes conditions of use had the highest central tendency and high-end worker inhalation exposure values, respectively. For PET byproduct, worker inhalation exposures were estimated using OSHA monitoring data, which resulted in central tendency exposure of 4.7 mg/m^3 and high-end exposure of 47 mg/m^3. For textile dyes, worker inhalation exposures were also estimated using OSHA monitoring data, which resulted in central tendency exposure of 0.066 mg/m^3 and high-end exposure of 74 mg/m^3.

The monitoring data sources and GSs and ESDs used to estimate occupational inhalation exposures all had overall data quality determinations of either medium or high. The basis for determining overall data quality is described in this section. In summary, each source is evaluated on multiple metrics based on defined criteria. The individual metric ratings are used to obtain an overall study rating. All metrics have an equal weight in determining the overall study rating. The resulting values are converted to an overall data quality determination of “high,” “medium,” “low,” “critically deficient,” or “not rated/not applicable.” For more details on this process, see Section 5 of the 2021 Draft Systematic Review Protocol

This section also includes information on the weight of the scientific evidence conclusions for these estimates, and a summary of the strengths, limitations, assumptions, and key sources of uncertainty for these estimates.

3.1.2.2 Summary of Dermal Exposures Assessment

Table 3-1 presents the estimated dermal absorbed dose for workers in various OES. The dose estimates assume one dermal exposure event (applied dose) per workday and that approximately 78 or 86 percent of the applied dose is absorbed through the skin (depending on whether the OES is industrial or commercial). The exposure estimates are provided for each OES, where the OES are “binned” based on characteristics known to effect dermal exposure such as the maximum weight fraction of 1,4-dioxane that could be present in that OES, open or closed system use of 1,4-dioxane, and large or small-scale use. For a more detailed description of EPA’s dermal assessment approach and each bin, see Appendix F.3.

As shown in the Table 3-1, the calculated dermal absorbed dose for workers is lower in comparison to those presented in the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). This is due to the relatively lower concentrations of 1,4-dioxane found for the OES included in this draft supplement than for those included in the 2020 RE. As noted previously, EPA did not assess dermal exposures to ONUs as the Agency does not expect ONUs to have routine dermal exposures in the course of their work. Depending on the OES, ONUs may have incidental dermal exposures due to surface contamination. However, data (*e.g.*, frequency and amount of liquid on the skin after contact) were not identified to assess this exposure.

Table 3-1. Estimated Dermal Absorbed Dose (mg/day) for Workers in Various Conditions of Use

OES	Bin	Use Setting	Weight Fraction (Max Y_{derm})	No Gloves (PF = 1)	Exposures Due to Glove Permeation/Chemical Breakthrough (mg/day)		
					Protective Gloves (PF = 5)	Protective Gloves (PF = 10)	Protective Gloves (Industrial Uses Only, PF = 20)
Textile Dye	7	Industrial and Commercial	4.7E−06	0.003 (CT) 0.009 (HE)	0.001 (CT) 0.002 (HE)	3.0E−4 (CT) 0.001 (HE)	1.5E−4 (CT) 4.5E−4 (HE)
Antifreeze	8	Commercial	8.6E−05	0.055 (CT) 0.165 (HE)	0.011 (CT) 0.033 (HE)	0.006 (CT) 0.017 (HE)	N/A
Surface Cleaner	9	Commercial	9.0E−06	0.006 (CT) 0.017 (HE)	0.001 (CT) 0.003 (HE)	0.001 (CT) 0.002 (HE)	N/A
Dish Soap	10	Commercial	2.04E−04	0.131 (CT) 0.393 (HE)	0.026 (CT) 0.079 (HE)	0.013 (CT) 0.039 (HE)	N/A
Dishwasher Detergent	11	Commercial	5.1E−05	0.033 (CT) 0.098 (HE)	0.007 (CT) 0.020 (HE)	0.003 (CT) 0.010 (HE)	N/A

OES	Bin	Use Setting	Weight Fraction (Max Y_{derm})	No Gloves (PF = 1)	Exposures Due to Glove Permeation/Chemical Breakthrough (mg/day)		
					Protective Gloves (PF = 5)	**Protective Gloves (PF = 10)**	**Protective Gloves (Industrial Uses Only, PF = 20)**
Laundry Detergent (Industrial and Institutional)	12	Industrial and Commercial	1.4E−05	0.009 (CT) 0.027 (HE)	0.002 (CT) 0.005 (HE)	0.001 (CT) 0.003 (HE)	4.5E−4 (CT) 0.001 (HE)
Paint and Floor Lacquer	13	Industrial and Commercial	3.0E−05	0.019 (CT) 0.058 (HE)	0.004 (CT) 0.012 (HE)	0.002 (CT) 0.006 (HE)	0.001 (CT) 0.003 (HE)
Polyethylene Terephthalate (PET) Byproduct	14	Industrial	1.4E−03	0.827 (CT) 2.48 (HE)	0.165 (CT) 0.496 (HE)	0.083 (CT) 0.248 (HE)	0.041 (CT) 0.124 (HE)
Ethoxylation Process Byproduct							
Hydraulic Fracturing	15	Industrial and Commercial	0.05	32.1 (CT) 96.2 (HE)	6.41 (CT) 19.2 (HE)	3.21 (CT) 9.62 (HE)	1.60 (CT) 4.81 (HE)
CT = central tendency; HE = high-end; PF = protection factor							

3.1.2.3 Weight of the Scientific Evidence Conclusions for Occupational Exposure Information

Table 3-2 provides a summary of EPA's overall weight of the scientific evidence conclusions for its occupational exposure estimates for each of the assessed OES. These determinations are OES-specific. For a description of overall confidence in all inhalation exposures, see Section 3.3.1.1. For an explanation of EPA's judgement on the weight of the scientific evidence conclusion, see Section 2.2.1.2. Factors that increase and decrease the strength of the weight of the scientific evidence are listed in Table_Apx C-5.

Due to a lack of data, EPA was not able to estimate ONU inhalation exposure from monitoring data or models, so a qualitative assessment of potential ONU exposures was made. Similarly, EPA did not assess dermal exposures to ONUs as EPA does not expect ONUs to have routine dermal exposures in the course of their work. Depending on the COU, ONUs may have incidental dermal exposures due to surface contamination. However, data (*e.g.*, frequency and amount of liquid on the skin after contact) were not identified to assess this exposure. Finally, due to the absence of dermal monitoring data, these columns were omitted from Table 3-2.

Table 3-2. Summary of the Weight of the Scientific Evidence for Occupational Exposure Estimates by OES

OES	**Inhalation Exposure**						**Dermal Exposure**	
	Monitoring			**Modeling**		**Weight of the Scientific Evidence**	**Modeling**	**Weight of the Scientific Evidence**
	Worker	**# Data Points**	**Data Quality Rating**[a]	**Worker**	**Data Quality Rating**[a]	**Worker**	**Worker**[b]	**Worker**
Textile Dye	✓	14	H	x	N/A	Moderate	✓	Moderate
Antifreeze	x	N/A	N/A	✓	H	Moderate	✓	Moderate
Surface Cleaner	✓	49	H	x	N/A	Moderate to Robust	✓	Moderate
Dish Soap	✓	29	M	x	N/A	Slight to Moderate	✓	Moderate
Dishwasher Detergent	✓	29	M	x	N/A	Slight to Moderate	✓	Moderate
Laundry Detergent (Industrial and Institutional)	x	N/A	N/A	✓	M	Moderate	✓	Moderate
Paint and Floor Lacquer	✓	17	H	x	N/A	Moderate	✓	Moderate
PET Byproduct	✓	11	H	x	N/A	Moderate	✓	Moderate
Ethoxylation Process Byproduct	✓	1	H	x	N/A	Slight to Moderate	✓	Moderate
Hydraulic Fracturing	x	N/A	N/A	✓	M	Moderate to Robust	✓	Moderate

[a] Data quality ratings of modeling approaches are based on the GS/ESD that was used in tandem with Monte Carlo modeling.

[b] Data quality ratings are not applicable for the dermal modeling approach because this modeling was conducted with an already-developed EPA model.

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

3.1.2.4.1 Number of Workers

There are uncertainties surrounding the estimated number of workers potentially exposed to 1,4-dioxane. First, BLS employment data for each industry/occupation combination are only available at the 3-, 4-, or 5-digit NAICS level, rather than at the full 6-digit NAICS level. This lack of specificity could result in an overestimate of the number of exposed workers if some 6-digit NAICS are included in the less granular BLS estimates but are not likely to use 1,4-dioxane for the assessed applications. EPA addressed this issue by refining the OES estimates using total employment data from the U.S. Census' SUSB. However, this approach assumes that the distribution of occupation types (SOC codes) in each 6-digit NAICS is equal to the distribution of occupation types at the parent 5-digit NAICS level. If the distribution of workers in occupations with 1,4-dioxane exposure differs from the overall distribution of workers in each NAICS, then this approach will result in inaccuracy. The effects of this uncertainty on the number of worker estimates are unknown, as the uncertainties may result in either over or underestimation of the estimates depending on the actual distribution.

Second, EPA's determinations of industries (represented by NAICS codes) and occupations (represented by SOC codes) that are associated with the OES assessed in this report are based on EPA's understanding of how 1,4-dioxane is used in each industry. The designations of which industries and occupations have potential exposures is a matter of professional judgement; therefore, the possibility exists for the erroneous inclusion or exclusion of some industries or occupations. This may result in inaccuracy but would be unlikely to systematically either overestimate or underestimate the count of exposed workers.

3.1.2.4.2 Analysis of Inhalation Exposure Monitoring Data

The principal limitation of the monitoring data is the uncertainty in the representativeness of the data due to some scenarios having limited exposure monitoring data in literature. Where few data are available, the assessed exposure levels are unlikely to be representative of worker exposure across the entire job category or industry. This may particularly be the case when monitoring data were available for only one site. Differences in work practices and engineering controls across sites can introduce variability and limit the representativeness of monitoring data. Age of the monitoring data can also introduce uncertainty due to differences in workplace practices and equipment used at the time the monitoring data were collected compared those currently in use. Therefore, older data may overestimate or underestimate exposures, depending on these differences. The effects of these uncertainties on the occupational exposure assessment are unknown, as the uncertainties may result in either overestimation or underestimation of exposures depending on the actual distribution of 1,4-dioxane air concentrations and the variability of work practices among different sites.

In some scenarios where monitoring data were available, EPA did not find sufficient data to determine complete statistical distributions. Ideally, EPA will present 50th and 95th percentiles for each exposed population. In the absence of percentile data for monitoring, the mean or midpoint of the range may serve as a substitute for the 50th percentile of the actual distributions. Similarly, the highest value of a range may serve as a substitute for the 95th percentile of the actual distribution. However, these substitutes are uncertain. The effects of these substitutes on the occupational exposure assessment are unknown, as the substitutes may result in either overestimation or underestimation of exposures depending on the actual distribution.

3.1.2.4.3 Modeled Inhalation Exposures

EPA addressed variability in inhalation models by identifying key model parameters to apply a statistical distribution that mathematically defines the parameter's variability. EPA defined statistical distributions for parameters using documented statistical variations where available. Where the statistical variation was unknown, assumptions were made to estimate the parameter distribution using available literature data, such as GSs and ESDs. However, there is uncertainty as to the representativeness of the parameter distributions with respect to the modeled scenario because the data are often not specific to sites that use 1,4-dioxane. In general, the effects of these uncertainties on the exposure estimates are unknown, as the uncertainties may result in either overestimation or underestimation on exposures depending on the actual distributions of each of the model input parameters.

There is also uncertainty as to whether the model equations generate results that represent actual workplace air concentrations. Some activity-based modeling does not account for exposures from other activities. Another uncertainty is lack of consideration for engineering controls. The GS/ESDs assume that all activities occur without any engineering controls or PPE, and in an open-system environment where vapor and particulates freely escape and can be inhaled. Actual exposures may be less than estimated depending on engineering control and PPE use.

A strength of the assessment is the variation of the model input parameters as opposed to using a single static value. This parameter variation increases the likelihood of true occupational inhalation exposures falling within the range of modeled estimates. An additional strength is that all data that EPA used to inform the modeling parameter distributions have overall data quality determinations of either high or medium from EPA's systematic review process.

3.1.2.4.4 Modeled Dermal Exposures

The *Dermal Exposure to Volatile Liquids Model* used to estimate dermal exposure to 1,4-dioxane in occupational settings assumes a fixed fractional absorption of the applied dose; however, fractional absorption may be dependent on skin loading conditions. The model also assumes a single exposure event per day based on existing framework of the *EPA/OPPT 2-Hand Dermal Exposure to Liquids Model* and does not address variability in exposure duration and frequency. Additionally, dermal exposures to 1,4-dioxane vapor that may penetrate clothing and the potential for associated direct skin contact with clothing saturated with 1,4-dioxane vapor are not included in quantifying exposures, which could potentially result in underestimates of exposures. Although the extent of saturation of clothing with 1,4-dioxane vapors is unknown, it is expected to be minimal given the low concentrations of 1,4-dioxane in formulations for the conditions of use in the draft supplement.

A strength of the dermal assessment approach is the estimation of two different fractional absorption values specific to industrial and commercial use settings as opposed to applying only one fractional absorption value to both settings.

3.2 General Population Exposures

1,4-Dioxane – General Population Exposures (Section 3.2):
Key Points

- EPA estimated oral and inhalation exposures to the concentrations of 1,4-dioxane in drinking water and air estimated in Section 2.3 using equations and exposure factors described in Appendix G.1 and Appendix J.2.
- EPA estimated general population exposures to 1,4-dioxane in air and water with a particular focus on populations that may be highly exposed
    - Fenceline communities. For exposures through air, EPA considered potential exposures for communities within 10 km of a release site. For drinking water, EPA considered potential exposures for communities relying on drinking water collected downstream of release sites.
    - Lifestage. For drinking water, EPA evaluated lifestage-specific exposures for adults, formula-fed infants, and children. For air exposures, the impacts of lifestage differences were not able to be adequately quantified and so the air concentrations are used for all lifestages.
    - High-end exposure estimates. EPA evaluated exposures based on high-end exposure scenarios (*e.g.*, air exposures include a range of modeled concentration predictions [low-end, central tendency, and high-end]), although only high-end model predictions of air concentrations are presented in this section).

General population exposures occur when 1,4-dioxane is released into the environment and the media is then a pathway for exposure. Figure 3-1 below provides a graphic representation of where and in which media 1,4-dioxane may be found and the corresponding route of exposure.

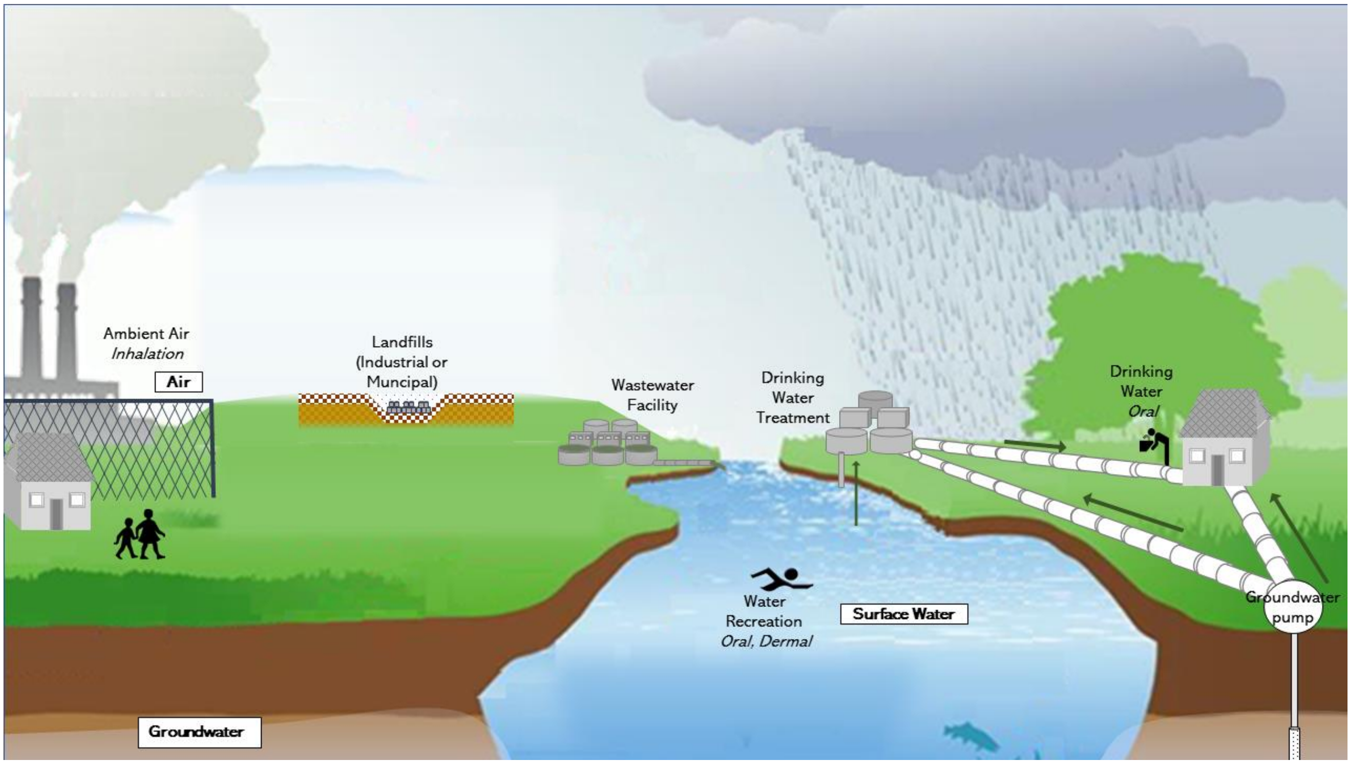

Figure 3-1. Potential Human Exposure Pathways to 1,4-Dioxane for the General Population[a]

[a] The diagram presents the media (white text boxes) and routes of exposure (italics for oral, inhalation, or dermal) for the general population. Sources of drinking water from surface or water pipes is depicted with grey arrows.

3.2.1 Approach and Methodology

EPA used modeled concentrations of 1,4-dioxane in air, surface water, and groundwater estimated in Section 2.3 to estimate acute and chronic general population exposures that could result from contact with environmental media. These acute and chronic exposure estimates are used to evaluate cancer and non-cancer risk described in Section 5. To estimate oral exposures to 1,4-dioxane in drinking water (including groundwater used as drinking water), EPA used equations and exposure factors described in Appendix G.1. To estimate inhalation exposures from 1,4-dioxane in air, EPA used equations and exposure factors described in Appendix J.2. Where possible, available monitored data within these environmental media were used to provide context for modeled results.

To estimate potential acute and chronic exposures through drinking water EPA calculated Acute Dose Rates (ADR) and Average Daily Doses (ADD) for adults, formula-fed infants, and children. To estimate lifetime exposures through drinking water, EPA calculated a Lifetime Average Daily Dose (LADD) based on 33 years[15] of exposure starting from birth or 33 years of exposure as an adult, averaged over a 78-year lifetime. Assumptions about drinking water intake and body weight for each age group were based on information in the *Exposure Factors Handbook*. EPA calculated ADs, ADDs, and LADDs based on the drinking water concentrations estimated under a range of conditions in Section 2.3.1.3. Details of these calculations are presented in Appendix I and *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Release to Surface Water from Individual Facilities* (U.S. EPA, 2023h).

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[15] Thirty-three years is the 95th percentile residential occupancy period (U.S. EPA *Exposure Factors Handbook*, Chapter 16, Table 16-5).

To estimate potential acute and chronic exposures through air, EPA calculated ADs and ADCs based on modeled air concentrations described in Section 2.3.3 To estimate potential lifetime exposures, EPA calculated LADCs based on 33 years of exposure. Methods adequate to quantify the impact of lifestage differences on 1,4-dioxane exposure are not available (see Section 4.3) and air concentration is used as the exposure metric for all lifestages per EPA guidance (U.S. EPA, 2012, 1994b). Specific equations, inputs and assumptions are described in detail in Appendix IJ.2.

3.2.2 Drinking Water Exposure Assessment

EPA assessed general population drinking water exposures that could result from surface water or groundwater used as drinking water. Exposures estimates presented below are based on surface water concentrations modeled in Section 2.3.1 or groundwater concentrations modeled in Section 2.3.2. Exposure estimates presented throughout this section focus on adults and formula-fed infants because these are lifestages with the greatest drinking water intake relative to body weights and therefore the greatest potential exposures.

3.2.2.1 Surface Water Exposure Assessment

EPA evaluated acute, chronic, and lifetime general population, exposures to 1,4-dioxane that may be present in drinking water based on modeled surface water concentrations estimated in Section 2.3.1. For drinking water exposures in this assessment, EPA focused on exposures in fenceline communities, defined in this context as members of the general population who rely on drinking water from water bodies receiving 1,4-dioxane releases from any industrial or DTD source.

Drinking water exposures were evaluated using a series of parallel analyses that provide information about the individual contributions of specific COUs as well as information about aggregate exposures that could result from multiple sources releasing to the same water body.

3.2.2.1.1 Exposures from Individual Facility Releases

To evaluate the individual contributions of releases associated with specific industrial and commercial COUs to general population exposures, EPA calculated ADRs, ADDs and LADDs based on modeled water concentrations estimated in Section 2.3.1.3.1 (Table 3-3). A total of 125 release scenarios were evaluated based on water concentrations estimated for annual releases that occur over a single day (a peak exposure scenario), over 30 days of release, or over 250 to 365 days of release. Exposure estimates are presented for both adults and formula-fed infants because these are lifestages with greatest drinking water intake relative to body weights and therefore greatest exposures. ADRs based on a single day release scenario range from 6.0×10^{-8} to 3,730 mg/kg for adults and 2.1×10^{-7} to 1.3×10^{4} mg/kg for infants. ADDs range from 1.1×10^{-11} to 0.5 mg/kg/day for adults and 2.7×10^{-11} to 1.3 mg/kg/day for infants and are not influenced by the days of release. LADDS range from 4.2×10^{-12} to 0.2 mg/kg/day for adults exposed for 33 years and 3.5×10^{-13} to 1.6×10^{-2} mg/kg/day for infants exposed for 1 year. Complete exposure calculations are available in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Release to Surface Water from Individual Facilities* (U.S. EPA, 2023h).

Table 3-3. Adult and Infant Exposures Estimated from Facility-Specific Releases

OES	No. of Releases Modeled	Age Group[b]	ADR (mg/kg-day)			ADD (mg/kg-day)			LADD (mg/kg-day)		
			Min Exposure[c]	Mean Exposure[d]	Max Exposure[e]	Min Exposure[c]	Mean Exposure[d]	Max Exposure[e]	Min Exposure[c]	Mean Exposure[d]	Max Exposure[e]
Disposal	25	Adult (21+ years)	6.03E−07	2.60E01	3.83E02	4.51E−10	1.44E−02	2.21E−01	1.91E−10	6.08E−03	9.36E−02
		Infant (birth to <1 year)	2.11E−06	9.11E01	1.34E03	1.15E−09	3.67E−02	5.65E−01	1.48E−11	4.71E−04	7.25E−03
Ethoxylation Byproduct	8	Adult (21+ years)	2.17E−07	1.04E02	8.31E02	9.06E−11	3.66E−02	2.93E−01	3.83E−11	1.55E−02	1.24E−01
		Infant (birth to <1 year)	7.61E−07	3.65E02	2.92E03	2.31E−10	9.36E−02	7.48E−01	2.97E−12	1.20E−03	9.59E−03
Functional Fluids (Open-System)	6	Adult (21+ years)	5.59E−04	6.33E−02	1.92E−01	1.83E−07	2.23E−05	6.66E−05	7.73E−08	9.44E−06	2.82E−05
		Infant (birth to <1 year)	1.96E−03	2.22E−01	6.75E−01	4.67E−07	5.70E−05	1.70E−04	5.99E−09	7.30E−07	2.18E−06
Import and Repackaging	12	Adult (21+ years)	4.35E−04	3.28E02	3.73E03	1.32E−07	3.05E−02	2.23E−01	5.59E−08	1.29E−02	9.43E−02
		Infant (birth to <1 year)	1.53E−03	1.15E03	1.31E04	3.38E−07	7.78E−02	5.69E−01	4.33E−09	9.97E−04	7.30E−03
Industrial Uses	31	Adult (21+ years)	5.34E−07	2.05E01	1.87E02	1.97E−10	1.36E−02	1.55E−01	8.31E−11	5.77E−03	6.56E−02
		Infant (birth to <1 year)	1.87E−06	7.21E01	6.55E02	5.02E−10	3.48E−02	3.96E−01	6.44E−12	4.47E−04	5.08E−03
Manufacture	2	Adult (21+ years)	3.35E00	6.56E01	1.28E02	2.50E−03	4.91E−02	9.57E−02	1.06E−03	2.08E−02	4.05E−02
		Infant (birth to <1 year)	1.17E01	2.30E02	4.48E02	6.40E−03	1.25E−01	2.44E−01	8.20E−05	1.61E−03	3.13E−03
PET Manufacturing	19	Adult (21+ years)	1.11E−04	4.32E01	6.67E02	3.86E−08	3.16E−02	5.00E−01	1.63E−08	1.34E−02	2.11E−01
		Infant (birth to <1 year)	3.91E−04	1.52E02	2.34E03	9.86E−08	8.07E−02	1.28E00	1.26E−09	1.04E−03	1.64E−02
Printing Inks	1	Adult (21+ years)	8.26E−02	8.26E−02	8.26E−02	6.18E−05	6.18E−05	6.18E−05	2.62E−05	2.62E−05	2.62E−05
		Infant (birth to <1 year)	2.90E−01	2.90E−01	2.90E−01	1.58E−04	1.58E−04	1.58E−04	2.02E−06	2.02E−06	2.02E−06
Remediation	16	Adult (21+ years)	6.04E−08	7.36E−02	7.19E−01	1.07E−11	4.56E−05	4.12E−04	4.51E−12	1.93E−05	1.74E−04
		Infant (birth to <1 year)	2.12E−07	2.58E−01	2.52E00	2.72E−11	1.17E−04	1.05E−03	3.49E−13	1.49E−06	1.35E−05

OES	**No. of Releases Modeled**	**Age Group**[b]	**ADR (mg/kg-day)**			**ADD (mg/kg-day)**			**LADD (mg/kg-day)**		
			Min Exposure[c]	**Mean Exposure**[d]	**Max Exposure**[e]	**Min Exposure**[c]	**Mean Exposure**[d]	**Max Exposure**[e]	**Min Exposure**[c]	**Mean Exposure**[d]	**Max Exposure**[e]
Overall	120	Adult (21+ years)	6.04E−08	5.84E01	3.73E03	1.07E−11	1.78E−02	5.00E−01	4.51E−12	7.55E−03	2.11E−01
		Infant (birth to <1 year)	2.12E−07	2.05E02	1.31E04	2.72E−11	4.56E−02	1.28E00	3.49E−13	5.84E−04	1.64E−02

Adult LADDs presented in this table were used to derive cancer risk estimates presented in Figure 5-1, Figure 5-2 and Figure 5-4.

ADRs presented here are calculated based on the assumption that all releases could occur on a single day of release (peak exposure scenario); ADDs and LADDs are based on chronic exposure scenarios and are the same regardless of the number of days of release assumed. LADDs for adults are based on 33 years of exposure averaged over a 78-year lifetime while LADDs for infant-specific exposures are based on 1 year of exposure averaged over a 78-year lifetime.

[a] Adult refers to 21+ years; infant refers to birth to <1 year.

[b] These COUs are added since the 2020 RE was published.

[c] The minimum exposure for the identified days of release, within the identified OES, and for the identified age group.

[d] The arithmetic mean exposure for the identified days of release, within the identified OES, and for the identified age group.

[e] The maximum exposure for the identified days of release, within the identified OES, and for the identified age group.

3.2.2.1.2 Exposures from Down-the-Drain Releases

To evaluate the potential contribution of DTD consumer and commercial releases, EPA calculated ADRs, ADDs, and LADDs using modeled water concentrations estimated as described in Section 2.3.1.2.2. Water concentrations of 1,4-dioxane resulting from DTD releases depend on the population size (an indicator of the number of people using products and contributing the releases) and the stream flows of the receiving water bodies. Therefore, the adult LADDs presented in Table 3-4 are based on the range of water concentrations estimated by Monte Carlo modeling of DTD release scenarios with varying population size and stream flows. LADDs range from 1.7×10^{-10} to 5.1×10^{-4} mg/kg/day for adults exposed for 33 years. Complete exposure calculations for adults and infants are available in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Surface Water Concentrations Predicted with Probabilistic Modeling* (U.S. EPA, 2023i).

Table 3-4. Adult LADD Exposures (mg/kg/day) Estimated from 1,4-Dioxane DTD Consumer and Commercial Releases

		Population Contributing to DTD Releases				
		100	1,000	10,000	100,000	1,000,000
Stream Flow (cfs)	100	5.1E–08	5.1E–07	5.1E–06	5.1E–05	5.1E–04
	300	1.7E–08	1.7E–07	1.7E–06	1.7E–05	1.7E–04
	1,000	5.1E–09	5.1E–08	5.1E–07	5.1E–06	5.1E–05
	3,000	1.7E–09	1.7E–08	1.7E–07	1.7E–06	1.7E–05
	10,000	5.1E–10	5.1E–09	5.1E–08	5.1E–07	5.1E–06
	30,000	1.7E–10	1.7E–09	1.7E–08	1.7E–07	1.7E–06

The frequencies of each of these combinations of population size and flow rate are presented Table 2-10. Adult LADDs presented in this table were used to derive the cancer risk estimates presented in Table 5-4.
LADDs for adults are based on 33 years of exposure averaged over a 78-year lifetime.

3.2.2.1.3 Disposal of Hydraulic Fracturing Produced Waters

To evaluate the potential contribution of disposal of hydraulic fracturing produced waters to surface water, EPA calculated ADRs, ADDs, and LADDs using the range of modeled water concentrations estimated in Section 2.3.1.2.2. (Table 3-5). Water concentrations of 1,4-dioxane resulting from disposal of hydraulic fracturing produced water vary substantially across sites. The estimated exposures presented here are based on the range of water concentrations estimated by Monte Carlo modeling for a range of site-specific factors. For this range of estimated surface water concentrations, Adult ADRs range from 2.45×10^{-16} to 7.96×10^{-3} mg/kg and adult ADDs range from 6.69×10^{-17} to 2.18×10^{-3} mg/kg/day. LADDs for adults exposed over 33 years over a 78-year lifetime range from 2.8×10^{-17} to 9.2×10^{-4} mg/kg/day. Complete exposure calculations for adults and infants are available in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Surface Water Concentrations Predicted with Probabilistic Modeling* (U.S. EPA, 2023i).

Table 3-5. Adult ADR, ADD, and LADD Exposures Estimated from Disposal of Hydraulic Fracturing Produced Waters to Surface Water

Monte Carlo Distribution	Adult Acute Dose Rate (mg/kg)	Adult Average Daily Dose (mg/kg/day)	Adult Lifetime Average Daily Dose (mg/kg/day)
Maximum	7.96E−03	2.18E−03	9.2E−04
99th percentile	3.13E−04	8.56E−05	3.6E−05
95th percentile	1.05E−04	2.86E−05	1.2E−05
Median	2.57E−06	7.03E−07	3.0E−07
5th percentile	9.31E−09	2.54E−09	1.1E−09
Minimum	2.45E−16	6.69E−17	2.8E−17
Adult LADDs presented in this table were used to derive cancer risk estimates presented in Table 5-5.			

3.2.2.1.4 Aggregate Exposure

Because multiple sources of 1,4-dioxane contribute to surface water and drinking water concentrations, EPA also estimated aggregate general population exposures that could occur because of combined contributions from DTD releases from consumer and commercial uses, upstream sources, and direct and indirect industrial releases. EPA calculated ADRs, ADDs, and LADDs based on modeled water concentrations estimated in Section 2.3.1.3.4 using probabilistic modeling of aggregate 1,4-dioxane surface water concentrations that could occur downstream of industrial release sites for each COU. LADDs estimated for adults exposed over 33 years over a 78-year lifetime range from 8.07×10^{-7} to 7.4×10^{-3} mg/kg/day based on median modeled water concentrations across COUs (Table 3-6). Complete exposure calculations for adults and infants are available in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Surface Water Concentrations Predicted with Probabilistic Modeling* (U.S. EPA, 2023i).

Table 3-6. Adult LADD Exposures from Aggregate Concentrations Estimated Downstream of Release Sites (Including DTD Releases and Direct and Indirect Industrial Releases)

COU	LADDs (mg/kg/day) Based on Modeled Aggregate Surface Water Concentrations Estimated across the Monte Carlo Distribution						
	Min	5th Percentile	25th Percentile	Median	75th Percentile	95th Percentile	Max
Disposal	6.00E−09	6.82E−07	1.63E−06	3.93E−06	9.00E−06	9.64E−04	1.21E−01
Ethoxylation byproduct	7.17E−09	3.72E−07	1.05E−06	1.93E−06	8.11E−06	1.98E−02	2.63E−01
Functional Fluids (Open-System)	5.62E−10	2.58E−07	4.99E−07	8.91E−07	7.20E−06	4.13E−05	6.22E−05
Import and Repackaging	1.82E−08	4.21E−07	2.10E−04	1.03E−03	4.53E−03	1.34E−02	1.18E00
Industrial Uses	4.61E−10	1.65E−07	3.90E−07	8.07E−07	4.66E−05	4.90E−03	5.91E−02

COU	**LADDs (mg/kg/day) Based on Modeled Aggregate Surface Water Concentrations Estimated across the Monte Carlo Distribution**						
	Min	**5th Percentile**	**25th Percentile**	**Median**	**75th Percentile**	**95th Percentile**	**Max**
Manufacture	3.51E−04	1.06E−03	2.29E−03	7.40E−03	1.75E−02	4.04E−02	4.04E−02
PET Manufacturing	1.73E−08	4.54E−07	1.48E−06	1.43E−05	6.06E−04	2.53E−02	2.11E−01
Printing Inks	4.80E−07	1.01E−06	1.27E−05	2.04E−05	2.38E−05	2.66E−05	2.71E−05
Remediation	2.48E−09	2.74E−07	6.29E−07	1.27E−06	3.10E−05	9.61E−05	2.29E−04
Adult LADDs presented in this table were used to derive cancer risk estimates presented in Figure 5-5. Percentiles reflect concentrations estimated at various points in the Monte Carlo distribution.							

3.2.2.2 Groundwater Exposure Assessment

EPA evaluated general population exposures that could occur from disposals of 1,4-dioxane that contaminate groundwater used as a primary source of drinking water. To estimate chronic exposures through this drinking water pathway, EPA calculated ADDs and LADDs for adults and formula-fed infants based on modeled groundwater concentrations of 1,4-dioxane estimated in Section 2.3.2. EPA did not evaluate acute exposures because methods used to estimate groundwater concentrations provide an indication of potential concentrations occuring over many years, rather than peak concentrations.

3.2.2.2.1 Disposal to Landfills

To evaluate general population exposure, EPA calculated ADDs and LADDs based on modeled groundwater concentrations estimated in Section 2.3.2.3. Potential groundwater concentrations resulting from disposal of 1,4-dioxane to municipal solid waste landfills vary across landfill loading rates and concentrations of 1,4-dioxane in leachate. Estimated exposures presented here are therefore based on the range of groundwater concentrations estimated under varying landfill conditions. Table 3-7 summarizes LADD exposure estimates estimated for 33 years of exposure as an adult. Under the range of landfill scenarios considered, adult LADDs range from 2.5×10^{6} to 2.4×10^{-2} mg/kg/day. The highest LADDs occur when leachate concentrations are above 100 mg/L and loading rates are above 10,000 lb. The complete set of exposure estimates for adults and infants relying on groundwater as a primary drinking water source are presented in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Land Releases to Landfills* (U.S. EPA, 2023f).

Table 3-7. Adult LADD Exposures Estimated from Groundwater Contamination from Landfills under Varying Landfill Conditions

	Loading Rate (lb)							
Leachate Concentration (mg/L)	0.1	1	10	100	1,000	10,000	100,000	1,000,000
0.0001	2.5E−16	2.4E−15	3.0E−14	2.9E−13	2.7E−12	2.6E−11	2.5E−10	2.4E−09
0.001	2.5E−15	2.4E−14	3.0E−13	2.9E−12	2.7E−11	2.6E−10	2.5E−09	2.4E−08
0.01	2.5E−14	2.4E−13	3.0E−12	2.9E−11	2.7E−10	2.6E−09	2.5E−08	2.4E−07
0.1	2.5E−13	2.4E−12	3.0E−11	2.9E−10	2.7E−09	2.6E−08	2.5E−07	2.4E−06
1	2.5E−12	2.4E−11	3.0E−10	2.9E−09	2.7E−08	2.6E−07	2.5E−06	2.4E−05
10	2.5E−11	2.4E−10	3.0E−09	2.9E−08	2.7E−07	2.6E−06	2.5E−05	2.4E−04
100	2.5E−10	2.4E−09	3.0E−08	2.9E−07	2.7E−06	2.6E−05	2.5E−04	2.4E−03
1,000	2.5E−09	2.4E−08	3.0E−07	2.9E−06	2.7E−05	2.6E−04	2.5E−03	2.4E−02
10,000	2.5E−08	2.4E−07	3.0E−06	2.9E−05	2.7E−04	2.6E−03	2.5E−02	2.4E−01
Adult LADDs presented in this table were used to derive cancer risk estimates presented in Table 5-6.								

3.2.2.2.2 Disposal of Hydraulic Fracturing Produced Waters

To evaluate general population exposure resulting from disposal of hydraulic fracturing produced waters to groundwater, EPA calculated ADDs and LADDs estimated in Section 2.3.2.4 (Table 3-8). Potential groundwater concentrations resulting from disposal of hydraulic fracturing produced waters vary substantially across sites. Estimated exposures presented here are based on the range of groundwater concentrations estimated through Monte Carlo modeling. Under the range of hydraulic fracturing scenarios considered, adult LADDs range from 4.9×10^{-9} to 2.1×10^{-4} mg/kg/day. The complete set of exposure estimates for adults and infants relying on groundwater as a primary drinking water source are presented in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Land Releases to Surface Impoundments* (U.S. EPA, 2023g).

Table 3-8. Estimated Exposures Resulting from Groundwater Contamination from Disposal of Hydraulic Fracturing Produced Water

Monte Carlo Distribution	Modeled Groundwater Concentration (mg/L)	Adult ADD (mg/kg/day)	Adult LADD (mg/kg/day)	Infant ADD (mg/kg/day)
Max	1.9E−02	2.1E−04	8.8E−05	5.3E−04
99th	1.5E−02	1.7E−04	7.1E−05	4.3E−04
95th	1.5E−02	1.7E−04	7.1E−05	4.3E−04
Mean	7.1E−04	7.9E−06	3.3E−06	2.0E−05
50th	1.2E−04	1.3E−06	5.6E−07	3.4E−06
5th	1.2E−04	1.3E−06	5.6E−07	3.4E−06
Min	4.4E−07	4.9E−09	2.1E−09	1.2E−08
Adult LADDs presented in this table were used to derive cancer risk estimates presented in Table 5-7.				

3.2.3 Air Exposure Assessment

EPA evaluated acute, chronic and lifetime general population, exposures to 1,4-dioxane in air. This analysis focuses on potential fenceline community exposures that may occur within 10 km of release sites.

3.2.3.1 Industrial COUs Reported to TRI

To evaluate general population exposures from industrial fugitive and stack emissions, EPA calculated ACs, ADCs, and LADCs based on modeled air concentrations estimated in Section 2.3.3. The LADCs presented in Table 3-9 are based on the maximum 95th percentile air concentrations estimated for the facilities within each COU. LADCs within 10 km of release types considered here range from 1.1×10^{-11} to 6.9×10^{-3} ppm. These lifetime exposure estimates are based on 33 years of exposure over a 78-year lifetime and are relevant to all lifestages. The complete set of inhalation exposure estimates for fenceline communities are presented in *1,4-Dioxane Supplemental Information File: Air Exposures and Risk Estimates for Single Year Analysis* (U.S. EPA, 2023e).

Table 3-9. Lifetime Average Daily Concentrations Estimated within 10 km of 1,4-Dioxane Releases to Air

1,4-Dioxane OES	# Facilities	LADCs Estimated within 5–10,000 m of Facilities with Greatest Exposures (ppm)								
		5 m	10 m	30 m	60 m	100 m	100 to 1,000 m	2,500 m	5,000 m	10,000 m
Disposal	15	1.8E−03	2.1E−03	7.6E−04	2.9E−04	1.3E−04	1.3E−05	8.0E−07	2.7E−07	8.8E−08
Dry Film Lubricant	8	6.8E−11	3.0E−09	2.2E−07	1.6E−06	2.7E−06	4.2E−07	1.2E−08	3.6E−09	1.6E−09
Ethoxylation Byproduct	6	2.8E−03	5.8E−03	3.1E−03	1.3E−03	6.9E−04	1.6E−04	9.3E−06	3.8E−06	1.5E−06
Film Cement	1	5.3E−05	5.5E−05	1.9E−05	9.7E−06	5.3E−06	9.7E−07	5.8E−08	2.0E−08	6.4E−09
Functional Fluids (Open-System)	2	5.4E−06	1.0E−05	4.4E−06	4.6E−06	7.7E−06	3.1E−06	2.9E−07	1.1E−07	3.7E−08
Import and Repackaging	1	1.1E−11	2.4E−10	2.3E−08	1.8E−07	3.7E−07	1.4E−07	2.8E−08	1.7E−08	9.4E−09
Industrial Uses	12	1.8E−03	2.0E−03	6.5E−04	2.4E−04	1.2E−04	3.0E−05	3.7E−06	1.4E−06	4.8E−07
Laboratory Chemical Use	1	8.7E−04	9.1E−04	3.1E−04	1.6E−04	8.7E−05	1.6E−05	9.6E−07	3.2E−07	1.1E−07
Manufacturing	1	3.7E−03	6.9E−03	3.3E−03	1.4E−03	6.7E−04	6.0E−05	3.5E−06	1.1E−06	3.4E−07
PET Manufacturing	13	3.4E−03	4.0E−03	1.5E−03	5.9E−04	2.7E−04	4.5E−05	8.8E−06	5.3E−06	2.8E−06
Spray Foam Application	1	3.3E−07	3.6E−07	1.2E−07	6.4E−08	3.6E−08	6.6E−09	7.3E−10	2.7E−10	1.0E−10

LADCs are based on the maximum 95th percentile air concentration predictions for the facility in each COU with the greatest exposures. Adult LADCs presented in this table were used to derive the cancer risk estimates presented in Table 5-8.

3.2.3.2 Hydraulic Fracturing

To evaluate general population exposures to fugitive emissions from hydraulic fracturing operations, EPA calculated ACs, ADCs, and LADCs based on modeled air concentrations estimated in Section 2.3.3.2.4 under a range of different release scenarios and topographical conditions (Table 3-10). LADCs within 1,000 m of hydraulic fracturing operations range from 8.7×10^{-4} to 5.2 ppm. These lifetime exposure estimates are based on 33 years of exposure over a 78-year lifetime and are relevant to all lifestages. The complete set of inhalation exposure estimates from fugitive emissions of hydraulic fracturing operations are presented in *1,4-Dioxane Supplemental Information File: Air Exposure and Risk Estimates for 1,4-Dioxane Emissions from Hydraulic Fracturing Operations* (U.S. EPA, 2023b).

Table 3-10. Exposures from Fugitive Emissions Estimated within 1,000 m of Hydraulic Fracturing Operations[a]

Fugitive Emissions Release Scenario	Exposure Duration	Air Concentrations for 95th Percentile Modeled Releases (ppm)						Air Concentrations for 50th Percentile Modeled Releases (ppm)					
		High-End Modeled Air Concentrations			Central Tendency (Mean) Modeled Air Concentrations			High-End Modeled Air Concentrations			Central Tendency (Mean) Modeled Air Concentrations		
		100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m
South (Coastal)-Rural-24	AC	5.3E−02	2.4E−03	6.2E−03	4.2E−02	1.8E−03	4.8E−03	3.1E−03	1.4E−04	3.6E−04	2.4E−03	1.0E−04	2.7E−04
	ADC	1.1E−02	4.8E−04	1.2E−03	8.3E−03	3.6E−04	9.4E−04	6.0E−04	2.7E−05	7.0E−05	4.7E−04	2.0E−05	5.4E−05
	LADC	4.5E−03	2.0E−04	5.2E−04	3.5E−03	1.5E−04	4.0E−04	2.6E−04	1.2E−05	3.0E−05	2.0E−04	8.6E−06	2.3E−05
West North Central-Rural-24	AC	4.1E−02	2.4E−03	5.5E−03	3.1E−02	1.5E−03	3.7E−03	2.3E−03	1.3E−04	3.1E−04	1.8E−03	8.3E−05	2.1E−04
	ADC	8.0E−03	4.6E−04	1.1E−03	6.1E−03	2.9E−04	7.3E−04	4.6E−04	2.7E−05	6.2E−05	3.5E−04	1.6E−05	4.2E−05
	LADC	3.4E−03	2.0E−04	4.6E−04	2.6E−03	1.2E−04	3.1E−04	1.9E−04	1.1E−05	2.6E−05	1.5E−04	6.9E−06	1.8E−05
South (Coastal)-Urban-24	AC	2.6E−02	5.8E−04	1.8E−03	2.3E−02	5.0E−04	1.6E−03	1.5E−03	3.3E−05	1.0E−04	1.3E−03	2.9E−05	9.0E−05
	ADC	5.1E−03	1.1E−04	3.5E−04	4.5E−03	9.9E−05	3.1E−04	2.9E−04	6.5E−06	2.0E−05	2.6E−04	5.7E−06	1.8E−05
	LADC	2.2E−03	4.8E−05	1.5E−04	1.9E−03	4.2E−05	1.3E−04	1.2E−04	2.8E−06	8.5E−06	1.1E−04	2.4E−06	7.5E−06
West North Central-Urban-24	AC	2.4E−02	6.2E−04	1.9E−03	1.9E−02	4.6E−04	1.4E−03	1.4E−03	3.6E−05	1.1E−04	1.1E−03	2.6E−05	8.1E−05
	ADC	4.8E−03	1.2E−04	3.7E−04	3.8E−03	9.0E−05	2.8E−04	2.7E−04	7.0E−06	2.1E−05	2.2E−04	5.2E−06	1.6E−05
	LADC	2.0E−03	5.2E−05	1.6E−04	1.6E−03	3.8E−05	1.2E−04	1.2E−04	3.0E−06	8.9E−06	9.3E−05	2.2E−06	6.7E−06
South (Coastal)-Rural-8	AC	9.6E−03	8.8E−05	3.8E−04	8.3E−03	6.8E−05	3.2E−04	5.5E−04	5.1E−06	2.2E−05	4.8E−04	3.9E−06	1.8E−05
	ADC	1.9E−03	1.7E−05	7.6E−05	1.6E−03	1.3E−05	6.2E−05	1.1E−04	1.0E−06	4.3E−06	9.4E−05	7.6E−07	3.6E−06
	LADC	8.0E−04	7.4E−06	3.2E−05	7.0E−04	5.6E−06	2.6E−05	4.6E−05	4.2E−07	1.8E−06	4.0E−05	3.2E−07	1.5E−06
West North Central-Rural-8	AC	2.0E−02	7.7E−04	1.9E−03	1.1E−02	2.1E−04	6.6E−04	6.1E−04	1.2E−05	3.8E−05	6.1E−04	1.2E−05	3.8E−05
	ADC	4.0E−03	1.5E−04	3.7E−04	2.1E−03	4.2E−05	1.3E−04	1.2E−04	2.4E−06	7.4E−06	1.2E−04	2.4E−06	7.4E−06
	LADC	1.7E−03	6.4E−05	1.5E−04	8.9E−04	1.8E−05	5.5E−05	5.1E−05	1.0E−06	3.1E−06	5.1E−05	1.0E−06	3.1E−06

Fugitive Emissions Release Scenario	Exposure Duration	Air Concentrations for 95th Percentile Modeled Releases (ppm)						Air Concentrations for 50th Percentile Modeled Releases (ppm)					
		High-End Modeled Air Concentrations			Central Tendency (Mean) Modeled Air Concentrations			High-End Modeled Air Concentrations			Central Tendency (Mean) Modeled Air Concentrations		
		100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m
South (Coastal)-Urban-8	AC	8.7E−03	6.8E−05	3.2E−04	8.0E−03	6.0E−05	2.9E−04	4.6E−04	3.4E−06	1.7E−05	4.6E−04	3.4E−06	1.7E−05
	ADC	1.7E−03	1.3E−05	6.4E−05	1.6E−03	1.2E−05	5.8E−05	9.0E−05	6.8E−07	3.3E−06	9.0E−05	6.8E−07	3.3E−06
	LADC	7.3E−04	5.6E−06	2.7E−05	6.7E−04	5.0E−06	2.4E−05	3.8E−05	2.9E−07	1.4E−06	3.8E−05	2.9E−07	1.4E−06
West North Central-Urban-8	AC	1.5E−02	2.9E−04	9.4E−04	9.2E−03	1.2E−04	4.5E−04	5.3E−04	6.8E−06	2.6E−05	5.3E−04	6.8E−06	2.6E−05
	ADC	2.9E−03	5.8E−05	1.8E−04	1.8E−03	2.4E−05	8.9E−05	1.0E−04	1.4E−06	5.1E−06	1.0E−04	1.4E−06	5.1E−06
	LADC	1.2E−03	2.4E−05	7.8E−05	7.7E−04	1.0E−05	3.8E−05	4.4E−05	5.7E−07	2.2E−06	4.4E−05	5.7E−07	2.2E−06

[a] Lifetime Average Daily Concentrations (LADCs) presented in this table correspond to the cancer risk estimates presented in Table 5-7.
AC = Acute Concentration; ADC = Average Daily Concentration; LADC = Lifetime Average Daily Concentration

3.2.3.3 Industrial and Institutional Laundry Facilities

To evaluate exposures to emissions from industrial and institutional laundry facilities, EPA calculated ACs, ADCs, and LADCs based on vapor and particular air concentrations estimated in Section 2.3.3.2.4. High-end and central tendency air exposures estimated under the more conservative exposure scenario evaluated (rural south coastal topography, assuming 24 hours of releases each day) are presented for each type of laundry in Table 3-11. LADCs estimated within 1,000 m of laundry facilities operations range from 8.7×10^{-4} to 2.4×10^{-6} ppm. These lifetime exposure estimates are based on 33 years of exposure over a 78-year lifetime and are relevant to all lifestages. The complete set of inhalation exposure estimates from fugitive emissions of commercial laundry facilities are presented in *1,4-Dioxane Supplemental Information File: Air Exposures and Risk Estimates for Industrial Laundry* (U.S. EPA, 2023c).

Table 3-11. Exposures from Fugitive Emissions Estimated near Industrial and Institutional Laundry Facilities[a]

Facility Type	Detergent and Emission Type	Exposure Duration	Modeled Air Concentrations for Maximum Release Estimates (ppm)					
			High-End			Central Tendency (Mean)		
			100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m
Industrial laundry	Liquid-Vapor	AC	6.9E−06	3.3E−07	8.2E−07	5.0E−06	2.1E−07	5.6E−07
		ADC	5.4E−06	2.5E−07	6.3E−07	4.9E−06	2.1E−07	5.6E−07
		LADC	2.3E−06	1.1E−07	2.7E−07	2.1E−06	9.0E−08	2.4E−07
	Powder-Vapor	AC	6.9E−06	3.3E−07	8.2E−07	4.9E−06	2.1E−07	5.6E−07
		ADC	5.4E−06	2.5E−07	6.2E−07	4.9E−06	2.1E−07	5.6E−07
		LADC	2.3E−06	1.1E−07	2.6E−07	2.1E−06	8.9E−08	2.3E−07
	Powder-PM10	AC	7.2E−06	1.6E−07	5.9E−07	5.1E−06	1.2E−07	4.2E−07
		ADC	5.6E−06	1.3E−07	4.7E−07	5.0E−06	1.2E−07	4.2E−07
		LADC	2.4E−06	5.5E−08	2.0E−07	2.1E−06	4.9E−08	1.8E−07
	Powder-PM2.5	AC	6.9E−06	3.1E−07	8.0E−07	4.9E−06	2.0E−07	5.5E−07
		ADC	5.4E−06	2.4E−07	6.1E−07	4.9E−06	2.0E−07	5.4E−07
		LADC	2.3E−06	1.0E−07	2.6E−07	2.1E−06	8.5E−08	2.3E−07
Institutional laundry	Liquid-Vapor	AC	3.6E−06	1.6E−07	4.1E−07	3.1E−06	1.3E−07	3.5E−07
		ADC	3.4E−06	1.6E−07	4.0E−07	3.1E−06	1.3E−07	3.5E−07
		LADC	1.4E−06	6.7E−08	1.7E−07	1.3E−06	5.7E−08	1.5E−07
	Powder-vapor	AC	1.1E−07	4.8E−09	1.2E−08	9.2E−08	4.0E−09	1.0E−08
		ADC	1.0E−07	4.7E−09	1.2E−08	9.2E−08	3.9E−09	1.0E−08
		LADC	4.2E−08	2.0E−09	4.9E−09	3.9E−08	1.7E−09	4.4E−09

Facility Type	Detergent and Emission Type	Exposure Duration	Modeled Air Concentrations for Maximum Release Estimates (ppm)					
			High-End			Central Tendency (Mean)		
			100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m
	Powder-PM10	AC	1.1E−07	2.5E−09	8.9E−09	9.4E−08	2.2E−09	7.9E−09
		ADC	1.0E−07	2.4E−09	8.7E−09	9.4E−08	2.2E−09	7.9E−09
		LADC	4.4E−08	1.0E−09	3.7E−09	4.0E−08	9.2E−10	3.3E−09
	Powder-PM2.5	AC	1.1E−07	4.6E−09	1.2E−08	9.2E−08	3.8E−09	1.0E−08
		ADC	1.0E−07	4.5E−09	1.1E−08	9.2E−08	3.8E−09	1.0E−08
		LADC	4.2E−08	1.9E−09	4.8E−09	3.9E−08	1.6E−09	4.3E−09

[a] LADCs presented in this table correspond to the cancer risk estimates presented in Table 5-10.
AC = Acute Concentration; ADC = Average Daily Concentration; LADC = Lifetime Average Daily Concentration

3.3 Weight of the Scientific Evidence Conclusions

As described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), the weight of the scientific evidence supporting exposure assessments is evaluated based on the availability and strength of exposure scenarios and exposure factors, measured and monitored data, estimation methodology and model input data, and, if appropriate, comparisons of estimated and measured exposures. The strength of each of these evidence streams can be ranked as either robust, moderate, slight, or indeterminate. For each component of this exposure assessment, EPA evaluated the weight of the scientific evidence for individual evidence streams and then used that information to evaluate the overall weight of evidence supporting each set of exposure estimates. General considerations for evaluating the strength of evidence for each evidence stream are summarized in Table_Apx C-5. Specific examples of how these considerations are applied to overall weight of evidence conclusions are provided in Table_Apx C-6.

Overall confidence descriptions of high, medium, or low are assigned to the exposure assessment based on the strength of the underlying scientific evidence. When the assessment is supported by robust evidence, overall confidence in the exposure assessment is high; when supported by moderate evidence, overall confidence is medium; when supported by slight evidence, overall confidence is low.

3.3.1 Occupational Exposures

The weight of the scientific evidence for occupational exposure estimates is determined by several different evidence streams, including:

- Evidence supporting the exposure scenarios (Section 3.1.1 and Appendix F.4)
- The quality and representativeness of available monitoring data (Appendix F.4)
- Evidence supporting modeling approaches (Section 3.1.1 and Appendix F.4)
- Evidence supporting model input data (Appendix F.4)

3.3.1.1 Inhalation Exposure

Occupational inhalation exposure estimates are supported by moderate to robust evidence (see Appendix F.6).

- **Exposure Scenarios and Exposure Factors.** The exposure scenarios and exposure factors underlying the inhalation assessment are supported by moderate to robust evidence.

Occupational inhalation exposure scenarios and exposure factors, including duration of exposure, body weight, and breathing rate, were informed by sources of data with medium to high data quality ratings, increasing the strength of evidence. For most OESs/COUs, EPA used information directly relevant to the evaluated exposure scenarios; however, for some OESs/COUs, EPA used information from surrogate scenarios, decreasing the strength of evidence for those scenarios. Additionally, there is uncertainty in the extent to which the entire population of workers within an OES/COU are represented by the available data.

- **Measured and Monitored Data.** Measured/monitored data are supported by moderate to robust evidence. EPA used sources of data such as OSHA and NIOSH, which have medium to high data quality ratings, increasing the strength of the evidence. For the OESs/COUs with available monitoring data, the data was directly applicable to the assessed exposure scenario, as opposed to from a surrogate exposure scenario. However, the available monitoring data was limited to a single source for each OES/COU and often consisted of a small or dated dataset. Additionally, these data often only included one or a limited number of sites at which the data were measured, decreasing the strength of evidence for those OESs/COUs.

- **Modeling Methodologies.** The modeling methodologies are supported by moderate to robust evidence. Modeling was implemented to assess occupational inhalation exposures for three of the OESs/COUs, using methodologies from GS/ESD that are generally well described. The modeling incorporates Monte Carlo simulation to allow for variation in the model input data, which increases the representativeness of the approach towards the true population of potentially exposed workers and increases the strength of the evidence. However, EPA was unable to develop distributions for all input parameters, increasing the uncertainty in the parameterization and applicability.

- **Model Input Data.** Model input data are supported by moderate to robust evidence. For some model input data, EPA used 1,4-dioxane-specific data from sources such as process information, product concentration information, and FracFocus 3.0. For other model input parameters, generic data from the GS/ESD used for the modeling methodology was used due to lack of 1,4-dioxane data.

- **Comparison of Modeled and Monitored Data.** The comparison of modeled and measured occupational inhalation exposures is not rated because no comparisons between modeled and measured exposures were made.

Overall Confidence in Occupational Inhalation Exposure Estimates

The overall confidence in the occupational inhalation exposure estimates (Section 3.3.1.1) ranges from low to high, depending on the OES/COU. Measured/monitored data are supported by moderate to robust evidence. Additionally, the modeling methodologies and underlying model input data is supported by moderate to robust evidence. However, there is uncertainty in the representativeness of the assessed exposure scenarios towards all potential exposures for the given OES/COU, limitations in the amount and age of monitoring data, and limitations in the modeling approaches towards 1,4-dioxane-specific use within the OES/COU. Therefore, while the underlying data and methods used to estimate occupational inhalation exposures is supported by moderate to robust evidence, the overall confidence of these estimates is low to high depending on the OES/COU. OES/COU-specific discussions of the available inhalation exposure data and overall confidence are presented in Appendix F.6.

3.3.1.2 Dermal Exposure

Occupational dermal exposure estimates are supported by slight to robust evidence (see Appendix F.3).

- **Exposure Scenarios and Exposure Factors.** The exposure scenarios and exposure factors underlying the dermal assessment are supported by moderate to robust evidence. Dermal exposure scenarios were informed by process information and GS/ESD with medium to high data quality ratings, increasing the strength of evidence. Exposure factors, including amount of material on skin, surface area of skin exposed, and absorption of 1,4-dioxane through the skin, were informed by literature sources, the ChemSTEER User Guide (U.S. EPA, 2015a) for standard exposure parameters, and a European model, which have medium to high data quality ratings. EPA used information directly relevant to the evaluated exposure scenarios; however, there is uncertainty in the extent to which the entire population of workers within an OES/COU are represented by the available data.

- **Measured and Monitored Data.** No measured/monitored dermal exposure data were used in the occupational dermal exposure assessment. EPA did use measured data on 1,4-dioxane concentrations in various products from process information and other literature sources, which have medium to high data quality ratings, depending on the data source.

- **Modeling Methodologies.** The modeling methodologies are supported by moderate evidence. EPA used the *EPA Dermal Exposure to Volatile Liquids* to calculate the dermal retained dose for each OES/COU. This model modifies the *EPA/OPPT 2-Hand Dermal Exposure to Liquids Model* by incorporating a “fraction absorbed (f_{abs})” parameter to account for the evaporation of volatile chemicals and a “protection factor (PF)” to account for glove use. These modifications improve the modeling methodology and allow EPA to differentiate dermal exposures between commercial and industrial settings by varying the absorption and dermal protection factors. However, the modeling approach is still limited by the low variability for different worker activities/exposure scenarios.

- **Model Input Data.** Model input data are supported by slight to moderate evidence. As discussed above, model parameters were informed by sources with medium to high data quality ratings. However, a limitation is that some of the model input data was generic and not specific to 1,4-dioxane OES/COU.

- **Comparison of Modeled and Monitored Data.** All occupational dermal exposures were modeled, and no measured dermal exposures were used in this assessment, therefore there is no comparison.

Overall Confidence in Occupational Dermal Exposure Estimates

The overall confidence in the occupational dermal exposure estimates (Section 3.1.2.2) is medium for all OES/COU because the same modeling approach was used for all OES/COU. The modeling methodology is supported by moderate evidence, with model input parameters from literature sources, a European model, standard defaults from the ChemSTEER User Guide (U.S. EPA, 2015a), and 1,4-dioxane product concentration data from process information. These sources range from slight to robust, depending on factors such as age and applicability to OES/COU. The modeling is limited by the use of standard input parameters that are not specific to 1,4-dioxane and a lack of variability in dermal exposure for different worker activities. Therefore, EPA’s overall confidence in the occupational dermal exposure estimates is medium.

3.3.2 Drinking Water

3.3.2.1 Drinking Water Exposure Estimates Based on Surface Water Concentrations

The weight of evidence for drinking water exposure estimates is determined by several different evidence streams, including:

- Evidence supporting the general population exposure scenarios (Section 3.2.1)
- The quality and representativeness of available surface water and drinking water monitoring data (Section 2.3.1.1)
- Evidence supporting modeling approaches (Section 2.3.1.3 and Appendix G.2)
- Evidence supporting release data used as model input data (Section 2.2 and Appendix E.3)
- Concordance between modeled and monitored water concentrations (Section 2.3.1.4)

As described in Section 2.3.1, multiple approaches were used to predict surface water concentrations resulting from several sources. These included the evaluation of facility-specific releases, down the drain releases to surface water, hydraulic fracturing releases and aggregation of surface water releases. The associated strengths, limitations and confidence in these estimated environmental concentrations are described in Section 2.3.1.4. The general population drinking water exposure scenarios and exposure factors used to estimate exposures that could result from estimated water concentrations are described in Section 3.2.

Drinking water exposure estimates based on modeled surface water concentrations are supported by overall moderate to robust evidence, with the strength of the evidence varying across analysis approaches and COUs/OESs.

- **Exposure Scenarios and Exposure Factors.** The exposure scenarios and exposure factors underlying all drinking water exposure estimates are supported by moderate to robust evidence. Exposure factors for drinking water are based on robust data on drinking water intake rates and body weights as derived from exposure factors from the EPA's *Exposure Factors Handbook* (U.S. EPA, 2011). However, the drinking water exposure scenarios generally rely on the assumption that little or no dilution occurs prior to drinking water intakes. That assumption may not be representative of exposures at all locations. While there are locations where this assumption is expected to the accurate, the extent of downstream dilution that occurs prior to drinking water intakes is highly variable across locations. The proximity of facility releases to actual drinking water intakes is evaluated in Section 2.3.1.2.4 and Appendix G.2.4. Uncertainties related to downstream dilution decrease the overall strength of evidence for these exposure scenarios. However, EPA has performed several analyses that calculate exposures and risks under alternate assumptions about downstream dilution and illustrate the quantitative impact of those assumptions (see Section 5.2.2.1.2), increasing the overall strength of evidence. Drinking water exposure scenarios also rely on the data-driven assumption that 1,4-dioxane is not removed through treatment. Moderate to robust data provide support for this assumption under many treatment scenarios. These assumptions may over-estimate exposure for some locations, but provide an overall distribution that is generally expected to be representative of exposure scenarios.
- **Measured and Monitored Data.** The measured/monitored data are supported by moderate evidence. The high number of monitoring data points for surface water and drinking water from high quality sources in multiple locations over multiple years increases the strength of the evidence from monitoring data. Monitoring data confirm that 1,4-dioxane is present in some surface water and drinking water in some locations. However, evidence from monitoring data may not be representative of all sites where 1,4-dioxane is released to surface water from TSCA sources, decreasing the strength of evidence from monitoring data. The lack of temporal and/or spatial concurrence between most monitoring data and reported release locations makes direct comparison challenging for most locations. However, a limited number of sites with monitoring data are co-located with sites where 1,4-dioxane releases are reported, supporting comparisons of monitoring and modeled estimates that increase the overall strength of the evidence. In addition,

as described in Section 2.3.1.4, monitoring data for surface water directly downstream from releases show concentrations multiple orders of magnitude greater than typical ambient surface water concentrations, aligning with patterns of modeled results.

- **Modeling Methodologies.** The modeling methodologies are supported by moderate to robust evidence.
    - The methodology for deriving exposure estimates for facility releases is moderate and is applicable to the populations included in the exposure scenarios. This approach makes some conservative assumptions about flow rates and release frequency and amount. Additionally, the modeling does not take into account downstream fate or transport, but the physical chemical properties of 1,4-dioxane are expected to moderate the impact these influences could have on the modeled instream concentrations. The model is designed to estimate possible higher end water concentrations expected at specific locations.
    - The probabilistic methodology used for deriving exposure estimates for DTD releases, hydraulic fracturing releases, and aggregate releases from all sources is robust. This approach incorporates the full distribution of facility releases over multiple years and corresponding instream flow rate data rather than relying on the most conservative model inputs. It is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions.
- **Model Input Data.** Model input data are supported by slight to robust evidence, with the strength of the evidence varying across individual COUs/OESs. The strength of evidence supporting modeled water concentrations relies heavily on the quality of the facility or OES-specific release data used as inputs for the model, including both the amount of release, location of the release, and the corresponding flow in the receiving water body. A summary of sources of flow and release data for facility release modeling is presented in Table 2-6. A more detailed OES-specific discussion of the confidence in sources of release information is presented in Appendix E.3.4.
    - For overall distributions of industrial releases across sites, model input data are supported by robust evidence. As illustrated in Section 5.2.2.1.2, EPA estimated exposures and risks across the full distribution of facility releases both for the whole dataset and for a subset of facilities with high quality reporting information. Comparison of these distributions demonstrates that inclusion of locations relying on more limited release information had limited impact on the overall distributions of exposures.
    - For COUs/OESs that rely primarily on release data reported to TRI via Form R, or reported to ICIS-NPDES via DMR, site-specific release estimates are supported by moderate to robust evidence. As described in Appendix E.3.1, these release estimates are based on release amounts reported by facilities. Most COUs/OESs are included in this group.
    - For COUs/OESs that rely primarily on release data reported to TRI via Form A, site-specific release estimates are supported by moderate evidence. As described in Appendix E.3.1, Form A simply indicates that releases are below the reporting thresholds and specific release estimates require assumptions about amounts, locations, and media of release. The Import and Repackaging OES releases used in this analysis are entirely based on Form A reporting of releases, and just under half of the Industrial Uses OES releases were reported via Form A.

- For COUs/OESs that rely primarily on other sources of release information or generic scenarios, site-specific release estimates are supported by slight to moderate evidence. For these scenarios, EPA estimated daily wastewater discharges by using various modeling approaches, including the use of surrogate TRI and DMR data and modeling using data from literature, GSs, and ESDs.
    - For DTD sources, release information is supported by slight to moderate evidence. Although confidence in the individual contribution from some specific COUs (ie specific consumer or commercial product categories) is lower, confidence in estimates of overall DTD releases is moderate. The presented model is intended to inform the total contribution of DTD releases to overall aggregate instream concentration as well as providing evidence of individual COUs that may be most influential. Presented results should be taken in relation to one another qualitatively rather than discrete quantitative values. Distributions of DTD releases of consumer and commercial products were estimated for each COU on a per capita basis using the SHEDS-HT model.
    - For hydraulic fracturing releases, release information is supported by moderate evidence. Releases were estimated using Monte Carlo modeling with information from the Draft ESD on Hydraulic Fracturing and FracFocus 3.0.

- **Comparison of Modeled and Monitored Data.** Comparisons of estimated and measured exposures provide moderate evidence. Because most of the available monitoring data are not collected in locations with known releases temporally or spatially, it is difficult to make direct comparisons in most locations. However, in case study locations where monitoring data re located near release sites, comparisons demonstrate that there is concordance between measured and/or reported and modeled estimates (Section 2.3.1.4), increasing the overall strength of the evidence. Monitoring data confirm that 1,4-dioxane is present in some surface water and drinking water. Uncertainty as to whether trends observed in case study locations are representative of all of the sites decreases overall confidence in these comparisons.

Overall Confidence in Exposure Estimates

Overall confidence in drinking water exposure estimates for surface water concentrations modeled from facility releases (Section 3.2.2.1.1) is high across the overall distribution, particularly when limited to sites with high quality sources of release data. For individual facilities and COUs, overall confidence in exposure estimates varies depending on the confidence in source-specific release data. The modeling methodology used for this analysis is supported by moderate evidence. This approach makes some conservative assumptions about flow rates and release frequency and amount. It is designed to estimate water concentrations expected at specific locations. Available monitoring data confirm that 1,4-dioxane is present in some surface water and drinking water, though most of the available data were not collected near release sites are therefore not directly comparable. The overall level of confidence in OES/COU-specific exposure estimates depends on the source of OES/COU-specific release data described in Appendix E.3:

- Overall confidence in drinking water exposure estimates is medium to high for OESs/COUs that rely primarily on site-specific release data reported to DMR or to TRI via Form R.
- Overall confidence in site-specific drinking water exposure estimates is medium for OESs/COUs for which site-specific release estimates are based on reporting to TRI via Form A
- Overall confidence in drinking water exposure estimates is low to medium for OESs/COUs for which site-specific release estimates are based on surrogate or modeled information.

Overall confidence in drinking water exposure estimates for DTD releases under varying conditions (Section 3.2.2.1.2), is medium. The modeling methodology used for this analysis is supported by robust evidence. This analysis is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. This analysis defines the conditions under which exposures are higher, but is not designed to predict the specific levels of exposure resulting from DTD releases at specific locations with precision. Exposure estimates rely on estimated distributions of DTD releases of specific consumer and commercial products categories associated with each COU. Distributions of DTD releases of consumer and commercial products were estimated for each COU on a per capita basis using the SHEDS-HT model. While confidence in the individual contribution from some specific COUs is lower, confidence in estimates of overall DTD releases is moderate.

Overall confidence in drinking water exposure estimates for hydraulic fracturing releases (Section 3.2.2.1.3) is medium. The modeling methodology used for this analysis is supported by robust evidence and is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. Releases used as inputs in the model were estimated using Monte Carlo modeling that captures variability across sites. However, the modeled exposure estimates are not directly tied to specific releases at known locations, decreasing the strength of the evidence related to the representativeness of the exposure estimates for actual exposures.

Overall confidence in drinking water exposure estimates for aggregate surface water concentrations predicted by probabilistic modeling (Section 3.2.2.1.4) is high across the overall distribution. For individual facilities and COUs, overall confidence in exposure estimates varies depending on the confidence in source-specific release data. The modeling methodology used for this analysis is supported by robust evidence and is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. The estimated drinking water concentrations modeled in this analysis incorporate contributions from direct and indirect industrial releases, DTD releases, and other upstream sources. Available monitoring data confirm that 1,4-dioxane is present in some surface water and drinking water, though most of the available data were not collected near release sites and are therefore not directly comparable. The overall level of confidence in resulting exposure estimates depends on the source of OES/COU-specific release data described in Appendix E.3:

- Overall confidence in drinking water exposure estimates is medium to high for OESs/COUs that rely primarily on release data reported to DMR or to TRI via Form R. Most COUs/OESs are included in this group
- Overall confidence in drinking water exposure estimates is medium for OESs/COUs for which release estimates are based on reporting to TRI via Form A. The Import and Repackaging OES releases used in this analysis are entirely based on Form A reporting of releases, and just under half of the Industrial Uses OES releases were reported via Form A.
- Overall confidence in drinking water exposure estimates is low to medium for OESs/COUs for which release estimates are based on surrogate or modeled information.

3.3.2.2 Drinking Water Exposure Estimates Based on Groundwater Concentrations

The weight of evidence for exposure estimates presented in this section is determined by several different evidence streams, including:

- Evidence supporting the exposure scenarios (Section 3.2.1)
- The quality and representativeness of available groundwater monitoring data (Section 2.3.2.1),
- Evidence supporting modeling approaches and input data (Sections 2.3.2.3.1 and 2.3.2.4.1)
- Evidence supporting release data used as model input data (Section 2.2 and Appendix E.4)
- Concordance between modeled and monitored water concentrations.

3.3.2.2.1 Groundwater Concentrations Resulting from Disposal to Landfill

Drinking water exposure estimates based on groundwater concentrations modeled for landfill disposal scenarios are supported by overall slight to moderate evidence.

- **Exposure Scenarios and Exposure Factors.** The exposure scenarios and exposure factors underlying these drinking water exposure estimates are supported by slight to moderate evidence. Exposure factors for drinking water are based on robust data on drinking water intakes, body weight, and other standard exposure factors from the EPA's *Exposure Factors Handbook* (U.S. EPA, 2011). However, the drinking water exposure scenario relies on the assumption that the groundwater concentrations estimated with the DRAS model may occur in locations where groundwater is used as a primary drinking water source. While there is uncertainty around this assumption, this scenario represents a sentinel exposure scenario

- **Measured and Monitored Data.** Measured/monitored data are supported by moderate evidence. Monitoring data were available to sufficiently cover most or all of the population groups included within the exposure scenarios but there are a limited number of studies to corroborate findings. Since little data is readily available on the concentration of 1,4-dioxane near or around landfills in groundwater, some caution is required when interpreting monitoring data as it may not be fully representative of conditions around all landfills.

- **Modeling Methodology.** The modeling methodology is supported by robust evidence. The DRAS methodology for deriving the estimate is well described. The underlying computational or scientific basis is robust and has an empirical basis considering chemical specific properties.

- **Model Input Data.** The release data relied on as a model input is supported by slight evidence. Model inputs for the DRAS model include chemical properties of 1,4-dioxane that are well-defined and reviewed and therefore supported by robust evidence. However, model inputs for leachate concentrations and loading rates are more uncertain. EPA does not have reasonably available information on actual concentrations of 1,4-dioxane in leachate for most landfills and therefore selected landfill leachate concentrations are based on potential for risk to human health. Loading rates are based on the range reported in TRI for RCRA subtitle C landfills and therefore may not be representative of nonhazardous landfills evaluated in this analysis. These uncertainties around landfill leachate concentrations and loading rates decrease the strength of the evidence for model input data.

- **Comparison of Modeled and Monitored Data.** Comparison of estimated and measured exposures provides moderate evidence because monitoring data confirm the presence of 1,4-dioxane in groundwater in some locations and modeled estimates and measured exposure values are comparable, however differences in methodology, collection, or context make it difficult to arrive at full concordance.

Overall Confidence in Exposure Estimates

Overall confidence in drinking water exposure estimates resulting from disposal to landfills (Section 3.2.2.2.1) is low to medium. The modeling methodology is robust. However, the release information relied on as model input data is supported by slight to moderate evidence, decreasing overall confidence. In addition, this drinking water exposure scenario relies on the assumption that the groundwater concentrations estimated with the DRAS model may occur in locations where groundwater is used as a primary drinking water source. While the substantial uncertainty around the extent to which these exposures occur decreases overall confidence in the exposure scenario, this scenario represents a sentinel exposure.

3.3.2.2.2 Groundwater Concentrations Resulting from Disposal of Hydraulic Fracturing Waste

Drinking water exposure estimates based on modeled groundwater concentrations estimated under a range of hydraulic fracturing waste disposal scenarios are supported by slight to moderate evidence.

- **Exposure Scenarios and Exposure Factors.** The exposure scenario factors underlying these exposure estimates are supported by slight to moderate evidence. Exposure factors for drinking water are based on robust data on drinking water intakes, body weight, and other standard exposure factors from the Exposure Factors Handbook. However, the drinking water exposure scenario relies on the assumption that the estimated groundwater concentrations may occur in locations where groundwater is used as a primary drinking water source.

- **Measured and Monitored Data.** The measured/monitored data are supported by indeterminate evidence. Available groundwater monitoring data are not located near hydraulic fracturing operations and do not provide information about the potential for hydraulic fracturing operations to contribute to groundwater contamination.

- **Modeling Methodologies.** The modeling methodology and input data are supported by robust evidence. The methodology for deriving the estimate is well described, the underlying computational or scientific basis is robust, and has an empirical basis considering chemical specific properties.

- **Model Input Data.** Hydraulic fracturing releases are supported by moderate evidence. As described in Appendix E.4.4, releases were estimated using Monte Carlo modeling with information from the Draft ESD on Hydraulic Fracturing and FracFocus 3.0. DRAS modeling was based on very limited data on concentrations of 1,4-dioxane in produced water as reported in the literature. Reliance on limited data and uncertainty around the representativeness of that data decrease the strength of the evidence for model input data.

- **Comparison of Modeled and Monitored Data.** The comparison of estimated and measured exposures is not rated because no comparisons between estimated and measured exposures were made.

Overall Confidence in Exposure Estimates

Overall confidence in drinking water exposure estimates resulting from disposal of hydraulic fracturing waste (Section 3.2.2.2.2) is low to medium. The modeling methodology is robust and the release information relied on as model input data is supported by moderate evidence. However, no monitoring data are available to confirm detection of 1,4-dioxane in groundwater near hydraulic fracturing operations. This drinking water exposure scenario relies on the assumption that the estimated groundwater concentrations may occur in locations where groundwater is used as a primary drinking water source. While the substantial uncertainty around the extent to which these exposures occur decreases overall confidence in the exposure scenario, this scenario represents a sentinel exposure.

3.3.3 Air

The weight of the scientific evidence for exposure estimates presented in this section is determined by several different evidence streams, including:

- Evidence supporting the exposure scenarios (Section 3.2.1)
- The quality and representativeness of available groundwater monitoring data (Section 2.3.3.1),
- Evidence supporting modeling approaches and input data (Section 2.3.3.2)
- Evidence supporting release data used as model input data (Section 2.3.3.2)
- Concordance between modeled and monitored water concentrations.

As described in Section 2.3.3, 1,4-dioxane concentrations in air were estimated for areas around industrial COUs reported to TRI, hydraulic fracturing operations, and institutional and industrial laundry facilities. The associated strengths and limitations of these estimated environmental concentrations are described in Section 2.3.3.3. The general population air exposure scenarios and exposure factors used to estimate exposures are described in Section 3.2.3.

3.3.3.1 Modeled Air Concentrations for Industrial COUs Reported to TRI

Inhalation exposure estimates resulting from 1,4-dioxane releases for industrial COUs reported to TRI are supported by overall moderate evidence.

- **Exposure Scenarios and Exposure Factors.** Exposure scenarios underlying these exposure estimates are supported by moderate evidence. The exposure factors used to build the exposure scenarios are directly relevant to general population exposures for communities living in close proximity to releasing facilities. While the long-term exposure scenarios are most directly relevant for individuals who reside in fenceline communities for many years, these scenarios are expected to be within the range of normal habits and exposure patterns expected in the general population. However, there is uncertainty around the extent to which people actually live and work around the specific facilities where exposures are highest, decreasing the overall strength of evidence for these exposure scenarios, particularly at the distances nearest to facilities.
- **Measured and Monitored Data.** The measured/monitored data are supported by indeterminate evidence. No measured or monitored data were available.
- **Modeling Methodologies.** The modeling methodology used to estimate exposure concentrations via the ambient air pathway is supported by robust evidence. Air concentrations were estimated using AERMOD and IIOAC. AERMOD is EPA's regulatory model and has been thoroughly peer reviewed; therefore, the general confidence in results from the model is high but reliant on the integrity and quality of the inputs used and interpretation of the results. Confidence in modeled air concentrations resulting from stack releases is lower at distances less than 100 m of release sites, but confidence in modeled concentrations for fugitive emissions is higher near release sites. While this is a source of uncertainty, air concentrations from fugitive emissions tend to peak within 10m of release sites while stack releases were found to peak around 100 m, indicating that air concentrations modeled at distances less than 100 m of release sites are generally driven by fugitive emissions. IIOAC is an Excel-based model with results based on pre-run AERMOD exposure scenarios under a variety of environmental and release conditions. There is a moderate to high confidence in air concentrations estimated using IIOAC because, although IIOAC results are based on pre-run AERMOD exposure scenarios (high confidence), some key sources of uncertainty identified in Section 2.3.3.3 (like limited set of distances evaluated (100, 100 to 1,000, and 1,000 m) and assumptions made about meteorological conditions necessary to provide a more conservative exposure estimate) can lead to a slightly lower confidence (moderate).
- **Model Input Data.** Model input data on air releases are supported by slight to robust evidence, with the strength of the evidence varying across COUs/OESs. A more detailed OES-specific discussion of the confidence in sources of release information is presented in Appendix E.5.4.
    - For COUs/OESs that rely primarily on release data reported to TRI via Form R, site-specific release estimates are supported by moderate to robust evidence. As described in Appendix E.5.4, these release estimates are based on specific release amounts and other source-specific information reported by facilities as a regulatory requirement.

  - For COUs/OESs that rely primarily on release data reported to TRI via Form A, site-specific release estimates are supported by moderate evidence. As described in Appendix E.5.4, Form A simply indicates that releases are below the reporting thresholds and specific release estimates require assumptions about exact amounts and locations of releases.
  - For COUs/OESs that rely primarily on other sources of release information or generic scenarios, release estimates are supported by evidence ranging from slight to moderate evidence. For these scenarios, EPA estimated daily and annual air releases using various modeling approaches, including the use of surrogate TRI data and modeling using data from literature, GSs, and ESDs.
- **Comparison of Modeled and Monitored Data.** Comparison of estimated and measured exposures provide indeterminate evidence. No measured or monitored data were available for comparison.

Overall Confidence in Exposure Estimates

Overall confidence in inhalation exposure estimates resulting for air concentrations modeled based on industrial releases (Section 3.2.3.1) varies across COUs. The AERMOD modeling methodology used for this analysis is robust and considers contributions from both stack and fugitive emissions. The exposure scenarios considered are most relevant to long-term residents in fenceline communities. There is uncertainty around the extent to which people live and work in the specific locations where exposures are highest, decreasing confidence in the exposure scenarios, particularly at distances nearest to facilities. Overall confidence varies due to variable levels of confidence in underlying release information used to the support the analysis:

- Overall confidence in site-specific inhalation exposure estimates is medium to high for OESs/COUs that rely primarily on release data reported to TRI via Form R.
- Overall confidence in site-specific inhalation exposure estimates is medium for OESs/COUs for which release estimates are based on data reported to TRI via Form A.
- Overall confidence in inhalation exposure estimates is low to medium for OESs/COUs for which release estimates are based on surrogate or modeled information.

3.3.3.2 Air Concentrations Modeled near Hydraulic Fracturing Operations and Industrial/Institutional Laundries

Inhalation exposure estimates resulting from 1,4-dioxane released to air from hydraulic fracturing operations and industrial/institutional laundries are supported by overall moderate evidence.

- **Exposure Scenarios and Exposure Factors.** Exposure scenarios underlying these exposure estimates are supported by moderate evidence. The factors used to build the exposure scenarios are directly relevant to general population exposures for communities living in close proximity to releasing facilities. While the long-term exposure scenarios are most directly relevant for individuals who reside in fenceline communities for many years, these scenarios are expected to be within the range of normal habits and exposure patterns expected in the general population. However, there is some uncertainty around the extent to which people actually live and work around the specific locations where exposures are highest, decreasing the overall strength of evidence for these exposure scenarios.
- **Measured and Monitored Data.** The measured/monitored data are supported by indeterminate evidence. No measured or monitored data were available.
- **Modeling Methodologies.** The modeling methodology used to estimate exposure concentrations via the ambient air pathway is supported by robust evidence. Air concentrations were estimated

using IIOAC. IIOAC is an Excel-based model with results based on pre-run AERMOD exposure scenarios under a variety of environmental and release conditions. There is a moderate to high confidence in air concentrations estimated using IIOAC because, although IIOAC results are based on pre-run AERMOD exposure scenarios (high confidence), some key sources of uncertainty identified in Section 2.3.3.3 (like limited set of distances evaluated (100, 100 to 1,000, and 1,000 m) and assumptions made about meteorological conditions necessary to provide a more conservative exposure estimate) can lead to a slightly lower confidence (moderate).

- **Model Input Data.** Input data used for modeling exposures from hydraulic fracturing operations and industrial/institutional laundries are supported by moderate evidence. As described in Appendix E.5.4, these modeled exposure estimates are based on alternative release estimates and scenario conditions found in the literature and derived with Monte Carlo models of release estimate, some of which have been peer reviewed, others which may not be peer reviewed. Since the modeled exposures are based on alternative release estimates, which in turn are based on modeled data and outputs, there is a lower overall confidence in the modeled exposures from such input data. Additionally, exposure estimates using this input data requires certain assumptions which can lead to a lower overall confidence in the estimated exposure concentrations.

- **Comparison of Modeled and Monitored Data.** Comparison of estimated and measured exposures provide indeterminate evidence. No measured or monitored data were available for comparison.

Overall Confidence in Exposure Estimates

Overall confidence in inhalation exposure estimates resulting for air concentrations modeled based on releases from hydraulic fracturing operations (Section 3.2.3.2) is medium. The modeling methodologies used to estimate air concentrations are robust. The distribution of air releases used as model input data were estimated using Monte Carlo modeling and rely on assumptions. No air monitoring data were available to confirm detection of 1,4-dioxane in air near hydraulic fracturing operations. There is uncertainty around the extent to which people live and work in the specific locations where exposures are highest, decreasing confidence in the exposure scenarios.

Overall confidence in inhalation exposure estimates resulting from air concentrations modeled based on releases from industrial and institutional laundries ion 0) is medium. The modeling methodologies are robust. The distribution of air releases used as model input data were estimated using Monte Carlo modeling and rely on assumptions. No air monitoring data were available to determine whether 1,4-dioxane is detected near industrial and institutional laundry facilities.

4 HUMAN HEALTH HAZARD

1,4-Dioxane – Human Health Hazards (Section 4):
Key Points

EPA previously evaluated reasonably available information for human health hazards and identified hazard endpoints for non-cancer effects and cancer effects following acute and chronic exposures. This section describes adjustments made to previously published hazard values to align with the exposure scenarios evaluated in this supplemental evaluation.

- EPA considered the potential for increased susceptibility across PESS factors throughout the hazard assessment and dose-response analysis. PESS categories identified in the assessment include lifestage, genetics, and preexisting disease.
- The primary acute/short-term, non-cancer endpoint for 1,4-dioxane is liver toxicity following inhalation exposure.
- The primary chronic, non-cancer endpoints for 1,4-dioxane are liver toxicity and systemic effects on the olfactory epithelium.
- Inhalation cancer endpoint for 1,4-dioxane is based on combined tumor risk at multiple sites.
- Oral and dermal cancer endpoints for 1,4-dioxane are based on liver tumors following oral exposures.

4.1 Summary of Hazard Endpoints Previously Identified in the 2020 Risk Evaluation

This draft supplement relies on the Hazard Identification and Dose-Response Assessment that was previously described in the 2020 RE. All hazard values used to calculate risks for 1,4-dioxane in this draft supplement were derived from the previously peer-reviewed PODs published in the 2020 RE and amended in the recent correction memo.

Hazard values used in the 2020 RE include human equivalent concentrations (HECs) and human equivalent doses (HEDs) for non-cancer endpoints. Additionally, an inhalation unit risk (IUR) and cancer slope factor (CSF) for lifetime cancer risk were derived for both occupational and consumer scenarios for COUs where it was applicable. The hazard values published in the 2020 RE and used as the basis for hazard values in this draft supplement were developed with consideration for potentially susceptible subpopulations. Several potential sources of susceptibility were discussed qualitatively including lifestage, genetic variability, liver disease, and other chronic diseases that may influence metabolism or target organ susceptibility. EPA applied a 10× uncertainty factor to non-cancer hazard values to account for these sources of human variability.

4.2 Summary of Adjustments to Previously Established Hazard Values

For many of the exposure scenarios evaluated in this draft supplement, the previously established peer-reviewed hazard values were applied without modification. For example, risks from occupational exposures to products containing 1,4-dioxane as a byproduct can be evaluated using the acute, chronic, and cancer hazard values previously developed for OESs.

Some of the exposure scenarios included in this draft supplement require duration adjustments to the previously established PODs. For example, to evaluate risks from ambient air exposures for fenceline communities, EPA assumes continuous exposure to air for 24 hours/day, 7 days/week. As described in more detail below, EPA adjusted the previously established HEC and IUR values (originally developed for 8 hours/day, 5 days/week exposures) to identify hazard values appropriate for continuous exposure scenarios.

In addition, acute and chronic non-cancer oral and dermal HEDs extrapolated from occupational HECs were corrected to apply consistent breathing rates assumptions.

The full set of hazard values used to evaluate risk from the exposure scenarios in this draft supplement are presented in Table 4-1.

Table 4-1. Hazard Values Used for 1,4-Dioxane in this Draft Supplement

Scenario (Population)	Endpoints	Inhalation HEC/IUR	Dermal HED/CSF	Oral HED/CSF	Total Uncertainty Factors	Reference(s)
Acute non-cancer (general population)	Systemic liver toxicity	26.2 ppm (94.5 mg/m^3) 24 hours	17.4 mg/kg-d (extrapolated from HEC)	17.4 mg/kg-d (extrapolated from HEC)	300	(Putz et al., 1979) (Mattie et al., 2012)
Acute non-cancer (occupational)	Systemic liver toxicity	78.7 ppm (284 mg/m^3) 8 hours	17.4 mg/kg-d (extrapolated from HEC)[a]	17.4 mg/kg-d (extrapolated from HEC)[a]	300	(Mattie et al., 2012)
Chronic non-cancer (general population)	Olfactory epithelium effects attributed to systemic delivery (inhalation) [a]; liver toxicity (oral)	0.846 ppm (3 mg/m^3) 24 hours, 7 days/week	0.56 mg/kg-d (extrapolated from HEC)	2.6 mg/kg-d	30	(Kano et al., 2009; Kasai et al., 2009)
Chronic non-cancer (occupational)	Olfactory epithelium effects attributed to systemic delivery (inhalation) [a]; liver toxicity (oral)	3.6 ppm (12.8 mg/m$^{3)}$ 8 hours, 5 days/week	0.56 mg/kg-d (extrapolated from HEC)[b]	2.6 mg/kg-d	30	(Kano et al., 2009; Kasai et al., 2009)
Cancer (general population)	Inhalation cancer risk based on combined tumor risk at multiple sites; oral/dermal cancer risk based on liver tumors	IUR: 1.6E−02 per ppm 4.3E−06 (µg/m^3)$^{-1}$ 24 hours, 365 days/ year	CSF: 1.2E−01(mg/kg-d)$^{-1}$ (extrapolated from oral CSF)	CSF: 1.2E−01(mg/kg-d)$^{-1}$		(Kano et al., 2009; Kasai et al., 2009; NTP, 1986)
Cancer (occupational)	Inhalation cancer risk based on combined tumor risk at multiple sites; oral/dermal cancer risk based on liver tumors	IUR: 3.7E−03 per ppm 1.0E−06 (µg/m^3)$^{-1}$ 8 hours, 5 days/week	CSF: 1.2E−01(mg/kg-d)$^{-1}$ (extrapolated from oral CSF)	CSF: 1.2E−01(mg/kg-d)$^{-1}$		(Kano et al., 2009; Kasai et al., 2009)

[a] Due to the uniform distribution of lesions (rather than a distribution consistent with airflow), EPA concluded that effects in the olfactory epithelium may be due to systemic delivery rather than portal of entry effects due to the (see discussion in the 2020 RE on p 183, p.188).

[b] Occupational HEDs extrapolated from occupational HECs were corrected as described in the correction memo.

HEC = Human Equivalent Concentration; HED = Human Equivalent Dose; CSF = Cancer Slope Factor; IUR = Inhalation Unit Risk

4.2.1 Derivation of Acute/Short-Term Hazard Values

4.2.1.1 Inhalation HEC

The acute/short-term HECs are based on the lowest observed adverse effect concentration (LOAEC) for systemic liver toxicity observed in a short-term inhalation toxicity study in rats exposed 6 hours/day for 5 days/week. In the 2020 RE, EPA derived an HEC for 8-hour occupational exposures by applying a duration adjustment for an 8-hour exposure and a dosimetric adjustment factor of 1 (the default value when the calculated ratio of animal to human blood:air partition coefficients is greater than 1 (U.S. EPA, 1994b). The occupational HEC derived in the 2020 RE is based on default breathing rate assumptions and did not use adjustments for occupational breathing rates. For this draft supplement, EPA also derived an HEC for continuous general population exposures by applying a 24-hour duration adjustment to the original HEC.

4.2.1.2 Oral and Dermal HEDs

In the absence of acute oral or dermal toxicity studies, the acute/short-term HED was derived from the acute HEC using route-to-route extrapolation. An acute HED for the general population was derived from the duration-adjusted 24-hour HEC using the following equation:

$$\text{dermal or oral HED (mg/kg-d)} = \text{POD}_{\text{HEC}}\ (\text{mg/m}^3) \times \text{inhalation volume} \times 100\%\ \text{inhalation absorption} \div \text{body weight}$$

where the inhalation volume for the general population is 14.7 m^3/day and body weight is 80 kg, based on EPA's *Exposure Factors Handbook* (U.S. EPA, 2011). Inhalation absorption was estimated based on experimental data from inhalation exposures in humans (Young et al., 1977; Young et al., 1976) that indicated that 1,4-dioxane is readily absorbed; however, the available studies did not measure the parameters needed to generate a quantitative estimate of the fraction absorbed. Given this qualitative indication of rapid systemic uptake and the absence of quantitative inhalation absorption data, 100 percent inhalation absorption is assumed.

In the 2020 RE, an occupational acute HED was derived from the occupational HEC using the same equation but with an inhalation volume for workers based on higher breathing rates. As described in the correction memo, that derivation was incorrect. Because the occupational HEC was derived based on a normal general population breathing rate, the HED derivation should apply the same breathing rate assumptions. This draft supplement for 1,4-dioxane uses the revised acute occupational HED, which is equal to the general population HED.

4.2.2 Derivation of Chronic Hazard Values

4.2.2.1 Inhalation HEC

The chronic HECs are based on BMCL_{10} (*i.e.*, the lower confidence limit of the benchmark concentrations associated with a benchmark response of 10 percent) for effects in the olfactory epithelium following inhalation exposures to rats for 6 hours/day, 5 days/week for 2 years. In the 2020 RE, EPA derived an HEC for chronic worker exposures by applying a duration adjustment for 8 hours/day and a dosimetric adjustment factor of 1 (the default value using the RGDR approach for systemic effects when the calculated ratio of animal to human blood:air partition coefficients is greater than 1 (U.S. EPA, 1994b)). The occupational HEC derived in the 2020 RE used default breathing rate assumptions and did not adjust for occupational breathing rates. For this draft supplement, EPA also derived an HEC for continuous general population exposures by applying an alternate duration adjustment for 24 hours/day, 7 days/week.

4.2.2.2 Oral HEDs

A chronic oral HED was calculated based on a benchmark dose level (BMDL) for liver toxicity observed following chronic drinking water exposures to male rats and a nearly identical NOAEL value for liver toxicity in male rats in a similar chronic toxicity study. In the 2020 RE, EPA derived an HED by multiplying the nearly identical rodent BMDL and NOAEL values by $(BW_A/BW_H)^{0.25}$, where BW_A is the bioassay-specific rodent body weight, and BW_H is the default human body weight of 70 kg. Because the chronic HED is based on a daily dose rate (as opposed to an intermittent exposure concentration), it is equally applicable to both occupational and general population exposures and no additional conversion is required.

4.2.2.3 Dermal HEDs

In the absence of chronic dermal toxicity studies, chronic dermal HEDs were derived from both the chronic HEC and from the oral HED using route-to-route extrapolation. In the 2020 RE, the dermal HED used for occupational risk calculations was extrapolated from the chronic worker HEC. For this draft supplement, EPA also derived an HED from the HEC for continuous general population exposure. The duration-adjusted chronic HEC for general populations was converted to a chronic HED for the general population using the following equation:

$$\text{dermal HED (mg/kg-d)} = \text{inhalation BMDL}_{\text{HEC}}\ (\text{mg/m}^3) \times \text{inhalation volume} \times 100\%\ \text{inhalation absorption} \div \text{body weight}$$

where the inhalation volume for the general population is 14.7 m^3/day (U.S. EPA, 2011) for a 24-hour general population exposure and the body weight is 80 kg. As described above for the acute hazard values, EPA assumed 100 percent inhalation absorption. In the 2020 RE, an occupational HED was derived from the occupational HEC using the same basic equation but with an inhalation volume for workers based on higher breathing rates. The difference in the HEDs derived from occupational and general population HECs reflect differences in breathing rate assumptions for the two populations.

In the 2020 RE, an occupational chronic dermal HED was derived from the occupational HEC using the same equation but with an inhalation volume for workers based on higher breathing rates. As described in the correction memo, that derivation was incorrect. Because the occupational HEC was derived based on a normal general population breathing rate, the HED derivation should apply the same breathing rate assumptions. This assessment uses the revised occupational chronic dermal HED, which is equal to the general population HED.

4.2.3 Derivation of Cancer Hazard Values

For cancer, the inhalation unit risk (IUR) value was derived using the MS-Combo model to evaluate the combined cancer risk for multiple tumor sites observed in male rats following inhalation exposure for 6 hours/day, 5 days/week for 2 years. Tumor types included in the MS-Combo model include nasal cavity squamous cell carcinoma, Zymbal gland adenoma, hepatocellular adenoma or carcinoma, renal cell carcinoma, peritoneal mesothelioma, mammary gland fibroadenoma, and subcutis fibroma. In the 2020 RE, EPA derived an IUR for chronic worker exposures by applying a dosimetric adjustment factor of 1 and a duration adjustment for 8 hours/day. The occupational IUR derived in the 2020 RE applied default breathing rate assumptions and did not use adjustments for occupational breathing rates. The occupational IUR was rounded to $1\times10^{-6}\ (\mu g/m^3)^{-1}$ for application in risk calculations. For this draft supplement, EPA used that rounded occupational IUR to derive an IUR for continuous general population exposures to 1,4-dioxane by applying a duration adjustment for 24 hours/day, 7 days/week.

The oral and dermal cancer slope factor was derived using the Multistage Weibull Model for the liver tumors in female mice that had been exposed continuously via drinking water. In the 2020 RE, EPA calculated an HED for each tumor type by multiplying rodent doses by $(BW_A/BW_H)^{0.25}$, where BW_A is the bioassay-specific rodent body weight and BW_H is the default human body weight of 70 kg. The CSF was then calculated by dividing the benchmark response rate (0.5) by the HED. This CSF was applied to both occupational and consumer/general population scenarios using scenario-specific risk benchmarks and lifetime exposure estimates.

4.3 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty in the Hazard and Dose-Response Analysis

All assumptions or uncertainties inherent to the human health hazard assessment and dose-response analysis peer-reviewed in the 2020 RE are still applicable for this draft supplement. As described in the 2020 RE, EPA has medium confidence in the acute non-cancer PODs and high confidence in the chronic non-cancer PODs for oral, dermal, and inhalation exposures. EPA has high confidence in the cancer inhalation unit risk and medium-high confidence in the oral and dermal cancer slope factor. These conclusions are based on the fact that there is a robust set of high quality chronic and sub-chronic inhalation and oral exposure studies in rats and mice. The available evidence demonstrates consistent systemic toxicity and tumor formation in rats exposed via inhalation and in both rats and mice exposed via drinking water. Key sources of uncertainty include limited data on some sensitive reproductive and developmental endpoints, reliance on route-to-route extrapolation, uncertainty around the mode of action for 1,4-dioxane carcinogenicity, and the potential for subpopulations or lifestages with increased biological susceptibility to 1,4-dioxane. Available methods indicate potential higher inhaled doses in young children than adults, consistent with 1,4-dioxane specific studies integrating lifestage differences in ventilation, anatomy and metabolism via CYP2E1 (U.S. EPA, 2012). The preferred method to quantify these lifestage differences is a 1,4-dioxane specific PBPK model; however, the available PBPK models for 1,4-dioxane are not adequate and there are not generally accepted default methods not specific to 1,4-dioxane. Therefore, the air concentration is used as the exposure metric for all lifestages and the 10× uncertainty factor accounts for these lifestage differences per EPA guidance (U.S. EPA, 2012, 1994b).

5 HUMAN HEALTH RISK CHARACTERIZATION

1,4-Dioxane – Human Health Risk Characterization (Section 5):
Key Points

EPA estimated cancer and non-cancer risks for each exposure pathway for a range of central tendency and high-end exposure scenarios. Overall confidence in risk estimates varies across exposure pathways and COUs, depending on the data and assumptions used to derive exposure and risk estimates. Differences in estimates between central tendency and high-end exposure scenarios may reflect both variability across the population and uncertainty in the exposure assessment.

- Cancer and non-cancer risks were evaluated for occupational inhalation and dermal exposures to 1,4-dioxane present as a byproduct.
  - Cancer risk estimates for inhalation exposure range from 8.3×10^{-12} to $1.8\text{E}\times10^{-3}$ for central tendency exposures and 5.4×10^{-11} to 2.3×10^{-2} for high-end exposures.
  - Cancer risk estimates for dermal exposure range from 8.1×10^{-7} to 8.6×10^{-4} for central tendency exposures and from 5.0×10^{-6} to 1.5×10^{-2} for high-end exposures.
- Cancer and non-cancer risks were evaluated for drinking water exposures resulting from releases to surface water, including facility releases, down-the-drain releases, hydraulic fracturing releases, and aggregate releases from multiple sources.
  - Risk from individual facilities vary substantially within and across COUs, with cancer risk estimates ranging from 5.4×10^{-13} to 0.025.
  - Cancer risk estimates from modeled down-the-drain releases are highest in locations where large populations are contributing to these releases and where they are ultimately discharged to streams with low flow.
  - Cancer risk estimates from modeled hydraulic fracturing waste releases to surface water are 3.57×10^{-8} for median modeled releases and 1.45×10^{-6} for 95th percentile modeled releases.
  - Probabilistic modeling provides a distribution of risk estimates reflecting a range of drinking water scenarios that account for aggregate sources of 1,4-dioxane in water.
- Cancer risks were evaluated for drinking water exposures resulting from releases to land with potential to reach groundwater.
  - Risk estimates from landfill leachate are highest under disposal scenarios resulting in higher 1,4-dioxane concentrations in leachate and higher landfill loading rates.
  - Cancer risk estimates for drinking water exposures resulting from hydraulic fracturing waste released to land/groundwater range from 4.0×10^{-7} for median modeled releases to 8.6×10^{-6} for 95th percentile modeled releases.
- Cancer and non-cancer risks were evaluated for general population exposure to 1,4-dioxane in air.
  - Cancer risk estimates for industrial air releases reported to TRI were generally highest within 1,000 m of the facilities and lower at greater distances.
  - Cancer risk estimates within 1,000 m of hydraulic fracturing operations range from 0.2×10^{-8} to 7.1×10^{-5} for a range of model scenarios across a range of high-end and central tendency release scenarios.
  - Cancer risk estimates within 1,000 m of industrial and institutional laundries range from 1.5×10^{-11} to 3.8×10^{-8} across a range of high-end and central tendency air concentrations modeled for maximum release scenarios.

5.1 Risk Characterization Approach

The exposure scenarios, populations of interest, and toxicological endpoints used for evaluating risks from acute and chronic exposures are summarized below in Table 5-1. To estimate risks from occupational and general population exposure scenarios evaluated in this draft supplement, EPA used the same methods described in the 2020 RE, as summarized below.

Table 5-1. Use Scenarios, Populations of Interest, and Toxicological Endpoints Used for Acute and Chronic Exposures

Populations of Interest and Exposure Scenarios	**Workers** [a] Acute – Adolescent (≥16 years old) and adult workers exposed to 1,4-dioxane for a single 8-hour exposure Chronic – Adolescent (≥16 years old) and adult workers exposed to 1,4-dioxane for the entire 8-hour workday for 260 days per year for 40 working years
	General Population Drinking Water Exposures [b] Acute – Adults, children, and formula-fed infants exposed to 1,4-dioxane through drinking water over a 24-hour period Chronic – Adults, children, and formula-fed infants exposed to 1,4-dioxane through drinking water up to 33 years [d]
	General Population Ambient Air Exposure [c] Acute – People exposed to 1,4-dioxane through ambient air over a 24-hour period Chronic – People exposed to 1,4-dioxane through ambient air continuously up to 33 years [d]
Health Effects, Hazard Values and Benchmarks	**Non-cancer Acute/Short-term Hazard Values** Sensitive acute/short-term health effect: liver toxicity Acute Uncertainty Factors (Benchmark MOE) = 300 (UF_A = 3; UF_H = 10; UF_L = 10) • 8-hour HEC (occupational exposure) = 78.7 ppm • 24-hour HEC (continuous general population exposure) = 26.2 ppm • Acute Oral and Dermal HED (occupational and general population exposure) = 17.4 mg/kg **Non-cancer Chronic Hazard Values** Sensitive chronic health effects: • Liver toxicity (oral) • Effects on the olfactory epithelium due to systemic exposures (inhalation and dermal) Chronic Uncertainty Factors (Benchmark MOE) = 30 (UF_A = 3; UF_H = 10) • HEC (8-hour occupational exposure) = 3.6 ppm • HEC (continuous exposure general population exposure) = 0.846 ppm • Oral HED (for both occupational and general population scenarios) = 2.6 mg/kg/day • Dermal HED (extrapolated from HECs for both occupational and general population scenarios) = 0.56 mg/kg/day **Cancer Hazard Values** Inhalation cancer hazard for 1,4-dioxane is based on combined tumor hazard at multiple sites • IUR (occupational) = 3.7E−03 per ppm • IUR (continuous) = 1.6E−02 per ppm Oral and dermal cancer hazards for 1,4-dioxane are based on liver tumors following oral exposures • Oral/dermal slope factor = 1.2E−01 $(mg/kg/day)^{-1}$

[a] Adult workers (≥16 years old) include both female and male workers. Risks to ONUs were not calculated separately because exposure data were not available for ONUs for the OESs being evaluated. Risks to ONUs are assumed to be equal to or less than risks to workers who handle materials containing 1,4-dioxane as part of their job.
[b] These scenarios are used to evaluate potential risks from 1,4-dioxane in surface water, drinking water sources and groundwater that may be used as drinking water.
[c] Inhalation exposures are described in terms of air concentrations and do not include lifestage-specific adjustments; risk estimates based on air concentrations are intended to address risks to all lifestages (see Section 4.3).
[d] 33 years is the 95th percentile residential occupancy period. U.S. EPA *Exposure Factors Handbook* (U.S. EPA, 2011), Chapter 16, Table 16-5.
MOE = margin of exposure; UF_A = Interspecies uncertainty factor for animal-to-human extrapolation; UF_H = Intraspecies uncertainty factor for human variability; UF_L = LOAEC-to-NOAEC uncertainty factor for reliance on a LOAEC as the POD

5.1.1 Estimation of Non-cancer Risks

EPA used a margin of exposure (MOE) approach to identify potential non-cancer risks. The MOE is the ratio of the non-cancer POD divided by a human exposure dose. Acute and chronic MOEs for non-cancer inhalation and dermal risks were calculated using the following equation:

$$\mathrm{MOE}_{\text{acute or chronic}} = \frac{\text{Non} - \text{cancer Hazard value (POD)}}{\text{Human Exposure}}$$

Where:

MOE = Margin of exposure (unitless)
Hazard value (POD)= HEC (ppm) or HED (mg/kg-d)
Human Exposure = Exposure estimate (in ppm or mg/kg-d)

MOE risk estimates may be interpreted in relation to benchmark MOEs. Benchmark MOEs are typically the total UF for each non-cancer POD. The MOE estimate is interpreted as indicating a human health risk if the MOE estimate is less than the benchmark MOE (*i.e.*, the total UF). On the other hand, if the MOE estimate is equal to or exceeded the benchmark MOE, risk is not indicated. Typically, the larger the MOE, the more unlikely it is that a non-cancer adverse effect occurs relative to the benchmark. When determining whether a chemical substance presents unreasonable risk to human health or the environment, calculated risk estimates are not "bright-line" indicators of unreasonable risk, and EPA has discretion to consider other risk-related factors apart from risks identified in risk characterization.

5.1.2 Estimation of Cancer Risks

Extra cancer risks for repeated exposures to a chemical were estimated using the following equations:

$$\text{Inhalation Cancer Risk} = \text{Human Exposure } \times \text{IUR}$$

or

$$\text{Dermal/Oral Cancer Risk} = \text{Human Exposure } \times \text{CSF}$$

Where:

Risk = Extra cancer risk (unitless)
Human exposure = Exposure estimate (LADC in ppm)
IUR = Inhalation unit risk
CSF = Cancer slope factor

Estimates of extra cancer risks are interpreted as the incremental probability of an individual developing cancer over a lifetime following exposure (*i.e.*, incremental, or extra individual lifetime cancer risk).

5.2 Human Health Risk Characterization

5.2.1 Summary of Risk Estimates for Occupational Exposures

EPA estimated cancer and non-cancer risks for workers exposed to 1,4-dioxane based on the occupational exposure estimates that were described in Section 3.1. Risks to ONUs were not calculated separately because exposure data were not available for ONUs for the OESs being evaluated. Risks to ONUs are assumed to be equal to or less than risks to workers who handle materials containing 1,4-dioxane as part of their job.

Below are summaries of the cancer risk estimates for the inhalation and dermal exposures as well as key sources of uncertainty for all occupational exposure scenarios assessed in this draft supplement. These risk estimates are based on exposures to workers in the absence of PPE such as gloves or respirators. Section 3.1.2.4 contains an overall discussion on strengths, limitations, assumptions, and key sources of uncertainty for the occupational exposure assessment. Additionally, Appendix F contains a comprehensive weight of the scientific evidence summary table which presents an OES-by-OES discussion of the key factors that contributed to each weight of the scientific evidence conclusion. Complete risk calculations and results for occupational OES/COUs from the current analysis as well as those previously presented in the 2020 RE are available in *1,4-Dioxane Supplemental Information File: Occupational Exposure and Risk Estimates* (U.S. EPA, 2023r).

Risk estimates vary across OES/COUs. Because cancer risk is the primary risk driver in most exposure scenarios, this summary of results focuses on cancer risk estimates. For 7 of the 10 COU subcategories evaluated, high-end cancer risk estimates were above 1 in 10,000. For many of those COUs, acute and/or chronic non-cancer risk estimates were below the corresponding benchmark MOEs, indicating that non-cancer risks may also be a concern. Cancer risk estimates for inhalation exposure range from 8.3×10^{-12} to 1.8×10^{-3} for central tendency exposures and 5.4×10^{-11} to 2.3×10^{-2} for high end exposures. Cancer risk estimates for dermal exposure range from 8.1×10^{-7} to 8.6×10^{-4} for central tendency exposures and from 5.0×10^{-6} to 1.5×10^{-2} for high end exposures. Risks are highest for PET manufacture, hydraulic fracturing operations, ethoxylation processes, and dish soap/dishwashing detergent. For these OESs, cancer risk estimates were greater than 1 in 10,000 for both central tendency and high-end exposures. For these OESs, the key uncertainties include limited exposure monitoring data, age of data, representativeness of key modeling parameters, and the extent to which the data collected under past practices and operations are representative of modern practice and operations.

Overall confidence in risk estimates for occupational inhalation exposures ranges from low to high, depending on the confidence in exposure assessment for each OES/COU. As described in Section 4.3, overall confidence in the cancer inhalation unit risk underlying these risk estimates is high. As described in Section 3.3.1.1, the measured and monitored inhalation exposure data are supported by moderate to robust evidence. Additionally, the exposure modeling methodologies and underlying model input data is supported by moderate to robust evidence. However, there is uncertainty in the representativeness of the assessed exposure scenarios towards all potential exposures for the given OES/COU, limitations in the amount and age of monitoring data, and limitations in the modeling approaches towards 1,4-dioxane-specific use within the OES/COU. Therefore, while the underlying data and methods used to estimate occupational inhalation risk is supported by moderate to robust evidence, the overall confidence of these estimates ranges from low to high depending on the OES/COU. Key exposure considerations along with the corresponding risk estimates are below.

- **Industrial/Commercial Use of Textile Dye.** Risk estimates were derived using personal breathing zone and area monitoring data collected from 1991 to 2010 at four facilities linked to the use of textile dyes. Cancer risk estimates for inhalation exposure range from 1.6×10^{-5} for central tendency exposures to 3.6×10^{-2} for high-end exposures. However, there is uncertainty in the risk estimates. The monitoring data used in this analysis are limited (*i.e.*, 14 samples from four sites). It also is not known how manufacturing processes and workplace conditions have changed since the 1990s, when approximately half of the data was collected. For instance, EPA does not have information available about the actual activities of the sampled workers and the representativeness of the facility engineering controls to the modern practice.
- **Industrial/Commercial Use of Antifreeze.** Risk estimates were derived from occupational exposures modeled using Monte Carlo simulations for the worker activity of container

unloading. Cancer risk estimates for inhalation exposure range from 8.3×10^{-12} for central tendency exposures to 5.4×10^{-11} for high-end exposures. However, there is uncertainty in the risk estimates. Specifically, there is uncertainty as to the representativeness of some of the model input data and, therefore, subsequent calculated exposures to the actual distribution of antifreeze occupational exposures. This is due to limitations of using generic industry values identified for the automotive industry. Also contributing to the uncertainty is that EPA used consumer antifreeze use rates that were scaled up for commercial/industrial use in the Monte Carlo modeling.

- **Industrial/Commercial Use of Surface Cleaner.** Risk estimates were derived using 49 personal breathing zone samples taken in 2019 during the use of surface cleaners in domestic kitchens and bathrooms. Cancer risk estimates for inhalation exposure range from 1.1×10^{-7} for central tendency exposures to 1.8×10^{-6} for high-end exposures. However, there is uncertainty in the risk estimates. Specifically, the monitoring data summary did not provide discrete monitoring points and only provided summary statistics such as the geometric mean and maximum. Therefore, EPA could not calculate the 50th and 95th percentile exposures. Also, it is uncertain the extent to which the cleaning activities captured in this study reflect all occupational surface cleaning scenarios, as they were measured in a consumer setting.

- **Industrial/Commercial Use of Dish Soap and Dishwasher Detergent.** Risk estimates were derived using 29 personal breathing zone and area exposure monitoring samples from 1980 taken during the transfer of detergents to/from storage, liquid mixing, and detergent bottling. Cancer risk estimates for inhalation exposure range from 4.0×10^{-4} for central tendency exposures to 1.0×10^{-3} for high-end exposures. However, there is uncertainty in these risk estimates. Although the percent of 1,4 dioxane in the detergents used at the site where monitoring was conducted was up to 0.423 percent, all monitoring samples were non-detect for 1,4-dioxane. Therefore, EPA used the estimated LOD and LOD/2 for the worker high-end and central-tendency exposure estimates. These values are two to four orders of magnitude (for dish soap and dishwasher detergent, respectively) greater than the 2020 RE consumer exposure inhalation estimate. This difference can reasonably be expected considering occupational users of dish soap are potentially exposed to higher concentrations of 1,4-dioxane in concentrated commercial formulations and use these dish soaps at higher frequencies for longer durations than consumers. However, there is uncertainty as to the representativeness of these estimates due to the age of the monitoring data, number of non-detects, and the limited sample size.

- **Industrial/Commercial Use of Laundry Detergent.** Risk estimates were derived from occupational exposures modeled using Monte Carlo simulations for the worker activity of unloading detergent into machines, container cleaning, and laundry operations. For industrial laundries, cancer risk estimates for vapor inhalation exposure range from 2.0×10^{-7} for central tendency exposures to 9.2×10^{-7} for high-end exposures. For institutional laundries, cancer risk estimates for vapor inhalation exposure range from 1.6×10^{-7} for central tendency exposures to 7.1×10^{-7} for high-end exposures. In both cases, cancer risk estimates for total particulates inhalation range from 4.0×10^{-8} for central tendency exposures to 9.8×10^{-8} for high-end exposures. Cancer risk estimates for respirable particulates inhalation range from 1.3×10^{-8} for central tendency exposures to 3.3×10^{-8} for high-end exposures. However, there is uncertainty in the risk estimates. Specifically, there is uncertainty as to the representativeness of some of the model inputs and, therefore, subsequent calculated exposures to the actual distribution of laundry detergent occupational exposures. This is due to limitations of using generic industry values identified for institutional and industrial laundries.

- **Industrial/Commercial Use of Paint and Floor Lacquer.** Risk estimates were derived using 17 personal breathing zone samples collected by NIOSH in 1987 at a military vehicle painting site. Cancer risk estimates for inhalation exposure range from 8.0×10^{-5} for central tendency exposures to 5.9×10^{-4} for high-end exposures. However, there is uncertainty in the risk estimates. The monitoring data used in this analysis are limited (*i.e.*, 17 samples taken at one site). It also is not known how processes and workplace conditions have changed since 1987. For instance, EPA does not have information available about the actual activities of the sampled workers and the representativeness of the facility engineering controls to modern practice.
- **Polyethylene Terephthalate (PET) Manufacturing.** Risk estimates were derived using personal breathing zone monitoring data collected from 1985 to 1994 at five facilities linked to PET manufacturing. Cancer risk estimates for inhalation exposure range from 1.8×10^{-3} for central tendency exposures to 2.3×10^{-2} for high-end exposures. However, there is uncertainty in the risk estimates. The monitoring data used in this analysis are limited (*i.e.*, 11 samples from five sites). It also is not known how manufacturing processes and workplace conditions have changed since 1994. For instance, EPA does not have information available about the actual activities of the sampled workers and the representativeness of the facility engineering controls to modern practice.
- **Ethoxylation Processes.** Risk estimates were derived using one composite 8-hour time-weighted average personal breathing zone sample collected from one worker in 2000 at a soap and detergent manufacturing facility. Cancer risk estimates for inhalation exposure range from 4.6×10^{-4} for central tendency exposures to 5.9×10^{-4} for high-end exposures. However, there is uncertainty in the risk estimates. Specifically, EPA is unable to model these occupational exposures and cannot determine the statistical representativeness of the one monitoring data point (*e.g.*, high-end, central tendency) towards potential exposures from this OES. Additionally, there is uncertainty as to the worker activities covered by this monitoring data and whether all foreseeable activities, corresponding exposures, and workplace operations are represented.
- **Hydraulic Fracturing Operations.** Risk estimates were derived from occupational exposures modeled using Monte Carlo simulations for the worker activities of container unloading, container cleaning, and equipment cleaning. Cancer risk estimates for inhalation exposure range from 7.0×10^{-5} for central tendency exposures to 9.5×10^{-3} for high-end exposures. However, there is uncertainty in the risk estimates. Specifically, there is uncertainty as to the representativeness of some of the model input data and, therefore, the subsequent calculated exposures to the actual distribution of hydraulic fracturing occupational exposures. This is due to limitations of using generic industry values identified for the hydraulic fracturing industry as well as self-reported values from FracFocus as model parameters. FracFocus data may not fully represent operations across multiple sites throughout the United States as only certain sites volunteered to submit data.

Overall confidence in risk estimates for occupational dermal exposures is medium for all OES/COUs because the same modeling approach was used for all OES/COUs. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.1.2, the dermal exposure modeling methodology is supported by moderate evidence, with model input parameters from literature sources, a European model, standard defaults from the ChemSTEER User Guide (U.S. EPA, 2015a), and 1,4-dioxane product concentration data from process information. These sources range from slight to robust, depending on factors such as age and applicability to OES/COU. The modeling is limited by the use of standard input parameters that are not specific to 1,4-dioxane and a lack of variability in dermal exposure for different worker activities. Differences in the dermal exposure modeling across COUs are driven primarily by COU-specific weight fractions of 1,4 dioxane and the independent assessment of evaporative impacts in commercial and industrial settings. Therefore, EPA's overall confidence in the occupational dermal risk estimates is medium.

5.2.2 Summary of Risk Estimates for the General Population

5.2.2.1 Drinking Water – Surface Water Pathway

Risks from drinking water exposures were evaluated using a series of analyses that provide information about the specific contributions of releases associated with individual OESs as well as information about aggregate exposures and risks that could result from multiple sources releasing to the same water body. Because most reasonably available surface water and drinking water monitoring data are not co-located with 1,4-dioxane release sites, this analysis relies primarily on drinking water concentrations modeled based on reasonably available release information. Risks predicted based on reasonably available monitoring data are presented in 5.2.2.1.1.

EPA estimated cancer and non-cancer risks for adults, children, and formula-fed infants exposed to 1,4-dioxane in drinking water. All risk estimates presented in this summary focus on the scenario with the greatest potential exposure and risk. Because adult drinking water exposures relative to body weight are greater than exposures relative to body weight averaged over the course of childhood (as illustrated in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Release to Surface Water from Individual Facilities* (U.S. EPA, 2023h)) the drinking water exposure scenario with the greatest lifetime cancer risk is 33 years of drinking water exposure as an adult. Similarly, because cancer risk is the primary risk driver in most exposure scenarios, this summary of results focuses on cancer risk estimates. More comprehensive sets of risk estimates for non-cancer effects and other exposure scenarios are presented in the supplemental files referenced throughout this section.

Drinking water exposure and risk estimates are highly dependent on the amount of 1,4-dioxane released and the flow of the receiving water body. Both of these factors vary substantially across facilities within each COU/OES, making release amount and flow much more important predictors of risk than a facility's identified COU/OES. Exposure and risk estimates are also influenced by whether there is a drinking water intake downstream of a release and the degree of dilution that occurs between the point of release and the drinking water intake. Many of the risk estimates presented in the sections that follow (for facility-specific releases, DTD, hydraulic fracturing, and aggregate modeling) assume that no additional downstream dilution occurs prior to reaching drinking water intakes. This represents an upper end estimate of exposure and risk based on the available data and the potential for intakes to be directly downstream of a releasing facility. EPA conducted further analysis of the facility-specific releases to consider the potential impact of downstream dilution on actual concentrations at drinking water intakes and resulting risk estimates. Even when accounting for dilution between known releases and identified drinking water intake locations, water concentrations estimated at drinking water intakes, instances of cancer risks greater than 1 in 1 million for some public water systems are identified. Proximity of releases to drinking water intakes and dilution are further discussed in Section 2.3.1.2.4/Appendix G.2.4 and Section 5.2.2.1.2.

1,4-Dioxane is not readily removed through typical wastewater or drinking water treatment processes. Therefore, the drinking water risk estimates presented below are derived based on the assumptions that drinking water intakes are located near 1,4-dioxane release sites and that no 1,4-dioxane is removed by

POTWs or through drinking water treatment. Use of source water estimated concentrations of 1,4-dioxane to calculate cancer risk estimates is considered protective of all systems. These assumptions are further discussed in Section 2.3.1.1/Appendix G.1.2).

5.2.2.1.1 Risks from Exposure to Drinking Water Concentrations Indicated in Finished Drinking Water Monitoring Data

EPA evaluated risks for 1,4-dioxane concentrations reported in the reasonably available finished (treated) drinking water monitoring data. Monitoring data included in this analysis were from generalized, broad monitoring strategies, rather than targeted efforts to assess areas of known contamination. As previously illustrated in Figure 2-10, 1,4-dioxane was below limits of detection for 89 percent of finished drinking water samples included in UCMR3 and state databases. Table 5-2 summarizes the distribution of lifetime cancer risk estimates from 1,4-dioxane concentrations detected in finished drinking water reported in these databases (described in Section 2.3.1.1). This drinking water monitoring data provides evidence that 1,4-dioxane is present in some finished drinking water and may contribute to cancer risks in locations at the high end of monitored drinking water concentrations.

Monitoring data may not include the full range of 1,4-dioxane concentrations that result from industrial releases. As discussed in Section 2.3.1.1, available drinking water monitoring data do not necessarily capture locations that are most impacted by releases temporally or spatially and they often reflect concentrations at a single point in time rather than average concentrations. However, as described in Appendix G.2.3.2, in locations where monitoring data are available near release sites, comparisons demonstrate strong concordance between modeled concentrations and monitoring data. EPA's evaluation of drinking water risks therefore primarily relied on modeled estimates of 1,4-dioxane concentrations that occur near release sites.

Table 5-2. Lifetime Cancer Risk Estimates for 1,4-Dioxane Concentrations Detected in Finished Drinking Water

	Percentile Drinking Water Monitoring Data							
	Min	**5%**	**25%**	**Median**	**75%**	**90%**	**95%**	**Max**
Water conc. (µg/L)	2.00E−03	3.50E−02	3.50E−02	3.50E−02	3.50E−02	7.93E−02	0.16	13.3
Lifetime Cancer Risk	4.02E−11	1.95E−08	1.95E−08	1.95E−08	8.37E−08	2.79E−07	3.46E−07	7.42E−06

Lifetime cancer risks based on 33 years of oral exposure through drinking water as an adult. Percentiles reflect concentrations across the distribution of available drinking water monitoring data (this distribution includes non-detects as half the detection limit).

5.2.2.1.2 Risks from Exposures to Water Concentrations Modeled from Industrial Releases

To estimate the contribution of industrial releases to general population risks from drinking water, EPA calculated cancer and non-cancer risk estimates based on modeled surface water concentrations in receiving water bodies described in Section 2.3.1.3.1 and the resulting drinking water exposures calculated as described in Section 3.2.2. Because there is substantial variation and uncertainty around the extent of dilution that may occur in the receiving water body between the point of release and the locations of drinking water intakes, EPA calculated cancer risk estimates under a range of reasonable downstream dilution assumptions.

Figure 5-1 shows the distribution of cancer risk estimates for industrial releases reported to TRI and DMR, assuming that concentrations at drinking water intakes are the same as concentrations that occur at the point of release after initial mixing in the receiving water body. Based on available data, this is a plausible scenario in some locations. Lifetime cancer risk estimates are based on 33 years of exposure as an adult and range from 5.41×10^{-13} to 2.54×10^{-2}. The median cancer risk estimate for these modeled concentrations is 2.32×10^{-6} and the 95th percentile risk estimate is 4.92×10^{-3}. Acute and chronic non-cancer risk estimates for some facilities (data not shown) also indicate potential for non-cancer risk relative to benchmark MOEs. This analysis represents an upper bound drinking water exposure scenario in which intakes are located near the point of release or in which minimal additional dilution occurs downstream. Complete cancer and non-cancer risk estimates for facility and OES-specific releases are presented in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Release to Surface Water from Individual Facilities* (U.S. EPA, 2023h).

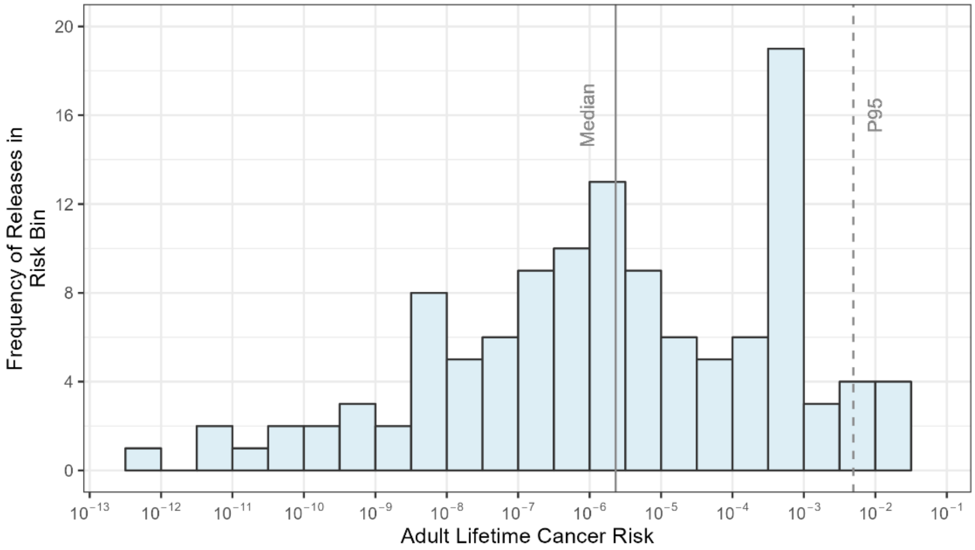

Figure 5-1. Distribution of Adult Lifetime Cancer Risk across all Facilities, Assuming No Additional Dilution Occurs between the Point of Release and the Location of Drinking Water Intakes

There is substantial variation in cancer risk estimates both within and across OESs. The large ranges of modeled water concentrations and corresponding risk estimates reflect the large differences in the amount of 1,4-dioxane released from facilities, the magnitude of flow within the receiving water body or both.

For facilities where specific release amounts or locations are not reported, release amounts and flow rates are based on conservative assumptions that may result in high risk estimates. There is uncertainty around risk estimates for those facilities with limited release information, but facility-specific information on release amounts and locations was available for most facilities. Therefore, while facility-specific risk estimates based on facilities with limited information should be interpreted with caution, most estimates are informed by moderate to robust modeling approaches and input data. To determine the extent to which inclusion of facilities with limited release information influences the overall distribution, EPA repeated this risk estimate analysis presented in Figure 5-1 using only facilities for which high quality release data are available (Figure 5-2). Specifically, this additional analysis is limited to facilities for which the annual release amount was sourced from either TRI Form R or DMR, and the receiving water body reach code was identified in the facility's NPDES permit. Out of the 120 total direct and indirect releases evaluated in this section, 80 met these strict data criteria. The resulting distribution of risk estimates are similar to the results of the analysis including all facilities, ranging from 5.41×10^{-13} to 2.54×10−2, with a median of 8.51×10^{-7} and 95th percentile of 4.92×10^{-3}.

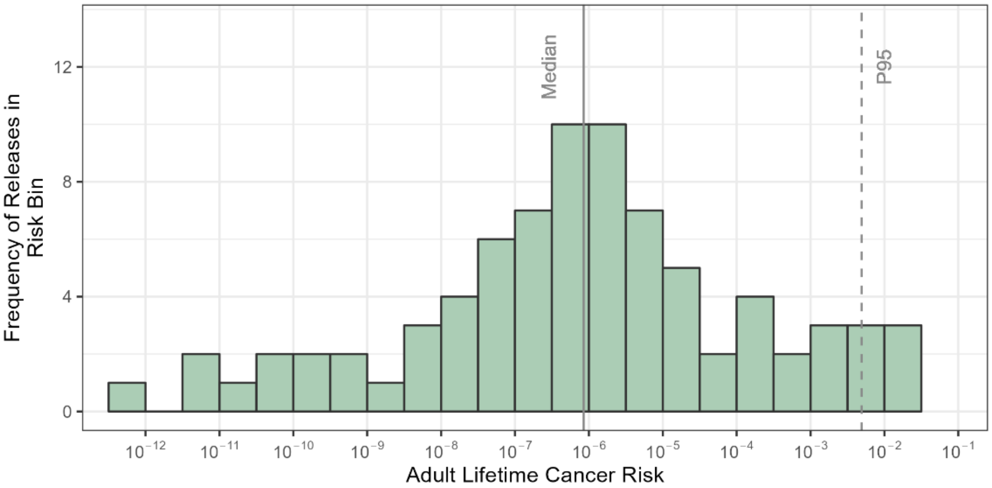

Figure 5-2. Distribution of Adult Lifetime Cancer Risk across Facilities with High Quality Release Data, Assuming No Additional Dilution Occurs between the Point of Release and the Location of Drinking Water Intakes

The risk estimates summarized in Figure 5-1 and Figure 5-2 rely on the assumption that concentrations at drinking water intakes are the same as concentrations estimated near the point of release. To evaluate the validity of that assumption, EPA considered the proximity of release sites to downstream drinking water intake locations for community and non-community non-transient PWSs. As shown in Table 5-3, of the 69 facilities with cancer risk greater than 1×10^{-6}, 22 (32 percent) have a downstream drinking water intake within 250 km and 4 of those have a drinking water intake within 10 km. A detailed description of this analysis is provided in Appendix G.2.4.

Table 5-3. Proximity of Nearest Downstream Drinking Water Intakes to Facilities Resulting in Cancer Risk Greater than 1×10^{-6}

Total Facilities Evaluated	Facilities with Cancer Risk above 1E−06	DWI within 250 km	DWI within 100 km	DWI within 50 km	DWI within 25 km	DWI within 10 km
120	69	22	17	11	7	4

The portion of 1,4-dioxane that remains after the additional dilution that occurs as it travels downstream is highly variable based on site-specific characteristics, ranging from less than 1 percent to nearly 100 percent of the original concentrations (Figure 5-3). The site-specific factors that influence this additional downstream dilution may not be fully captured in a national-scale assessment. Based on available site-specific information for each facility, the mean modeled dilution predicted at downstream drinking water intakes is diluted to 1 percent of original concentrations estimated in receiving water bodies near the point of release.

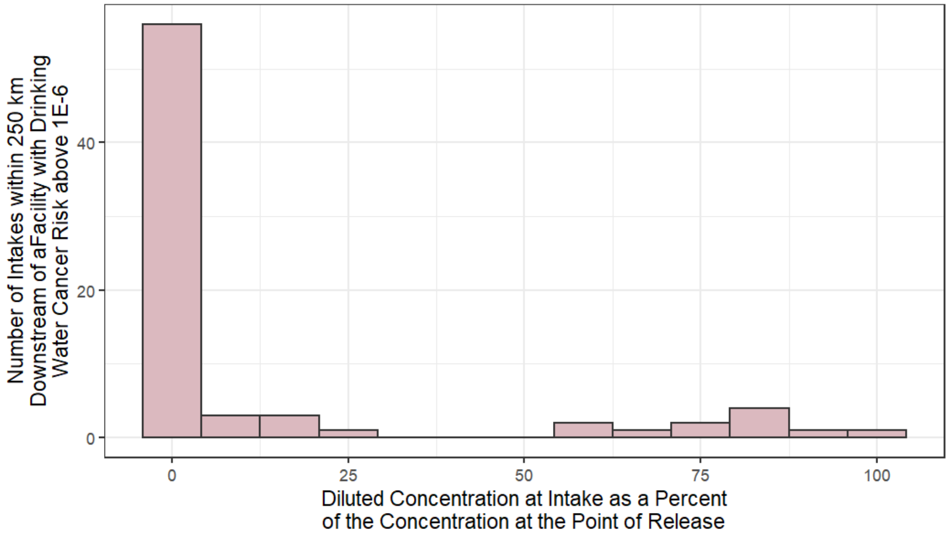

Figure 5-3. Distribution of Dilution of 1,4-Dioxane Concentrations at Downstream Drinking Water Intakes

Figure 5-4 shows the distribution of cancer risk estimates for industrial releases, assuming that concentrations at drinking water intake locations are diluted to 1 percent of the original 1,4-dioxane concentrations in surface water estimated at the point of release. Lifetime cancer risk estimates for these modeled concentrations range from 5.41×10^{-15} to 2.54×10^{-4}. The median cancer risk estimate is 8.51×10^{-9} and the 95th percentile risk estimate is 4.92×10^{-5}. This represents a plausible drinking water exposure scenario consistent with mean modeled downstream dilution predicted across all facilities based on available site-specific information.

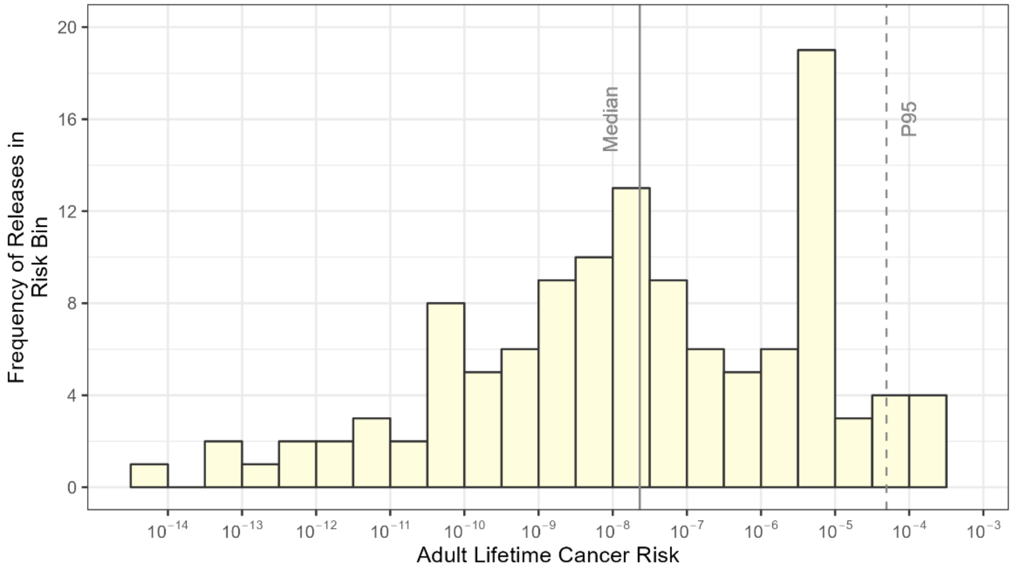

Figure 5-4. Distribution of Adult Lifetime Cancer Risk across all Facilities, Assuming Dilution to 1% of Initial Concentrations in the Receiving Water Body

In addition to estimating how the overall distribution of cancer risk estimates would shift based on a standard assumption of downstream dilution to 1 percent of original concentrations (as shown in Figure 5-4), EPA also estimated water concentrations and risks that may occur at specific drinking water intakes located downstream of releasing facilities, based on location-specific estimates of dilution. For the 22 facilities with cancer risk greater than 1 in 1 million and drinking water intakes located within 250 km downstream, EPA identified downstream intakes associated with 73 distinct PWSs.

Even when accounting for site-specific influences on dilution, EPA modeled concentrations that would result in adult lifetime cancer risk in excess of 1 in 1 million at intakes for 20 of the PWSs identified through this assessment, serving a combined population of 2,124,000 people. Adult lifetime cancer risk estimates were greater than 1 in 100,000 for 5 of these public water systems, serving a combined population of 834,000 people. This analysis also identified locations with multiple releasing facilities upstream of the same drinking water intake; however, in all such cases the aggregated adult lifetime cancer risk calculated at the intake for the aggregated diluted concentration was less than 1 in a million. A detailed description of this analysis is provided in Appendix G.2.4. Overall confidence in these dilution-adjusted risk estimates is high for drinking water intakes located at or near the point of release, but confidence decreases substantially with increasing distance downstream. This analysis does not provide a comprehensive survey of modeled 1,4-dioxane concentrations at all drinking water intakes. There may be additional drinking water intakes downstream of facilities releasing 1,4-dioxane that are not accounted for in the intake database used in this analysis.

Overall, these analyses indicate that in many locations, downstream dilution may be expected to substantially reduce 1,4-dioxane concentrations at the point of drinking water intakes. However, even when accounting for dilution, upstream industrial releases reported to TRI or DMR contribute to cancer risk estimates greater than 1 in a million or 1 in 100,000 at known drinking water intake locations.

The set of distributions presented in Figure 5-1, Figure 5-2, and Figure 5-4 indicate that high risks can occur in specific locations downstream of release sites due to factors such as the size of the releasing event(s), stream flow volume, proximity of the release site to drinking water intake, and limited drinking water treatment removal from typical treatment methods.

Overall confidence in the overall distribution of risk estimates for drinking water exposures resulting from facility releases is medium-high. Overall confidence in site-specific risk estimates for individual facility releases varies both within and across OES, depending on the confidence in the source-specific release data. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.1, the overall exposure modeling methodology used for this analysis is supported by moderate evidence. It is designed to estimate water concentrations expected at specific locations. Exposure estimates for this scenario are based on some conservative assumptions about flow rates and release frequency and amount. For most COUs, this analysis is limited to facilities that report via TRI and/or DMR. Other sources releasing smaller amounts of 1,4-dioxane are not directly captured. Available monitoring data confirm that 1,4-dioxane is present in some surface water and drinking water, though most of the available data were not collected near release sites are therefore not directly comparable.

The overall level of confidence in facility-specific release estimates and resulting risk estimates depends on the source of the release data described in Appendix E.3:

- Overall confidence in drinking water exposure estimates is medium to high for OESs/COUs that rely primarily on release data reported to DMR or to TRI via Form R. Most COUs/OESs are included in this group.
- Overall confidence in drinking water exposure estimates is medium for OESs/COUs for which release estimates are based on reporting to TRI via Form A. The Import and Repackaging OES

releases used in this analysis are entirely based on Form A reporting of releases, and just under half of the Industrial Uses OES releases were reported via Form A.

While confidence in facility-specific risk estimates varies, estimates for most facilities are informed by moderate to robust modeling approaches and input data. Furthermore, the overall distribution is not meaningfully altered by exclusion of facility-specific data based on more limited release information (as illustrated in Figure 5-1 and Figure 5-2). There is some uncertainty around the proximity of releases to drinking water intake locations and the extent to which 1,4-dioxane is further diluted prior to reaching intake locations. EPA therefore estimated distributions of cancer risk estimates under a range of assumptions about downstream dilution, reflecting the range of plausible drinking water intake scenarios, as indicated by available site-specific information.

5.2.2.1.3 Risks from Exposures to Water Concentrations Modeled from DTD Releases (from POTWs), Assuming No Downstream Dilution

EPA evaluated the potential contribution of DTD releases of consumer and commercial products to drinking water exposure and risk. Surface water concentrations at the point of DTD releases via POTWs are primarily determined by the size of the population contributing to DTD releases and the flow rates of receiving water bodies. Risk estimates presented in this section are not tied to known releases at specific locations. Rather, this analysis defines the conditions under which DTD releases would result in varying levels of risk.

Cancer risk estimates shown in Table 5-4 were calculated based on drinking water exposure estimates presented in Section 3.2.2.1.2, which correspond to surface water concentrations estimated by probabilistic modeling of DTD releases under varying population sizes and stream flows. The resulting risk estimates indicate that risk is highest in locations where large populations are contributing to DTD releases and those releases are ultimately discharged to streams with low flow. Cancer risk estimates greater than 1 in a million were seen in combinations of population size and flow rates that are likely to occur in some locations. As described in Section 2.3.1.3.2, EPA considered the frequency of the varying combinations of population sizes and flow rates. For communities with single POTWs treating wastewater, most fell into the range of 100 to 10,000 people, with the annual average flow of the receiving water body less than 300 cfs (Table 2-10). Cancer risk estimates for communities in this range of population sizes are as low as 2.04×10^{-8} at flows of 300 cfs and increase at lower flows. For example, cancer risk estimates for releases from a population size of 10,000 could be as high as 2.04×10^{-6} at a flow of 30 cfs. Acute and chronic non-cancer risk estimates for these scenarios do not indicate non-cancer risk relative to benchmark MOEs. For reference, stream flows of 100 cfs might be considered a small river, while anything less than 100 cfs would be considered a stream or creek. Complete cancer and non-cancer risk estimates for the range of water concentrations from DTD releases estimated under varying conditions using probabilistic modeling are presented in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Surface Water Concentrations Predicted with Probabilistic Modeling* (U.S. EPA, 2023i).

Overall confidence in risk estimates for drinking water exposures resulting from DTD releases under varying population and stream flow conditions is medium. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.1, the exposure modeling methodology used for this analysis is supported by robust evidence and is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. Exposure estimates rely on estimated distributions of DTD releases of consumer and commercial products for each COU. Distributions of DTD releases of consumer and commercial products were estimated for each COU on a per capita basis using the

SHEDS-HT model. Because this analysis is not tied to specific sites, there is uncertainty around the proximity of releases to drinking water intake locations and the extent to which 1,4-dioxane is further diluted prior to reaching intake locations. For this analysis, EPA assumed that no additional dilution occurs prior to reaching drinking water intakes. While confidence in the individual contribution from some specific COUs is lower, confidence in estimates of overall DTD releases is moderate.

Table 5-4. Lifetime Cancer Risk[a] Estimates from DTD Releases Alone (at the Point of Release) under a Range of Population and Flow Rate Scenarios

		Population Contributing to Down-the-Drain Releases				
		100	1,000	10,000	100,000	1,000,000
Stream Flow (cfs)	100	6.11E–09	6.11E–08	6.11E–07	6.11E–06	6.11E–05
	300	2.04E–09	2.04E–08	2.04E–07	2.04E–06	2.04E–05
	1,000	6.11E−10	6.11E−09	6.11E−08	6.11E−07	6.11E−06
	3,000	2.04E−10	2.04E−09	2.04E−08	2.04E−07	2.04E−06
	10,000	6.11E−11	6.11E−10	6.11E−09	6.11E−08	6.11E−07
	30,000	2.04E−11	2.04E−10	2.04E−09	2.04E−08	2.04E−07

[a] Lifetime cancer risks based on 33 years of oral exposure through drinking water as an adult.
The frequencies of each of these combinations of population size and flow rate are presented in Table 2-10.

5.2.2.1.4 Risks from Exposure to Drinking Water Concentrations Modeled from Disposal of Hydraulic Fracturing Produced Waters to Surface Water, Assuming No Downstream Dilution

EPA evaluated the potential contribution of the disposal of hydraulic fracturing produced waters to surface water by aggregating exposures and risks. The range of water concentrations that may result from releases of hydraulic fracturing waste to surface water were estimated using probabilistic modeling. Risk estimates presented in this section are not tied to known releases at specific locations. Rather, this analysis defines the conditions under which releases from hydraulic fracturing would result in varying levels of risk. These risk estimates are based on the assumption that 1,4-dioxane is not removed by POTWs or through drinking water treatment.

Cancer risk estimates across the full distribution of modeled releases are presented in Table 5-5. Cancer risk estimates are 3.57×10^{-8} for median modeled releases and 1.45×10^{-6} for 95th percentile modeled releases. Acute and chronic non-cancer risk estimates based on 95th percentile modeled releases do not indicate risk relative to benchmark MOEs. The maximum water concentration estimated by the model reflects a scenario in which waste is released to a stream with very low flow. EPA does not have site-specific information to indicate that such a scenario actually occurs. Complete risk estimates for the range of water concentrations from DTD releases estimated under varying conditions using probabilistic modeling are presented in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Surface Water Concentrations Predicted with Probabilistic Modeling* (U.S. EPA, 2023i).

Overall confidence in risk estimates for drinking water exposures resulting from hydraulic fracturing releases is medium. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.1, the exposure modeling methodology used for this analysis is supported by robust evidence and is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. Releases used as inputs in the model were estimated using Monte Carlo modeling that captures variability across sites. However, the modeled exposure estimates are not directly tied to specific releases at known locations, decreasing the strength of the evidence related to the representativeness of the exposure estimates for actual exposures. There is some uncertainty around the proximity of releases to drinking water intake locations and the extent to which 1,4-dioxane is further diluted prior to reaching intake locations. For this analysis, EPA assumed that no additional dilution occurs prior to reaching drinking water intakes.

Table 5-5. Lifetime Cancer Risks Estimated from Hydraulic Fracturing Produced Waters Disposed to Surface Water under a Range of Scenarios

Monte Carlo Distribution	Adult Lifetime Cancer Risk
Maximum	1.10E−04
99th Percentile	4.35E−06
95th Percentile	1.45E−06
Median	3.57E−08
5th Percentile	1.29E−10
Minimum	3.40E−18
LADDs used to calculate these cancer risk estimates are presented in Table 3-5.	

5.2.2.1.5 Aggregate Risks from Drinking Water Exposures Modeled from Multiple Sources Releasing to Surface Water, Assuming No Downstream Dilution

Multiple sources may contribute to 1,4-dioxane concentrations in drinking water sourced from surface water in a single location. EPA therefore estimated aggregate general population exposures and risks that could occur as a result of combined contributions from multiple sources. As described in Section 2.3.1.3.4, EPA used probabilistic modeling to predict aggregate surface water concentrations that could occur when accounting for DTD releases, indirect releases, and other upstream sources. EPA estimated cancer and non-cancer risks for the drinking water exposure estimates in Section 3.2.2, which correspond to the modeled aggregate surface water concentrations described in Section 2.3.1.3.4 and assume that no 1,4-dioxane is removed through treatment. This analysis also assumes that concentrations at drinking water intakes are not further diluted from the concentrations modeled near the point of release. There is wide variation in both cancer and non-cancer risk within and across OESs/COUs when taking into account aggregate contributions from other sources. This variation is illustrated in the cancer risk estimates shown in the distributions of cancer risk estimates for exposures modeled for each OES/COU in Figure 5-5. The large ranges of risk estimates for some OESs/COUs reflect substantial variation in releases and characteristics of receiving water bodies across the set facilities associated with those OESs. High-end cancer risk estimates in this analysis are very similar to high-end risk estimates for individual facility releases alone, indicating that high-end estimates are driven primarily by high-end industrial releases. Complete cancer and non-cancer risk estimates for the range of aggregate water concentrations estimated for each COU using probabilistic modeling are presented in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates* *for 1,4-Dioxane Surface Water Concentrations Predicted with Probabilistic Modeling* (U.S. EPA, 2023i).

Overall confidence in distributions of risk estimates for drinking water exposures resulting from for aggregate surface water concentrations predicted by probabilistic modeling varies across OES/COU. While confidence is not uniform for all facilities within an OES, overall confidence ratings for each OES are intended to communicate how the factors that contribute to confidence and uncertainty vary across COUs. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.1, the exposure modeling methodology used for this analysis is supported by robust evidence and is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. For most COUs, this analysis is limited to facilities that report releases via TRI and/or DMR. Other sources releasing smaller amounts of 1,4-dioxane are not directly captured, though the distribution of surface water monitoring data used to represent background concentrations in the model is intended to capture these other upstream sources. Available monitoring data confirm that 1,4-dioxane is present in some surface water and drinking water, though most of the available data were not collected near release sites and are therefore not directly comparable. In release locations where monitoring data are available, case studies demonstrate strong concordance between modeled estimates and measured surface water concentrations.

There is some uncertainty around the proximity of releases to drinking water intake locations and the extent to which 1,4-dioxane is further diluted prior to reaching intake locations. For this analysis, EPA assumed that no additional dilution occurs prior to reaching drinking water intakes. The characterization of downstream dilution presented in 5.2.2.1.2 for individual facility releases illustrates the extent to which downstream dilution may impact overall risk estimates.

The overall level of confidence in resulting exposure estimates depends on the source of OES/COU-specific release data described in Appendix E.3:

- Overall confidence in drinking water exposure estimates is medium to high for OESs/COUs that rely primarily on release data reported to DMR or to TRI via Form R. Most COUs/OESs are included in this group.
- Overall confidence in drinking water exposure estimates is medium for OESs/COUs for which release estimates are based on reporting to TRI via Form A. The Import and Repackaging OES releases used in this analysis are entirely based on Form A reporting of releases, and just under half of the Industrial Uses OES releases were reported via Form A.
- Overall confidence in drinking water exposure estimates is low to medium for OESs/COUs for which release estimates are based on surrogate or modeled information.

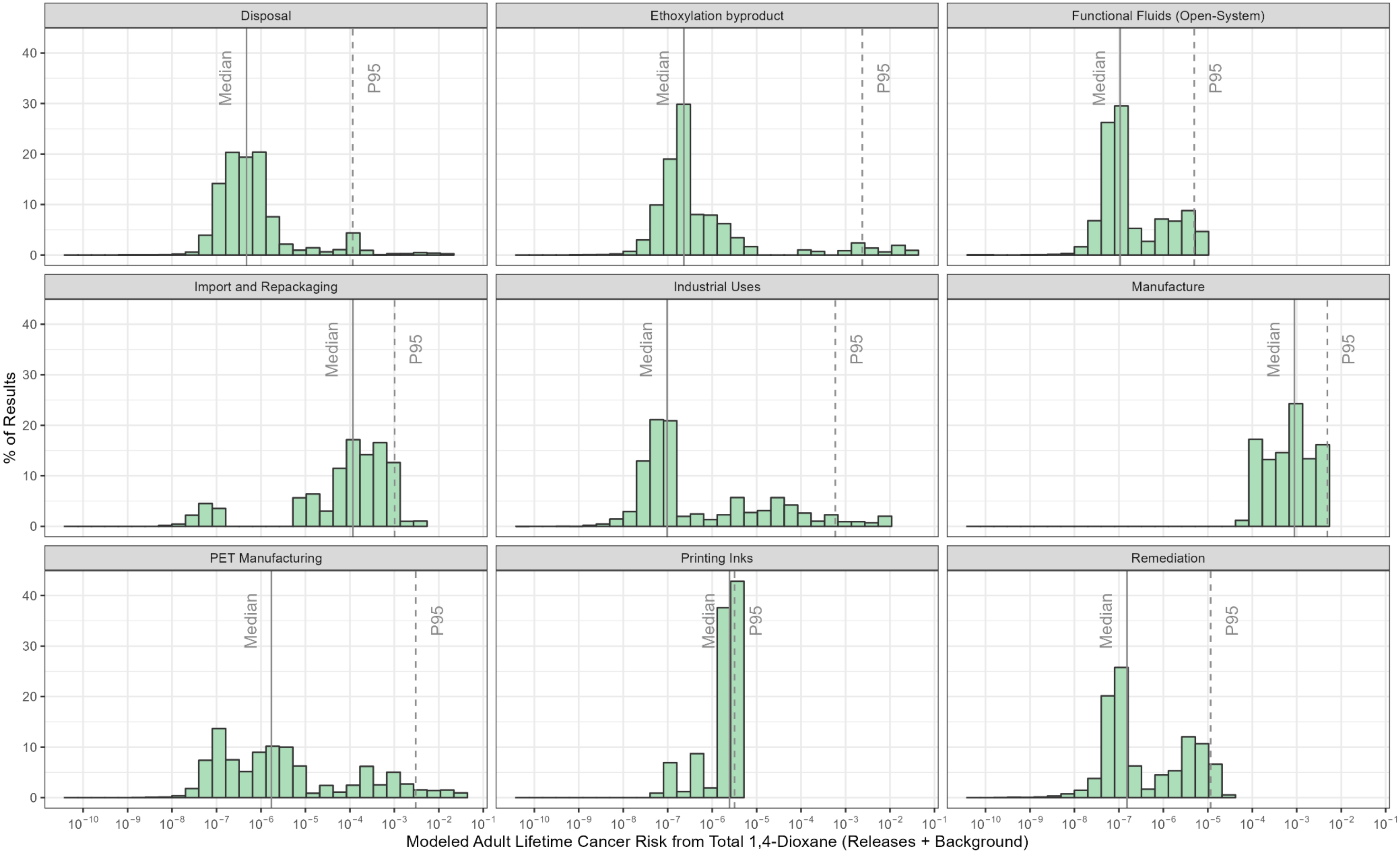

Figure 5-5. Histograms of Lifetime Cancer Risk Estimates for Aggregate Water Concentrations Estimated Downstream of COUs with Vertical Lines Showing the Median and 95th Percentile (P95) Values

5.2.2.1.6 Integrated Summary of Drinking Water Risk Estimates across Multiple Lines of Evidence for Surface Water

Risks from drinking water exposures were evaluated using a series of analyses that provided information about the specific contributions of releases associated with individual sources as well as aggregate exposures and risks to the general population. This analysis finds cancer risk estimates greater than 1 in 1 million from drinking water exposures informed by both monitoring data and modeled surface water concentrations. Modeled concentrations result in cancer risk estimates greater than 1 in 1 million across a range of individual sources and aggregate sources utilizing plausible drinking water exposure scenarios.

Monitoring data demonstrates that 1,4-dioxane is present in some source water and finished drinking water samples. Measured concentrations in finished drinking water samples resulted in cancer risk estimates greater than 1 in 1 million at the high end of the distribution of monitoring samples. Most drinking water treatment systems are not expected to remove 1,4-dioxane from water, suggesting that concentrations detected in source water can also be an indication of concentrations in drinking water.

Available monitoring data provided information about general population exposures but did not capture high concentrations occuring in specific locations or at specific times from direct and indirect releases into water bodies. Therefore, EPA relied on estimated concentrations modeled for a range of specific release scenarios, including direct and indirect industrial releases, DTD releases, disposal of hydraulic fracturing waste, and aggregate concentrations resulting from varying combinations of multiple sources to characterize risks from the water pathway. EPA evaluated the performance of the models used to estimate water concentrations with monitoring data from site-specific locations serving as cases studies. These case study comparisons demonstrated strong concordance between modeled concentrations and monitoring data, thereby increasing confidence in risk estimates based on modeled concentrations.

Across all modeled scenarios, 1,4-dioxane concentrations in water are primarily determined by the amount of release from varying sources and the flow of the receiving water body. These two factors are highly location and source-specific, resulting in very wide ranges of modeled water concentrations and risk estimates for each set of analyses presented in the previous section above.

Risk estimates based on 1,4-dioxane concentrations modeled in the receiving water bodies at the point of release show potential for risk greater than 1 in 1 million or 1 in 100,000 from each of the sources assessed.

As described in Section 5.2.2.1.2, dilution that occurs between the point of release and drinking water intake locations may be expected to reduce 1,4-dioxane concentrations in some locations. However, even when accounting for dilution, upstream releases contribute to cancer risk estimates greater than 1 in a million or 1 in 100,000 at some drinking water intake locations. EPA evaluated risks based on modeled water concentrations for a sample of drinking water intake locations downstream of releases where risk was greater than 1 in 1 million. After accounting for additional dilution, cancer risk estimates remained greater than 1 in 1 million for 27 percent of the public water systems evaluated, serving a combined population of over 2 million people.

The potential relative contribution from different sources varies under different conditions and is likely to be site-specific. For example, high-end risk estimates in the aggregate model (presented in Section 5.2.2.1.5) are very similar to high end risk estimates for facility releases alone (presented in Section 5.2.2.1.2), suggesting that in cases where industrial releases are high, those releases will be the dominant source of 1,4-dioxane in water. On the other hand, under some conditions (presented in Section 5.2.2.1.3), plausible DTD release scenarios may present risk greater than 1 in 1 million in the absence of industrial releases. Taken together, the analyses presented throughout this section demonstrate that each of these sources may contribute to drinking water risks under some conditions. These analyses define the conditions under which different levels of risk may occur.

5.2.2.2 Drinking Water – Groundwater and Disposal Pathways

EPA estimated risks from general population exposures that could occur if groundwater containing 1,4-dioxane is used as a source of drinking water. Risk estimates presented in this section are not tied to known releases at specific locations. Rather, this analysis defines the conditions under which 1,4-dioxane disposal to landfills or from hydraulic fracturing operations could result in varying levels of risk.

Cancer and non-cancer risk estimates were calculated based on modeled groundwater concentrations described in Section 2.3.1.4 and corresponding drinking water exposures estimates described in Section 3.2.2.2. All risk estimates presented in this summary focus on the scenario with the greatest potential exposure and risk. Because adult drinking water exposures relative to body weight are greater than exposures relative to body weight that occur over the course of childhood (as illustrated in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Release to Surface Water from Individual Facilities* (U.S. EPA, 2023h)), the scenario with the greatest lifetime cancer risk is 33 years of exposure as an adult.

For potential groundwater concentrations resulting from landfill leachate, EPA estimated cancer and non-cancer risks for adults and formula-fed infants at concentrations estimated under varying hypothetical combinations of leachate concentrations and loading rates. As shown in Table 5-6, lifetime cancer risk estimates increase under scenarios with higher leachate concentrations and loading rates. Chronic non-cancer risk estimates (not shown) indicate risk relative to the benchmark MOE only at the highest leachate concentrations and loading rates. These concentrations and loading rates represent a scenario where 1,4-dioxane is either delisted and released to a municipal solid waste landfill or when trace concentrations present in consumer and commercial products are disposed to those same landfills. Though the higher concentrations of 1,4-dioxane in leachate and higher loading rates are less likely, they may represent a sentinel PESS exposure. Complete results for cancer and non-cancer risk are available in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Land Releases to Landfills* (U.S. EPA, 2023f).

Overall confidence in risk estimates for drinking water exposures resulting from disposal to landfills is low to medium. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.2.1 the modeling methodology is robust. However, the release information relied on as model input data is supported by slight to moderate evidence, decreasing overall confidence. In addition, this drinking water exposure scenario relies on the assumption that the estimated groundwater concentrations may occur in locations where groundwater is used as a primary drinking water source. While the substantial uncertainty around the extent to which these exposures occur decreases overall confidence in the exposure scenario, this scenario represents a sentinel exposure.

Table 5-6. Lifetime Cancer Risks[a] Estimated for Modeled Groundwater Concentrations Estimated under Varying Landfill Conditions

Leachate Concentration (mg/L)	Loading Rate (lb)							
	0.1	1	10	100	1,000	10,000	100,000	1,000,000
0.0001	3.1E−17	2.9E−16	3.6E−15	3.4E−14	3.3E−13	3.1E−12	3.0E−11	2.9E−10
0.001	3.1E−16	2.9E−15	3.6E−14	3.4E−13	3.3E−12	3.1E−11	3.0E−10	2.9E−09
0.01	3.1E−15	2.9E−14	3.6E−13	3.4E−12	3.3E−11	3.1E−10	3.0E−09	2.9E−08
0.1	3.1E−14	2.9E−13	3.6E−12	3.4E−11	3.3E−10	3.1E−09	3.0E−08	2.9E−07
1	3.1E−13	2.9E−12	3.6E−11	3.4E−10	3.3E−09	3.1E−08	3.0E−07	2.9E−06
10	3.1E−12	2.9E−11	3.6E−10	3.4E−09	3.3E−08	3.1E−07	3.0E−06	2.9E−05
100	3.1E−11	2.9E−10	3.6E−09	3.4E−08	3.3E−07	3.1E−06	3.0E−05	2.9E−04
1,000	3.1E−10	2.9E−09	3.6E−08	3.4E−07	3.3E−06	3.1E−05	3.0E−04	2.9E−03
10,000	3.1E−09	2.9E−08	3.6E−07	3.4E−06	3.3E−05	3.1E−04	3.0E−03	2.9E−02

[a] Lifetime cancer risks based on 33 years of oral exposure through drinking water as an adult.

For groundwater concentrations resulting from disposal of hydraulic fracturing produced water, EPA calculated cancer and non-cancer risks for adults and formula-fed infants. Cancer risk estimates across the full distribution of modeled releases are presented in Table 5-7. Cancer risk estimates are 4.0E−07 for median modeled releases and 8.6×10^{-6} for 95th percentile modeled releases. Chronic non-cancer risk estimates are above the corresponding benchmark MOE for all modeled groundwater concentrations, indicating lower non-cancer risk from non-cancer effects. Complete cancer and noncancer risk calculations are available in *1,4-Dioxane Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Land Releases to Surface Impoundments* (U.S. EPA, 2023g).

Overall confidence in risk estimates for drinking waters resulting from disposal of hydraulic fracturing waste is low to medium. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.2.2, the modeling methodology is robust and the release information relied on as model input data is supported by moderate evidence. However, no monitoring data are available to confirm detection of 1,4-dioxane in groundwater near hydraulic fracturing operations. This drinking water exposure scenario relies on the assumption that the estimated groundwater concentrations may occur in locations where groundwater is used as a primary drinking water source. While the substantial uncertainty around the extent to which these exposures occur decreases overall confidence in the exposure scenario, this scenario represents a sentinel exposure.

Table 5-7. Lifetime Cancer Risks[a] Estimated for Modeled Groundwater Concentrations Resulting from Disposal of Hydraulic Fracturing Produced Water

Monte Carlo Distribution	Modeled Groundwater Concentration (mg/L)	Adult LADD (mg/kg/day)	Adult Cancer Risk Estimate
Max	1.9E−02	8.8E−05	1.1E−05
99th	1.5E−02	7.1E−05	8.6E−06
95th	1.5E−02	7.1E−05	8.6E−06
Mean	7.1E−04	3.3E−06	4.0E−07
50th	1.2E−04	5.6E−07	6.8E−08
5th	1.2E−04	5.6E−07	6.8E−08
Min	4.4E−07	2.1E−09	2.5E−10

[a] Lifetime cancer risks based on 33 years of oral exposure through drinking water as an adult.

5.2.2.3 Air Pathway

EPA estimated risks from general population exposures to 1,4-dioxane released to air, with a focus on exposures in fenceline communities. Risks were evaluated for air releases from industrial COUs, hydraulic fracturing operations, and industrial and institutional laundry facilities based on exposure estimates in Section 3.2.3.

5.2.2.3.1 Industrial COUs Reported to TRI

EPA estimated risks from general population exposures that could occur in communities neighboring industrial releases associated with stack and fugitive emissions. Cancer and non-cancer risk estimates for general population exposures within 10,000 m of industrial releases were calculated for the 10th, 50th, and 95th percentiles of modeled exposure concentrations estimated in Section 3.2.3.1. Table 5-8 summarizes the cancer risk estimates for 95th percentile exposure concentrations within 1,000 m of the facilities with the greatest risk in each OES/COU, ranging from 1.05×10^{-10} to 1.1×10^{-4}. Cancer risk estimates based on 50th percentile modeled exposure concentrations within 1,000 m of the highest risk facilities range from 2.5×10^{-11} to 8.3×10^{-5} (data not shown). Risk estimates were generally highest within 1,000 m of the facilities and lower at greater distances. As discussed in Section 2.3.3.3, exposure estimates very near facilities (5 to 10 m) may be impacted by assumptions made for modeling around an area source (10×10 area source places people at 5 m on top of the release point). This, in combination with other factors like meteorological data, release heights, and plume characteristics can result in lower or higher exposures at 5 m than just off the release point at 10 m. Air concentrations from fugitive emissions tend to peak within 10 m of release sites while contributions from stack releases generally peak around 100 m, meaning that risks nearest to release sites are often driven by fugitive releases. Acute and chronic non-cancer risk estimates (not shown) do not indicate risk relative to benchmark MOES for any of the estimated exposure concentrations at any facilities evaluated. Complete cancer and non-cancer risk results for air concentrations modeled from stack, fugitive and combined air emissions are provided in *1,4-Dioxane Supplemental Information File: Air Exposures and Risk Estimates for Single Year Analysis* (U.S. EPA, 2023e).

Air exposure and risk estimates are dependent on release amounts, stack heights, contributions from stack releases and fugitive emissions, topography, and meteorological conditions. These factors vary substantially across facilities within each OES/COU, making release amount, stack height, and meteorological conditions more important predictors of risk than a facility's identified OES/COU.

Overall confidence in site-specific risk estimates for inhalation exposure resulting from industrial releases varies across OES/COUs. As described in Section 4.3, overall confidence in the cancer inhalation unit risk underlying these risk estimates is high. As described in Section 3.3.3.1, the AERMOD modeling methodology used for this analysis is robust and accounts for both stack and fugitive emissions. The exposure scenarios considered are most relevant to long-term residents in fenceline communities. There is some uncertainty around the extent to which people actually live and work around the specific facilities where risks are highest, decreasing overall confidence in the exposure scenario, particularly at distances nearest release sites. Overall confidence varies due to variable levels of confidence in underlying release information used to estimate exposures. An OES-specific discussion of the confidence in sources of release information is presented in Appendix E.5E.5.4, but in general terms:

- Overall confidence in risk estimates is medium to high for OESs/COUs that rely primarily on release data reported to TRI via Form R.
- Overall confidence in risk estimates is medium for OESs/COUs for which release estimates are based on data reported to TRI via Form A.
- Overall confidence in risk estimates is low to medium for OESs/COUs for which release estimates are based on surrogate or modeled information.

Table 5-8. Inhalation Lifetime Cancer Risks[a] within 10 km of Industrial Air Releases Based on 95th Percentile Modeled Exposure Concentrations

OES	Corresponding COUs			# Facilities		Distance from Facility with Greatest Risk (m)[b]						Overall Confidence
	Life Cycle Stage	Category	Subcategory	Total	Risk >1E−06	5	10	30	60	100	100–1,000	
Disposal	Disposal	Disposal	Hazardous waste incinerator Off-site waste transfer Underground injection Hazardous landfill	15	5	2.88E−05	3.42E−05	1.22E−05	4.67E−06	2.13E−06	2.00E−07	Medium to High
Dry Film Lubricant	Industrial use, commercial use	Other uses	Dry film lubricant	8	0	1.09E−12	4.83E−11	3.46E−09	2.62E−08	4.26E−08	6.72E−09	Low to Medium
Ethoxylation Byproduct	Processing	Byproduct	Byproduct produced during the ethoxylation process to make ethoxylated ingredients for personal care products	6	3	4.42E−05	9.21E−05	4.96E−05	2.09E−05	1.11E−05	2.58E−06	Medium to High
Film Cement	Industrial use, commercial use	Adhesives and sealants	Film cement	1	0	8.46E−07	8.86E−07	2.99E−07	1.54E−07	8.46E−08	1.55E−08	Low to Medium
Functional Fluids (Open-System)	Industrial use	Functional fluids (open and closed systems)	Polyalkylene glycol lubricant Synthetic metalworking fluid Cutting and tapping fluid	2	0	8.67E−08	1.60E−07	6.98E−08	7.31E−08	1.23E−07	5.02E−08	Medium to High
Import and Repackaging	Manufacturing	Import	Import Repackaging	1	0	1.82E−13	3.78E−12	3.74E−10	2.82E−09	5.89E−09	2.18E−09	Medium to High

OES	Corresponding COUs			# Facilities		Distance from Facility with Greatest Risk (m)^{b}						Overall Confidence
	Life Cycle Stage	Category	Subcategory	Total	Risk >1E−06	5	10	30	60	100	100–1,000	
Industrial Uses	Processing	Processing as a reactant	Polymerization Catalyst	12	6	2.84E−05	3.24E−05	1.04E−05	3.84E−06	1.89E−06	4.85E−07	Medium to High
	Processing	Non-incorporative	Basic organic chemical manufacturing (process solvent)									
	Industrial use	Intermediate use	Plasticizer intermediate Catalysts and reagents for anhydrous acid reactions, brominations, and sulfonations									
Laboratory Chemical Use	Industrial use, commercial use	Laboratory chemicals	Chemical reagent Reference material Spectroscopic and photometric measurement Liquid scintillation counting medium Stable reaction medium Cryoscopic solvent for molecular mass determinations Preparation of histological sections for microscopic examination	1	1	1.40E−05	1.46E−05	4.91E−06	2.54E−06	1.40E−06	2.54E−07	Low to Medium
Manufacturing	Manufacturing	Domestic manufacture	Domestic manufacture	1	1	5.91E−05	1.10E−04	5.20E−05	2.18E−05	1.08E−05	9.62E−07	Medium to High

OES	Corresponding COUs			# Facilities		Distance from Facility with Greatest Risk (m)[b]						Overall Confidence
	Life Cycle Stage	**Category**	**Subcategory**	**Total**	**Risk >1E−06**	**5**	**10**	**30**	**60**	**100**	**100–1,000**	
PET Manufacturing	Processing	Byproduct	Byproduct produced during the production of polyethlene terephtalate	13	10	5.42E−05	6.48E−05	2.37E−05	9.47E−06	4.35E−06	7.25E−07	Medium to High
Spray Foam Application	Industrial use, commercial use	Other uses	Spray polyurethane foam	1	0	5.28E−09	5.68E−09	1.94E−09	1.02E−09	5.79E−10	1.05E−10	Low to Medium

[a] Lifetime cancer risks based on 33 years of continuous inhalation exposure averaged over a 78-year lifetime.
[b] Cancer risks were also calculated at 2,500, 5,000 and 10,000 m from all facilities.

Land Use Analysis

For locations where lifetime cancer risk is greater than 1×10^{-6}, EPA evaluated land use patterns to determine whether fenceline community exposures may be reasonably anticipated. Detailed results of this analysis are described in Appendix J.3 and are consistent with the methods described in the draft 2022 Fenceline Report. In short, EPA determined whether residential, industrial/commercial businesses, or other public spaces are present within those radial distances where cancer risk estimates for 95th percentile modeled air concentrations are greater than 1×10^{-6} for each facility. In all cases, risks greater than 1×10^{-6} were within 1,000 m or less of releasing facilities. This analysis was limited to facilities that could be mapped to a GIS location. Based on this characterization of land use patterns, fenceline community exposures have the potential to occur at 50 percent of facilities (11 of 22 GIS-mapped facilities) where cancer risk is greater than 1×10^{-6} based on modeled fenceline air concentrations.

Aggregate Risk

EPA also evaluated potential risks from aggregate exposures from multiple neighboring facilities using a conservative screening methodology. EPA identified five groups of two to four facilities reporting 1,4-dioxane releases in proximity to each other, *i.e.*, within 10 km. Aggregating risks estimated for these groups of facilities were generally dominated by the facility with the greatest risk. This aggregate analysis did not identify locations with cancer risk greater than 1×10^{-6} that did not already have cancer risk above that level from an individual facility. Details of the methods and results of this aggregate analysis are described in Appendix J.4.

5.2.2.3.2 Hydraulic Fracturing

Cancer and non-cancer risk estimates for potential general population exposures within 1,000 m of hydraulic fracturing operations were calculated for a range of air concentrations modeled across the distribution of release estimates, as described in Section 3.3.3.2. Table 5-9 presents lifetime cancer risk estimates for exposure to high-end and central tendency air concentrations modeled for both high end (95th percentile) and central tendency (50th percentile) modeled releases for a range of topographical and meteorological scenarios. Lifetime cancer risk estimates for distances within 1,000 m of hydraulic fracturing operations range from 3.9×10^{-7} to 7.1×10^{-5} for high end release estimates and 2.2×10^{-8} to 4.1×10^{-6} for central tendency release estimates across a range of model scenarios. Acute and chronic non-cancer risk estimates (not shown) do not indicate risk relative to benchmark MOEs for any exposure concentrations estimated for hydraulic fracturing operations. Complete results are provided in *1,4-Dioxane Supplemental Information File: Air Exposure and Risk Estimates for 1,4-Dioxane Emissions from Hydraulic Fracturing Operations* (U.S. EPA, 2023b).

Overall confidence in risk estimates for inhalation exposures resulting for air concentrations modeled based on releases from hydraulic fracturing operations is medium. As described in Section 4.3, overall confidence in the cancer inhalation unit risk underlying these risk estimates is high. As described in Section 3.3.3.2 the modeling methodologies used to estimate air concentrations are robust. The distribution of air releases used as model input data were estimated using Monte Carlo modeling and rely on assumptions. No air monitoring data were available to confirm detection of 1,4-dioxane is air near hydraulic fracturing operations. Because the air concentrations underlying this analysis are based on releases estimated using probabilistic modeling, they are not tied to specific locations that can be evaluated for land use patterns. There is therefore substantial uncertainty around the extent to which people actually live and work around the specific locations where risks are highest, decreasing overall confidence in the exposure scenario.

Table 5-9. Lifetime Cancer Risk Estimates for Fugitive Emissions from Hydraulic Fracturing[a] [b]

Fugitive Emissions Release Scenario	Cancer Risk Estimates for 95th Percentile Modeled Releases						Cancer Risk Estimates for 50th Percentile Modeled Releases					
	High-End Modeled Air Concentrations			Central Tendency Modeled Air Concentrations			High-End Modeled Air Concentrations			Central Tendency Modeled Air Concentrations		
	100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m
South (Coastal)-Rural-24	7.1E−05	3.2E−06	8.3E−06	5.6E−05	2.4E−06	6.4E−06	4.1E−06	1.8E−07	4.7E−07	3.2E−06	1.4E−07	3.6E−07
West North Central-Rural-24	5.4E−05	3.1E−06	7.3E−06	4.1E−05	1.9E−06	4.9E−06	3.1E−06	1.8E−07	4.2E−07	2.4E−06	1.1E−07	2.8E−07
South (Coastal)-Urban-24	3.4E−05	7.7E−07	2.4E−06	3.0E−05	6.7E−07	2.1E−06	2.0E−06	4.4E−08	1.4E−07	1.7E−06	3.8E−08	1.2E−07
West North Central-Urban-24	3.2E−05	8.3E−07	2.5E−06	2.6E−05	6.1E−07	1.9E−06	1.8E−06	4.8E−08	1.4E−07	1.5E−06	3.5E−08	1.1E−07
South (Coastal)-Rural-8	1.3E−05	1.2E−07	5.1E−07	1.1E−05	9.0E−08	4.2E−07	7.3E−07	6.8E−09	2.9E−08	6.4E−07	5.2E−09	2.4E−08
West North Central-Rural-8	2.7E−05	1.0E−06	2.5E−06	1.4E−05	2.9E−07	8.8E−07	1.5E−06	5.9E−08	1.4E−07	8.2E−07	1.6E−08	5.0E−08
South (Coastal)-Urban-8	1.2E−05	9.0E−08	4.3E−07	1.1E−05	8.0E−08	3.9E−07	6.7E−07	5.2E−09	2.5E−08	6.1E−07	4.6E−09	2.2E−08
West North Central-Urban-8	1.9E−05	3.9E−07	1.2E−06	1.2E−05	1.6E−07	6.0E−07	1.1E−06	2.2E−08	7.1E−08	7.0E−07	9.1E−09	3.4E−08

[a] Lifetime cancer risks based on 33 years of continuous inhalation exposure averaged over a 78-year lifetime.
[b] Cancer risk estimates shown here are based on modeled releases and air concentrations estimated for 72 days of release.

5.2.2.3.3 Industrial and Institutional Laundry Facilities

Cancer and non-cancer risk estimates for potential general population exposures within 1,000 m of industrial and institutional laundry facilities were calculated for a range of air concentrations modeled for a range of releases, as described in Section 3.2.3.3. Table 5-10 presents lifetime cancer risk estimates for exposures estimated from both high-end and central tendency air concentrations modeled based on the maximum release scenario for each type of laundry under the most conservative exposure scenario evaluated (rural south coastal topography, assuming 24 hours of releases each day). Lifetime cancer risk estimates for distances within 1,000 m of laundry facilities range from 1.5×10^{-11} to 3.8×10^{-8} across a range of high end and central tendency exposure scenarios. Acute and chronic non-cancer risk estimates (not shown) do not indicate risk for any estimated exposure concentrations for laundries relative to the benchmark MOEs. Complete results are provided in *1,4-Dioxane Supplemental Information File: Air Exposures and Risk Estimates for Industrial Laundry* (U.S. EPA, 2023c).

Overall confidence in risk estimates from inhalation exposures resulting from industrial and institutional laundries is medium. As described in Section 4.3, overall confidence in the cancer inhalation unit risk underlying these risk estimates is high. As described in Section 3.3.3.2, the modeling methodologies are robust. The distribution of air releases used as model input data were estimated using Monte Carlo modeling and rely on assumptions. No air monitoring data were available to determine whether 1,4-dioxane is detected near industrial and institutional laundry facilities. Because the air concentrations underlying this analysis are based on probabilistic modeling, they are not tied to specific locations that can be evaluated for land use patterns. There is therefore substantial uncertainty around the extent to which people actually live and work around the specific locations where risks are highest, decreasing overall confidence in the exposure scenario.

Table 5-10. Lifetime Cancer Risk Estimates for Fugitive Emissions from Industrial and Institutional Laundry Facilities[a]

Facility Type	Detergent and Emissions Type	Cancer Risk Estimates for Maximum Modeled Releases					
		High-End Modeled Air Concentrations			Central Tendency Modeled Air Concentrations		
		100 m	1,000 m	100 to 1,000 m	100 m	1,000 m	100 to 1,000 m
Industrial Laundry	Liquid – vapor	3.7E−08	1.7E−09	4.3E−09	3.3E−08	1.4E−09	3.8E−09
	Powder – vapor	3.6E−08	1.7E−09	4.2E−09	3.3E−08	1.4E−09	3.8E−09
	Powder – PM10	3.8E−08	8.8E−10	3.2E−09	3.4E−08	7.9E−10	2.9E−09
	Powder – PM2.5	3.6E−08	1.6E−09	4.1E−09	3.3E−08	1.4E−09	3.7E−09
Institutional Laundry	Liquid – vapor	2.3E−08	1.1E−09	2.7E−09	2.1E−08	9.0E−10	2.4E−09
	Powder – vapor	6.8E−10	3.2E−11	7.9E−11	6.2E−10	2.7E−11	7.0E−11
	Powder – PM10	7.1E−10	1.6E−11	5.9E−11	6.4E−10	1.5E−11	5.3E−11
	Powder – PM2.5	6.8E−10	3.0E−11	7.7E−11	6.2E−10	2.6E−11	6.9E−11

[a] Lifetime cancer risks based on 33 years of continuous inhalation exposure averaged over a 78-year lifetime.

5.2.2.4 Potentially Exposed or Susceptible Subpopulations

EPA considered PESS throughout the exposure assessment presented in this draft supplement and throughout the hazard identification and dose-response analysis described in the 2020 RE. Table 5-11 summarizes how PESS were incorporated into the draft supplement through consideration of increased exposures and/or increased biological susceptibility. The table also summarizes the remaining sources of uncertainty related to consideration of PESS.

Table 5-11. Summary of PESS Considerations Incorporated throughout the Analysis and Remaining Sources of Uncertainty

PESS Categories	Potential Exposures Identified in Specific Subpopulations and Incorporated into Exposure Assessment	Potential Sources of Biological Susceptibility Identified and Incorporated into Hazard Assessment
Lifestage	General population drinking water exposure scenarios include lifestage-specific exposure factors for adults, children, and formula-fed infants (Section 5.2.2.1); Inhalation exposures are based on air concentrations and are therefore consistent across lifestages (Section 5.2.2.3). Based on pchem properties and a lack of studies evaluating potential for accumulation in milk, EPA did not quantitatively evaluate the milk pathway and this is a source of uncertainty.	EPA qualitatively described the potential for biological susceptibility due to lifestage differences and developmental toxicity but did not identify quantitative evidence of lifestage-specific susceptibilities to 1,4-dioxane; A 10× UF was applied for human variability. The magnitude of potential lifestage differences in metabolism and toxicity are not well quantified and are a remaining source of uncertainty.
Pre-existing Disease	EPA did not identify health conditions that may influence exposure. The potential for pre-existing disease to influence exposure (due to altered metabolism, behaviors, or treatments related to the condition) is a source of uncertainty.	EPA qualitatively described the potential for pre-existing health conditions, such as liver disease, to increase susceptibility or alter toxicokinetics, but did not identify direct quantitative evidence. A 10× UF was applied for human variability. The potential impact of pre-existing diseases on susceptibility to 1,4-dioxane is a remaining source of uncertainty.
Lifestyle Activities	EPA did not identify specific lifestyle activities that expected to increase 1,4-dioxane exposure. This is a remaining source of uncertainty.	EPA did not identify lifestyle factors that influence biological susceptibility to 1,4-dioxane. This is a remaining source of uncertainty.
Occupational Exposures	EPA evaluated a range of occupational exposure scenarios in manufacturing, hydraulic fracturing and use of commercial products that increase exposure to 1,4-dioxane present as a byproduct. EPA evaluated risks for high-end exposure estimates for each of these scenarios (Section 5.2.1).	EPA did not identify occupational factors that increase biological susceptibility to 1,4-dioxane. This is a remaining source of uncertainty.
Geographic Factors	EPA evaluated risks to fenceline communities from 1,4-dioxane in ambient air (Section 5.2.2.3) and in drinking water downstream of release sites (Section 5.2.2.1). EPA mapped tribal lands in relation to air, surface water and ground water releases of 1,4-dioxane to identify potential for	EPA did not identify geographic factors that increase biological susceptibility to 1,4-dioxane. This is a remaining source of uncertainty.

PESS Categories	Potential Exposures Identified in Specific Subpopulations and Incorporated into Exposure Assessment	Potential Sources of Biological Susceptibility Identified and Incorporated into Hazard Assessment
	increased exposures for tribes due to geographic proximity (Section 2.3).	
Socio-demographic Factors	EPA did not identify specific sociodemographic factors that influence exposure to 1,4-dioxane. This is a remaining source of uncertainty.	EPA did not identify sociodemographic factors that influence biological susceptibility to 1,4-dioxane. This is a remaining source of uncertainty.
Nutrition	EPA did not identify nutritional factors influencing exposure to 1,4-dioxane. This is a remaining source of uncertainty.	EPA did not identify nutritional factors that influence biological susceptibility to 1,4-dioxane. This is a remaining source of uncertainty.
Genetics	EPA did not identify genetic factors influencing exposure to 1,4-dioxane. This is a remaining source of uncertainty.	Indirect evidence that genetic variants may increase susceptibility of the target organ was addressed through a 10× UF for human variability. The magnitude of the impact of genetic variants is unknown and is a source of uncertainty.
Unique Activities	Some tribes may have increased exposure to drinking water due to tribal activities such as sweat lodges. EPA has identified upper bound drinking water estimates of 2 to 4 L/day associated with tribal lifeways for some tribes (Harper, 2017; Harper and Ranco, 2009; Harper et al., 2007; Harper et al., 2002). Risk calculations in this draft supplement assume an acute adult drinking water intake of 3.2 L/day and a chronic drinking water intake of 0.88 L/day. Other potential sources of increased exposure to 1,4-dioxane due to specific tribal lifeways or other unique activity patterns are a source of uncertainty.	EPA did not identify unique activities that influence susceptibility to 1,4-dioxane. This is a remaining source of uncertainty.
Aggregate Exposures	EPA evaluated risk from aggregate sources of exposure contributing to 1,4-dioxane in water (Section 5.2.2.1) or from multiple sources in proximity releasing to air (Section 5.2.2.3, Appendix J.4). Risks from aggregate exposures across routes or pathways were evaluated qualitatively and are a remaining source of uncertainty.	EPA does not identify ways that aggregate exposures would influence susceptibility to 1,4-dioxane. This is a remaining source of uncertainty.
Other Chemical and Non-chemical Stressors	EPA did not identify chemical and nonchemical factors influencing exposure to 1,4-dioxane. This is a remaining source of uncertainty.	EPA did not identify chemical or nonchemical factors that influence susceptibility to 1,4-dioxane. There is insufficient data to quantitatively address potential increased susceptibility due to chemical or nonchemical stressors and this is a remaining source of uncertainty.

5.2.2.5 Aggregate and Sentinel Exposures

In this draft supplement, EPA considers the combined 1,4-dioxane exposure an individual may experience due to releases from multiple facilities in proximity releasing to air or multiple releases contributing to drinking water concentrations in a particular location. For general population drinking water exposure scenarios, EPA evaluated combined exposure and risks from multiple sources of 1,4-dioxane in surface water, including direct and indirect industrial releases, DTD releases, and upstream background contamination (Section 5.2.2.1). For general population air exposure scenarios, EPA evaluated combined exposure and risk across multiple facilities in proximity releasing to air (Section 5.2.2.3 and Appendix J.4).

EPA considered aggregating cancer risks across inhalation, oral, and/or dermal routes of exposure. There is uncertainty around the extent to which cancer risks across routes are additive for 1,4-dioxane. Liver tumors are the primary site of cancer risk from oral exposures. Inhalation exposure in rats is associated with multiple tumor types, including liver. The IUR used to calculate inhalation cancer risk reflects combined risks from multiple tumor types. While EPA concluded that nasal cavity lesions are likely to be primarily the result of systematic delivery (as discussed on p.192 of the 2020 RE), there is uncertainty around the degree to which those effects could be partially due to portal of entry effects following inhalation exposure. It is therefore unclear the extent to which it is appropriate to quantitatively aggregate cancer risks based on the IUR with liver tumor risks associated with oral or dermal exposures. EPA considers the potential aggregate cancer risk across routes to be a source of uncertainty for 1,4-dioxane cancer risk estimates.

EPA also considered aggregating cancer risks across dermal and oral exposures. The dermal cancer slope factor is derived from the oral cancer slope factor by route-to-route extrapolation. Because the systemic effect is assumed to be the same for both routes, EPA determined that it could be biologically appropriate to aggregate risk from dermal and oral exposures. General population scenarios included inhalation and oral not dermal exposures and occupational and consumer exposure scenarios included inhalation and dermal not oral exposures. However, this draft supplement does not include COUs or pathways in which both oral and dermal exposure routes are considered.

EPA also considered potential for aggregate exposures across groups. For example, there may be some individuals who are exposed at work as well as through general population air and drinking water pathways or through consumer product use. This as a source of uncertainty. These types of aggregate risks were not quantified and risks for individual exposure scenarios should be interpreted with an appreciation for potential aggregate exposures and risks.

Section 2605(b)(4)(F)(ii) of TSCA requires EPA, as a part of the risk evaluation, to describe whether aggregate or sentinel exposures under the conditions of use were considered and the basis for their consideration. EPA defines sentinel exposure as "the exposure to a single chemical substance that represents the plausible upper bound of exposure relative to all other exposures within a broad category of similar or related exposures (40 CFR § 702.33)." In this draft supplement, EPA considered sentinel exposures by considering risks to populations who may have upper bound exposures, including workers and ONUs who perform activities with higher exposure potential and fenceline communities. EPA characterized high-end exposures in evaluating exposure using both monitoring data and modeling approaches. Where statistical data are available, EPA typically uses the 95th percentile value of the available dataset to characterize high-end exposure for a given COU.

5.2.2.6 Summary of Overall Confidence and Remaining Uncertainties in Human Health Risk Characterization

The overall level of confidence in each set of risk estimates depends on the level of confidence in the underlying hazard values summarized in Section 4.3 and the level of confidence in exposure estimates described in more detail in Section 3.3.

For all risk estimates, EPA has medium to high confidence in the underlying hazard PODs used as the basis for this risk characterization. Sources of confidence in each of the hazard values were described in the 2020 RE and are summarized in Section 4.3. Cancer risk is the primary risk driver for each of the scenarios evaluated in this draft supplement and is therefore the basis of overall confidence levels described herein. There is remaining uncertainty for all risk estimates around the potential impact of 1,4-dioxane on potentially exposed or susceptible subpopulations (as discussed in Section 5.2.2.4). EPA applied an intraspecies uncertainty factor of 10 to all non-cancer PODs to account for variation across gender, age, health status, or genetic makeup, and other factors that may increase susceptibility, but the actual magnitude of the impact of these factors on susceptibility is unknown. Similarly, EPA evaluated risks from exposure scenarios that are intended to reflect the most highly exposed populations (including fenceline communities and highly exposed workers), but the potential for other highly exposed populations that were not identified in this analysis is a source of uncertainty. Potential for aggregate risks across routes or pathways that are not quantitatively evaluated in this assessment is another source of uncertainty.

5.2.2.6.1 Risks from Occupational Exposures

Overall confidence in risk estimates for occupational inhalation exposures ranges from low to high, depending on the confidence in exposure assessment for each OES/COU. As described in Section 4.3, overall confidence in the cancer inhalation unit risk underlying these risk estimates is high. As described in Section 3.3.1.1, the measured and monitored inhalation exposure data are supported by moderate to robust evidence. Additionally, the exposure modeling methodologies and underlying model input data is supported by moderate to robust evidence. However, there is uncertainty in the representativeness of the assessed exposure scenarios towards all potential exposures for the given OES/COU, limitations in the amount and age of monitoring data, and limitations in the modeling approaches towards 1,4-dioxane-specific use within the OES/COU. Therefore, while the underlying data and methods used to estimate occupational inhalation risk is supported by moderate to robust evidence, the overall confidence of these estimates ranges from low to high depending on the OES/COU. OES/COU-specific discussions of the available inhalation exposure data and overall confidence are presented in Appendix F.6 and summarized in Table 3-2.

Overall confidence in risk estimates for occupational dermal exposures is medium for all OES/COUs because the same modeling approach was used for all OES/COUs. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.1.2, the dermal exposure modeling methodology is supported by moderate evidence, with model input parameters from literature sources, a European model, standard defaults from the ChemSTEER User Guide (U.S. EPA, 2015a), and 1,4-dioxane product concentration data from process information. These sources range from slight to robust, depending on factors such as age and applicability to OES/COU. The modeling is limited by the use of standard input parameters that are not specific to 1,4-dioxane and a lack of variability in dermal exposure for different worker activities. Therefore, EPA's overall confidence in the occupational dermal risk estimates is medium

5.2.2.6.2 Risks from General Population Exposures through Drinking Water

Overall confidence in the overall distribution of risk estimates for drinking water exposures resulting from facility releases is medium-high. Overall confidence in site-specific risk estimates for individual facility releases varies both within and across OES, depending on the confidence in the source-specific release data. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.1, the exposure modeling methodology used for this analysis is supported by moderate evidence. It is designed to estimate water concentrations expected at specific locations. Exposure estimates for this scenario are based on some conservative assumptions about flow rates and release frequency and amount. A summary of sources of flow and release data for facility release modeling is presented in Table 2-6. Available monitoring data confirm that 1,4-dioxane is present in some surface water and drinking water, though most of the available data were not collected near release sites are therefore not directly comparable. The overall level of confidence depends on the source of OES/COU-specific release data described in Appendix E.3:

- Overall confidence in drinking water exposure estimates is medium to high for OESs/COUs that rely primarily on release data reported to DMR or to TRI via Form R. Most COUs/OESs are included in this group.
- Overall confidence in drinking water exposure estimates is medium for OESs/COUs for which release estimates are based on reporting to TRI via Form A. The Import and Repackaging OES releases used in this analysis are entirely based on Form A reporting of releases, and just under half of the Industrial Uses OES releases were reported via Form A.
- Overall confidence in drinking water exposure estimates is low to medium for OESs/COUs for which release estimates are based on surrogate or modeled information.

Overall confidence in risk estimates for drinking water exposures resulting from DTD releases under varying population and stream flow conditions is medium. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.1, the exposure modeling methodology used for this analysis is supported by robust evidence and is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. Exposure estimates rely on estimated distributions of DTD releases of consumer and commercial products for each COU.

Overall confidence in risk estimates for drinking water exposures resulting from hydraulic fracturing releases is medium. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.1, the exposure modeling methodology used for this analysis is supported by robust evidence and is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. Releases used as inputs in the model were estimated using Monte Carlo modeling that captures variability across sites. However, the modeled exposure estimates are not directly tied to specific releases at known locations, decreasing the strength of the evidence related to the representativeness of the exposure estimates for actual exposures.

Overall confidence in risk estimates for drinking water exposures resulting from for aggregate surface water concentrations predicted by probabilistic modeling varies across OES/COU. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.1, the exposure modeling methodology used for this analysis is supported by robust evidence and is designed to provide a nationally representative distribution of estimated water concentrations under varying conditions. Available monitoring data confirm that 1,4-dioxane is present in some surface water and drinking water, though most of the available data were not collected near release sites and are therefore not directly comparable. In release locations where monitoring data are available, case studies demonstrate strong concordance between modeled estimates and measured surface water concentrations. There is some uncertainty around the proximity of releases to drinking water intake locations and the extent to which 1,4-dioxane is further diluted prior to reaching intake locations. The characterization of downstream dilution presented in 5.2.2.1.2 for individual facility releases illustrates the extent to which downstream dilution may impact overall risk estimates. The overall level of confidence in resulting exposure estimates depends on the source of OES/COU-specific release data described in Appendix E.3:

- Overall confidence in drinking water exposure estimates is medium to high for OESs/COUs that rely primarily on release data reported to DMR or to TRI via Form R. Most COUs/OESs are included in this group.
- Overall confidence in drinking water exposure estimates is medium for OESs/COUs for which release estimates are based on reporting to TRI via Form A. The Import and Repackaging OES releases used in this analysis are entirely based on Form A reporting of releases, and just under half of the Industrial Uses OES releases were reported via Form A.
- Overall confidence in drinking water exposure estimates is low to medium for OESs/COUs for which release estimates are based on surrogate or modeled information.

5.2.2.6.3 Risks from General Population Exposures through Groundwater and Land Disposal Pathways

Overall confidence in risk estimates for drinking water exposures resulting from disposal to landfills is low to medium. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.2.1 the modeling methodology is robust. However, the release information relied on as model input data is supported by slight to moderate evidence. In addition, this drinking water exposure scenario relies on the assumption that the estimated groundwater concentrations may occur in locations where groundwater is used as a primary drinking water source. While the substantial uncertainty around the extent to which these exposures occur decreases overall confidence in the exposure scenario, this scenario represents a sentinel exposure.

Overall confidence in risk estimates for drinking waters resulting from disposal of hydraulic fracturing waste is low to medium. As described in Section 4.3 overall confidence in the oral and dermal cancer slope factor underlying these risk estimates is medium-high. As described in Section 3.3.2.2.2, the modeling methodology is robust, and the release information relied on as model input data is supported by moderate evidence. However, no monitoring data are available to confirm detection of 1,4-dioxane in groundwater near hydraulic fracturing operations. This drinking water exposure scenario relies on the assumption that the estimated groundwater concentrations may occur in locations where groundwater is used as a primary drinking water source. While the substantial uncertainty around the extent to which these exposures occur decreases overall confidence in the exposure scenario, this scenario represents a sentinel exposure.

5.2.2.6.4 Risks from General Population Exposures through Air

Overall confidence in risk estimates for inhalation exposure resulting from industrial releases varies across COUs. As described in Section 4.3, overall confidence in the cancer inhalation unit risk underlying these risk estimates is high. As described in Section 3.3.3.1, the AERMOD modeling methodology used for this analysis is robust. The exposure scenarios considered are most relevant to long-term residents in fenceline communities. There is some uncertainty around the extent to which people actually live and work around the specific facilities where risks are highest, decreasing overall confidence in the exposure scenario. Overall confidence varies due to variable levels of confidence in underlying release information used to estimate exposures. An OES-specific discussion of the confidence in sources of release information is presented in Appendix E.5.4, but in general terms:

- Overall confidence in risk estimates is medium to high for OESs/COUs that rely primarily on release data reported to TRI via Form R.
- Overall confidence in risk estimates is medium for OESs/COUs for which release estimates are based on data reported to TRI via Form A.
- Overall confidence in risk estimates is low to medium for OESs/COUs for which release estimates are based on surrogate or modeled information.

Overall confidence in risk estimates for inhalation exposures resulting for air concentrations modeled based on releases from hydraulic fracturing operations is medium. As described in Section 4.3, overall confidence in the cancer inhalation unit risk underlying these risk estimates is high. As described in Section 3.3.3.2 the modeling methodologies used to estimate air concentrations are robust. The distribution of air releases used as model input data were estimated using Monte Carlo modeling and rely on assumptions. No air monitoring data were available to confirm detection of 1,4-dioxane is air near hydraulic fracturing operations. Because the air concentrations underlying this analysis are based on probabilistic modeling, they are not tied to specific locations that can be evaluated for land use patterns. There is therefore substantial uncertainty around the extent to which people actually live and work around the specific locations where risks are highest, decreasing overall confidence in the exposure scenario.

Overall confidence in risk estimates from inhalation exposures resulting from industrial and institutional laundries is medium. As described in Section 4.3, overall confidence in the cancer inhalation unit risk underlying these risk estimates is high. As described in Section 3.3.3.2, the modeling methodologies are robust. The distribution of air releases used as model input data were estimated using Monte Carlo modeling and rely on assumptions. No air monitoring data were available to determine whether 1,4-dioxane is detected near industrial and institutional laundry facilities. Because the air concentrations underlying this analysis are based on probabilistic modeling, they are not tied to specific locations that can be evaluated for land use patterns. There is therefore substantial uncertainty around the extent to which people actually live and work around the specific locations where risks are highest, decreasing overall confidence in the exposure scenario.

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APPENDICES

Appendix A ABBREVIATIONS AND ACRONYMS

30Q5	Lowest 30-day average flow that occurs in a 5-year period
7Q10	Lowest 7-day average flow that occurs in a 10-year period
AC	Acute concentrations
ACA	American Coatings Association
ACGIH	American Conference of Governmental Industrial Hygienists
ADC	Average daily concentration
ADD	Average daily dose
ADR	Acute Dose Rate
AEC	Acute Exposure Concentration
APF	Assigned protection factor
ASTDR	Agency for Toxic Substances and Disease Registry
BHET	Bishydroxyethyl terephthalate
BLS	Bureau of Labor Statistics
BMD	Benchmark dose
BMDL	Benchmark dose level
CASRN	Chemical Abstracts Service Registry Number
CDR	Chemical Data Reporting
CEB	Chemical Engineering Branch
CEHD	Chemical Exposure Health Data
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CFR	Code of Federal Regulations
COU	Condition of use
CSF	Cancer slope factor
CT	Central tendency
CWA	Clean Water Act
DAF	Dilution attenuation factor
DHHR	Department of Health and Human Services
DIY	Do-it-yourself
DMR	Discharge monitoring report
DRAS	Delisting Risk Assessment Software
DTD	Down-the-drain
DWI	Drinking water intake
DWT	Drinking water treatment
ECHA	European Chemicals Agency
ECHO	Environmental Compliance History Online database
EPA	Environmental Protection Agency
EPACMTP	Environmental Protection Agency Composite Model for Leachate Migration with Transformation Products
EPCRA	Emergency Planning and Community Right-to-Know Act
EROM	Enhanced Runoff Method (database)
ESD	Emission Scenario Document
FRS	Facility Registry Service
FT	Full-text (screening)
GS	Generic Scenario
HAWC	Health Assessment Workplace Collaborative (tool)

HE	High-end
HEC	Human equivalent concentration
HED	Human equivalent dose
HERO	Health and Environmental Research Online (EPA Database)
HHE	Health hazard evaluation
HSDB	Hazardous Substances Data Bank
ICIS	Integrated Compliance Information System
IFC	Industrial Function Category
IIOAC	Integrated Indoor/Outdoor Air Calculator (EPA)
IRIS	Integrated Risk Information System
IUR	Inhalation unit risk
K_{OC}	Soil organic carbon: water partitioning coefficient
K_{OW}	Octanol: water partition coefficient
LADC	Lifetime Average Daily Concentration
LADD	Lifetime Average Daily Dose
LOAEC	Lowest-observed-adverse-effect-concentration
LOD	Limit of detection
Log K_{OC}	Logarithmic organic carbon: water partition coefficient
Log K_{OW}	Logarithmic octanol: water partition coefficient
LOQ	Limit of quantitation
MLD	Million liters per day
MOE	Margin of exposure
MRD	Methodology Review Draft (EPA)
MW	Molecular weight
NAICS	North American Industry Classification System
ND	Non-detect
NEI	National Emissions Inventory
NIOSH	National Institute for Occupational Safety and Health
NOAEL	No-observed-adverse-effect-level
NPDES	National Pollutant Discharge Elimination System
OAQPS	Office of Air Quality Planning and Standards
OCF	One-component foam
OCSPP	Office of Chemical Safety and Pollution Prevention
OD	Operating days
OECD	Organisation for Economic Co-operation and Development
OEHHA	Office of Environmental Health Hazard Assessment
OES	Occupational exposure scenario
ONU	Occupational non-user
OPPT	Office of Pollution Prevention and Toxics
OSHA	Occupational Safety and Health Administration
PBZ	Personal breathing zone
PECO	Population, exposure, comparator, and outcome
PEL	Permissible exposure limit
PESS	Potentially exposed or susceptible subpopulations
PET	Polyethylene terephthalate
PF	Protection factor
PNOR	Particulates not otherwise regulated
POD	Point of departure
POTW	Publicly owned treatment works (wastewater)

PPE	Personal protective equipment
PV	Production volume
PWS	Public water system
QA/QC	Quality assurance/quality control
QE	NHDPlus V2.1 flow values representing “the best EROM estimate of actual mean flow”
RE	(2020 RE) Risk Evaluation
RCRA	Resource Conservation and Recovery Act
REACH	Registration, Evaluation, Authorisation and Restriction of Chemicals (European Union)
RESO	Receptors, exposure, setting or scenario, and outcomes
SACC	Science Advisory Committee on Chemicals
SDS	Safety data sheet
SDWA	Safe Drinking Water Act
SHEDS-HT	Stochastic Human Exposure and Dose Simulation-High Throughput
SIC	Standard Industrial Classification
SOC	Standard Occupational Classification
SpERC	Specific Environmental Release Categories
SPF	Spray polyurethane foam
STORET	STOrage and RETrieval and Water Quality exchange
SUSB	Statistics of United States Businesses
SWIFT	Sciome Workbench for Interactive Computer-Facilitated Text-mining
TIAB	Title/abstract (screening)
TRI	Toxics Release Inventory
TSCA	Toxic Substances Control Act
TWA	Time-weighted average
UCMR	Unregulated Contaminant Monitoring Rule
U.S.	United States
USGS	U.S. Geological Survey
VOC	Volatile organic compound
VP	Vapor pressure
WQP	Water Quality Portal
WWT	Wastewater treatment

Appendix B LIST OF SUPPLEMENTAL DOCUMENTS

Associated **Systematic Review Data Quality Evaluation and Data Extraction Documents** – Provide additional detail and information on individual study evaluations and data extractions including criteria and data quality results.

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure – Provides a compilation of tables for the data extraction and data quality evaluation information for 1,4-dioxane. Each table shows the data point, set, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation of environmental release and occupational exposure (U.S. EPA, 2023t). This supplemental file may also be referred to as the 1,4-Dioxane Supplement to the Risk Evaluation Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure.

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Systematic Review Supplemental File: Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure – Provides a compilation of tables for the data quality evaluation information for 1,4-dioxane. Each table shows the data point, set, or information element that was evaluated from a data source that has information relevant for the evaluation of general population, consumer, and environmental exposure (U.S. EPA, 2023r). This supplemental file may also be referred to as the 1,4-Dioxane Supplement to the Risk Evaluation Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure.

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Systematic Review Supplemental File: Data Extraction Information for General Population, Consumer, and Environmental Exposure – Provides a compilation of tables for the data extraction for 1,4-dioxane. Each table shows the data point, set, or information element that was extracted from a data source that has information relevant for the evaluation of general population, consumer, and environmental exposure (U.S. EPA, 2023h). This supplemental file may also be referred to as the 1,4-Dioxane Supplement to the Risk Evaluation Data Extraction Information for General Population, Consumer, and Environmental Exposure.

Associated **Supplemental Information Files** – Provide additional details and information on exposure, hazard, and risk assessments.

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Environmental Releases to Air – Provides a summary of stack and fugitive air emissions for each occupational exposure scenario (OES) in the 1,4-Dioxane Supplemental Risk Evaluation (U.S. EPA, 2023j).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Environmental Releases to Land for all OES Except Disposal – Provides a summary of land releases for each 1,4-dioxane OES except for the Disposal OES (U.S. EPA, 2023k).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Environmental Releases to Land for the Disposal OES – This spreadsheet contains a summary of land releases for the 1,4-dioxane occupational Disposal OES (U.S. EPA, 2023l).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Environmental Releases to Water for OES without TRI or DMR data – Provides a summary of direct and indirect water releases for each 1,4-dioxane OES for which Toxics Release Inventory (TRI) and Discharge Monitoring Report (DMR) data were not available (U.S. EPA, 2023n).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Environmental Releases to Water for OES with TRI and DMR – Provides a summary of direct and indirect water releases for each 1,4-dioxane OES for which TRI or DMR data were available (U.S. EPA, 2023m).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Occupational Exposure and Risk Estimates – Provides a summary of occupational exposures and risks estimated for all conditions of use (COUs; (including those evaluated in this supplemental evaluation as well as those previously evaluated in the 2020 RE) (U.S. EPA, 2023r).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Release to Surface Water from Individual Facilities – Provides water concentrations estimated from individual facility releases reported to TRI and calculates corresponding drinking water exposures and risks (U.S. EPA, 2023h).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Surface Water Concentrations Predicted with Probabilistic Modeling – Provides water concentrations estimated by probabilistic modeling for DTD releases, disposal of hydraulic fracturing waste to surface water, and for aggregate concentrations estimated downstream of industrial release sites; calculates corresponding drinking water exposures and risks. This file also calculates drinking water exposures and risks estimated from drinking water monitoring data (U.S. EPA, 2023i).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: EWISRDXL BrunswickCountyNC Case Study – Provides the Excel workbook file for the Brunswick County, NC surface water case study (U.S. EPA, 2023o).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: EWISRDXL ColumbiaTN Case Study – Provides the Excel workbook file for the Columbia, TN surface water case study (U.S. EPA, 2023p).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: EWISRDXL LiverpoolOH Case Study – Provides the Excel workbook file for the Liverpool, OH surface water case study (U.S. EPA, 2023q).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane –Supplemental Information File: EWISRD-XL-R probabilistic model code – Provides the R script used to perform the probabilistic surface water modeling by OES/COU (U.S. EPA, 2023a).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: WQP Processed Surface Water Data – Provides the processed monitoring data in surface water retrieved from the Water Quality Portal (U.S. EPA, 2023s).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Land Releases to Landfills – Provides calculations of groundwater concentration derived from the waste adjusted dilution attenuation factor extracted from Delisting Risk Assessment Software (DRAS) for Landfills and the corresponding risk calculations (U.S. EPA, 2023f).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Drinking Water Exposure and Risk Estimates for 1,4-Dioxane Land Releases to Surface Impoundments – Provides calculations of groundwater concentration derived from the waste adjusted dilution attenuation factor extracted from DRAS for release of hydraulic fluid produced water to surface impoundments and the corresponding risk calculations (U.S. EPA, 2023g).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Air Exposures and Risk Estimates for Single Year Analysis – Provides air concentrations estimated by American Meteorological Society/Environmental Protection Agency Regulatory Model (AERMOD) for air releases reported to TRI in 2019 and calculates corresponding exposure concentrations and risk estimates (U.S. EPA, 2023e).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Air Exposure and Risk Estimates for 1,4-Dioxane Emissions from Hydraulic Fracturing Operations – Provides air concentrations estimated by Integrated Indoor/Outdoor Air Calculator (IIOAC) based on Monte Carlo modeling of air releases from hydraulic fracturing operations and calculates corresponding exposure concentrations and risks (U.S. EPA, 2023b).

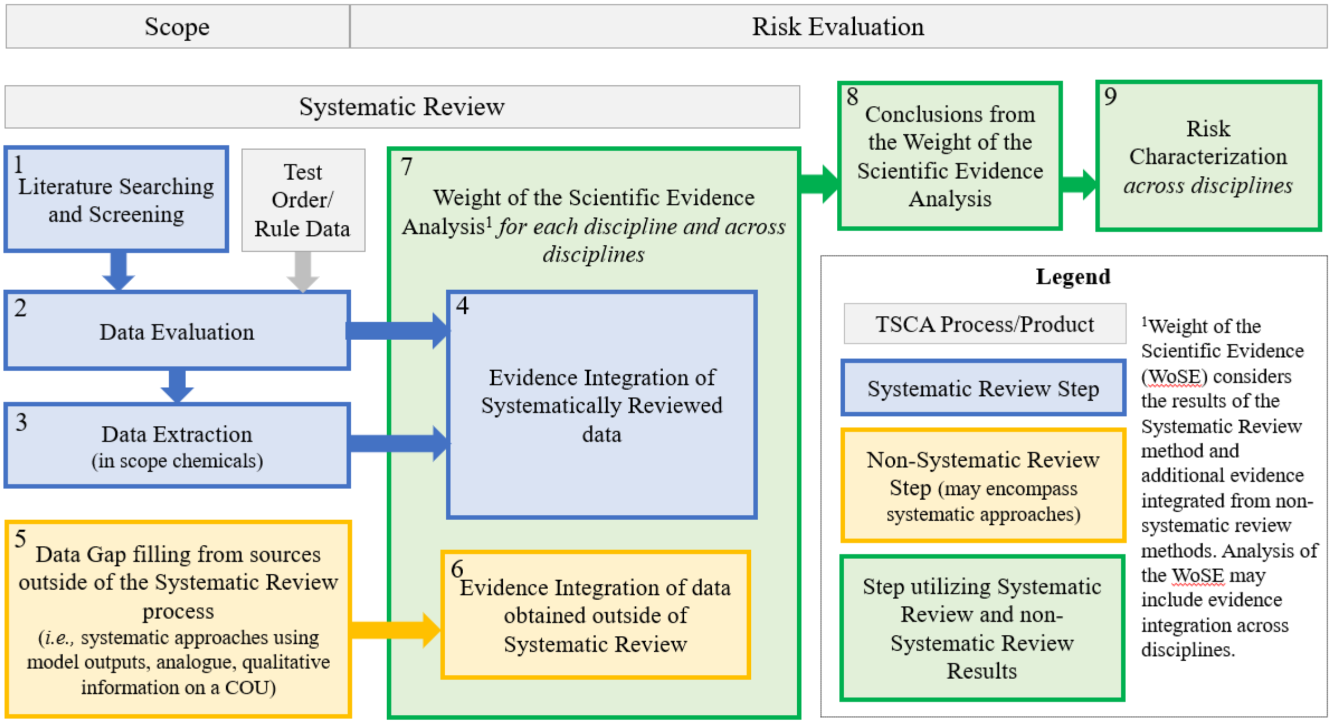
Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Air Exposures and Risk Estimates for Industrial Laundry – Provides air concentrations estimated by IIOAC based on Monte Carlo modeling of air releases from hydraulic fracturing operations and calculates corresponding exposure concentrations and risks (U.S. EPA, 2023c).

Draft Supplement to the Risk Evaluation for 1,4-Dioxane – Supplemental Information File: Air Exposures and Risk Estimates for Multi-Year Analysis – Provides air concentrations estimated by IIOAC for 6 years (2015 to 2020) of air releases reported to TRI and calculates the corresponding exposure concentrations and risk estimates (U.S. EPA, 2023d).

Appendix C SYSTEMATIC REVIEW PROTOCOL FOR THE DRAFT SUPPLEMENT TO THE RISK EVALUATION FOR 1,4-DIOXANE

The U.S. EPA's Office of Pollution Prevention and Toxics (EPA/OPPT) applies systematic review principles in the development of risk evaluations under the amended TSCA. TSCA section 26(h) requires EPA to use scientific information, technical procedures, measures, methods, protocols, methodologies, and models consistent with the best available science and base decisions under section 6 on the weight of scientific evidence. Within the TSCA risk evaluation context, the weight of the scientific evidence is defined as "a systematic review method, applied in a manner suited to the nature of the evidence or decision, that uses a pre-established protocol to comprehensively, objectively, transparently, and consistently identify and evaluate each stream of evidence, including strengths, limitations, and relevance of each study and to integrate evidence as necessary and appropriate based upon strengths, limitations, and relevance" (40 CFR 702.33).

To meet the TSCA section 26(h) science standards, EPA used the TSCA systematic review process described in the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances* (U.S. EPA, 2021a) (hereinafter referred to as "2021 Draft Systematic Review Protocol"). Table_Apx C-1 (Section 3 of the 2021 Draft Systematic Review Protocol) depicts the steps in which information is identified and whether it undergoes the formal systematic review process (U.S. EPA, 2021a). Information attained via the systematic review process is integrated with information attained from sources of information that do not undergo systematic review (*e.g.*, EPA-generated model outputs) to support a weight of the scientific evidence analysis.



Figure_Apx C-1. Overview of the TSCA Risk Evaluation Process with Identified Systematic Review Steps

The process complements the risk evaluation process in that are used to develop the exposure and hazard assessments based on reasonably available information. EPA defines "reasonably available information" to mean information that EPA possesses or can reasonably obtain and synthesize for use in risk evaluations, considering the deadlines for completing the evaluation (40 CFR 702.33).

C.1 Clarifications and Updates to the 2021 Draft Systematic Review Protocol

In 2021, EPA released the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances* (U.S. EPA, 2021a), a framework of systematic review approaches under TSCA, to address comments received on a precursor systematic review approaches framework, the Application of Systematic Review in TSCA Risk Evaluations (U.S. EPA, 2018c). In April 2022, the SACC provided comments on the 2021 Draft Systematic Review Protocol while additional comments on OPPT's systematic review approaches were garnered during the public comment period. In lieu of an update to the 2021 Draft Systematic Review Protocol, this systematic review protocol for the *Draft Supplement to the Risk Evaluation for 1,4-Dioxane* describes some clarifications and different approaches that were implemented than those described in the 2021 Draft Systematic Review Protocol in response to (1) SACC comments, (2) public comments, or (3) to reflect chemical-specific risk evaluation needs.

C.1.1 Clarifications and Updates

Throughout the 2021 Draft Systematic Review Protocol, there were some terms used that were not explicitly defined, resulting in their different uses within the document (U.S. EPA, 2021a). Table_Apx C-1 lists the terms that were updated to resolve some of the confusion expressed by the public and SACC comments regarding the implementation of the respective systematic review-related step. One main clarification is that *all references that undergo systematic review are considered for use in the risk evaluation*, even those that do not meet the various discipline and sub-discipline screening criteria (*i.e.*, RESO, PESO, PECO) or that are categorized as supplemental information at title and abstract (TIAB) or full-text screening.

Section 4.2.5 of the 2021 Draft Systematic Review Protocol describes how data sources (*e.g.*, individual references, databases) may be tagged and linked in epidemiological cohort studies when information is present in multiple studies (U.S. EPA, 2021a). References will generally undergo data quality evaluation and extraction if there are data that pass screening criteria; however, to prevent the same data from being represented multiple times and conflating the amount of available information on a subject area, EPA selects the reference(s) that most appropriately describes the extractable results (indicated as the parent reference in DistillerSR). For example, if two references portray the same information from the same dataset, only one is counted in the overall dataset (*i.e.*, deduplication). If two references contain information about the same dataset, but only one provides additional contextual information or summary statistics (*e.g.*, mean), both data sources are linked but the extractable information from both may be combined in DistillerSR. This allows the capture of key information while avoiding double counting the data of interest, which may be the case whether or not one reference contains original or extractable data that passes screening criteria. The linked reference containing the majority of the data, which are evaluated and extracted, is identified in DistillerSR as the parent reference; the "complementary child reference" in DistillerSR does not undergo data evaluation and extraction. Linking the references in DistillerSR allows the reference with more limited information or only contextual information to be tracked and utilized to evaluate the extracted data in the other related studies. The child reference may undergo data quality evaluation and extraction if there are additional unique and original data that pass screening criteria. One clarification is that this procedure of identifying potential duplicative information *applies to all information that is considered in a risk evaluation under TSCA (not just epidemiological* *cohort studies)*. Also, this procedure may apply when there is duplicative information in two references, even if it is more than just "contextual."

Section 5 of the 2021 Draft Systematic Review Protocol describes how EPA conducts data quality evaluation of data/information sources considered for a respective chemical risk evaluation, with Section 5.2 specifically explaining the terminology used to describe both metric and overall data/information source quality determinations (U.S. EPA, 2021a). To respond to both SACC and public comments regarding the inappropriate use of quantitative methodologies to calculate both "Metric Rankings" and "Overall Study Rankings", *EPA decided to not implement quantitative methodologies to attain either metric and overall data/information source quality determinations* and therefore updated the terminology used for both metric ("Metric Ranking") and overall data/information source ("Overall Study Ranking") quality determinations (Table_Apx C-1). Specifically, metric and overall data/information source quality determination terminology have been updated to "Metric Rating" and "Overall Quality Determination", respectively. The word "level" was also often used synonymously and inconsistently with the word "ranking" in the 2021 Draft Systematic Review Protocol; that inconsistency has been rectified, resulting in the word "level" no longer being used to indicate either metric or overall data/information source quality determinations (U.S. EPA, 2021a).

Sections 4.3.2.1.3 and 6 of the 2021 Draft Systematic Review Protocol describes when EPA may reach out to authors of data/information sources to obtain raw data or missing elements that are important to support the data evaluation and data integration steps (U.S. EPA, 2021a). In such cases, the request(s) for additional data/information, number of contact attempts, and responses from the authors are documented. EPA's outreach is considered unsuccessful if those contacted do not respond to email or phone requests within 1 month of initial attempt(s) of contact. One important clarification to this guidance is that *EPA may reach out to authors anytime during the systematic review process for a given data/information source or reference, and that contacting authors does not explicitly happen during the data quality evaluation or extraction steps.*

Table_Apx C-1. Terminology Clarifications between the 2021 Draft Systematic Review Protocol and the 1,4-Dioxane Systematic Review Protocol

2021 Draft Systematic Review Protocol Term	1,4-Dioxane Systematic Review Protocol Term Update	Clarification
"Title and abstract" or "Title/abstract"	Title and abstract (TIAB)	To increase consistency, the term "title and abstract" will be used to refer to information specific to "title and abstract" screening.
Variations of how "include," "on topic" or "PECO/PESO/RESO relevant" implied a reference was considered for use in the risk evaluation, whereas "exclude," "off topic" or "not PECO/PESO/RESO relevant" implied a reference was *not* considered for use in the risk evaluation.	Meets/does not meet PECO/PESO/RESO screening criteria	The term "include" or "exclude" falsely suggests that a reference was or was not, respectively, considered in the risk evaluation. There was also confusion regarding whether "on topic" and "PECO/PESO/RESO relevant" were synonymous and suggested those references were explicitly considered for use in the risk evaluation (and by default, "off topic" and "not PECO/PESO/RESO relevant" references were not). References that meet the screening criteria (*e.g.*, PECO, PESO, RESO) proceed to the next systematic review step; however, all references that undergo systematic review at any time are considered in the risk evaluation. Information that is categorized as

2021 Draft Systematic Review Protocol Term	1,4-Dioxane Systematic Review Protocol Term Update	Clarification
		supplemental or does not meet screening criteria are generally less relevant for quantitative use in the risk evaluation but may be considered if there is a data need identified. For instance, mechanistic studies are generally categorized as supplemental information at either title and abstract or full-text screening steps but may undergo the remaining systematic review steps if there is a relevant data need for the risk evaluation (*e.g.*, dose response, mode of action).
Database source not unique to a chemical	Database	Updated term and definition of "Database": Data obtained from databases that collate information for the chemical of interest using methods that are reasonable and consistent with sound scientific theory and/or accepted approaches and are from sources generally using sound methods and/or approaches (*e.g.*, state or federal governments, academia). Example databases include STORET and the Massachusetts Energy and Environmental Affairs Data Portal. The term in the 2021 Draft Systematic Review Protocol (Table_Apx N-1) incorrectly suggested that databases that contain information on a singular chemical are not considered (U.S. EPA, 2021a). Furthermore, the wording "large" was removed to prevent confusion and the incorrect suggestion that there is a data size requirement for databases that contain information that may be considered for systematic review.
Metric Ranking or Level	Metric Rating	As explained above, EPA is not implementing quantitative methodologies to indicate metric quality determinations, therefore the term "ranking" is inappropriate. The term "level" was inconsistently used to indicate metric quality determinations previously, therefore EPA is removing the use of this term to reduce confusion when referring to metric quality determinations. The term "Rating" is more appropriate to indicate the use of professional judgement to determine a quality level for individual metrics.
Overall Study Ranking or Level	Overall Quality Determination (OQD)	As explained above, EPA is not implementing quantitative methodologies to indicate overall data/information source quality determinations, therefore the term "ranking" is inappropriate. The term "level" was inconsistently used to indicate overall data/information source quality determinations previously, therefore EPA is removing the use of this term to reduce confusion

2021 Draft Systematic Review Protocol Term	1,4-Dioxane Systematic Review Protocol Term Update	Clarification
		when referring to overall data/information source quality determinations. The term "Rating" is more appropriate to indicate the use of professional judgement to determine a quality level for the overall data/information source quality determination.

C.2 Data Search

To expand upon the previous analysis conducted in the 2020 RE, this draft Supplement to the Risk Evaluation for 1,4-Dioxane addresses additional COUs in which 1,4-dioxane is present as a byproduct of the manufacturing process and evaluates risks from general population exposures to 1,4-dioxane released to water, air, and land. This draft supplement focuses on evaluating additional exposure pathways that were not addressed in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). Therefore, the data search focused on prioritizing updated literature search results to characterize environmental releases and occupational exposure, and general population, consumer, and environmental exposure information to evaluate the exposure pathways in scope for this draft supplement. Data sources may also contain information that may be used to evaluate exposure pathways already addressed in the 2020 RE (U.S. EPA, 2020c) (*i.e.*, consumer exposure). Below are the four additional exposure pathways being assessed in this Draft Supplement to the Risk Evaluation for 1,4-Dioxane (Section 1.2).

- Occupational exposure to
    - 1,4-dioxane present as a byproduct in commercial products during ethoxylation processing or polyethylene terephthalate (PET) manufacturing and in hydraulic fracturing waste (Sections 3.1, 5.2.1)
- General population exposures to
    - 1,4-dioxane present in drinking water sourced from surface water as a result of direct and indirect industrial releases and DTD releases of consumer and commercial products (Sections 2.3.1, 3.2.2 and 5.2.2.1);
    - 1,4-dioxane present in drinking water sourced from groundwater contaminated as a result of disposals (Sections 2.3.2, 3.2.2.2 and 5.2.2.1.6); and,
    - 1,4-dioxane released to air from industrial and commercial sources (Sections 2.3.3, 3.2.3, and 5.2.2.3).

C.2.1 Multi-Disciplinary Updates to the Data Search

For this *Draft Supplement to the Risk Evaluation for 1,4-Dioxane*, the updated literature search was conducted as described in Section 4 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), where the peer-reviewed and gray literature updated search followed the approach outlined in Sections 4.2 and 4.3 of the 2021 Draft Systematic Review Protocol, respectively (U.S. EPA, 2021a). The updated search for peer-reviewed and gray literature relevant references was completed in October 2021 and January 2022, respectively, which also considered information found for the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). Occasionally additional data sources relevant for the risk evaluation may be identified after the initial search for peer-reviewed and gray literature; these data sources will then undergo systematic review for the relevant discipline(s). Additionally, each discipline utilizes different strategies (*e.g.*, search strings) to attain their discipline-specific pools of data sources that undergo systematic review

As mentioned in Section 4.2.2 of the 2021 Draft Systematic Review Protocol, a supplemental literature search is conducted to fill data gaps, but in this draft supplement, the supplemental search was conducted to update the literature search conducted to identify any potentially relevant environmental release and occupation exposure and general population, consumer, and environmental exposure information (U.S. EPA, 2021a). Rather than utilizing positive and negative seed references as described in Section 4.2.4.2 of the 2021 Draft Systematic Review Protocol, search strings were used in SWIFT[16]-Review to better identify relevant references to evaluate exposure pathways addressed in this draft supplement (U.S. EPA, 2021a). The language describing the new exposure pathways and COUs that are in scope for this draft supplement was used to derive the search strings listed below in Sections C.2.3.1 and C.2.3.2. When the search strings are identified in the title, abstract, keyword, or Medical Subject Heading (MeSH) fields of a given reference in SWIFT-Review, those references proceeded with TIAB screening.

The evaluation of physical and chemical properties, fate properties and environmental and human health hazard information did not differ from the respective information provided in the Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c) to address the additional exposure pathways in this draft supplement, therefore no additional references were identified for these respective topics or underwent systematic review for these disciplines. One minor clarification to what was described in the 2021 Draft Systematic Review Protocol is that the PECO statement used to screen general population, consumer, and environmental exposure information considered for this draft supplement, currently resides in Appendix Section H.5 (which was intended to encompass PECO statements regarding environmental and human health hazard information), rather than in Appendix Section H.4 (U.S. EPA, 2021a). Please see Appendix C.3.2 below for additional updates specific to the implementation of the PECO statement.

C.2.2 Additional Data Sources Identified

As mentioned above in Appendix C.2, additional data sources containing potentially relevant information for a respective risk evaluation may be identified. For this draft supplement, additional gray literature data sources were identified for the characterization of environmental release and occupational exposure and general population, consumer, and environmental exposure, as explained below in Sections C.2.2.1 and C.2.2.2, respectively.

C.2.2.1 Additional Data Sources Identified for Environmental Release and Occupational Exposure

As explained in Appendix E of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), generic scenarios and emission scenarios documents are listed as part of the initial gray literature sources. Some generic scenarios and a draft emission scenario document became available after the gray literature search was completed in January 2022 and were considered for the environmental release and occupational exposure assessment. This includes the Draft OECD ESD on Hydraulic Fracturing (U.S. EPA, 2022d), Draft GS on Furnishing Cleaning Products (U.S. EPA, 2022a), EPA Methodology Review Draft (MRD) on Commercial Use of Automotive Detailing Products (U.S. EPA, 2022b), and Draft GS on Use of Laboratory Chemicals (U.S. EPA, 2022h). The updated sources were added to EPA's Health and Environmental Research Online (HERO) database in 2022 as well as the systematic review process.

In addition to the gray literature sources listed above, an online database called FracFocus 3.0 (GWPC and IOGCC, 2022) was included in the pool of references EPA considered for environmental release and occupational exposure through backward searching. These are described in Section 4.4 of the 2021 Draft

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[16] SWIFT is an acronym for "Sciome Workbench for Interactive Computer-Facilitated Text-mining." SWIFT-Active Screener uses machine learning approaches to save screeners' time and effort.

Systematic Review Protocol (U.S. EPA, 2021a). Backward searching from the Draft OECD ESD on Hydraulic Fracturing (U.S. EPA, 2022d) led to EPA's identification of the FracFocus data. The Agency gathered the data directly from the source and only pulled data specific to sites that reported using 1,4-dioxane in fracturing fluids. This source was added to the HERO database as well as the systematic review process.

C.2.2.2 Additional Data Sources Identified for General Population, Consumer, and Environmental Exposure

In addition to the gray literature sources listed in Appendix E of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), several other gray literature sources were considered for inclusion the general population, consumer, and environmental exposure assessment and added to the HERO database in 2022. The Water Quality Portal (WQP) database, the successor of EPA's STORET (STOrage and RETrieval) database, was incorporated because it includes a large variety of chemical-specific data. Also, WQP is a portal that combines data from multiple databases—not just STORET—such as the U.S. Geological Survey's National Water Information System. Information from WQP was collected in July 2022.

A few additional gray literature sources (databases) were included in the pool of references EPA considered on general population, consumer, and environmental exposure through backwards searching, which is described in Section 4.3.3 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Backwards searching from the Third Unregulated Contaminant Monitoring Rule (UCMR3) database (U.S. EPA, 2017d) led to EPA's identification of data from a few states collecting data on 1,4-dioxane for longer periods of time than reported in UCMR3. EPA was able to secure data from three state databases, which as added. In addition, elevated levels of 1,4-dioxane in samples from UCMR3 and a reference found in the pool of peer-reviewed articles led to addition to databases with data on 1,4-dioxane levels measured in drinking water and surface water in the Cape Fear Watershed in North Carolina. Six of the highest concentrations of 1,4-dioxane in the UCMR3 database were reported in water from this watershed, which were also discussed in one of the references found in the literature search for peer-reviewed sources. All gray literature database sources added to the search, including the datasets from North Carolina, were added to HERO and the systematic review process (see Section G.1 for further information).

C.2.3 Search Strings

As explained above in Section C.2.1, the search strings below were used to identify references relevant to evaluating environmental releases and occupational exposure, as well as general population, consumer, and environmental exposure.

C.2.3.1 Environmental Release and Occupational Exposure Search Strings

Life Cycle:
TIAB: ("1,4-dioxane" AND ("MFG" OR "import" OR "processing" OR "manufactur*" OR "releases" OR "waste disposal" OR "reaction product" OR "repackaging" OR "recycling" OR "throughput" OR "operating days" OR "batch" OR "production speed"))

Treatment Efficiencies:
TIAB: ("1,4-dioxane" AND ("GAC" OR "granular activated carbon" OR "reverse osmosis" OR "advanced oxidation" OR "hydrogen peroxide with ultraviolet" OR ("hydrogen peroxide" AND "UV") OR "hydrogen peroxide with ozone" OR ("hydrogen peroxide" AND "ozone") OR "AOP" OR "Fenton's reagent" OR "bioremediation"))

Occupational Workers:
TIAB: (“1,4-dioxane” AND (“janitor*” OR “mechanic” OR “laborer” OR “custodia*” OR “painter*” OR “laboratory technician” OR “laboratory employee*” OR (“pharmaceutical” AND (“employee” OR “worker” OR “technician”)) OR “residential construction” OR “industrial construction”))

General:
TIAB: (“1,4-dioxane” AND (“surfactant” OR “ethoxylat*” OR “nonylphenol ethoxylate” OR “alkylphenol ethoxylate” OR “sulfated” OR “industrial laundr*” OR “commercial laundr*” OR “institutional laundr*” OR “institutional laundr*” OR “advanced oxidation” OR “ozone-peroxide advanced oxidation” OR “low dioxane” OR “low dioxane ether sulfates” OR “low dioxane ethoxylated surfactants” OR “low 1,4 dioxane ether sulfates” OR “low 1,4 dioxane ethoxylated surfactants” OR “safety data sheet” OR “material safety data sheet”))

Process Uses:
TIAB: (“1,4-dioxane” AND (“stabilizer” AND (“chlorinated solvents” OR “degreasing” OR “electronics manufacturing” OR “metal finishing”)) OR (“solvent” AND (“histology” OR “cellulose acetate membrane” OR “microscopy” OR “organic chemical manufacturing” OR “organic chemical”)) OR (“textile” AND (“wetting” OR “dispersing”)) OR (“esterification” AND (“by-product” OR “byproduct”)))

Product Uses:
TIAB: (“1,4-dioxane” AND (“solvent” AND (“paint*” OR “lacquer*” OR “varnish remover” OR “stain” OR “printing” OR “scintillation” OR “resin*” OR “oil*” OR “rubber chemicals” OR “rubber” OR “sealant*” OR “adhesive*” OR “wax*” OR “cement*”)))
TIAB: (“1,4-dioxane” AND (“artificial leather” OR “purifying agent” OR “antifreeze” OR “de-icing” OR “pesticide*” OR “fumigant*”))

CASRNs of Ethoxylated Chemicals:
TIAB: (“9005-65-6” OR “3088-31-1” OR “68081-98-1” OR “68439-50-9” OR “68551-12-2” OR “68439-49-6” OR “9043-30-5” OR “26183-52-8” OR “9002-92-0” OR “9004-82-4” OR “9005-64-5” OR “68131-40-8” OR “68991-48-0” OR “37251-67-5” OR “5274-68-0” OR “864529-51-1” OR “84133-50-6” OR “68439-45-2” OR “68987-81-5” OR “9003-11-6” OR “61791-29-5” OR “9005-08-7” OR “61791-13-7” OR “166736-08-9” OR “3055-99-0” OR “66455-14-9” OR “68131-39-5” OR “68213-23-0” OR “68951-67-7” OR “66455-15-0” OR “61791-26-2” OR “9004-95-9” OR “9005-00-9” OR “61827-42-7” OR “68081-91-4” OR “68585-40-0” OR “68815-56-5” OR “61788-85-0” OR “3055-97-8” OR “120313-48-6” OR “68439-46-3” OR “69227-22-1” OR “68002-97-1”)

C.2.3.2 General Population, Consumer, and Environmental Exposure Search Strings

Population:
TIAB: (“general population” OR “bystanders” OR “near-facility” OR “industrial facilit*” OR “commercial facilit*” OR “employee” OR “employees” OR “worker*” OR “manufacturer” OR “near-disposal” OR “near surface disposal” OR “child*” OR “teenage*” OR “susceptible population” OR “immunocompromised” OR “preschool” OR “senior*” OR “older adults” OR “elderly” OR “pregnant women” OR “preexisting condition*” OR “lactating women” OR “childbearing” OR “prenatal” OR “infant*” OR “adolescen*”)

Landfills:

TIAB: (“dioxane” AND (“landfill” OR “leach*” OR “incineration” OR “wastewater” OR “GAC” OR “granular activated carbon” OR “reverse osmosis” OR “waste site” OR “land disposal” OR “waste disposal” OR “landfill leach*”))

Indoor Air and Water:
TIAB: (“dioxane” AND (“inhal*” OR “tap water” OR “water well” OR “indoor air” OR “surface water” OR “groundwater” OR “outdoor air” OR “ambient air” OR “drinking water” OR (“biomonitoring” OR “monitoring” AND (“air” OR “water”)) OR “drinking” OR “aquifer” OR “leach*” OR “municipal water”)) NOT (“spill”)

Consumer and Industrial Use:
TIAB: (“1,4-dioxane” AND (“ingest*” OR “swallow*” OR “showering” OR “bathing” OR “swimming” OR “wading” OR “inhal*” OR “paint*” OR “industrial manufactur*” OR “residential construction” OR “commercial construction” OR “cleaning” OR “dishwasher” OR “printing” OR “food supplement*” OR “packaging” OR “breast milk” OR “human milk” OR “intake rates” OR “launder*” OR “surface cleaner” OR “automotive”))

Concentration and Dose:
TIAB: (“reference concentration” OR “RfC” OR “NOAEL” OR “LOAEL” OR “benchmark concentration” OR “reference dose” OR “RfD” OR “chronic oral” OR “chronic inhalation” OR “oral slope factor” OR “soil screening level” OR “PEL” OR “permissible exposure limit” OR “weighted average” OR “weight fraction” OR “emission rate*” OR “inhalation unit risk” OR “IUR” OR “dose-response” OR “reverse dosimetry” OR “biomonitoring” OR “media concentration*” OR (“estimate*” AND (“acute” OR “subchronic” OR “chronic”)) OR “single-dose” OR “repeated-dose” OR “daily intake”)

C.3 Data Screening

Sections 4.2.5 and 4.3.2 of the 2021 Draft Systematic Review Protocol describe how title and abstract (TIAB) and full-text (FT) screening, respectively, are conducted to identify references that may contain relevant information for use in risk evaluations under TSCA using discipline-specific screening criteria (defined below in Sections C.3.1.1 and C.3.2.1 (U.S. EPA, 2021a). Specifically, TIAB screening efforts may be conducted using the specialized web-based software programs DistillerSR[17] and SWIFT-Active-Screener[18]; however, for this draft supplement, EPA used SWIFT-Active-Screener exclusively. Additional details on how SWIFT Active-Screener utilizes a machine-learning algorithm to automatically compute which unscreened documents are most likely to be relevant[19] are available in Section 4.2.5 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). When TIAB screening is completed, references that meet screening criteria (*i.e.*, PECO/RESO/PESO statements) then undergo FT screening if the full reference is able to be retrieved and generated into a Portable Document Format (PDF).

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[17] As noted on the DistillerSR web page, this systematic review software “automates the management of literature collection, triage, and assessment using AI and intelligent workflows...to produce transparent, audit ready, and compliant literature reviews.” EPA uses DistillerSR to manage the workflow related to screening and evaluating references; the literature search is conducted external to DistillerSR.

[18] SWIFT-Active Screener is another systematic review software that EPA is adopting in the TSCA systematic review process. From Sciome’s SWIFT-Active Screener web page: “As screening proceeds, reviewers include or exclude articles while an underlying statistical model in SWIFT-Active Screener automatically computes which of the remaining unscreened documents are most likely to be relevant. This ‘Active Learning’ model is continuously updated during screening, improving its performance with each reference reviewed. Meanwhile, a separate statistical model estimates the number of relevant articles remaining in the unscreened document list.”

[19] Description comes from the SWIFT-Active Screener web page.

Literature inventory trees were introduced in the scoping process for the risk evaluations that began systematic review in 2019 in response to comments received from the SACC and public to better illustrate how references underwent various systematic review steps (*e.g.*, TIAB and FT screening). As explained in various final scope documents (*e.g.*, Section 2.1.2 in the *Final Scope of the Risk Evaluation for 4,4'-(1-Methylethylidene)bis[2, 6-dibromophenol]* (U.S. EPA, 2020b)), literature inventory trees demonstrate how references that meet screening criteria progress to the next systematic review step. EPA used the Health Assessment Workplace Collaborative (HAWC) tool to develop web-based literature inventory trees to enhance the transparency of the decisions resulting from the screening processes. Updates made to the available literature considered for the draft supplement that are made between publishing the draft and final Supplement to the Risk Evaluation for 1,4-Dioxane (*e.g.*, additional references may be provided to the EPA through public comment) will be reflected in HAWC (see also hyperlinks to HAWC in the figure captions below for each respective literature inventory tree).

The web-based literature inventory trees in HAWC also allow users to directly access the references in the HERO database (more details available in Section 1 of the 2021 Draft Systematic Review Protocol) by selecting appropriate nodes, which indicate whether a reference has met screening criteria at different screening steps and/or types of content that may be discerned at that respective systematic review step (U.S. EPA, 2021a). Furthermore, as mentioned in the various final scope documents, the sum of the numbers for the various nodes in the literature inventory trees may be smaller or larger than the preceding node because some studies may have unclear relevance or be relevant for many categories of information. The screening process for each discipline varies and the nodes in the literature tree indicate the screening decisions determined for each reference and whether specific content could be determined; if no references had a specific screening decision and/or contained specific content relevant for a respective discipline, a node will not be present on the literature tree to depict this.

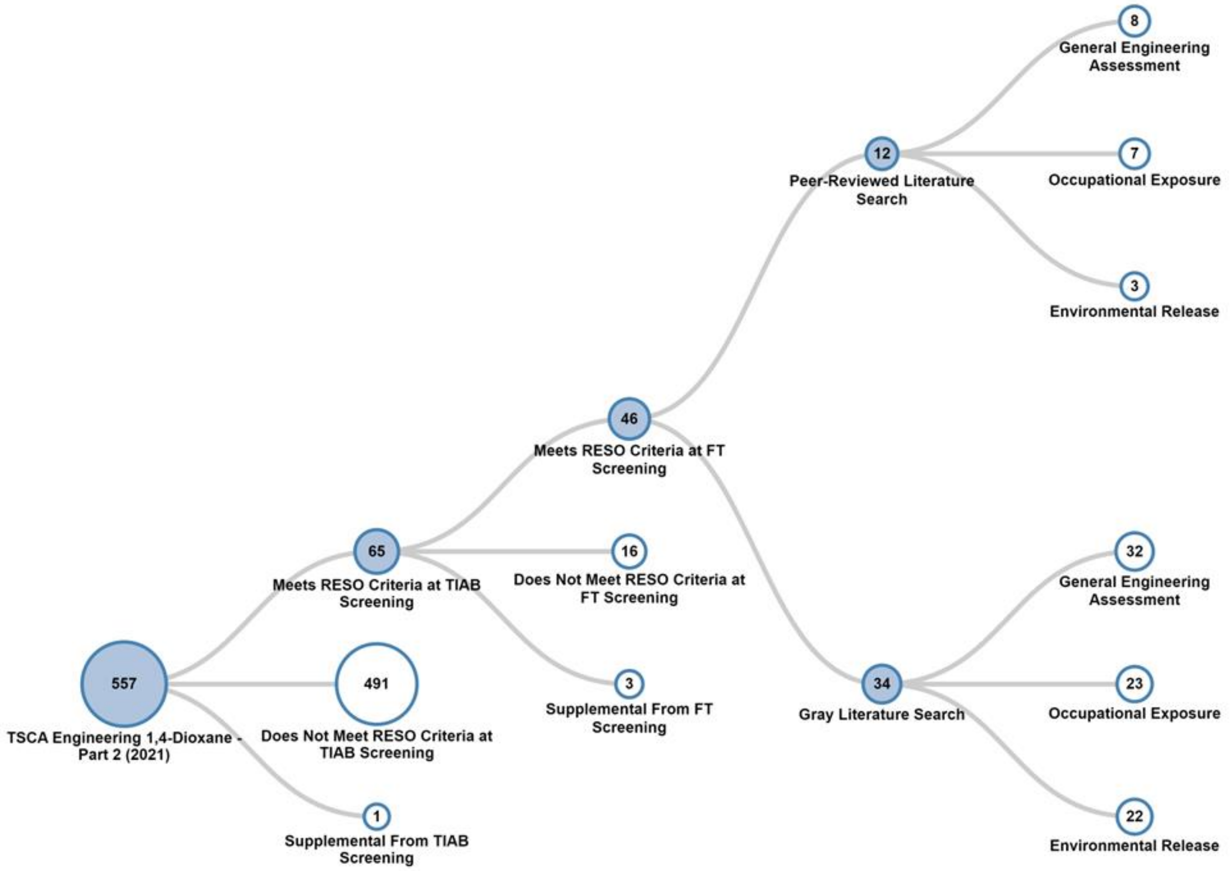
In the literature inventory trees below, which depict systematic review search results used to evaluate the new exposure pathways in this draft supplement, some references were unattainable for FT screening. The “PDF not available” node refers to references or sources of information for which EPA was unable to obtain the entire reference or source of data/information but were identified in the literature search because of the availability of the title and abstract. For the references considered to evaluate environmental release and occupational exposure, all references that passed screening criteria were found and underwent FT screening. For the references considered to evaluate general population, consumer, and environmental exposure, one PDF could not be obtained through interlibrary loan or through other channels available to EPA to obtain reprints of published sources.

As mentioned in Section C.1, although all information contained in references that enter systematic review are considered for use in the risk evaluation, the references that satisfy the screening criteria are generally deemed to contain the most relevant and useful information for characterizing the uses, exposure, and hazard of a chemical of interest and are generally utilized in the risk evaluation (and can be used later on to identify further data needs). On the other hand, data or information sources that do not satisfy the screening criteria outlined below may undergo data quality evaluation and extraction should a data need arise for the risk evaluation.

C.3.1 Environmental Release and Occupational Exposure

During data screening, EPA followed the process described in Appendix H, Section H-3 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) to conduct TIAB and FT screening for 1,4-dioxane literature search results guided by the RESO statement. RESO stands for **R**eceptors, **E**xposure, **S**etting or Scenario, and **O**utcomes. Data or information sources that comply with the screening criteria specified in the RESO statement then undergo data quality evaluation and extraction. Figure_Apx C-2 presents the number of references that report general engineering data, environmental release, and occupational exposure data that passed RESO screening criteria at TIAB and FT screening.

C.3.1.1 Environmental Release and Occupational Exposure Literature Tree



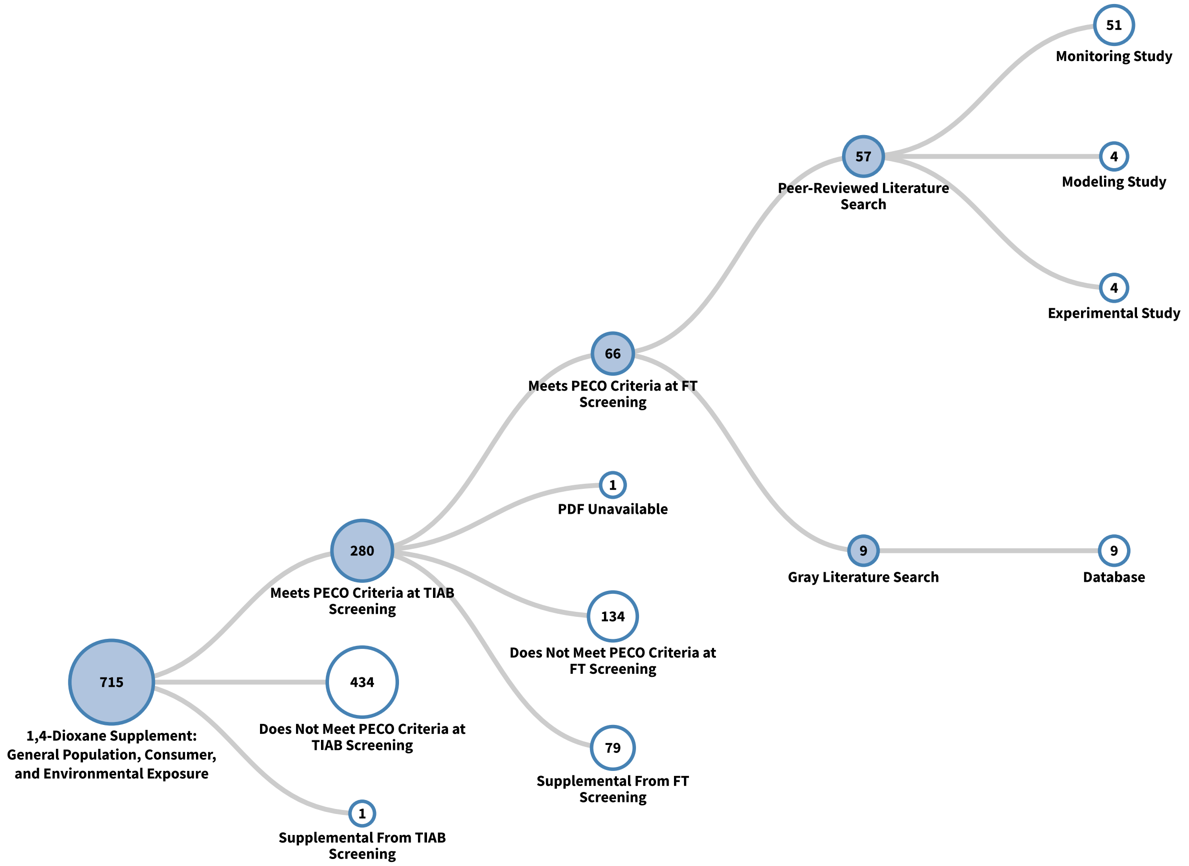
Figure_Apx C-2. Literature Inventory Tree – Environmental Releases and Occupational Exposure Search Results for 1,4-Dioxane

View the interactive literature inventory tree in HAWC. Data in this figure represent all references obtained from the publicly available databases and gray literature references searches that were included in systematic review as of March 30, 2023. Additional data may be added to the interactive version as they become available.

C.3.2 General Population, Consumer, and Environmental Exposure

The TIAB and FT screening process was consistent with what EPA previously outlined in Sections 4.2.5 and 4.3.2 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). PECO stands for **P**opulation, **E**xposure, **C**omparator or Scenario, and **O**utcomes for Exposure Concentration or Dose. The PECO statement, as depicted in Appendix H.5.14 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), was refined to better identify references that may contain information relevant for this draft supplement. Specifically, data that are relevant for characterizing exposure to 1,4-dioxane in food, including biota that humans consume, was not evaluated and extracted because 1,4-dioxane is not expected to bioaccumulate in organisms likely to be consumed by humans. During TIAB screening, if it is unclear if a reference will meet the PECO screening criteria without having the full reference to review, or if a reference is determined to meet the PECO screening criteria, that reference will advance to FT screening. Studies containing potentially relevant supplemental material were also tracked and categorized during the literature screening process. Relevant supplemental material may be reviewed, evaluated for data quality, and incorporated into risk evaluations, as needed. For example, references were considered supplemental if they contained data from countries outside of North America on 1,4-dioxane levels associated with landfills because different countries have very different waste management policies (including requirements for landfills), and local hydrogeology in other regions may not be relevant to sites in the United States. Figure_Apx C-3 presents the number of references that report general population, consumer, and environmental exposure data that passed PECO screening criteria at TIAB and FT screening.

C.3.2.1 General Population, Consumer, and Environmental Exposure Literature Tree



Figure_Apx C-3. Literature Inventory Tree – General Population, Consumer, and Environmental Exposure Search Results for 1,4-Dioxane

View the interactive literature inventory tree in HAWC. Data in this figure represent all references obtained from the publicly available databases and gray literature references searches that were included in systematic review as of November 28, 2022. Additional data may be added to the interactive version as they become available.

C.4 Data Evaluation and Data Extraction

Data evaluation and extraction for this draft supplement are as described in Sections 5 and 6 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Data evaluation is the systematic review step in which EPA assesses quality of the individual data sources using the evaluation strategies and criteria for each topic area (*e.g.*, physical and chemical property data, fate and transport data, occupational exposure and environmental release data, general population, consumer, and environmental exposure data). The evaluation method uses a structured framework with predefined criteria for each type of data/information source. The goal of the method used by EPA is to provide transparency, consistency, and as much objectivity as possible to the evaluation process along with meeting the TSCA science standards. Data extraction is the systematic review step in which EPA identifies quantitative and qualitative data/information from data/information sources that meet screening criteria and extract the data/information using structured forms or templates.

As explained above in Section C.1, terminology updates were made regarding the description of both metric and overall data/information source quality determinations from what was originally described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Specifically, metric and overall data/information source quality determination terminology have been updated to "Metric Rating" and "Overall Quality Determination", respectively. For additional clarifications regarding these updates, please see Table_Apx C-1.

Although references that meet screening criteria following FT screening will generally proceed to data quality evaluation and extraction steps, one clarification to the procedures outlined in Section 6 of the 2021 Draft Systematic Review Protocol is that in situations where EPA is unable to extract data/information from sources that meet screening criteria (*e.g.*, formatting prohibits accurate extraction), such sources may not have extracted data to present in the risk evaluation or the respective supplemental documents. Systematic Review Supplemental Files for the draft supplement contain results from the data quality evaluation and extraction systematic review steps. Also, the template used to display the data may be modified from those that were provided in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) because the purpose of these supplemental documents is to accommodate the data needs for each respective risk evaluation. The following sections provide specific information about the data quality and extraction process followed to address the exposure pathways in scope for this draft supplement and any clarifications or updates regarding these systematic review steps as described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a).

C.4.1 Environmental Release and Occupational Exposure

As described in the 2021 Draft Systematic Review Protocol, evaluation and extraction followed the steps outlined in Sections 5, 6, and 6.2 (U.S. EPA, 2021a). The data extraction and data quality results are summarized in Table_Apx E-8 for air, Table_Apx E-4 for water, Table_Apx E-6 for land, and Table_Apx F-33 for occupational exposure. The 1,4-Dioxane Supplement to the Risk Evaluation Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure provides the results from the data extraction and quality evaluation, including metric rating and the overall quality determination for each data source (U.S. EPA, 2023t).

C.4.2 General Population, Consumer, and Environmental Exposure

As described in the 2021 Draft Systematic Review Protocol, evaluation and extraction generally followed the steps outlined in Section 5 and 6 (U.S. EPA, 2021a). However, a few updates were made to the data quality evaluation metrics for a few evidence streams since the metrics were published in the 2021 Draft Systematic Review Protocol. Most of the changes were editorial or minor clarifications, including the standardization of some metrics that apply to multiple evidence streams, where appropriate. For example, in the Quality Assurance/Quality Control (QA/QC) metric for evaluating monitoring and experimental evidence streams, the acronym QA/QC was defined and replaced all references to quality assurance and quality control when occurring separately or together, and the term QA/QC techniques was changed to QA/QC measures, which already appeared in the metrics.

A few metrics applicable to multiple evidence streams were modified slightly to better fit some of the unique situations that frequently arise for a certain type of evidence stream (*e.g.*, databases). For example, some metrics were updated to clarify the intent of the metric and better account for variation in types of evidence included in one grouping (*e.g.*, experiments involving chamber studies vs. product concentration assessments). The domains did not change; however, see below for the changes and updates made to the data evaluation metrics for the respective evidence types (*i.e.*, monitoring, experimental studies and databases) as presented in Sections C.4.2.2, C.4.2.3 and C.4.2.4. No changes were made to the data evaluation metrics for modeling data, as described in Appendix N Section N.6.2 in the 2021 Draft Systematic Review Protocol. The 1,4-Dioxane Supplement to the Risk Evaluation Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure provides details of the data quality evaluation results, including metric rating and the overall quality determination for each data source (U.S. EPA, 2023r).

Data extraction is the process in which quantitative and qualitative data/information are identified from each relevant data/information source and extracted using structured forms or templates. Data extraction was conducted as described in Section 6 of the 2021 Draft Systematic Review Protocol for all evidence streams relevant for this supplement. However, with respect to information stored within databases, EPA does not conduct a separate data extraction because the data are more accessible and have additional context in the original database format. Both the date and data present in the database when the database underwent FT screening are available in the HERO database (HERO IDs: 10365582, 10365609, 10365665, 10365667, 10365696, 10365698, 10368680, 10410586, and 10501014). If a reference or data/information source (*e.g.*, a peer-reviewed reference) presents data from a database that did not undergo systematic review (*e.g.*, a foreign database that is not publicly accessible), the data would be extracted from the reference or data/information source to the extent possible; this did not apply to references or sources of data or information that underwent systematic review for this supplement.

As mentioned above in Section C.4, references may not undergo data extraction, regardless of data quality rating, if they contain no extractable data points (*e.g.*, values are contained in a non-digitizable figure or are representative of unspecified media or treatment processes). This constitutes an update to Section 6 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Extraction forms and templates are tailored to fit the data extraction needs for each risk evaluation.

The types of fields extracted vary by evidence stream and generally followed Section 6.3 of the 2021 Draft Systematic Review Protocol with regard to the data characteristics captured (U.S. EPA, 2021a). Examples of types of data extracted and the extraction formats for the four evidence streams identified through systematic review to evaluate environmental, general population, and consumer exposure data are listed in the extraction tables provided in the 1,4-Dioxane Supplement to the Risk Evaluation Data Extraction Information for General Population, Consumer, and Environmental Exposure (U.S. EPA, 2023h).

C.4.2.1 Data Quality Evaluation Metric Updates

Shown below are the data evaluation metrics for three evidence streams, presenting which data evaluation metrics changed since the publication of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). For evidence streams not listed below, there were no changes to the data evaluation metrics since the 2021 Draft Systematic Review Protocol was published. Other data quality criteria for studies on consumer, general population, and environmental exposure appear in Appendix N of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). For example, the criteria for modeling studies appear in Table_Apx N9. Data quality criteria for other types of studies (*e.g.*, environmental release and occupational exposure assessment) are published in other appendices to the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a).

For the below tables in Sections C.4.2.2, C.4.2.3 and C.4.2.4, in order to make it easier for the reader to see what the changes were to the data evaluation metrics, the following convention is used: text inserted is underlined, and text deleted is in strikeout.

C.4.2.2 Data Evaluation Criteria for Monitoring Data, as Revised

Table_Apx C-2. Evaluation Criteria for Sources of Monitoring Data

Quality Rating	**Description**
Domain 1. Reliability	
Metric 1. Sampling methodology	
High	Samples were collected according to publicly available SOPs that are scientifically sound and widely accepted (*i.e.*, from a source generally ~~using~~ known to use sound methods and/or approaches) for the chemical and media of interest. Example SOPs include U.S. Geological Survey (USGS') "National Field Manual for the Collection of Water-Quality Data," EPA's "Ambient Air Sampling" (SESDPROC-303-R5), etc. OR The sampling protocol used was not a publicly available SOP from a source generally known to use ~~using~~ sound methods and/or approaches, but the sampling methodology is clear, appropriate (*i.e.*, scientifically sound), and similar to widely accepted protocols for the chemical and media of interest. All pertinent sampling information is provided in the data source or companion source. Examples include: • Sampling equipment • Sampling procedures/regime • Sample storage conditions/duration • Performance/calibration of sampler • Study site characteristics • Matrix characteristics
Medium	Sampling methodology is discussed in the data source or companion source and is generally appropriate (*i.e.*, scientifically sound) for the chemical and media of interest; however, **one or more pieces of sampling information is not described.** The missing information is unlikely to have a substantial impact on results. OR Standards, methods, protocols, or test guidelines may not be widely accepted, but a successful validation study for the new/unconventional procedure was conducted prior to the sampling event and is consistent with sound scientific theory and/or accepted approaches. Or a review of information indicates the methodology is acceptable and differences in methods are not expected to lead to lower quality data.
Low	Sampling methodology is only briefly discussed; therefore, **most sampling information is missing** and likely to have a substantial impact on results. AND/OR The sampling methodology **does not represent best sampling methods, protocols, or guidelines** for the chemical and media of interest (*e.g.*, outdated [but still valid] sampling equipment or procedures, long storage durations). AND/OR There are **some inconsistencies** in the reporting of sampling information (*e.g.*, differences between text and tables in data source, differences between standard method and actual procedures reported to have been used, etc.) that led to a low confidence in the sampling methodology used.

Critically Deficient	The sampling methodology is not discussed in the data source or companion source. AND/OR Sampling methodology is not scientifically sound or is not consistent with widely accepted methods/approaches for the chemical and media being analyzed (*e.g.*, inappropriate sampling equipment, improper storage conditions). AND/OR There are **numerous inconsistencies** in the reporting of sampling information, resulting in high uncertainty in the sampling methods used.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 2. Analytical methodology	
High	Samples were analyzed according to publicly available analytical methods that are scientifically sound and widely accepted (*i.e.*, from a source generally ~~using~~ known to use sound methods and/or approaches) and are appropriate for the chemical and media of interest. Examples include EPA SW-846 Methods, NIOSH Manual of Analytical Methods 5th Edition, etc. OR The analytical method used was not a publicly available method from a source generally ~~using~~ known to use sound methods and/or approaches, but the methodology is clear and appropriate (*i.e.*, scientifically sound) and similar to widely accepted protocols for the chemical and media of interest. All pertinent sampling information is provided in the data source or companion source. Examples include: • Extraction method • Analytical instrumentation (required) • Instrument calibration • Limit of quantitation (LOQ), LOD, detection limits, and/or reporting limits • Recovery samples • Biomarker used (if applicable) • Matrix-adjustment method (*i.e.*, creatinine, lipid, moisture)
Medium	Analytical methodology is discussed in detail and is clear and appropriate (*i.e.*, scientifically sound) for the chemical and media of interest; however, **one or more pieces of analytical information is not described**. The missing information is unlikely to have a substantial impact on results. AND/OR The analytical **method may not be standard/widely accepted, but a method validation study was conducted** prior to sample analysis and is expected to be consistent with sound scientific theory and/or accepted approaches. AND/OR Samples were collected at a site and immediately analyzed using an on-site mobile laboratory, rather than shipped to a stationary laboratory.
Low	Analytical methodology is only briefly discussed. Analytical instrumentation is provided and consistent with accepted analytical instrumentation/methods. However, **most analytical information is missing** and likely to have a substantial impact on results. AND/OR Analytical method **is not s**tandard/widely accepted, and method validation is limited or not available. AND/OR Samples were analyzed using field screening techniques. AND/OR

	LOQ, LOD, detection limits, and/or reporting limits not reported. AND/OR There are **some inconsistencies or possible errors** in the reporting of analytical information (*e.g.*, differences between text and tables in data source, differences between standard method and actual procedures reported to have been used, etc.) which leads to a lower confidence in the method used.
Critically Deficient	Analytical methodology is not described, **including analytical instrumentation** (*i.e.*, HPLC, GC). AND/OR Analytical methodology is not scientifically appropriate for the chemical and media being analyzed (*e.g.*, method not sensitive enough, not specific to the chemical, out of date). AND/OR There are numerous inconsistencies in the reporting of analytical information, resulting in high uncertainty in the analytical methods used.
Not Rated/ Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 3. Selection of biomarker of exposure	
High	Biomarker in a specified matrix is known to have an accurate and precise quantitative relationship with external exposure, internal dose, or target dose (*e.g.*, previous studies (or the current study) have indicated the biomarker of interest reflects external exposures). AND Biomarker (parent chemical or metabolite) is derived from exposure to the chemical of interest.
Medium	Biomarker in a specified matrix has accurate and precise quantitative relationship with external exposure, internal dose, or target dose. AND Biomarker is derived from multiple parent chemicals, not only the chemical of interest, **but** there is a stated method to apportion the estimate to only the chemical of interest
Low	Biomarker in a specified matrix has accurate and precise quantitative relationship with external exposure, internal dose, or target dose. AND Biomarker is derived from multiple parent chemicals, not only the chemical of interest, **and** there is NOT an accurate method to apportion the estimate to only the chemical of interest. OR Biomarker in a specified matrix is a poor surrogate (low accuracy and precision) for exposure/dose.
Critically Deficient	Not applicable. A study will not be deemed critically deficient based on the use of biomarker of exposure.
Not rated/ applicable	Metric is not applicable to the data source.
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Domain 2. Representative	
Metric 4. Geographic area	

High	Geographic location(s) is reported, discussed, or referenced.
Medium	Not applicable. This metric is dichotomous (*i.e.*, high vs. critically deficient).
Low	Not applicable. This metric is dichotomous (*i.e.*, high vs. critically deficient).
Critically Deficient	Geographic location is not reported, discussed, or referenced.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 5. Temporality	
High	Timing of sample collection for monitoring data is consistent with current or recent exposures **(within 5 years)** may be expected.
Medium	Timing of sample collection for monitoring data **is less consistent** with current or recent exposures **(>5 to 15 years)** may be expected.
Low	Timing of sample collection for monitoring data is not consistent with when current exposures **(>15 years old)** may be expected and likely to have a substantial impact on results.
Critically Deficient	Timing of sample collection for monitoring data is **not reported, discussed, or referenced.**
Not Rated/ Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 6. Spatial and temporal variability	
High	Sampling approach accurately captures variability of environmental contamination in population/scenario/media of interest based on the heterogeneity/homogeneity and dynamic/static state of the environmental system. For example: • **Large sample size** (*i.e.*, ≥10 or more samples for a single scenario). • Use of replicate samples. • Use of systematic or continuous monitoring methods. • Sampling over a sufficient period of time to characterize trends. • For urine, 24-hour samples are collected (vs. first morning voids or spot). • For biomonitoring studies, the timing of sample collected is appropriate based on chemical properties (*e.g.*, half-life), the pharmacokinetics of the chemical (*e.g.*, rate of uptake and elimination), and when the exposure event occurred.
Medium	Sampling approach likely captures variability of environmental contamination in population/scenario/media of interest based on the heterogeneity/homogeneity and dynamic/static state of the environmental system. Some uncertainty may exist, but it is unlikely to have a substantial impact on results. For example: • **Moderate sample size** (*i.e.*, 5–10 samples for a single scenario), or • Use of judgmental (non-statistical) sampling approach, or • No replicate samples. • For urine, first morning voids or pooled spot samples.
Low	Sampling approach poorly captures variability of environmental contamination in population/scenario/media of interest. For example:

	• **Small sample size** (*i.e.*, <5 samples), or • Use of haphazard sampling approach, or • No replicate samples, or • Grab or spot samples in single space or time, or • Random sampling that does not include all periods of time or locations, or • For urine, un-pooled spot samples.
Critically Deficient	Sample size is not reported. **Single sample** collected per data set. For biomonitoring studies, the timing of sample collected is not appropriate based on chemical properties (*e.g.*, half-life), the pharmacokinetics of the chemical (*e.g.*, rate of uptake and elimination), and when the exposure event occurred.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 7. Exposure scenario	
High	The data closely represent relevant exposure scenario (*i.e.*, the population/scenario/media of interest). Examples include: • Amount and type of chemical/product used • Source of exposure • Method of application or by-stander exposure • Use of exposure controls • Microenvironment (location, time, climate)
Medium	The data likely represent the relevant exposure scenario (*i.e.*, population/scenario/media of interest). **One or more key pieces of information may not be described** but the deficiencies are unlikely to have a substantial impact on the characterization of the exposure scenario. AND/OR If surrogate data, activities seem similar to the activities within scope.
Low	The data lack multiple key pieces of information, and the deficiencies are likely to have a substantial impact on the characterization of the exposure scenario. AND/OR There are **some inconsistencies or possible errors** in the reporting of scenario information (*e.g.*, differences between text and tables in data source, differences between standard method and actual procedures reported to have been used, etc.) which leads to a lower confidence in the scenario assessed. AND/OR If surrogate data, activities have lesser similarity but are still potentially applicable to the activities within scope.
Critically Deficient	If reported, the exposure scenario discussed in the monitored study does not represent the exposure scenario of interest for the chemical.
Not rated/ Not applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Domain 3. Accessibility/clarity	
Metric 8. Reporting of results	

High	Supplementary or raw data (*i.e.*, individual data points) are reported, allowing summary statistics to be calculated or reproduced. AND • Description of data set summarized (*i.e.*, location, population, dates, etc.) • Range of concentrations or percentiles • Number of samples in data set • Frequency of detection • Measure of variation (coefficient of variation [CV], standard deviation) • Measure of central tendency (mean, geometric mean, median) • Test for outliers (if applicable) Summary statistics are detailed and complete. Example parameters include: AND Both adjusted and unadjusted results are provided (*i.e.*, correction for void completeness in urine biomonitoring, whole-volume or lipid adjusted for blood biomonitoring, wet or dry weight for environmental tissue samples or soil samples) [only if applicable].
Medium	Supplementary or raw data (*i.e.*, individual data points) are not reported, and therefore summary statistics cannot be reproduced. AND/OR Summary statistics are reported but are missing one or more parameters (see description for high). AND/OR Only adjusted or unadjusted results are provided, but not both [only if applicable].
Low	Supplementary data are not provided, and summary statistics are missing most parameters (see description for high). AND/OR There are some inconsistencies or errors in the results reported, resulting in low confidence in the results reported (*e.g.*, differences between text and tables in data source, less appropriate statistical methods).
Critically Deficient	There are numerous inconsistencies or errors in the calculation and/or reporting of results, resulting in highly uncertain reported results.
Not Rated/ Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 9. Quality assurance	
High	The study quality assurance/quality control (QA/QC) measures and all pertinent QA/QC information is provided in the data source or companion source. Examples include: • Field, laboratory, and/or storage recoveries • Field and laboratory control samples • Baseline (pre-exposure) samples • Biomarker stability • Completeness of sample (*i.e.*, creatinine, specific gravity, osmolality for urine samples) AND No QA/QC issues were identified, or any identified issues were minor and adequately addressed (*i.e.*, correction for low recoveries, correction for completeness).
Medium	The study applied and documented QA/QC measures; however, **one or more pieces of QA/QC information is not described.** Missing information is unlikely to have a substantial impact on results.

	AND No QA/QC issues were identified, or any identified issues were minor and addressed (*i.e.*, correction for low recoveries, correction for completeness).
Low	QA/QC measures and results were not directly discussed but <u>are</u> implied through the study's use of standard field and laboratory protocols. AND/OR Deficiencies were noted in QA/QC control measures that are likely to have a substantial impact on results. AND/OR There are some inconsistencies in the QA/QC measures reported, resulting in low confidence in the QA/QC measures taken and results (*e.g.*, differences between text and tables in data source).
Critically Deficient	QA/QC issues have been identified which significantly interfere with the overall reliability of the study.
Not Rated/ Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
<u>Domain 4</u>. Variability and uncertainty	
<u>Metric 10</u>. Variability and uncertainty	
High	The study characterizes variability in the population/media studied. AND Key uncertainties, limitations, and data gaps have been identified. AND The uncertainties are minimal and have been characterized.
Medium	The study has limited characterization of variability in the population/media studied. AND/OR The study has limited discussion of key uncertainties, limitations, and data gaps. AND/OR Multiple uncertainties have been identified but are unlikely to have a substantial impact on results.
Low	The characterization of variability is absent. AND/OR Key uncertainties, limitations, and data gaps are not discussed. AND/OR Uncertainties identified may have a substantial impact on the exposure the exposure assessment
Critically Deficient	Estimates are highly uncertain based on characterization of variability and uncertainty.
Not Rated/ Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*

C.4.2.3 Data Evaluation Criteria for Experimental Data, as Revised

Table_Apx C-3. Evaluation Criteria for Sources of Experimental Data

Data Quality Rating	**Metric Description**
Domain 1. Reliability	
Metric 1. Sampling Methodology and Conditions	
High	Samples were collected according to publicly available SOPs, methods, protocols, or test guidelines that are scientifically sound and widely accepted from a source generally known to use sound methods and/or approaches such as EPA, NIST, American Society for Testing and Materials, ISO, and ACGIH. OR The sampling protocol used was not a publicly available SOP from a source generally known to use sound methods and/or approaches, but the sampling methodology is clear, appropriate (*i.e.*, scientifically sound), and similar to widely accepted protocols for the chemical and media of interest. All pertinent sampling information is provided in the data source or companion source. Examples include: • Sampling conditions (*e.g.*, temperature, humidity) • Sampling equipment and procedures • Sample storage conditions/duration • Performance/calibration of sampler
Medium	Sampling methodology is discussed in the data source or companion source and is generally appropriate (*i.e.*, scientifically sound) for the chemical and media of interest, however, one or more pieces of sampling information is not described. The missing information is unlikely to have a substantial impact on results. OR Standards, methods, protocols, or test guidelines may not be widely accepted, but a successful validation study for the new/unconventional procedure was conducted prior to the sampling event and is consistent with sound scientific theory and/or accepted approaches.
Low	Sampling methodology is only briefly discussed. Therefore, most sampling information is missing and likely to have a substantial impact on results. AND/OR The sampling methodology does not represent best sampling methods, protocols, or guidelines for the chemical and media of interest (*e.g.*, outdated (but still valid) sampling equipment or procedures, long storage durations). AND/OR There are some inconsistencies in the reporting of sampling information (*e.g.*, differences between text and tables in data source, differences between standard method and actual procedures reported to have been used, etc.) which lead to a low confidence in the sampling methodology used.
Critically Deficient	The sampling methodology is not discussed in the data source or companion source. AND/OR Sampling methodology is not scientifically sound or is not consistent with widely accepted methods/approaches for the chemical and media being analyzed (*e.g.*, inappropriate sampling equipment, improper storage conditions). AND/OR There are numerous inconsistencies in the reporting of sampling information, resulting in high uncertainty in the sampling methods used.

Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
High	Samples were analyzed according to publicly available analytical methods that are scientifically sound and widely accepted (*i.e.*, from a source generally using sound methods and/or approaches) and are appropriate for the chemical and media of interest. Examples include EPA SW-846 Methods, NIOSH Manual of Analytical Methods 5th Edition, etc. OR The analytical method used was not a publicly available method from a source generally known to use sound methods and/or approaches, but the methodology is clear and appropriate (*i.e.*, scientifically sound) and similar to widely accepted protocols for the chemical and media of interest. All pertinent analytical ~~sampling~~ information is provided in the data source or companion source. Examples include: • Extraction method • Analytical instrumentation (required) • Instrument calibration • LOQ, LOD, detection limits, and/or reporting limits • Recovery samples • Biomarker used (if applicable) • Matrix-adjustment method (*B* creatinine, lipid, moisture)
Medium	Analytical methodology is discussed in detail and is clear and appropriate (*i.e.*, scientifically sound) for the chemical and media of interest; however, one or more pieces of analytical information is not described. The missing information is unlikely to have a substantial impact on results. AND/OR The analytical method may not be standard/widely accepted, but a method validation study was conducted prior to sample analysis and is expected to be consistent with sound scientific theory and/or accepted approaches. AND/OR Samples were collected at a site and immediately analyzed using an on-site mobile laboratory, rather than shipped to a stationary laboratory.
Low	Analytical methodology is only briefly discussed. Analytical instrumentation is provided and consistent with accepted analytical instrumentation/methods. However, most analytical information is missing and likely to have a substantial impact on results. AND/OR Analytical method is not standard/widely accepted, and method validation is limited or not available. AND/OR Samples were analyzed using field screening techniques. AND/OR LOQ, LOD, detection limits, and/or reporting limits not reported. AND/OR There are some inconsistencies or possible errors in the reporting of analytical information (*e.g.*, differences between text and tables in data source, differences between standard method and actual procedures reported to have been used, etc.) which leads to a lower confidence in the method used.
Critically Deficient	Analytical methodology is not described, including analytical instrumentation (*i.e.*, HPLC, GC). AND/OR

	Analytical methodology is not scientifically appropriate for the chemical and media being analyzed (*e.g.*, method not sensitive enough, not specific to the chemical, out of date). AND/OR There are numerous inconsistencies in the reporting of analytical information, resulting in high uncertainty in the analytical methods used.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
<u>Metric 3</u>. Selection of biomarker of exposure	
High	Biomarker in a specified matrix is known to have an accurate and precise quantitative relationship with external exposure, internal dose, or target dose (*e.g.*, previous studies (or the current study) have indicated the biomarker of interest reflects external exposures). AND Biomarker (parent chemical or metabolite) is derived from exposure to the chemical of interest.
Medium	Biomarker in a specified matrix has accurate and precise quantitative relationship with external exposure, internal dose, or target dose. AND Biomarker is derived from multiple parent chemicals, not only the chemical of interest, **but** there is a stated method to apportion the estimate to only the chemical of interest
Low	Biomarker in a specified matrix has accurate and precise quantitative relationship with external exposure, internal dose, or target dose. AND Biomarker is derived from multiple parent chemicals, not only the chemical of interest, **and** there is NOT a stated method to apportion the estimate to only the chemical of interest. OR Biomarker in a specified matrix is a poor surrogate (low accuracy and precision) for exposure/dose.
Critically Deficient	Not applicable. A study will not be deemed critically deficient based on the use of biomarker of exposure.
Not Rated/Not Applicable	Metric is not applicable to the data source.
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
<u>Metric 4</u>. Testing scenario	
High	Testing conditions closely represent relevant exposure scenarios (*i.e.*, population/scenario/media of interest). Examples include: • Amount and type of chemical/product used • Source of exposure/test substance • Method of application or by-stander exposure • Use of exposure controls • Microenvironment (location, time, climate, temperature, humidity, pressure, airflow) AND Testing conducted under a broad range of conditions for factors such as temperature, humidity, pressure, airflow, and chemical mass/weight fraction (if appropriate).

Medium	The data likely represent the relevant exposure scenario (*i.e.*, population/scenario/media of interest). One or more key pieces of information may not be described but the deficiencies are unlikely to have a substantial impact on the characterization of the exposure scenario. AND/OR If surrogate data, activities seem similar to the activities within scope.
Low	The data lack multiple key pieces of information and the deficiencies are likely to have a substantial impact on the characterization of the exposure scenario. AND/OR There are some inconsistencies or possible errors in the reporting of scenario information (*e.g.*, differences between text and tables in data source, differences between standard method and actual procedures reported to have been used, etc.) which leads to a lower confidence in the scenario assessed. AND/OR If surrogate data, activities have lesser similarity but are still potentially applicable to the activities within scope. AND/OR Testing conducted under a single set of conditions, except for experiments to determine a weight fraction or concentration in a product.
Critically Deficient	Testing conditions are not relevant to the exposure scenario of interest for the chemical.
Not Rated/Not Applicable	
Reviewer’s comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 5. Sample size and variability	
High	Sample size is reported and large enough (*i.e.*, ≥ 10 samples) to be reasonably assured that the samples represent the scenario of interest. AND Replicate tests performed and variability across tests is characterized (if appropriate).
Medium	Sample size is moderate (*i.e.*, 5 to <10 samples), thus the data are likely to represent the scenario of interest. AND Replicate tests performed and variability across tests is characterized (if appropriate).
Low	Sample size is small (*i.e.*, <5 samples for most types of experiments or 1 per product for experiments to determine a weight fraction or concentration in a product), thus the data are likely to poorly represent the scenario of interest. AND/OR Replicate tests were not performed.
Critically Deficient	Sample size is not reported. AND/OR Single sample collected per data set, except for experiments to determine a weight fraction or concentration in a product. AND/OR For biomonitoring studies, the timing of sample collected is not appropriate based on chemical properties (*e.g.*, half-life), the pharmacokinetics of the chemical (*e.g.*, rate of uptake and elimination), and when the exposure event occurred.
Not Rated/Not Applicable	

Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 6. Temporality	
High	Source(s) of tested items appears to be current (within 5 years).
Medium	Source(s) of tested items is less consistent with when current or recent exposures (>5 to 15 years) are expected.
Low	Source(s) of tested items is not consistent with when current or recent exposures (>15 years) are expected or is not identified.
Critically Deficient	Temporality of tested items is not reported, discussed, or referenced.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Domain 3. Accessibility/clarity	
Metric 7. Reporting of results	
High	Supplementary or raw data (*i.e.*, individual data points) are reported, allowing summary statistics to be calculated or reproduced. AND Summary statistics are detailed and complete. Example parameters include: • Description of data set summarized (*i.e.*, location, population, dates, etc.) • Range of concentrations or percentiles • Number of samples in data set • Frequency of detection • Measure of variation (CV, standard deviation) • Measure of central tendency (mean, geometric mean, median) • Test for outliers (if applicable) AND Both adjusted and unadjusted results are provided (*i.e.*, correction for void completeness in urine biomonitoring, whole-volume or lipid adjusted for blood biomonitoring) [only if applicable].
Medium	Supplementary or raw data (*i.e.*, individual data points) are not reported, and therefore summary statistics cannot be reproduced. AND/OR Summary statistics are reported but are missing one or more parameters (see description for high). AND/OR Only adjusted or unadjusted results are provided, but not both [only if applicable].
Low	Supplementary data are not provided, and summary statistics are missing most parameters (see description for high). AND/OR There are some inconsistencies or errors in the results reported, resulting in low confidence in the results reported (*e.g.*, differences between text and tables in data source, less appropriate statistical methods).
Critically Deficient	There are numerous inconsistencies or errors in the calculation and/or reporting of results, resulting in highly uncertain reported results.

Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 8. Quality assurance	
High	The study applied quality assurance/quality control (QA/QC) measures and all pertinent QA/QC information is provided in the data source or companion source. Examples include: • Laboratory, and/or storage recoveries. • Laboratory control samples. • Baseline (pre-exposure) samples. • Biomarker stability • Completeness of sample (*i.e.*, creatinine, specific gravity, osmolality for urine samples) AND No QA/QC issues were identified, or any identified issues were minor and adequately addressed (*i.e.*, correction for low recoveries, correction for completeness).
Medium	The study applied and documented QA/QC measures; however, one or more pieces of QA/QC information is not described. Missing information is unlikely to have a substantial impact on results. AND No QA/QC issues were identified, or any identified issues were minor and addressed (*i.e.*, correction for low recoveries, correction for completeness).
Low	QA/QC measures and results were not directly discussed but are implied through the study's use of standard field and laboratory protocols. AND/OR Deficiencies were noted in QA/QC measures that are likely to have a substantial impact on results. AND/OR There are some inconsistencies in the QA/QC measures reported, resulting in low confidence in the QA/QC measures taken and results (*e.g.*, differences between text and tables in data source).
Critically Deficient	QA/QC issues have been identified which significantly interfere with the overall reliability of the study.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Domain 4. Variability and uncertainty	
Metric 9. Variability and uncertainty	
High	The study characterizes variability in the population/media studied. AND Key uncertainties, limitations, and data gaps have been identified. AND The uncertainties are minimal and have been characterized.
Medium	The study has limited characterization of variability in the population/media studied. AND/OR

	The study has limited discussion of key uncertainties, limitations, and data gaps. AND/OR Multiple uncertainties have been identified but are unlikely to have a substantial impact on results.
Low	The characterization of variability is absent. AND/OR Key uncertainties, limitations, and data gaps are not discussed. AND/OR Uncertainties identified may have a substantial impact on ~~the exposure~~ the exposure assessment
Critically Deficient	Estimates are highly uncertain based on characterization of variability and uncertainty.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*

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C.4.2.4 Data Evaluation Criteria for Databases, as Revised

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6144 **Table_Apx C-4. Evaluation Criteria for Sources of Database Data**

Data Quality Rating	**Description**
Domain 1. Reliability	
Metric 1. Sampling methodology	
High	Widely accepted sampling methodologies (*i.e.*, from a source generally known to use ~~using~~ sound methods and/or approaches) were used to generate the data presented in the database. Example SOPs include USGS's "National Field Manual for the Collection of Water-Quality Data," EPA's "Ambient Air Sampling" (SESDPROC-303-R5), etc.
Medium	One or more pieces of sampling methodology information is not described, but missing information is unlikely to have a substantial impact on results. OR The sampling methodologies were consistent with sound scientific theory and/or accepted approaches based on the reported sampling information but may not have followed published procedures from a source generally known to use sound methods and/or approaches.
Low	The sampling methodology was not reported in data source or readily available companion data source.
Critically Deficient	The sampling methodologies used were not appropriate for the chemical/media of interest in the database (*e.g.*, inappropriate sampling equipment, improper storage conditions).
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 2. Analytical methodology	

High	Widely accepted analytical methodologies (*i.e.*, from a source generally using sound methods and/or approaches) were used to generate the data presented in the database. Example SOPs include EPA SW-846 Methods, NIOSH Manual of Analytical Methods 5th Edition, etc.
Medium	The analytical methodologies were consistent with sound scientific theory and/or accepted approaches based on the reported analytical information but may not have followed published procedures from a source generally known to use sound methods and/or approaches.
Low	The analytical methodology was not reported in data source or companion data source.
Critically Deficient	The analytical methodologies used were not appropriate for the chemical/media of interest in the database (*e.g.*, method not sensitive enough, not specific to the chemical, out of date).
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Domain 2. Representative	
Metric 3. Geographic area	
High	Geographic location(s) is reported, discussed, or referenced.
Medium	Not applicable. This metric is dichotomous (*i.e.*, high vs. critically deficient).
Low	Not applicable. This metric is dichotomous (*i.e.*, high vs. critically deficient).
Critically Deficient	Geographic location is not reported, discussed, or referenced.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 4. Temporal	
High	The data reflect current conditions (within 5 years) AND/OR Database contains robust historical data for spatial and temporal analyses (if applicable).
Medium	The data are less consistent with current or recent exposures (>5 to 15 years) AND/OR Database contains sufficient historical data for spatial and temporal analyses (if applicable).
Low	Data are not consistent with when current exposures (>15 years old) may be expected AND/OR Database does not contain enough historical data for spatial and temporal analyses (if applicable).
Critically Deficient	Timing of sample data is not reported, discussed, or referenced.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*

Metric 5. Exposure scenario	
High	The data closely represent relevant exposure scenario (*i.e.*, the population/scenario/media of interest). Examples include: • Amount and type of chemical/product used • Source of exposure • Method of application or by-stander exposure • Use of exposure controls • Microenvironment (location, time, climate)
Medium	The data likely represent the relevant exposure scenario (*i.e.*, population/scenario/media of interest). **One or more key pieces of information may not be described** but the deficiencies are unlikely to have a substantial impact on the characterization of the exposure scenario. AND/OR If surrogate data, activities seem similar to the activities within scope.
Low	The data lack multiple key pieces of information and the deficiencies are likely to have a substantial impact on the characterization of the exposure scenario. AND/OR There are **some inconsistencies or possible errors** in the reporting of scenario information (*e.g.*, differences between text and tables in data source, differences between standard method and actual procedures reported to have been used, etc.) which leads to a lower confidence in the scenario assessed. AND/OR If surrogate data, activities have lesser similarity but are still potentially applicable to the activities within scope.
Critically Deficient	If reported, the exposure scenario discussed in the monitored study does not represent the exposure scenario of interest for the chemical.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Domain 3. Accessibility/clarity	
Metric 6. Availability of database and supporting documents	
High	Database is widely accepted and/or from a source generally known to use sound methods and/or approaches (*e.g.*, raw data from NHANES, STORET).
Medium	The database may not be widely known or accepted (*e.g.*, state-maintained databases), but the database is adequately documented with most or all of the following information: • Within the database, metadata is present (sample identifiers, annotations, flags, units, matrix descriptions, etc.) and-data fields are generally clear and defined. • A user manual and other supporting documentation is available, or there is sufficient documentation in the data source or companion source. • Database quality assurance and data quality control measures are defined and/or a QA/QC protocol was followed.
Low	The database may not be widely known or accepted, and only limited database documentation is available (see the medium rating).
Critically Deficient	No information is provided on the database source or availability to the public.

Not Rated/ Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Metric 7. Reporting of results	
High	The database or information source reporting the analysis of the database data is well organized and understandable by the target audience. AND Summary statistics in the data source are detailed and complete. Example parameters include: • Description of data set summarized (*i.e.*, location, population, dates, etc.) • Range of concentrations or percentiles • Number of samples in data set • Frequency of detection • Measure of variation (CV, standard deviation) • Measure of central tendency (mean, geometric mean, median) • Test for outliers (if applicable)
Medium	The database or information source reporting the analysis of the database data is well organized and understandable by the target audience. AND/OR Summary statistics are missing one or more parameters (see description for high).
Low	The database or information source reporting the analysis of the database data is unclear or not well organized. AND/OR Summary statistics are missing most parameters (see description for high) AND/OR There are some inconsistencies or errors in the results reported, resulting in low confidence in the results reported (*e.g.*, differences between text and tables in data source, less appropriate statistical methods).
Critically Deficient	There are numerous inconsistencies or errors in the calculation and/or reporting of results, resulting in highly uncertain reported results. AND/OR The information source reporting the analysis of the database data is missing key sections or lacks enough organization and clarity to locate and extract necessary information.
Not Rated/Not Applicable	
Reviewer's Comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*
Domain 4. Variability and uncertainty	
Metric 8. Variability and uncertainty	
High	Variability, key uncertainties, limitations, and/or data gaps have been identified. AND/OR The uncertainties are minimal and have been characterized.
Medium	The study has limited discussion of variability, key uncertainties, limitations, and/or data gaps. AND/OR Multiple uncertainties have been identified but are unlikely to have a substantial impact on results.

Low	Variability, key uncertainties, limitations, and data gaps are not discussed. AND/OR Uncertainties identified may have a substantial impact on the exposure the exposure assessment
Critically Deficient	Estimates are highly uncertain based on characterization of variability and uncertainty.
Not Rated/Not Applicable	
Reviewer's comments	*[Document concerns, uncertainties, limitations, and deficiencies and any additional comments that may highlight study strengths or important elements such as relevance]*

C.5 Evidence Integration

As described in Section 7 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), evidence integration refers to the consideration of evidence obtained from systematic review and scientific information obtained from sources that did not undergo systematic review to implement a weight of the scientific evidence approach. The weight of the scientific evidence is defined as "a systematic review method, applied in a manner suited to the nature of the evidence or decision, that uses a pre-established protocol to comprehensively, objectively, transparently, and consistently identify and evaluate each stream of evidence, including strengths, limitations, and relevance of each study and to integrate evidence as necessary and appropriate based upon strengths, limitations, and relevance" (40 CFR 702.33).The consideration of the quality and relevance of the data, while taking into account the strengths and limitations of the data, to appropriately evaluate the evidence for this supplement, is described in Section 7 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). Table_Apx C-5 and Table_Apx C-6, originally from Section 7.3 of the 2021 Draft Systematic Review Protocol, provide general considerations and examples of factors that contribute to the strength of evidence for each evidence stream and example weight of the scientific evidence judgments based on these general considerations, respectively, when evaluating *potentially relevant exposure data* for this draft supplement (U.S. EPA, 2021a).

Table_Apx C-5. Considerations that Inform Evaluations of the Strength of the Evidence

Considerations	Factors that Increase Strength	Factors that Decrease Strength
The overall weight of the scientific evidence judgment considers the general considerations below as well as chemical-specific considerations to designate each exposure scenario as robust, moderate, slight, or indeterminate. The designation is a measure of the weight of the evidence supporting the representativeness of the exposure estimates toward the true distribution of exposure (and releases) for the scenario.		
Relevance to exposure scenario	• Directly relevant to evaluated exposure scenario	• Data used is for an alternative or surrogate scenario
For modeled estimates	• Model used has been peer-reviewed and is being applied in a manner appropriate to its design and objective	• Evidence demonstrating implausibility • Model has not been peer-reviewed and no ground-truthing has been performed • Parameterization is not well described, documented or is not appropriate to the evaluated scenario
Data quality	• Medium or high data quality rating (via Data Evaluation)	• Low data quality rating (via Data Evaluation) • Imprecision or inaccuracy

Considerations	**Factors that Increase Strength**	**Factors that Decrease Strength**
Data points	• High number of data points	• Low number of data points • High proportion of data sampled prior to changes in industry or other relevant conditions (*e.g.*, OSHA PEL)
Representative of the whole industry (for occupational scenarios)	• Large proportion of sites included within the exposure scenario were measured	• Evidence may not be sufficiently representative of all of the sites for the exposure scenario
Representative of the sub-population	• Applicable to most or all of the different population groups included within the exposure scenario	• Information was not available to sufficiently cover most or all of population groups included within the exposure scenarios
Consistency	• Consistency and replication within a study and across studies	• Inexplicable contradictory findings across studies
Variability	• Variability is accounted in estimates • Full distributions of input parameters	• Variability unaccounted in estimates
Uncertainties	• Uncertainties are low and the uncertainties are unlikely to significantly impact exposure estimates	• Uncertainties that are likely to over- or under-estimate exposure from the actual exposures for the exposure scenario

Table_Apx C-6. Evaluation of the Weight of the Scientific Evidence for Exposure Assessments

Category	Robust	Moderate	Slight	Indeterminate	Overall Weight of the Scientific Evidence
Exposure Scenario Factors (*e.g.*, habits, worker activities, exposure factors)	• Directly relevant to evaluated exposure scenario • Applicable to most or all of the different population groups included within the exposure scenario • Full distributions of input parameters • High or medium quality data ratings • The habits, worker activities, and/or use patterns are accounted for, are current • Uncertainties are low and the uncertainties are unlikely to significantly impact exposure estimates	• Surrogate scenarios from similar chemicals are used to infer similar exposures or emissions. • Some distribution of input parameters • High or medium quality data ratings • There is some, but not complete, documentation or description of assumptions, limitations and uncertainties • Surrogate scenarios from similar uses are used to infer similar use patterns or habits and practices	• Medium or low quality data ratings • Partially supported by assumptions • Uncertainties are not fully known or documented • Habits and practices are not fully known and there is a high degree of uncertainty in defining use patterns	• Qualitative descriptions of exposure without additional context. • No supporting data on habits and practices are available	The consideration factors and the categories to the left result in an overall weight of the scientific evidence judgment as one of the following: • *Robust* • *Moderate* • *Slight* • *Indeterminate*
Measured/Monitored Data	There is measured information and the temporal and spatial aspect of the measurements are well described, relevant and reflect current conditions • Medium or high data quality rating (via Data Evaluation) • High number of data points • Multiple studies or a large number of data points which indicate similar findings • Large proportion of sites included within the exposure scenario were measured	There is measured information which does not reflect current environmental conditions or does not correspond to current activities but provides evidence of exposure • Limited number of studies or limited number of data points which indicate similar findings • Information was not available to sufficiently cover most or all of population groups included within the exposure scenarios • There is some, but not complete, documentation or	There is limited measured information and information and does not reflect exposure conditions and does not correspond to known activities • Information was not available to sufficiently cover most or all of population groups included within the exposure scenarios • Assumptions and uncertainties are not known or documented	No measured or monitored data are available	

Category	Robust	Moderate	Slight	Indeterminate	Overall Weight of the Scientific Evidence
	• Consistency and replication within a study and across studies • Uncertainties are low and the uncertainties are unlikely to significantly impact exposure estimates • Sensitivity of the exposure estimates has been described and quantified incorporating assumptions, limitations, and uncertainties	description of assumptions, limitations, and uncertainties			The consideration factors and the categories to the left result in an overall weight of the scientific evidence judgment as one of the following: • *Robust* • *Moderate* • *Slight* • *Indeterminate*
Estimation Methodology/Data	• The methodology for deriving the estimate is well described and the underlying computational and/or scientific basis is robust, has an empirical basis or well documented mathematical basis and considers chemical specificity (*e.g.*, physical and chemical properties and fate) • Applicable to most or all of the different population groups included within the exposure scenario (representative) • Sensitivity of the exposure estimates has been described and quantified incorporating assumptions, limitations, and uncertainties	• The methodology for deriving the estimate is well described and the underlying computational and/or scientific basis is robust, however there is uncertainty in the parameterization or applicability • There is some, but not complete, documentation or description of assumptions, limitations and uncertainties.	• Modeling approach used to estimate exposures is not rooted in scientific rigor or does not mathematically represent the exposure scenario; parameterization is not complete or does not utilize the best available science. • Assumptions and uncertainties are not known or documented	• Modeling approach is not available for the scenario or lack of information on parameters prohibits use of available models.	The consideration factors and the categories to the left results in an overall weight of the scientific evidence judgment as one of the following: • *Robust* • *Moderate* • *Slight* • *Indeterminate*
Comparison of Estimated and Measured Exposures (*if*	• There are comparable estimates using alternate approaches	• Modeled estimates and measured exposure values are comparable, however	• There is a lack of correspondence between measured exposures and	• Category does not have indeterminate criterion.	

Category	Robust	Moderate	Slight	Indeterminate	Overall Weight of the Scientific Evidence
both estimated and measured estimates are used)	• There is concordance between measured and/or reported and modeled estimates/predictions for the same exposure scenario • Sensitivity of the exposure estimates has been described and quantified incorporating assumptions, limitations, and uncertainties	differences in methodology, collection, or context make it difficult to arrive at full concordance • There is some, but not complete, documentation or description of assumptions, limitations and uncertainties	modeled exposure estimates even when uncertainty and variability are accounted for. • Assumptions and uncertainties are not known or documented		

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C.5.1 Environmental Release and Occupational Exposure

EPA evaluated environmental releases based on reported release data, modeling approaches, and industry sector information from standard engineering sources such as TRI and DMR. As described in Appendix E, EPA estimated COU-specific releases where supporting data existed and documented uncertainties where an absence of such data required a broader application of release estimates.

EPA evaluated occupational exposures based on monitoring data, modeling approaches, and worker activity information from standard engineering sources and systematic review as described in Appendix F. EPA used COU-specific assessment approaches where supporting data existed and documented uncertainties where supporting data were only applicable for broader assessment approaches.

C.5.2 General Population

General population exposures were evaluated for each exposure pathway based on environmental release data identified as described above in Section C.4.1, environmental monitoring data identified through available databases or as described in Section C.4.2, and any other relevant information identified through systematic review. As described in Section 1, all physical and chemical and fate properties evaluated in the 2020 RE were used to evaluate the in-scope exposure pathways of the supplement.

C.5.2.1 General Population: Surface Water

To evaluate the surface water pathway, EPA relied on modeled surface water concentrations based on environmental release data reported to TRI and DMR (Appendix E.3.1) and releases modeled for other release types, including DTD and hydraulic fracturing (Appendix E.3.2).

EPA identified ambient surface water monitoring data through the WQP, drinking water monitoring from PWSs through the UCMR3 database and three state-specific databases (Section 2.3.1.1). EPA used available surface water monitoring data to confirm the accuracy of model predictions in location-specific case-studies (Appendix G.2.3.2). In addition, available drinking water monitoring data (see 1,4-Dioxane Supplement to the Risk Evaluation Data Extraction Information for General Population, Consumer, and Environmental Exposure) were used to provide context and a point of reference for modeled drinking water concentrations and risk estimates (Section 5.2.2.1.5) (U.S. EPA, 2023h).

C.5.2.2 General Population: Groundwater

To evaluate the land pathway (groundwater) releases, EPA relied on environmental release data reported to TRI (Section 2.2.1.1 and Appendix E.4.1) and releases modeled for hydraulic fracturing operations (Appendix E.5.2).

EPA identified groundwater monitoring data for 1,4-dioxane through the WQP as presented in Section 2.3.2.1 and described in Appendix H.1. Furthermore, EPA contextualized potential groundwater concentrations identified in the literature through systematic review (see 1,4-Dioxane Supplement to the Risk Evaluation Data Extraction Information for General Population, Consumer, and Environmental Exposure) using search terms identified in Appendix C.2 (U.S. EPA, 2023h).

C.5.2.3 General Population Exposure: Ambient Air

EPA did not identify quantitative outdoor air monitoring data for 1,4-dioxane through systematic review. To evaluate the air pathway, EPA relied on modeled air concentrations based on industrial releases reported to TRI (Section 2.3.3.2.2 and Appendix E.5.1), releases modeled for laundry facilities

(Section 2.3.3.2.4 and Appendix E.5.2), and releases modeled for hydraulic fracturing operations (Section 2.3.3.2.4 and Appendix E.5.2).

Appendix D COU-OES MAPPING AND CROSSWALK

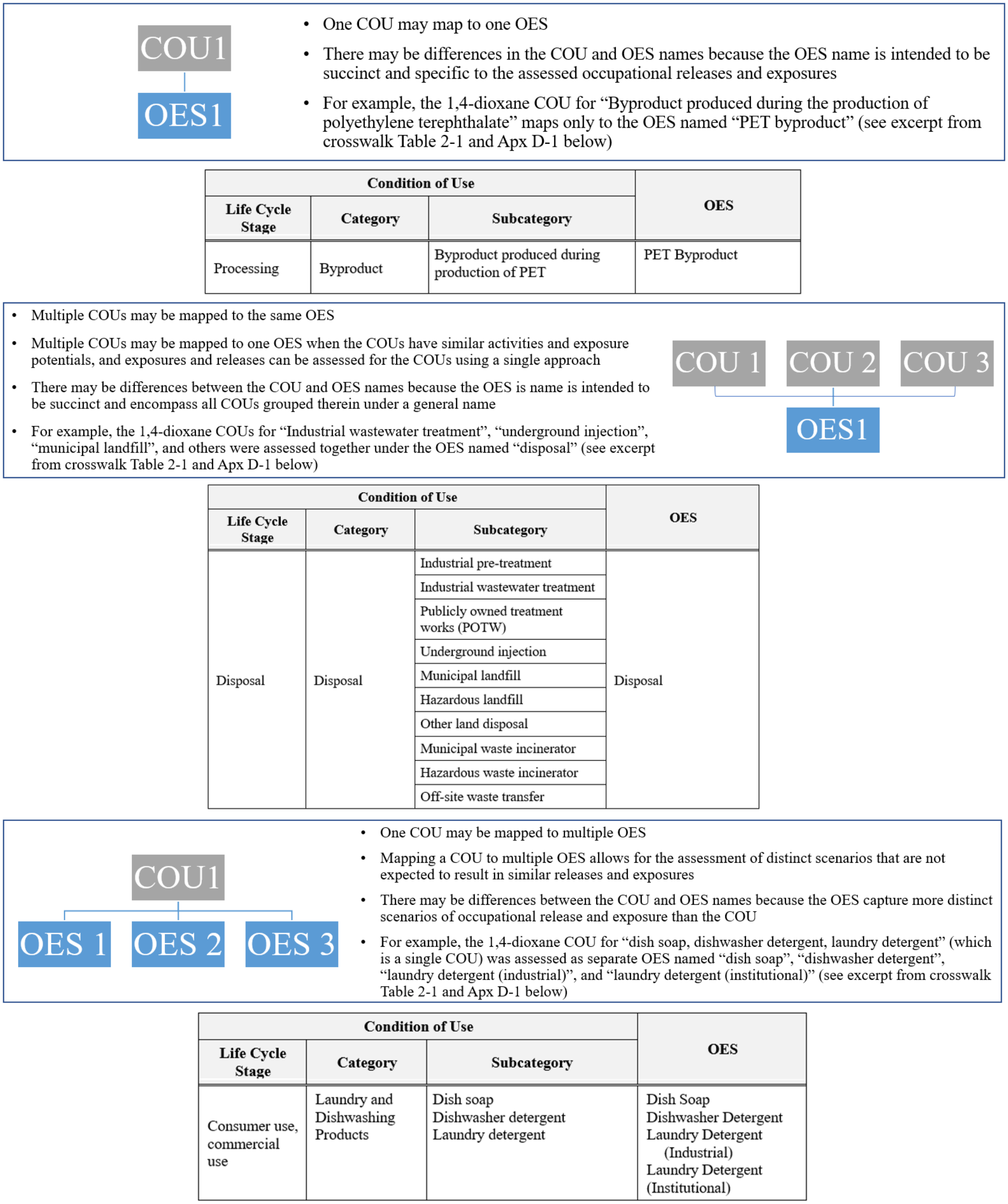
This appendix contains additional information about the relationship between the COUs and OESs determined for 1,4-dioxane.

Condition of Use (COU): TSCA § 3(4) defines COUs as "the circumstances, as determined by the Administrator, under which a chemical substance is intended, known, or reasonably foreseen to be manufactured, processed, distributed in commerce, used, or disposed of". COUs included in the scope of EPA's risk evaluations are typically tabulated in scope documents and risk evaluation documents as summaries of life cycle stages, categories, and subcategories of use. Therefore, a COU is composed of a combination of life cycle stage, category, and subcategory. COU development may include Chemical Data Reporting (CDR) information, market profile information, and literature sources. Early in the risk evaluation process, EPA maps each COU to an occupational exposure scenario for the environmental release and occupational exposure assessment.

Occupational Exposure Scenario (OES): This term is intended to describe the grouping or segmenting of COUs for assessment of releases and exposures. For example, EPA may assess a group of multiple COUs together as one OES due to similarities in release and exposure sources, worker activities, and use patterns. Alternatively, EPA may assess multiple OES for one COU because there are different release and exposure potentials for a given COU. OES determinations are also largely driven by the availability of data and modeling approaches to assess occupational releases and exposures. For example, even if there are similarities between multiple COUs, if there is sufficient data to separately assess releases and exposures for each COU, EPA would not group them into the same OES.

D.1 COU-OES Mapping

The details of an identified COU will determine the number of associated OES(s). Mapping OES to COUs may come in many forms, including a direct one-to-one mapping of a single OES to a single COU, mapping of one OES to multiple COUs, or mapping of multiple OES to a single COU, as shown in Figure_Apx D-1. The OES mapping is driven by similarities and differences in the expected occupational exposures and releases for a COU and the reasonably available data to estimates such exposures and releases, as discussed in Section 2.1.1. Further, there may be differences in the name of an OES from the name of the COU to which it is mapped. This is because OES names are intended to be succinct, capture all COUs where one OES is mapped to multiple COUs, and distinct enough to represent the specific occupational exposure and release scenario.



- One COU may map to one OES
- There may be differences in the COU and OES names because the OES name is intended to be succinct and specific to the assessed occupational releases and exposures
- For example, the 1,4-dioxane COU for "Byproduct produced during the production of polyethylene terephthalate" maps only to the OES named "PET byproduct" (see excerpt from crosswalk Table 2-1 and Apx D-1 below)

Condition of Use			**OES**
Life Cycle Stage	**Category**	**Subcategory**	
Processing	Byproduct	Byproduct produced during production of PET	PET Byproduct

- Multiple COUs may be mapped to the same OES
- Multiple COUs may be mapped to one OES when the COUs have similar activities and exposure potentials, and exposures and releases can be assessed for the COUs using a single approach
- There may be differences between the COU and OES names because the OES is name is intended to be succinct and encompass all COUs grouped therein under a general name
- For example, the 1,4-dioxane COUs for "Industrial wastewater treatment", "underground injection", "municipal landfill", and others were assessed together under the OES named "disposal" (see excerpt from crosswalk Table 2-1 and Apx D-1 below)

Condition of Use			**OES**
Life Cycle Stage	**Category**	**Subcategory**	
Disposal	Disposal	Industrial pre-treatment	Disposal
		Industrial wastewater treatment	
		Publicly owned treatment works (POTW)	
		Underground injection	
		Municipal landfill	
		Hazardous landfill	
		Other land disposal	
		Municipal waste incinerator	
		Hazardous waste incinerator	
		Off-site waste transfer	

- One COU may be mapped to multiple OES
- Mapping a COU to multiple OES allows for the assessment of distinct scenarios that are not expected to result in similar releases and exposures
- There may be differences between the COU and OES names because the OES capture more distinct scenarios of occupational release and exposure than the COU
- For example, the 1,4-dioxane COU for "dish soap, dishwasher detergent, laundry detergent" (which is a single COU) was assessed as separate OES named "dish soap", "dishwasher detergent", "laundry detergent (industrial)", and "laundry detergent (institutional)" (see excerpt from crosswalk Table 2-1 and Apx D-1 below)

Condition of Use			**OES**
Life Cycle Stage	**Category**	**Subcategory**	
Consumer use, commercial use	Laundry and Dishwashing Products	Dish soap Dishwasher detergent Laundry detergent	Dish Soap Dishwasher Detergent Laundry Detergent (Industrial) Laundry Detergent (Institutional)

Figure_Apx D-1. COU and OES Mapping

D.2 COU-OES Crosswalk

A crosswalk of the COU with the OES assessed is provided in Table_Apx D-1. As discussed in Section 2.1.1, a COU is a combination of life cycle stage, category, and subcategory and EPA mapped each COU to an OES. The purpose of an OES is to group, where appropriate, COUs based on similarity of the operations and data availability for each COU. EPA assessed environmental releases (air, water, and land) and occupational exposures (inhalation and dermal) to 1,4-dioxane for each of the OES listed in Table_Apx D-1. As noted in this table, some of these OESs were in scope of the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c) while others were in scope of this supplemental risk evaluation.

Table_Apx D-1. Categories and Subcategories of Conditions of Use Included in the Scope of the Risk Evaluation

Condition of Use			**OES**	**Risk Evaluation in Which Occupational Exposures Were Assessed**
Life Cycle Stage	**Category**[a]	**Subcategory**[b]		
Manufacturing	Domestic Manufacture	Domestic Manufacture	Manufacturing	2020 RE
	Import	Import	Import and Repackaging	2020 RE
		Repackaging		
Processing	Processing as a Reactant	Polymerization catalyst	Industrial Uses	2020 RE
	Non-incorporative	Basic organic chemical manufacturing (process solvent)		
	Byproduct	Byproduct produced during processes	Ethoxylation Process Byproduct	Supplemental RE
		Byproduct produced during production of PET	PET Byproduct	Supplemental RE
	Recycling	Recycling	Disposal	2020 RE
Distribution in commerce	Distribution	Distribution	Distribution activities (*e.g.*, loading) considered throughout life cycle, rather than using a single distribution scenario	N/A
Industrial use	Intermediate use	Plasticizer intermediate	Industrial Uses	2020 RE
		Catalysts and reagents for anhydrous acid reactions, brominations, and sulfonations		
	Processing aids, not otherwise listed	Wood pulping[c]		
		Extraction of animal and vegetable oils[c]		
		Wetting and dispersing agent in textile processing[c]		
		Polymerization catalyst		
		Purification of process intermediates		
		Etching of fluoropolymers		

Condition of Use			OES	Risk Evaluation in Which Occupational Exposures Were Assessed
Life Cycle Stage	**Category**[a]	**Subcategory**[b]		
	Functional fluids (open and closed system)	Polyalkylene glycol lubricant	Functional Fluids (Open-System)	2020 RE
		Synthetic metalworking fluid		
		Cutting and tapping fluid		
		Hydraulic fluid	Functional Fluids (Closed-System)	2020 RE
Industrial use, commercial use	Laboratory Chemicals	Chemical reagent	Laboratory Chemicals	2020 RE
		Reference material		
		Spectroscopic and photometric measurement		
		Liquid scintillation counting medium		
		Stable reaction medium		
		Cryoscopic solvent for molecular mass determinations		
		Preparation of histological sections for microscopic examination		
	Adhesives and Sealants	Film cement	Film Cement	2020 RE
	Other Uses	Spray polyurethane foam; Printing and printing compositions, including 3D printing; dry film lubricant; Hydraulic fracturing	Spray Foam Application	2020 RE
			Printing Inks (3D)	2020 RE
			Dry Film Lubricant	2020 RE
			Hydraulic Fracturing	Supplemental RE
Consumer use, commercial use	Paints and Coatings	Latex wall paint or floor lacquer	Paint and Floor Lacquer	Supplemental RE
	Cleaning and Furniture Care Products	Surface cleaner	Surface Cleaner	Supplemental RE
	Laundry and Dishwashing Products	Dish soap Dishwasher detergent Laundry detergent	Dish Soap Dishwasher Detergent Laundry Detergent (Industrial) Laundry Detergent (Institutional)	Supplemental RE
	Arts, Crafts, and Hobby Materials	Textile dye	Textile Dye	Supplemental RE
	Automotive Care Products	Antifreeze	Antifreeze	Supplemental RE
	Other Consumer Uses	Spray polyurethane foam	Spray Foam Application	2020 RE
Disposal	Disposal	Industrial pre-treatment	Disposal	2020 RE

Condition of Use			OES	Risk Evaluation in Which Occupational Exposures Were Assessed
Life Cycle Stage	Category[a]	Subcategory[b]		
		Industrial wastewater treatment		
		Publicly owned treatment works (POTW)		
		Underground injection		
		Municipal landfill		
		Hazardous landfill		
		Other land disposal		
		Municipal waste incinerator		
		Hazardous waste incinerator		
		Off-site waste transfer		

[a] These categories of conditions of use reflect Chemical Data Reporting (CDR) rule codes and broadly represent conditions of use for 1,4-dioxane in industrial and/or commercial settings.
[b] These subcategories reflect more specific uses of 1,4-dioxane.

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Appendix E INDUSTRIAL AND COMMERCIAL ENVIRONMENTAL RELEASES

This appendix contains additional information relevant to the assessment of industrial and commercial environmental releases.

E.1 Estimates of the Number of Industrial and Commercial Facilities with Environmental Releases

As a part of the assessment of industrial and commercial environmental releases, EPA estimated the number of facilities with releases for each OES. Where available, EPA used 2013 to 2019 TRI (U.S. EPA, 2022g) and 2013-2019 DMR (U.S. EPA, 2022c) data to provide a basis to estimate the number of sites using 1,4-dioxane within an OES. Additional information on how EPA utilized TRI and DMR to estimate the number of sites using 1,4-dioxane within a COU can be found in Section 2.2.1.2.2 of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Where the number of sites could not be determined using TRI or DMR or where these data were determined to not capture the entirety of sites within an OES, EPA supplemented the available data with U.S. economic data using the following methods:

- Identify the North American Industry Classification System (NAICS) codes for the industry sectors associated with these uses.
- Estimate total number of sites using the U.S. Census' Statistics of US Businesses (SUSB) (U.S. Census Bureau, 2015) data on total establishments by 6-digit NAICS.
- Review available ESDs and GSs for established facility estimates for each occupational exposure scenario.
- Combine the data generated in bullets 1 through 3 to produce an estimate of the number of sites using 1,4-dioxane in each 6-digit NAICS code and sum across all applicable NAICS codes for the COU, augmenting as needed with data from the ESDs and GSs, to arrive at a total estimate of the number of sites within the COU.

A summary of the number of facilities EPA determined for each OES and each type of release is shown in Table_Apx E-1. The number of facilities may be different for each type of release within the same OES if sufficient data were available to make this differentiation.

Table_Apx E-1. Summary of EPA's Estimates for the Number of Facilities for Each OES

OES	Type of Release	Number of Facilities	Notes
Manufacturing	Air, Land	1	Based on 2019 TRI reporting (U.S. EPA, 2022g).
	Surface Water	1	Based on 2013–2019 DMR and TRI reporting (U.S. EPA, 2022c, g).
	POTW or Non-POTW WWT	1	Based on 2013–2019 TRI reporting (U.S. EPA, 2022g).
Import and Repackaging	Air, Land	1	Based on 2019 TRI reporting (U.S. EPA, 2022g).
	Surface Water	6	Based on 2013–2019 DMR and TRI reporting (U.S. EPA, 2022c, g).
	POTW or Non-POTW WWT	6	Based on 2013–2019 TRI reporting (U.S. EPA, 2022g).

OES	Type of Release	Number of Facilities	Notes
Industrial Uses	Air, Land	12	Based on 2019 TRI reporting (U.S. EPA, 2022g).
	Surface Water	24	Based on 2013–2019 DMR and TRI reporting (U.S. EPA, 2022c, g).
	POTW or Non-POTW WWT	17	Based on 2013–2019 TRI reporting (U.S. EPA, 2022g).
Functional Fluids (Open-System)	Air, Land	2	Based on 2019 TRI reporting (U.S. EPA, 2022g).
	Surface Water	6	Based on 2013–2019 DMR and TRI reporting (U.S. EPA, 2022c, g).
	POTW or Non-POTW WWT	1	Based on 2013–2019 TRI reporting (U.S. EPA, 2022g).
Functional Fluids (Closed-System)	All	N/A	Assessed as a part of Industrial Uses OES.
Laboratory Chemical	All	132	Calculated using the GS on Use of Laboratory Chemicals (U.S. EPA, 2022h) and the amount of 1,4-dioxane used in laboratory uses per the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Film Cement	All	211	Based on the number of sites for this OES in the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c), which is a bounding estimate based on U.S. Census Bureau data for NAICS code 512199, Other Motion Picture and Video Industries.
Spray Foam Application	All	1,553,559	Based on the number of sites for this OES in the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c), which is a bounding estimate based on U.S. Census Bureau data for NAICS code 238310, Drywall and Insulation Contractors.
Printing Inks (3D)	Air, Land	N/A	Assessed as a part of Industrial Uses OES.
	Surface Water, POTW, non-POTW WWT	1	Based on 2013-2019 DMR and TRI reporting (U.S. EPA, 2022c, g).
Dry Film Lubricant	All	8	Based on the number of sites for this OES in the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c), which is based on conversations with the Kansas City National Security Campus (manufacturer and uses of dry film lubricants).
Disposal	Air	15	Based on 2019 TRI reporting (U.S. EPA, 2022g).
	Surface Water	24	Based on 2013–2019 DMR and TRI reporting (U.S. EPA, 2022c, g).
	POTW or Non-POTW WWT, Land	4	Based on 2013–2019 TRI reporting (U.S. EPA, 2022g).

OES	Type of Release	Number of Facilities	Notes
Textile Dye	All	783	Bounding estimate based on U.S. Census Bureau data for NAICS code 313310, Textiles and Fabric Finishing Mills.
Antifreeze	All	84,383	Bounding estimate based on U.S. Census Bureau data for NAICS codes 811111, General Automotive Repair, and 811198, All Other Automotive Repair and Maintenance.
Surface Cleaner	All	Unknown within Liverpool OH; 55,998 (industry bounding estimate)	Land release estimates for this OES were developed for the Liverpool, OH case study and the number of sites within this locality is unknown (*e.g.*, the release estimates are not per site but for the entire locality). Bounding estimate for the industry is based on U.S. Census Bureau data for NAICS code 561720, Janitorial Services.
Dish Soap	All	Unknown within Liverpool OH; 773,851 (industry bounding estimate)	Land release estimates for this OES were developed for the Liverpool, OH, case study and the number of sites within this locality is unknown (*e.g.*, the release estimates are not per site but for the entire locality). Bounding estimate for the industry is based on U.S. Census Bureau data for NAICS codes 623300, 713900, 721100, 721300, 722300, 722400, and 722500.
Dishwasher Detergent	All	Unknown within Liverpool OH; 773,851 (industry bounding estimate)	Land release estimates for this OES were developed for the Liverpool, OH, case study and the number of sites within this locality is unknown (*e.g.*, the release estimates are not per site but for the entire locality). Bounding estimate for the industry is based on U.S. Census Bureau data for NAICS codes 623300, 713900, 721100, 721300, 722300, 722400, and 722500.
Laundry Detergent (Institutional)	All	95,533	Bounding estimate based on industry information as described in the ESD on Water Based Washing operations at Industrial and Institutional Laundries (OECD, 2011b).
Laundry Detergent (Industrial)	All	2,453	Bounding estimate based on U.S. Census Bureau data for NAICS code 812330, Linen and Uniform Supply.
Paints and Floor Lacquer	All	33,648	Bounding estimate based on U.S. Census Bureau data for NAICS code 811121, Automotive Body, Paint, and Interior Repair and Maintenance.
Polyethylene Terephthalate (PET) Byproduct	Air, Land	13	Based on 2019 TRI reporting (U.S. EPA, 2022g).
	Surface Water	19	Based on 2013–2019 DMR and TRI reporting (U.S. EPA, 2022c, g).
	POTW or Non-POTW WWT	14	Based on 2013-2019 TRI reporting (U.S. EPA, 2022g).
Ethoxylation Process Byproduct	Air, Land	8	Based on 2019 TRI reporting (U.S. EPA, 2022g).
	Surface Water	7	Based on 2013–2019 DMR and TRI reporting (U.S. EPA, 2022c, g).

OES	Type of Release	Number of Facilities	Notes
	POTW or Non-POTW WWT	6	Based on 2013-2019 TRI reporting (U.S. EPA, 2022g).
Hydraulic Fracturing	All	411	Based on the number of sites that reported using 1,4-dioxane to FracFocus 3.0 (GWPC and IOGCC, 2022).

E.2 Estimates of Number of Release Days for Industrial and Commercial Releases

As a part of the assessment of industrial and commercial environmental releases, EPA also estimated the number of release days for each OES. EPA referenced the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), Generic Scenarios (GS), Emission Scenario Documents (ESDs), or made assumptions when estimating release days for each OES. In summary, EPA estimated the number of operating days using the below sources of data:

1. **Facility-Specific Data:** Use facility-specific data if available. If facility-specific data is not available, estimate the days/year using one of the following approaches:
    a. If facilities have known or estimated average daily use rates, calculate the days/year as: Days/year = Estimated Annual Use Rate for the Site (kg/year) / average daily use rate from sites with available data (kg/day).
    b. If sites with days/year data do not have known or estimate average daily use rates, use the average number of days/year from the sites with such data.
2. **Industry-Specific Data:** Industry-specific data may be available in the form of GSs, ESDs, trade publications, or other relevant literature. In such cases, these estimates should take precedent over other approaches, unless facility-specific data are available.
3. **Manufacture of Lower-PV Specialty Chemicals:** For the manufacture of lower-PV specialty chemicals like 1,4-dioxane, the chemical is not expected to be manufactured continuously throughout the year. Therefore, a value of 250 days/year should be used. This assumes the plant manufactures the chemical 5 days/week and 50 weeks/year (with 2 weeks down for turnaround). For the manufacture of 1,4-dioxane as a byproduct (*e.g.*, ethoxylation process, PET manufacturing), 250 days/year is also used, assuming these industrial manufacturing facilities have a similar operating schedule of 5 days/week and 50 weeks/year.
4. **Processing as Reactant (Intermediate Use) in the Manufacture of Specialty Chemicals:** Similar to #3, the manufacture of specialty chemicals is not expected to occur continuously throughout the year. Therefore, a value of 250 days/year can be used.
5. **Other Chemical Plant OES (*e.g.*, Industrial Uses):** For these OESs, it is reasonable to assume that 1,4-dioxane is not always in use at the facility, even if the facility operates 24/7. Therefore, in general, a value of 300 days/year can be used based on the "SpERC [Specific Environmental Release Categories] fact sheet – Formulation & (re)packing of substances and mixtures – Industrial (Solvent-borne)" which uses a default of 300 days/year for the chemical industry. However, in instances where the OES uses a low volume of the chemical of interest, 250 days/year can be used as a lower estimate for the days/year.
6. **POTWs:** Although POTWs are expected to operate continuously over 365 days/year, the discharge frequency of 1,4-dioxane from a POTW will be dependent on the discharge patterns of the chemical from the upstream facilities discharging to the POTW. However, there can be multiple upstream facilities (possibly with different OES) discharging to the same POTW and

information to determine when the discharges from each facility occur on the same day or separate days is typically not available. Therefore, an exact number of days/year the 1,4-dioxane is discharged from the POTW cannot be determined and a value of 365 days/year should be used.

7. **All Other OESs:** Regardless of the facility operating schedule, other OESs are unlikely to use 1,4-dioxane every day. Therefore, a value of 250 days/year should be used for these OESs.

A summary along with a brief explanation is presented in Table_Apx E-2 below. These estimates of release days are applicable to the air and water release estimates for each OES; however, there is a high level of variability and uncertainty associated with the number of days of release associated with land releases. Therefore, EPA could not estimate the number of days of release for land releases.

Table_Apx E-2. Summary of EPA's Estimates for Air and Water Release Days Expected for Each OES

OES	Release Days	Notes
Manufacturing	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities, consistent with the 2020 RE (U.S. EPA, 2020c).
Import and Repackaging	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities, consistent with the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Industrial Uses	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities, consistent with the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Functional Fluids (Open-System)	247	Per the 2011 OECD Emission Scenario Document on the Use of Metalworking Fluids, consistent with the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Functional Fluids (Closed-System)	N/A	Assessed as a part of Industrial Uses OES.
Laboratory Chemical	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities, consistent with the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Film Cement	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities, consistent with the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Spray Foam Application	3	Per the 2018 EPA generic scenario Application of Spray Polyurethane Foam Insulation, consistent with the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Printing Inks (3D)	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities, consistent with the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Dry Film Lubricant	48	Per process description information provided in the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).

OES	Release Days	Notes
Disposal	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities, consistent with the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c).
Textile Dye	31 to 295	Based on the 2015 OECD on Textile Dyes (OECD, 2017) and Monte Carlo Modeling.
Antifreeze	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities.
Surface Cleaner	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities.
Dish Soap	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities.
Dishwasher Detergent	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities.
Laundry Detergent (Institutional)	250 to 365	Based on the 2011 OECD ESD on Industrial and Institutional Laundries (OECD, 2011b) and Monte Carlo Modeling.
Laundry Detergent (Industrial)	20 to 365	Based on the 2011 OECD ESD on Industrial and Institutional Laundries (OECD, 2011b) and Monte Carlo Modeling.
Paints and Floor Lacquer	250	Based on the 2011 OECD ESD on Coating Application via Spray Painting in the Automotive Refinishing Industry (OECD, 2011a).
Polyethylene Terephthalate (PET) Byproduct	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities.
Ethoxylation Process Byproduct	250	Assumed 5 days per week and 50 weeks per year with 2 weeks per year for shutdown activities.
Hydraulic Fracturing	1 to 72	Based on the reported number of days for sites that use 1,4-dioxane in FracFocus 3.0 (GWPC and IOGCC, 2022). This range of release days refers to only the hydraulic fracturing and not post-fracturing production stages. EPA's estimates for flowback and produced water releases during production stages occur over 350 days/year (U.S. EPA, 2022d).

E.3 Water Release Assessment

This section describes EPA's methodology for estimating daily wastewater discharges from industrial and commercial facilities manufacturing, processing, or using 1,4-dioxane. Facilities report wastewater discharges either via Discharge Monitoring Reports (DMRs) under the NPDES or TRI. EPA used 2013 to 2019 DMR (U.S. EPA, 2022c) and 2013 to 2019 TRI (U.S. EPA, 2022g) data to estimate daily wastewater discharges for the OES where available; however, EPA did not have these data for every OES. For OES without DMR and TRI data, EPA used alternate assessment approaches to estimate wastewater discharges. Both approaches, that for OESs with DMR and TRI data and that for OESs without these data, are described below.

E.3.1 Assessment Using TRI and DMR

EPA found 2013 to 2019 DMR and/or 2013 to 2019 TRI data for facilities within the following OESs:

- Manufacturing,
- Import and repackaging,
- Industrial uses,
- Functional fluids (open-system),
- 3D printing,
- Disposal,
- PET byproduct, and
- Ethoxylation byproduct.

The 2013 to 2019 TRI data were rated "medium" in EPA's systematic review process and the 2013 to 2019 DMR were rated "medium." EPA estimated daily discharges using TRI and DMR data for these OESs, with the following general stages as described in the rest of this section:

1. Collect wastewater discharge data from 2013 to 2019 DMR and TRI data,
2. Map wastewater discharge data to occupational exposure scenarios,
3. Estimate the number of facility operating days per year, and
4. Estimate daily wastewater discharges and summarize wastewater discharges for each OES.

Note that EPA compared the TRI and DMR data used to estimate water releases for the PET byproduct OES in this risk evaluation to information from a life cycle analysis on the PET manufacturing process in Appendix E.6.

Step 1: Collect Wastewater Discharge Data from DMR and TRI

The first step in estimating daily releases was to obtain 2013 through 2019 DMR and TRI data. Under the CWA, EPA regulates the discharge of pollutants into receiving waters through NPDES. A NPDES permit authorizes discharging facilities to discharge pollutants to specified limits. NPDES permits apply pollutant discharge limits to each outfall at a facility. For risk evaluation purpose, EPA is interested only in the outfalls to surface water body. NPDES permits also include internal outfalls, but they aren't included in this analysis. This is because these outfalls are internal monitoring points within the facility wastewater collection or treatment system, so they do not represent discharges from the facility. The permits require facilities to monitor their discharges and report the results to EPA and the state regulatory agency. Facilities report these results in DMRs. EPA makes these reported data publicly available via EPA's ECHO system and EPA's Water Pollutant Loading Tool (Loading Tool). The Loading Tool is a web-based tool that obtains DMR data through ECHO, presents data summaries and calculates pollutant loading (mass of pollutant discharged). EPA queried the ECHO Loading Tool to pull data for each of years 2013 through 2019. EPA removed facilities reporting zero discharges for 1,4-dioxane in DMR from the analysis because EPA cannot confirm if the pollutant is present at the facility.

Each facility subject to the TRI reporting rule must report annually the volume of chemical released to the environment and/or managed through recycling, energy recovery, and treatment. Unlike DMR, TRI includes both reports of annual direct discharges to surface water and annual indirect discharges to off-site publicly owned treatment works (POTW) and wastewater treatment (WWT) facilities (non-POTW WWT). Similar to the air release assessment, EPA included both TRI reporting Form R and TRI reporting Form A submissions in the water release assessment. Where sites reported to TRI with Form A, EPA used the Form A threshold for total releases of 500 lb/year. EPA used the entire 500 lb/year for both direct and indirect wastewater discharges; however, since this threshold is for total site releases, these 500 lb/year are attributed either to direct discharges or indirect discharges for this analysis, not both (since that would double count the releases and exceed the total release threshold for Form A). EPA pulled the TRI Basic Plus Data Files for each of years 2013 through 2019.

In summary, wastewater discharges reported to DMR and TRI include the following:

- DMR:
  - On-site releases to surface water (direct discharges).
- TRI:
  - On-site releases to surface water (direct discharges),
  - Off-site transfers to POTWs (indirect discharges), and
  - Off-site transfers to non-POTW WWT (indirect discharges).

Note that the two datasets are not updated concurrently. The Loading Tool automatically and continuously checks ICIS-NPDES for newly submitted DMRs. The Loading Tool processes the data weekly and calculates pollutant loading estimates; therefore, water discharge data (DMR data) are available on a continual basis. Although the Loading Tool process data weekly, each permitted discharging facility is only required to report their monitoring results for each pollutant at a frequency specified in the permit (*e.g.*, monthly, every two months, quarterly). TRI data is reported annually for the previous calendar year and is typically released in October (*i.e.*, 2020 TRI data is released in October 2021).

Step 2: Map Wastewater Discharge Data to Occupational Exposure Scenarios
The next step in estimating daily releases was to map 2013 through 2019 DMR and TRI data to the 1,4-dioxane OES. EPA used the same mapping methodology for the water assessment as that described in Appendix E.5.1. EPA ensured consistency in the OES mapping for sites that reported to both TRI and DMR. EPA also ensured consistency in the OES mapping between the air, water, and land assessments.

Step 3: Estimate the Number of Facility Operating Days per Year
EPA then estimated the number of operating days (days/year) for each facility reporting wastewater discharges to DMR and TRI. EPA generally used the same number of operating days for the same OES for both the air and water analysis, which is based on the general methodology described previously in Appendix E.2.

Step 4: Estimate Daily Wastewater Discharges and Summarize Wastewater Discharges for each OES
After the initial steps of selecting and mapping of the water discharge data and estimating the number of facility operating days/year were completed, the next step was to summarize annual and daily wastewater discharges for each OES. EPA summarized annual wastewater discharges reported in DMR and TRI for each facility. EPA estimated daily wastewater discharges separately for direct and indirect discharges, as discussed below.

EPA estimated the median and maximum daily direct wastewater discharges at each facility, using the steps below. EPA presented the calculated median and maximum daily direct wastewater discharged separately for the DMR and TRI datasets because these data do not always agree/match.

1. Obtained total annual loads calculated from the Loading Tool and reported annual surface water discharges in TRI for years 2013 through 2019.
2. Divided the annual direct discharge over the number of estimated operating days for the OES to which the facility has been mapped. The number of operating days differ for each OES, as summarized in Appendix E.2.
3. Calculated the median daily direct wastewater discharge across all years of data for each facility, separately for both DMR and TRI data.

4. Identified the maximum daily direct wastewater discharge across all years of data for each facility. EPA also noted which reporting year had this maximum daily direct wastewater discharge, separately for both DMR and TRI data.

For *indirect* discharges to POTW or non-POTW WWT, EPA estimated the average daily indirect discharges for each facility and each reporting year (2013 through 2019) in TRI using steps #1 and #2 above. DMR data do not include indirect discharges. EPA did not estimate the median or maximum daily indirect discharges across all years.

A summary of the estimated daily discharges using 2013 to 2019 DMR and TRI is included in *1,4-Dioxane Supplemental Information File: Environmental Releases to Water for OES with TRI and DMR* (U.S. EPA, 2023m).

E.3.2 Assessment for OES without TRI and DMR

EPA did not find DMR or TRI data for any of the years included in this analysis for the following OESs:

- Functional Fluids (Closed-Systems)
- Laboratory Chemicals
- Film Cement
- Spray Polyurethane Foam
- Dry Film Lubricant
- Textile Dye
- Antifreeze
- Surface Cleaner
- Dish Soap
- Dishwasher Detergent
- Laundry Detergent,
- Paints and Floor Lacquer
- Hydraulic Fracturing

For these OESs, EPA estimated daily wastewater discharges by using various modeling approaches—including the use of surrogate TRI and DMR data and modeling using data from literature, GSs, and ESDs. EPA's assessment of daily wastewater discharges for each of these OESs is described below.

Functional Fluids (Closed-Systems)

Wastewater discharge data were not available for this OES and EPA did not find any information to model wastewater discharges for this OES using literature, GSs, or ESDs. EPA expects that the sources of release for this OES to be similar to those for the Industrial Uses OES, based on the process information in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). Therefore, EPA grouped the water release assessment for Functional Fluids (Closed-Systems) into that for Industrial Uses. However, there is uncertainty in this assumption of similar release sources between these OESs.

Laboratory Chemicals

EPA estimated daily wastewater discharges for facilities within the laboratory chemicals OESs using the Draft GS on Use of Laboratory Chemicals (U.S. EPA, 2022h). The GS on Use of Laboratory Chemicals was rated "high" during EPA's systematic review process.

Per the GS on Use of Laboratory Chemicals, water releases are not expected for hazardous chemicals. Because 1,4-dioxane is considered a hazardous substance under CERCLA (40 CFR Part 302.4) and the PubChem Hazardous Substances Data Bank (HSDB), there are no water releases for this OES. This is consistent with the water release assessment for this OES in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), which indicates that water releases are not expected for laboratory uses of 1,4-dioxane.

Film Cement

EPA estimated daily wastewater discharges for facilities within the Film Cement OES using process information from the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). The underlying process information for this assessment was rated "high" during EPA's systematic review process.

Per the risk evaluation, EPA does not expect water releases of 1,4-dioxane for this OES. EPA expects the glue bottles to be disposed of as solid waste without. There is some uncertainty as to whether and how much 1,4-dioxane may remain in the glue bottles when disposed. However, due to the small quantities of the glue and high volatility of the 1,4-dioxane, EPA expects any residual 1,4-dioxane to evaporate to the air or remain in the solid waste stream (U.S. EPA, 2020c).

Spray Polyurethane Foam

EPA estimated daily wastewater discharges for facilities within the Spray Polyurethane Foam OES using the same approach described for this OES in Appendix E.5.2, which is the use of the GS on Application of Spray Polyurethane Foam Insulation (U.S. EPA, 2020c). The GS on the Application of Spray Polyurethane Foam Insulation was rated "medium" during EPA's systematic review process.

The GS indicates that there are six release points:

1. Releases to fugitive air for volatile chemicals during unloading of raw materials from transport containers;
2. Releases to water, incineration, or landfill from cleaning or disposal of transport containers;
3. Releases to fugitive air for volatile chemicals during transport container cleaning;
4. Releases to incineration or landfill from spray polyurethane foam application equipment cleaning;
5. Releases to fugitive air for volatile chemicals during equipment cleaning; and
6. Releases to landfill of scrap foam from trimming applied foam.

Based on the GS, only release point #2 has the potential for wastewater discharges. To estimate this release, EPA used the equations specified in the GS (U.S. EPA, 2020c). Apart from weight fraction in spray polyurethan foam, EPA did not find any data specific to 1,4-dioxane in this OES. Therefore, the calculation of releases using this GS are for a "generic site," using the default input parameter values from the GS. Specifically, EPA used the same input parameter values that were used in the original risk evaluation for estimates of occupational exposure; see Appendix G of the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Using this methodology, EPA calculated a range of wastewater releases for this OES. For the low-end, EPA assumed there are no water releases, which is consistent with the GS explanation that containers may be disposed of without rinsing. For the high-end, EPA assumed the containers may be rinsed / poured down drains such that the entire release point #2 is to POTW. Direct water discharges are not likely given the setting (construction/ renovation sites).

EPA's calculation of wastewater discharges for this OES, including all calculation inputs, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Water for OES without TRI or DMR data* (U.S. EPA, 2023n).

Dry Film Lubricant

EPA estimated daily wastewater discharges for facilities within the Dry Film Lubricant OES using process information from the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). The underlying process information for this assessment was rated "high" during EPA's systematic review process.

Per the risk evaluation, EPA does not expect water releases of 1,4-dioxane for this OES. Based on conversations the with only known user, EPA expects wastes to be drummed and sent to a waste handler with residual wastes releasing to air or being disposed to landfill. (U.S. EPA, 2020c).

Textile Dye

EPA estimated daily wastewater discharges for facilities within the Textile Dye OES using the OECD ESD on Textile Dyes (OECD, 2017) and Monte Carlo modeling. The ESD on Textile Dyes was rated "medium" during EPA's systematic review process. The use of Montel Carlo modeling allows for variation of calculation input parameters such that a distribution of environmental releases can be calculated, from which EPA can estimate the 50th and 95th percentile releases. An explanation of this modeling approach is included in Appendix E.11.

Antifreeze

EPA did not find any information to model wastewater discharges for this OES using literature, GSs, or ESDs, nor does EPA expect this OES to be similar to other OES such that surrogate data may be used. EPA evaluated the potential for releases using the OECD ESD on Chemical Additives used in Automotive Lubricants (OECD, 2020) and the EPA MRD on Commercial Use of Automotive Detailing Products (U.S. EPA, 2022b). The ESD and MRD were both rated "high" during EPA's systematic review process.

For the use of antifreeze, EPA expects releases may occur from volatilizations of 1,4-dioxane, disposal or cleaning of empty antifreeze containers, and spent antifreeze. Both the ESD and MRD indicate that containers of automotive maintenance fluids are typically small and are not rinsed, but rather disposed of as solid waste (U.S. EPA, 2022b; OECD, 2020). Additionally, the ESD on Chemical Additives used in Automotive Lubricants indicates that spent lubricants are disposed of via incineration, which EPA expects is similarly done for spent antifreeze (OECD, 2020). Therefore, based on this information, EPA does not expect water releases of 1,4-dioxane for this OES.

Surface Cleaner

EPA estimated daily wastewater discharges for facilities within the Surface Cleaner OES using the SHEDs-HT model, which is described in Section 2.1.1.2. This modeling was completed for one case study location (Liverpool OH) and only estimates indirect wastewater discharges. EPA does not expect direct wastewater discharges to surface water from the types of commercial facilities within this OES (*e.g.*, restaurants, office buildings, other locations with janitorial services).

Dish Soap

EPA estimated daily wastewater discharges for facilities within the Surface Cleaner OES using the SHEDs-HT model, which is described in Section 2.1.1.2. This modeling was completed for one case study location (Liverpool OH) and only estimates indirect wastewater discharges. EPA does not expect direct wastewater discharges to surface water from the types of commercial facilities within this OES (*e.g.*, restaurants, assisted living facilities, amusement, and recreation facilities).

Dishwasher Detergent

EPA estimated daily wastewater discharges for facilities within the Surface Cleaner OES using the SHEDs-HT model, which is described in Section 2.1.1.2. This modeling was completed for one case study location (Liverpool OH) and only estimates indirect wastewater discharges. EPA does not expect direct wastewater discharges to surface water from the types of commercial facilities within this OES (*e.g.*, restaurants, assisted living facilities, amusement, and recreation facilities).

Laundry Detergent

EPA estimated daily wastewater discharges for facilities within the Laundry Detergent OES using the OECD ESD on Industrial and Institutional Laundries (OECD, 2011b) and Monte Carlo modeling. The ESD on Industrial and Institutional Laundries was rated "medium" during EPA's systematic review process. The use of Montel Carlo modeling allows for variation of calculation input parameters such that a distribution of environmental releases can be calculated, from which EPA can estimate the 50th and 95th percentile releases. An explanation of this modeling approach is included in Appendix E.12.

Paints and Floor Lacquer

EPA estimated daily wastewater discharges for facilities within the Paints and Floor Lacquers OES using the OECD ESD on Coating Application via Spray-Painting in the Automotive Refinishing Industry (OECD, 2011a). The ESD was rated "medium" during EPA's systematic review process.

As described in the process description in Appendix F.4.7, 1,4-dioxane was identified by a public comment as present in automotive refinishing products, thereby allowing EPA to identify the above ESD as the most applicable GS/ESD available. This ESD indicates that releases are expected from

1. Releases to incineration or landfill from container cleaning/disposal,
2. Releases to incineration or landfill from equipment cleaning,
3. Releases to incineration or landfill from over sprayed coating that is captured by emission controls, and
4. Releases to air from over sprayed coating that is not captured by emission controls.

None of these releases are expected to water (OECD, 2011a). Therefore, based on this ESD, EPA does not expect water releases of 1,4-dioxane for this OES.

Hydraulic Fracturing

EPA estimated daily wastewater discharges for facilities within the Hydraulic Fracturing OES using the Draft OECD ESD on Hydraulic Fracturing (U.S. EPA, 2022d) and Monte Carlo modeling. The Draft ESD on Hydraulic Fracturing was rated "high" during EPA's systematic review process. The use of Montel Carlo modeling allows for variation of calculation input parameters such that a distribution of environmental releases can be calculated, from which EPA can estimate the 50th and 95th percentile releases. An explanation of this modeling approach is included in Appendix E.13.

E.3.3 Water Release Estimates Summary

A summary of industrial and commercial water releases estimated using the above methods is presented in Table_Apx E-3 below.

Table_Apx E-3. Summary of Daily Industrial and Commercial Water Release Estimates for 1,4-Dioxane

OES	Type of Water Discharge	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Manufacturing	Surface Water	1	1.21	21.4	250	Medium	TRI, DMR
	POTW or Industrial WWT	1	0	6.69		Medium	TRI
Import and Repackaging	Surface Water	6	0.91[m]		250	Medium	TRI, DMR
	POTW or Industrial WWT	6	0	0.91		Medium	TRI
Industrial Uses	Surface Water	24	0	24.5	250	Medium	TRI, DMR
	POTW or Industrial WWT	17	0	105		Medium	TRI
Functional Fluids (Open-System)	Surface Water	6	0	0.67	247	Medium	TRI, DMR
	POTW or Industrial WWT	1	4.67	70.9		Medium	TRI
Functional Fluids (Closed-System)	All	Assessed as a part of Industrial Uses OES				N/A	N/A
Laboratory Chemical	Surface Water, POTW, or Industrial WWT	132	0 (water releases not expected)		250	High	GS[d]
Film Cement	Surface Water, POTW, or Industrial WWT	211	0 (water releases not expected)		250	High	Process information[e]
Spray Foam Application	Surface Water	1,553,559	0 (surface water releases not expected)		3	Medium	GS[f]
	POTW	1,553,559	0	0.0036		Medium	GS[f]
Printing Inks (3D)	Surface Water	1	0.018	0.022	250	Medium	TRI, DMR
	POTW or Industrial WWT	1	0 (no indirect releases per TRI)			Medium	Medium

OES	Type of Water Discharge	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Dry Film Lubricant	Surface Water, POTW, or Industrial WWT	8	0 (water releases not expected)		48	High	Process information[e]
Disposal	Surface Water	24	0	31.8	250	Medium	TRI, DMR
	POTW or Industrial WWT	4	0	0.91		Medium	TRI
Textile Dye	POTW	783	1.50E−05	0.001	31 to 295	Medium	ESD[g] and Modeling[h]
	Land (unknown landfill type) or POTW (unknown partitioning)	783	2.09E−07	9.72E−05		Medium	ESD[g] and Modeling[h]
Antifreeze	Surface water, POTW, or Industrial WWT	84,383	0 (water releases not expected)		250	High	Process information[e] and Modeling[h]
Surface Cleaner	POTW	Unknown	0.072 (single value for all sites in Liverpool OH case study)		250	N/A	SHEDS-HT[l]
	Land (unknown landfill) or POTW	Unknown	18[n]			High	SHEDS-HT, Process information[e] Modeling[h]
Dish Soap	POTW	Unknown	0.064[n]		250	N/A	SHEDS-HT[m]
Dishwasher Detergent	POTW	Unknown	0.00144[n]		250	N/A	SHEDS-HT[l]
Laundry Detergent	Fugitive air, stack air, or POTW	95,533	1.51E−10	0.00714	250 to 365	Medium	ESD[i] and Modeling[h]

OES	Type of Water Discharge	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
(Institutional) – Liquid Detergents	(unknown partitioning)						
	Land (unknown landfill), incineration, or POTW (unknown partitioning)	95,533	4.05E−12	3.95E−05		Medium	ESD[i] and Modeling[h]
Laundry Detergent (Institutional) – Powder Detergents	Fugitive air, stack air, or POTW (unknown partitioning)	95,533	3.05E−08	2.10E−04	250 to 365	Medium	ESD[i] and Modeling[h]
	Land (unknown landfill), incineration, or POTW (unknown partitioning)	95,533	5.36E−08	0.0018		Medium	ESD[i] and Modeling[h]
Laundry Detergent (Industrial) – liquid detergents	Fugitive air, stack air, or POTW (unknown partitioning)	2,453	5.48E−12	0.011	20 to 365	Medium	ESD[i] and Modeling[h]
	Land (unknown landfill), incineration, or POTW (unknown partitioning)	2,453	4.78E−12	1.46E−04		Medium	ESD[i] and Modeling[h]

OES	Type of Water Discharge	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Laundry Detergent (Industrial) – powder detergents	Fugitive air, stack air, or POTW (unknown partitioning)	2,453	1.76E−09	0.0112	20 to 365	Medium	ESD[i] and Modeling[h]
	Land (unknown landfill), incineration, or POTW (unknown partitioning)	2,453	2.92E−11	3.92E−04		Medium	ESD[i] and Modeling[h]
Paints and Floor Lacquer	Surface water, POTW, or Industrial WWT	33,648	0 (water releases not expected)		250	Medium	ESD[j] and process information[e]
PET Byproduct	Surface water	19	0	10,050	250	Medium	TRI, DMR
	POTW or Industrial WWT	14	0	682		Medium	TRI
Ethoxylation Process Byproduct	Surface water	7	0	0.25	250	Medium	TRI, DMR
	POTW or Industrial WWT	6	0	448		Medium	TRI
Hydraulic Fracturing	Surface water, incineration, or landfill (unknown partitioning)	411	3.61E−10	4.59	1 to 72	Medium	ESD[k] and Modeling[h]
	Recycle/Reuse (48%), underground injection (43%), Surface water (6%), or land (3%)	411	1.85E−10	1.12		Medium	ESD[k] and Modeling[h]

[a] Where available, EPA used 2013–2019 TRI (U.S. EPA, 2022g) and 2013–2019 DMR (U.S. EPA, 2022c) data to provide a basis to estimate the number of sites using 1,4-dioxane within a COU.

OES	Type of Water Discharge	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			

[b] Where available, EPA used the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c), generic scenarios, and emission scenario documents to provide a basis to estimate the number of release days of 1,4-dioxane within a COU.

[c] Narrative descriptions of all release estimate sources are provided in Appendix E.3.2.

[d] The generic scenario used for this COU is the GS on Use of Laboratory Chemicals (U.S. EPA, 2022h).

[e] For this COU, EPA used process information, which is further described in Appendix E.3.2.

[f] The generic scenario used for this COU is the GS on Application of Spray Polyurethane Foam Insulation (U.S. EPA, 2018b).

[g] The emission scenario document used for this COU is the ESD on Textile Dyes (OECD, 2017).

[h] For this COU, EPA used various models and literature for model input parameters as described in Appendix E.3.2.

[i] The emission scenario document used for this COU is the ESD on Industrial and Institutional Laundries (OECD, 2011b).

[j] The emission scenario document used for this COU is the ESD on Coating Application via Spray Painting in the Automotive Refinishing Industry (OECD, 2011a).

[k] The emission scenario document used for this COU is the Draft ESD on Hydraulic Fracturing (U.S. EPA, 2022d).

[l] This value is the Commercial Upstream POTW releases estimated from the SHEDS-HT Down the Drain Model for the Liverpool OH case study (see Section 2.1.1.2).

[m] All sites for this OES reported under Form A in TRI.

[n] A single annual value was provided for all sites in the Liverpool OH case study.

E.3.4 Summary of Weight of the Scientific Evidence Conclusions in Water Release Estimates

Table_Apx E-4 provides a summary of EPA's weight of the scientific evidence conclusions in its water release estimates for each of the OES. Detailed descriptions of non-OES specific strengths, limitations, assumptions, and uncertainties (*e.g.*, general limitations for TRI, DMR, etc.) are provided in Appendix E.6.

Table_Apx E-4. Summary of Weight of the Scientific Evidence Conclusions in Water Release Estimates by OES

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Manufacturing	Wastewater discharges are assessed using reported discharges from 2013–2019 TRI and DMR. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI and DMR have medium overall data quality determinations, and consistency within the dataset (all reporters use the same or similar reporting forms). EPA included 7 years of TRI and DMR data in the analysis, which increases the variability of the dataset. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Strengths of DMR data are that it is based on monitoring data collected by facilities and the annual pollutant load is calculated by integrating release reports over shorter timeframes (*e.g.*, monthly, quarterly) and extrapolating over the course of the year. Factors that decrease the strength of the evidence for this OES include the low number of data points, uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Additionally, EPA made assumptions on the number of operating days to estimate daily releases, which introduces additional uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Import and Repackaging	Wastewater discharges are assessed using reported discharges from 2013–2019 TRI and DMR. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI and DMR have medium overall data quality determinations, and consistency within the dataset (all reporters use the same or similar reporting forms). EPA included seven years of TRI and DMR data in the analysis, which increases the variability of the dataset. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Strengths of DMR data are that it is based on monitoring data collected by facilities and the annual pollutant load is calculated by integrating release reports over shorter timeframes (*e.g.*, monthly, quarterly) and extrapolating over the course of the year. Factors that decrease the strength of the evidence for this OES include the low number of data points, uncertainty in the accuracy of reported releases, uncertainty in EPA's use of Form A submissions, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Some facilities within this OES reported to TRI using a Form A, which does not include any details on chemical release quantities. When a facility has submitted a Form A, there is no way to discern the quantity released. Therefore, where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media; however, there is uncertainty in this because the actual release quantity is unknown. Furthermore, the threshold represents an upper limit on total releases from the facility; therefore, assessing releases at the threshold value may overestimate actual releases from the facility. Additionally, uncertainty is introduced from EPA's assumptions on the number of operating days to estimate daily releases and in the

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	mapping of DMR-reporting facilities to this OES. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Industrial Uses	Wastewater discharges are assessed using reported discharges from 2013–2019 TRI and DMR. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI and DMR have medium overall data quality determinations, and consistency within the dataset (all reporters use the same or similar reporting forms). EPA included seven years of TRI and DMR data in the analysis, which increases the variability of the dataset. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Strengths of DMR data are that it is based on monitoring data collected by facilities and the annual pollutant load is calculated by integrating release reports over shorter timeframes (*e.g.*, monthly, quarterly) and extrapolating over the course of the year. Factors that decrease the strength of the evidence for this OES include uncertainty in the accuracy of reported releases, uncertainty in EPA's use of Form A submissions, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Some facilities within this OES reported to TRI using a Form A, which does not include any details on chemical release quantities. When a facility has submitted a Form A, there is no way to discern the quantity released. Therefore, where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media; however, there is uncertainty in this because the actual release quantity is unknown. Furthermore, the threshold represents an upper limit on total releases from the facility; therefore, assessing releases at the threshold value may overestimate actual releases from the facility. Additionally, uncertainty is introduced from EPA's assumptions on the number of operating days to estimate daily releases and in the mapping of DMR-reporting facilities to this OES. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Functional Fluids (Open-System)	Wastewater discharges are assessed using reported discharges from 2013–2019 TRI and DMR. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI and DMR have medium overall data quality determinations, and consistency within the dataset (all reporters use the same or similar reporting forms). EPA included seven years of TRI and DMR data in the analysis, which increases the variability of the dataset. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Strengths of DMR data are that it is based on monitoring data collected by facilities and the annual pollutant load is calculated by integrating release reports over shorter timeframes (*e.g.*, monthly, quarterly) and extrapolating over the course of the year. Factors that decrease the strength of the evidence for this OES include the low number of data points, uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. The assessment includes data from only two sites that reported to TRI (one of which reported zero water releases) and four that reported to DMR. Additionally, uncertainty is introduced from EPA's assumptions on the number of operating days to estimate daily releases and in the mapping of DMR-reporting facilities to this OES. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Functional Fluids (Closed-System)	No data was available to estimate releases for this OES. For the water release assessment, EPA grouped this OES with the Industrial Uses OES because the sources of release are expected to be similar between these OESs. Factors that increase the strength of evidence for this OES are that TRI and DMR have medium overall data quality determinations and consistency within the dataset (all reporters use the same or similar reporting forms). Additionally, EPA included seven years of TRI and DMR data in the analysis, which increases the variability of the dataset. Factors that decrease the strength of evidence for this OES are that the Industrial Releases OES release data are use as surrogate for this OES, uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Refer to the Industrial Uses OES discussion for additional discussion. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Laboratory Chemicals	Wastewater discharges are assessed using the Draft GS on Use of Laboratory Chemicals. Per the GS, water releases are not expected for hazardous chemicals. Because 1,4-dioxane is considered a hazardous chemical under CERCLA, no water releases are expected for this OES according to the GS. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the Draft GS on Use of Laboratory Chemicals has a high overall data quality determination, and there is a low level of uncertainty in the data. Factors that decrease the strength of the evidence for this OES include the that the GS has not been peer-reviewed, uncertainty in the representativeness of the GS towards all sites in this OES, and a lack of variability in the analysis. Specifically, because the default values in the ESD are generic, there is uncertainty in the representativeness of generic site estimates of actual releases from real-world sites that use 1,4-dioxane. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods, contributing to uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Film Cement	Wastewater discharges are assessed using process information from the *Final Risk Evaluation for 1,4-Dioxane*. Per the process information, EPA does not expect water releases of 1,4-dioxane for this OES because 1,4-dioxane volatilizes to air after application of the film cement and empty cement bottles are disposed of as solid waste without rinsing. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the underlying data sources for the process information have a high overall data quality determination, and there is a low level of uncertainty in the data because the process information comes directly from actual users of 1,4-dioxane in film cement. Factors that decrease the strength of the evidence for this OES include the uncertainty in the representativeness of evidence to all sites in this OES and a lack of variability. Specifically, the process information for the production and use of film cement is based on information from three use sites, one from Australia and two from the U.S. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Spray Foam Application	Wastewater discharges are assessed using the GS on Application of Spray Polyurethane Foam Insulation. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the underlying data sources for the process information have a medium overall data quality determination, and there is a low level of uncertainty in the data. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of the GS to all sites since it is generic and not specific to sites that use 1,4-dioxane and a lack of variability. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Printing Inks (3D)	Wastewater discharges are assessed using reported discharges from 2013–2019 DMR. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that DMR has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). Additionally, EPA used DMR data for seven years, which increases the variability of the dataset. Strengths of DMR data are that it is based on monitoring data collected by facilities and the annual pollutant load is calculated by integrating release reports over shorter timeframes (*e.g.*, monthly, quarterly) and extrapolating over the course of the year. Factors that decrease the strength of the evidence for this OES include the low number of data points, uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites. Additionally, no TRI data is available for this OES, EPA made assumptions on the number of operating days, and there is uncertainty in the mapping of DMR-reporting facilities to this OES. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Dry Film Lubricant	Wastewater discharges are assessed using process information from the *Final Risk Evaluation for 1,4-Dioxane*. Based on conversations the with only known user who supplied this process information, EPA expects wastes to be drummed and sent to a waste handler with residual wastes releasing to air or being disposed to landfill, such that there are no water releases. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the underlying data sources for the process information have a high overall data quality determination, and there is a low level of uncertainty in the data. Additionally, the process information comes directly from an actual user of 1,4-dioxane in dry film lubricants. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of evidence to all sites and a lack of variability. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Disposal	Wastewater discharges are assessed using reported discharges from 2013–2019 TRI and DMR. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI and DMR have medium overall data quality determinations, and consistency within the dataset (all reporters use the same or similar reporting forms). EPA included seven years of TRI and DMR data in the analysis, which increases the variability of the dataset. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Strengths of DMR data are that it is based on monitoring data collected by facilities and the annual pollutant load is

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	calculated by integrating release reports over shorter timeframes (*e.g.*, monthly, quarterly) and extrapolating over the course of the year. Factors that decrease the strength of the evidence for this OES include uncertainty in the accuracy of reported releases, uncertainty in EPA's use of Form A submissions, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Some facilities within this OES reported to TRI using a Form A, which does not include any details on chemical release quantities. When a facility has submitted a Form A, there is no way to discern the quantity released. Therefore, where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media; however, there is uncertainty in this because the actual release quantity is unknown. Furthermore, the threshold represents an upper limit on total releases from the facility; therefore, assessing releases at the threshold value may overestimate actual releases from the facility. Additionally, uncertainty is introduced from EPA's assumptions on the number of operating days to estimate daily releases and in the mapping of DMR-reporting facilities to this OES. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Textile Dye	Wastewater discharges are assessed using Monte Carlo modeling with information from the ESD on Textile Dyes. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), that the ESD on Textile Dyes has a medium overall data quality determination and was peer reviewed, the high number of data points (simulation runs), consistency within the dataset, and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential release values that represents a larger proportion of sites than a discrete value. Factors that decrease the strength of the evidence for this OES include uncertainties and limitations in the representativeness of the estimates for sites that specifically use 1,4-dioxane because the default values in the ESD are generic. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods, contributing to uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Antifreeze	Wastewater discharges are assessed using the OECD ESD on Chemical Additives used in Automotive Lubricants and the EPA MRD on Commercial Use of Automotive Detailing Products. Factors that increase the strength of evidence for this OES are that the ESD and MRD used have high overall data quality determinations, consistency within the sources used, and a low number of uncertainties. Both sources indicate that containers of automotive maintenance fluids are not typically rinsed, but rather disposed of as solid waste or incinerated, such that there are no water releases, contributing to consistency and a low level of uncertainty in the data. Factors that decrease the strength of the evidence for this OES include that the ESD and MRD are not directly applicable to antifreeze uses (used as surrogate), uncertainty in the representativeness of the ESD and MRD to all sites and sites that specifically use 1,4-dioxane since these documents contain generic values, and a lack of variability. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Surface Cleaner	Wastewater discharges are assessed using the SHEDS-HT model. Factors that increase the strength of evidence for this OES include that the release estimates are directly relevant to the OES (as opposed to surrogate) and variability in the model input parameters. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness to all sites because the estimate is based on one case study for Liverpool, OH and because the estimate is not site-specific (the release estimate is a total for all sites in Liverpool, OH). Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Dish Soap	Wastewater discharges are assessed using the SHEDS-HT model. Factors that increase the strength of evidence for this OES include that the release estimates are directly relevant to the OES (as opposed to surrogate) and variability in the model input parameters. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness to all sites because the estimate is based on one case study for Liverpool, OH and because the estimate is not site-specific (the release estimate is a total for all sites in Liverpool, OH). Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Dishwasher Detergent	Wastewater discharges are assessed using the SHEDS-HT model. Factors that increase the strength of evidence for this OES include that the release estimates are directly relevant to the OES (as opposed to surrogate) and variability in the model input parameters. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness to all sites because the estimate is based on one case study for Liverpool, OH and because the estimate is not site-specific (the release estimate is a total for all sites in Liverpool, OH). Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Laundry Detergent	Wastewater discharges are assessed using Monte Carlo modeling with information from the ESD on Industrial and Institutional Laundries. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), that the ESD on Industrial and Institutional Laundries has a medium overall data quality determination and was peer reviewed, there are a high number of data points (simulation runs), consistency within the dataset, and full distributions of input parameters. Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential release values that represents a larger proportion of sites than a discrete value. Additionally, EPA was able to separately estimate releases for industrial and institutional laundry settings. Factors that decrease the strength of the evidence for this OES include uncertainties and limitations in the representativeness of the estimates for sites that specifically use 1,4-dioxane because the default values in the ESD are generic. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Paint and Floor Lacquer	Wastewater discharges are assessed using OECD ESD on Coating Application via Spray-Painting in the Automotive Refinishing Industry. According to the ESD, no releases are expected to water. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the ESD has a medium overall data quality determination, and a low number of uncertainties. F Factors that decrease the strength of the evidence for this OES include a lack of variability and uncertainty in the representativeness of the ESD to all sites and sites that specifically use 1,4-dioxane since the ESD is generic. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Polyethylene Terephthalate (PET) Byproduct	Wastewater discharges are assessed using reported discharges from 2013–2019 TRI and DMR. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI and DMR have medium overall data quality determinations, consistency within the dataset (all reporters use the same or similar reporting forms), and consistency with the emission data from the related life cycle analysis discussed in Appendix E.6. EPA included seven years of TRI and DMR data in the analysis, which increases the variability of the dataset. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Strengths of DMR data are that it is based on monitoring data collected by facilities and the annual pollutant load is calculated by integrating release reports over shorter timeframes (*e.g.*, monthly, quarterly) and extrapolating over the course of the year. Factors that decrease the strength of the evidence for this OES 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. Additionally, EPA made assumptions on the number of operating days to estimate daily releases, which introduces additional uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Ethoxylation Process Byproduct	Wastewater discharges are assessed using reported discharges from 2013–2019 TRI and DMR. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI and DMR have medium overall data quality determinations, and consistency within the dataset (all reporters use the same or similar reporting forms). EPA included seven years of TRI and DMR data in the analysis, which increases the variability of the dataset. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Strengths of DMR data are that it is based on monitoring data collected by facilities and the annual pollutant load is calculated by integrating release reports over shorter timeframes (*e.g.*, monthly, quarterly) and extrapolating over the course of the year. Factors that decrease the strength of the evidence for this OES include uncertainty in the accuracy of reported releases and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Additionally, EPA made assumptions on the number of operating days to estimate daily releases, which introduces additional uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Hydraulic Fracturing	Wastewater discharges are assessed using Monte Carlo modeling with information from the Draft ESD on Hydraulic Fracturing and FracFocus 3.0. Factors that increase the strength of evidence for this OES are that the release estimates are

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	directly relevant to the OES (as opposed to surrogate), that the Draft ESD on Hydraulic Fracturing and FracFocus 3.0 have medium overall data quality determinations, the high number of data points (simulation runs), and full distributions of input parameters. Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential release values that represents a larger proportion of sites than a discrete value. Factors that decrease the strength of the evidence for this OES include that the Draft ESD has not been peer reviewed, uncertainties that may result in over-estimates of releases, and limitations in the representativeness of the estimates for all sites. Specifically, EPA used some input values from the Draft ESD; because the default values in the ESD are generic, there is uncertainty in the representativeness of the generic site estimates of real-world sites that use 1,4-dioxane. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods, contributing to uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

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E.4 Land Release Assessment

This section describes EPA's methodology for estimating annual land releases from industrial and commercial facilities manufacturing, processing, or using 1,4-dioxane. EPA did not estimate daily land releases due to the high level of uncertainty in the number of release days associated with land releases. Facilities report annual land releases to the Toxics Release Inventory (TRI), which include a variety of release mechanisms, including but not limited to underground injection, RCRA Subtitle C landfills, other landfills, surface impoundments, and land treatment. EPA used 2019 TRI (U.S. EPA, 2022g) data to estimate annual land releases for the OES where available; however, EPA did not have these data for every OES. For OES without TRI data, EPA used alternate assessment approaches to estimate annual land releases.

In addition, EPA did a more in-depth analysis of TRI for sites within the disposal OES. Specifically, EPA did an analysis of 2013 to 2019 TRI data for this OES. Operations at disposal sites are expected to be more complex than those at sites in other OES, which typically generate waste for land disposal off site. Additionally, the disposal OES includes the sites of ultimate disposal (*i.e.*, they are the landfills themselves) and EPA considered the impact of continuous years of land releases of 1,4-dioxane at these sites on general population and ecological exposures.

E.4.1 Assessment Using TRI

EPA found 2019 TRI data for facilities within the following OESs:

- Manufacturing
- Import and Repackaging
- Industrial Uses
- Functional Fluids (Open-System)
- Disposal
- PET Byproduct
- Ethoxylation Byproduct

The TRI data were rated "medium" in EPA's systematic review process. EPA estimated annual land releases using TRI for these OESs, with the following general stages as described in the rest of this section.

1. Collect land release data from the 2013 to 2019 TRI for the Disposal OES and 2019 TRI data for all other OES,
2. Map land release data to occupational exposure scenarios,
3. Analyze 2013 to 2019 TRI data for the disposal OES, and
4. Summarize 2019 annual land releases for the other OES.

Step 1: Collect Land Release Data from TRI

The first step in estimating land releases was to obtain TRI data. As previously discussed in Appendix E.3.1, each facility subject to the TRI reporting rule must report annually the volume of chemical released to the environment and/or managed through recycling, energy recovery, and treatment. Similar to the air release assessment, EPA included both TRI reporting Form R and TRI reporting Form A submissions in the land release assessment. Where sites reported to TRI with Form A, EPA used the Form A threshold for total releases of 500 lb/year. EPA used the entire 500 lb/year for each type of land release; however, since this threshold is for total site releases, these 500 lb/year are attributed one type of land release at a time (since assessing it for more than one land release media at a time would double count the releases and exceed the total release threshold for Form A). EPA pulled the TRI Basic Plus Data Files for each of years 2013 through 2019.

TRI data include both on- and off-site land releases. In summary, TRI includes the following land release media:

- On-site releases:
    - Underground injection
    - RCRA subtitle C landfills
    - Other landfills
    - Land treatment
    - RCRA surface impoundments
    - Other surface impoundments
    - Other land disposal
    - Waste rock
- Off-site releases:
    - Underground injection
    - RCRA subtitle C landfills
    - Other landfills
    - Land treatment
    - RCRA surface impoundments
    - Other surface impoundments
    - Other land disposal
    - Transfer to waste broker for disposal
    - Solidification/stabilization

Step 2: Map Land Release Data to Occupational Exposure Scenarios

The next step in estimating land releases was to map the 2013 to 2019 TRI data to the 1,4-dioxane OES. EPA used the same mapping methodology as that used for both the air and water assessments, which is described in Appendix E.5.1. EPA ensured consistency in the OES mapping between the air, water, and land assessments.

Step 3: Analyze and Summarize 2013 to 2019 TRI Data for the Disposal OES

For the sites that EPA mapped to the disposal OES in the 2013 to 2019 TRI data, EPA analyzed and summarized the land release data as follows:

- EPA summarized which of the reporting years that each disposal facility submitted data to TRI. This summary allows for visualization of which sites report recurring land disposal of 1,4-dioxane between 2013 and 2019.
- EPA differentiated between disposal sites that transferred 1,4-dioxane to other sites for disposal and the receiving sites that disposed of 1,4-dioxane on site. For the receiving sites at which 1,4-dioxane was disposed of to land, EPA summarized the number of unique sites from which the receiving sites received 1,4-dioxane for land disposal and the total amount of 1,4-dioxane received for land disposal between 2013 and 2019.
- EPA summarized the total amount of 1,4-dioxane released to each land release media between 2013 and 2019. In summary, 1,4-dioxane was disposed of from disposal OES sites via on-site and off-site RCRA subtitle C landfills, on-site and off-site underground injection, and off-site other landfills between 2013 and 2019.

EPA's analysis and summary of land releases for 2013 to 2019 TRI sites in the disposal OES can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for the Disposal OES* (U.S. EPA, 2023l).

Step 4: Summarize Annual Land Releases for Other OES with 2019 TRI data

For the remaining OES for which 2019 TRI data were available, EPA summarized the annual land releases by media type (*e.g.*, underground injection, RCRA subtitle C landfills, other landfills, land treatment) and site information, including site identity, city, state, zip code, TRI facility ID, and FRS ID. EPA did not estimate daily land releases due to the high level of uncertainty in the number of release days associated with land releases.

EPA's summary of land release for these OESs is included in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

E.4.2 Assessment for OES without TRI

EPA did not find 2019 TRI data for the following OES:

- Functional Fluids (Closed-Systems)
- Laboratory Chemicals
- Film Cement
- Spray Polyurethane Foam
- 3D Printing
- Dry Film Lubricant
- Textile Dye
- Antifreeze
- Surface Cleaner
- Dish Soap
- Dishwasher Detergent
- Laundry Detergent
- Paints and Floor Lacquer
- Hydraulic Fracturing

For these OESs, EPA estimated land releases by using various modeling approaches, including the use of surrogate TRI data and modeling using data from literature, GSs, and ESDs. EPA's assessment of land releases for each of these OESs is described below.

Functional Fluids (Closed-Systems)

Land release data were not available for this OES and EPA did not find any information to model land release for this OES using literature, GSs, or ESDs. EPA expects that the sources of release for this OES to be similar to those for the Industrial Uses OES, based on the process information in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). Therefore, EPA grouped the land release assessment for Functional Fluids (Closed-Systems) into that for Industrial Uses. However, there is uncertainty in this assumption of similar release sources between these OESs.

Laboratory Chemicals

EPA estimated land releases for facilities within the Laboratory Chemicals OES using the Draft GS on Use of Laboratory Chemicals (U.S. EPA, 2022h). The GS on Use of Laboratory Chemicals was rated "high" during EPA's systematic review process.

The GS indicates that there are eight release points:

1. Release to air from transferring volatile chemicals from transport containers.
2. Release to air, water, incineration, or landfill from transferring solid powders.
3. Release to water, incineration, or land from cleaning or disposal of transport containers.

4. Release to air from cleaning containers used for volatile chemicals.
5. Labware equipment cleaning residuals released to water, incineration, or landfill.
6. Release to air during labware equipment cleaning for volatile chemicals.
7. Release to air from laboratory analyses for volatile chemicals.
8. Release to water, incineration, or landfill from laboratory waste disposal.

Based on the GS, release points #2, 3, 5, and 8 have the potential for land releases; however, release point #2 is not applicable because 1,4-dioxane is not a solid powder. To estimate the remaining land releases, EPA used the equations specified in the Draft GS (U.S. EPA, 2022h). EPA did not find any data specific to 1,4-dioxane in this OES. Therefore, the calculation of releases using this GS are for a "generic site," using the default input parameter values from the GS.

Using this methodology, EPA calculated high-end and low-end annual land releases for this OES. The low- and high-end estimates are based on the low-end or typical and high-end or worst-case calculation input parameter defaults from the GS. EPA's calculation of land releases for this OES, including all calculation inputs, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

Film Cement

EPA estimated land releases for facilities within the Film Cement OES using process information from the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). The underlying process information for this assessment was rated "high" during EPA's systematic review process.

The process of using film cement involves applying the cement onto edges of photographic films by hand using a small brush, then joining the pieces of film by pressing and heating to dry the cement. Based on this process information, EPA expects land releases may result from disposal of empty film cement bottles that contain residual amounts of film cement containing 1,4-dioxane. EPA estimated this land release as a range, using a film cement use rate of 2.5 to 12 L/site-year and a concentration of 1,4-dioxane in the film cement of 45 to 50 percent from the process information in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), and the EPA/OPPT Small Container Residual Model central tendency loss fraction of 0.3 percent and high-end loss fraction of 0.6 percent. EPA is uncertain of the specific type of land disposal for the empty film cement bottles.

EPA's calculation of land releases for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

Spray Polyurethane Foam

EPA estimated land releases for facilities within the Spray Polyurethane Foam OES using the GS on Application of Spray Polyurethane Foam Insulation (U.S. EPA, 2020c). The GS on the Application of Spray Polyurethane Foam Insulation was rated "medium" during EPA's systematic review process.

The GS indicates that there are six release points:

1. Releases to fugitive air for volatile chemicals during unloading of raw materials from transport containers;
2. Releases to water, incineration, or landfill from cleaning or disposal of transport containers;
3. Releases to fugitive air for volatile chemicals during transport container cleaning;
4. Releases to incineration or landfill from spray polyurethane foam application equipment cleaning;

5. Releases to fugitive air for volatile chemicals during equipment cleaning; and
6. Releases to landfill of scrap foam from trimming applied foam.

Based on the GS, release points #2, 4, and 6 have the potential for land releases. To estimate these releases, EPA used the equations specified in the GS (U.S. EPA, 2020c). Apart from weight fraction in spray polyurethan foam, EPA did not find any data specific to 1,4-dioxane in this OES. Therefore, the calculation of releases using this GS are for a "generic site," using the default input parameter values from the GS. Specifically, EPA used the same input parameter values that were used in the original risk evaluation for estimates of occupational exposure; see Appendix G of the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Using this methodology, EPA calculated high-end and low-end annual land releases for this OES. The low- and high-end estimates are based on the low-end or typical and high-end or worst-case calculation input parameter defaults from the GS. EPA's calculation of land releases for this OES, including all calculation inputs, can be found *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

3D Printing

Land release data were not available for this OES and EPA did not find any information to model land releases for this OES using literature, GSs, or ESDs. EPA expects that industrial applications of this OES to be accounted for in the Industrial Uses TRI data. Per the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), 3D printing ink containing 1,4-dioxane is used in research labs to print biomedical products. Because the 2019 TRI data for the Industrial Uses OES include medicinal and pharmaceutical manufacturing NAICS codes, medical research labs that conduct 3D printing with 1,4-dioxane inks may be captured in that OES. Therefore, EPA grouped the land release assessment for 3D Printing into that for Industrial Uses. However, there is uncertainty in whether 3D printing sites are truly captured in the Industrial Uses TRI data.

Dry Film Lubricant

EPA estimated land releases for facilities within the Dry Film Lubricant OES using process information from the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). The underlying process information for this assessment was rated "high" during EPA's systematic review process.

The process for the production and use of dry film lubricant is described in the 2020 RE and is based on information provided to EPA by the one known user. In summary, the process entails first producing the concentrated dry film lubricant by mixing 1,4-dioxane with other additives, followed by the dilution of the concentrated dry film lubricant with additional 1,4-dioxane and the use of the dry film lubricant. The use involves spray application onto substrates in a vented paint booth and the subsequent curing in a vented oven and cleaning of the dried parts in a 1,4-dioxane bath (U.S. EPA, 2020c). Based on this process description, EPA expects land releases may result from

1. Residuals in empty containers of pure 1,4-dioxane used for mixing of the concentrated dry film lubricant,
2. Cleaning residuals for equipment used for mixing of the concentrated dry film lubricant,
3. Residuals in empty containers of pure 1,4-dioxane used for diluting the concentrated dry film lubricant,
4. Residuals in empty containers of concentrated dry film lubricant, and
5. Waste from cleaning spray application equipment and the parts onto which the dry film lubricant was applied.

EPA estimated land releases using 1,4-dioxane use rates derived from the process information and standard EPA models. Specifically, EPA estimated land releases from release points #1, 3 and 4 using the EPA/OPPT Small Container Residual Model, which has a central tendency loss fraction of 0.3 percent and a high-end loss fraction of 0.6 percent of the container volume. EPA used container volumes specified in the process information, which are either 1-gallon or 0.5-pint containers (U.S. EPA, 2020c). EPA estimated releases from release point #2 using the EPA/OPPT Single Process Vessel Residual Model, which has a central tendency loss fraction of 0.2 percent and a high-end loss fraction of 1 percent of the 1,4-dioxane throughput. EPA estimated land releases from release point #5 by assuming the entire volume of the cleaning bath used for equipment and parts is released to landfill. This is consistent with the process information, which indicates that spent 1,4-dioxane is disposed of as chemical waste, which EPA assumes may be to either incineration or RCRA subpart C landfills (U.S. EPA, 2020c).

EPA's calculation of land releases for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

Textile Dye

EPA estimated land releases for facilities within the Textile Dye OES using the OECD ESD on Textile Dyes (OECD, 2017) and Monte Carlo modeling. The ESD on Textile Dyes was rated "medium" during EPA's systematic review process. The use of Montel Carlo modeling allows for variation of calculation input parameters such that a distribution of environmental releases can be calculated, from which EPA can estimate the 50th and 95th percentile releases. An explanation of this modeling approach is included in Appendix E.11.

Antifreeze

EPA did not find any directly applicable GS/ESD or literature sources for this OES; however, EPA evaluated the potential for releases using the OECD ESD on Chemical Additives used in Automotive Lubricants (OECD, 2020) and the EPA MRD on Commercial Use of Automotive Detailing Products (U.S. EPA, 2022b). The ESD and MRD were both rated "high" during EPA's systematic review process.

For the use of antifreeze, EPA expects releases may occur from volatilizations of 1,4-dioxane during unloading/ pouring antifreeze into vehicles, disposal, or cleaning of empty antifreeze containers, and spent antifreeze. Both the ESD and MRD indicate that containers of automotive maintenance fluids are typically small and are not rinsed, but rather disposed of as solid waste (U.S. EPA, 2022b; OECD, 2020). Additionally, the ESD on Chemical Additives used in Automotive Lubricants indicates that spent lubricants are disposed of via incineration by blending with fuel oil (OECD, 2020). However, since spent antifreeze is unlikely to be blended with fuel oil, EPA expects spent antifreeze may be disposed of via incineration or landfills that take chemical waste. Therefore, EPA expects land releases result from disposal of empty antifreeze containers and spent antifreeze.

To estimate the use rate of 1,4-dioxane for this OES, EPA used the consumer use rate of antifreeze (0.15 kg antifreeze/job) from the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c) and scaled this value up to a commercial use rate based on a range of the number of cars serviced at auto shops from the Near-Field/Far-Field Brake Model and Automotive Detailing MRD (1 to 9 jobs/day). EPA used a range of concentration of 1,4-dioxane in antifreeze from the process description in Appendix F.4.2 and assumed antifreeze container sizes ranging from 16 ounces to 5 gallons per the default container sizes in the MRD and ESD, respectively (U.S. EPA, 2022b; OECD, 2020).

To estimate the land release from container disposal, EPA used the calculated 1,4-dioxane throughput based on the above batch parameters and the EPA/OPPT Small Container Residual Model, which has a central tendency loss fraction of 0.3 percent and a high-end loss fraction of 0.6 percent. To estimate the land release from spent antifreeze, EPA used the 1,4-dioxane throughput and a mass balance assuming 100 percent release minus upstream losses from container disposal and volatilizations during unloading (estimated with the EPA/OAQPS AP-42 Loading Model).

EPA's calculation of land releases for this OES, including all calculation inputs and assumptions, can be found in the supplemental attachment *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

Surface Cleaner

EPA did not find any directly applicable GS/ESD or literature sources for this OES; however, EPA estimated land releases using the SHEDs-HT modeling conducted for the one case study location (Liverpool OH) and the assumptions described here. EPA expects that the main release points from the use of surface cleaners are from

1. Disposal of empty containers containing residual cleaning solution,
2. Application of the cleaning solution, and
3. Disposal of cleaning solution by rinsing or wiping.

Because EPA did not find any directly applicable GSs or ESDs, EPA used the Draft GS on Furnishing Cleaning (U.S. EPA, 2022a) to inform these releases due to the similarities in surface cleaning and furnishing cleaning. The Draft GS on Furnishing Cleaning was rated "high" during EPA's systematic review process. Per this Draft GS, empty containers may be rinsed out in sinks or disposed of without rinsing, such that releases may be to wastewater or landfill; the GS uses the EPA/OPPT Small Container Residual Model to estimate this release. Application losses are to fugitive air from spray application; the GS uses literature data to estimate this release. Once applied, the cleaner may be rinsed off or wiped off with rags or towels, such that releases may be to wastewater or landfill; the GS assumes 100 percent release scenario, estimating this release by subtracting the upstream losses from the cleaner use rate (U.S. EPA, 2022a).

The SHEDs-HT modeling estimated wastewater discharges of 72 g of 1,4-dioxane per day for commercial uses of surface cleaners containing 1,4-dioxane in Liverpool OH. As described previously, because both release point #1 and #3 may also be to either wastewater or landfills, EPA assumes the same quantity of 72 g of 1,4-dioxane per day from the SHEDs-HT model may be released to unknown landfills for this OES. EPA notes that these 72 g is either entirely to wastewater or landfill or some split between the two media. The 72 g is not to both wastewater and landfill because that would double count the release,

EPA's calculation of land releases for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

Dish Soap

EPA did not find any directly applicable GS/ESD or literature sources for this OES; however, EPA estimated land releases using the SHEDs-HT modeling conducted for the one case study location (Liverpool OH) and the assumptions described here. EPA expects that the main release points from the use of dish soaps are from:

1. Disposal of empty containers containing residual dish soap, and
2. Cleaning and rinsing of dishes.

EPA expects that empty containers may be rinsed out in sinks or disposed of without rinsing, such that releases may be to wastewater or landfill. Further, EPA expects that the entire amount of dish soap used for cleaning dishes is rinsed down the drain of sinks during the cleaning and rinsing process. EPA uses the SHEDs-HT modeled estimated of wastewater discharges for this OES (64 g 1,4-dioxane per day for Liverpool OH) and back calculates a 1,4-dioxane use rate using the EPA/OPPT Small Container Residual Model central tendency and high-end loss fractions and an assumption of 100 percent release.

Using this back-calculated 1,4-dioxane use rate, EPA then applied the EPA/OPPT Small Container Residual Model to estimate land releases for the Liverpool OH case study. EPA expects that this land release is to unknown landfills. EPA's calculation of land releases for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

Dishwasher Detergent

EPA did not find any directly applicable GS/ESD or literature sources for this OES; however, EPA estimated land releases using the SHEDs-HT modeling conducted for the one case study location (Liverpool OH) and the same assumptions as that described for the dish soap OES above. EPA's calculation of land releases for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

Laundry Detergent

EPA estimated land releases for facilities within the Laundry Detergent OES using the OECD ESD on Industrial and Institutional Laundries (OECD, 2011b) and Monte Carlo modeling. The ESD on Industrial and Institutional Laundries was rated "medium" during EPA's systematic review process. The use of Montel Carlo modeling allows for variation of calculation input parameters such that a distribution of environmental releases can be calculated, from which EPA can estimate the 50th and 95th percentile releases. An explanation of this modeling approach is included in Appendix E.12.

Paints and Floor Lacquer

EPA estimated land releases for facilities within the Paints and Floor Lacquers OES using the OECD ESD on Coating Application via Spray-Painting in the Automotive Refinishing Industry (OECD, 2011a). The ESD was rated "medium" during EPA's systematic review process.

As described in the process description in Appendix F.4.7, 1,4-dioxane was identified by a public comment as present in automotive refinishing products, thereby allowing EPA to identify the above ESD as the most applicable GS/ESD available. This ESD indicates that releases are expected from

1. Releases to incineration or landfill from container cleaning/disposal,
2. Releases to incineration or landfill from equipment cleaning,
3. Releases to incineration or landfill from over sprayed coating that is captured by emission controls, and

4. Releases to stack air from over sprayed coating that is not captured by emission controls.

Based on the GS, release points #1 through 3 have the potential for land releases. To estimate these releases, EPA used the equations specified in the ESD (OECD, 2011a). Apart from weight fraction in coatings (see Appendix F.4.7), EPA did not find any data specific to 1,4-dioxane in this OES. Therefore, the calculation of releases using this GS are for a "generic site," using the default input parameter values from the ESD.

Using this methodology, EPA calculated the low-end and high-end land releases for this OES, which are expected to be to unknown landfills per the ESD (OECD, 2011a). The low- and high-end estimates are based on the low- and high-end calculation input parameter defaults from the ESD. EPA's calculation of land releases for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Land for all OES Except Disposal* (U.S. EPA, 2023k).

Hydraulic Fracturing

EPA estimated land releases for facilities within the Hydraulic Fracturing OES using the Draft OECD ESD on Hydraulic Fracturing (U.S. EPA, 2022d) and Monte Carlo modeling. The Draft ESD on Hydraulic Fracturing was rated "high" during EPA's systematic review process. The use of Montel Carlo modeling allows for variation of calculation input parameters such that a distribution of environmental releases can be calculated, from which EPA can estimate the 50th and 95th percentile releases. An explanation of this modeling approach is included in Appendix E.13.

E.4.3 Land Release Estimates Summary

A summary of industrial and commercial land releases estimated using the above methods is presented in Table_Apx E-5 below.

Table_Apx E-5. Summary of Daily Industrial and Commercial Land Release Estimates for 1,4-Dioxane

OES	Type of Land Release	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Manufacturing	Land (all types)	1	0		250	Medium	TRI
Import and Repackaging	Land (all types)	1	0		250	Medium	TRI
Industrial Uses	Land (all types)	12	0	227 (annually)	250	Medium	TRI
Functional Fluids (Open-System)	Land (all types)	2	0	0	247	Medium	TRI
Functional Fluids (Closed-System)	All	Assessed as a part of Industrial Uses OES				N/A	N/A
Laboratory Chemical	Land (unknown type)	132	0	489 (annually)	250	High	GS[d]
Film Cement	Land (unknown type)	211	0.0035 (annually)	0.037 (annually)	250	High	Process information[e]
Spray Foam Application	Land (unknown type)	1,553,559	0.032 (annually)	0.047 (annually)	3	Medium	GS[f]
Printing Inks (3D)	Fugitive Air, Stack Air, and Land (all types)	Assessed as a part of Industrial Uses OES			250	N/A	N/A
Dry Film Lubricant	Land (hazardous waste landfill)	8	0	188 (annually)	48	High	Process information[e]

OES	Type of Land Release	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Disposal	Land (RCRA Sub C landfill)	18	0	7,307 (annually)	250	Medium	TRI
	Land (Underground injection)	18	0	331,980 (annually)		Medium	TRI
	Land (Non-RCRA landfills)	18	0	890 (annually)		Medium	TRI
	Land (all other types)	18	0	0		Medium	TRI
Textile Dye	Land (unknown landfill type) or POTW (unknown partitioning)	783	2.09E−07	9.72E−05	31 to 295	Medium	ESD[g] and Modeling[h]
Antifreeze	Land (unknown landfill)	84,383	3.75E−07 (annually)	0.029 (annually)	250	High	Process information[e] and Modeling[h]
Surface Cleaner	Land (unknown landfill) or POTW	Unknown	18[m]		250	High	SHEDS-HT[l], Process information[e] Modeling[h]
Dish Soap	Land (unknown landfill)	Unknown	0.048 (annual value for all sites in Liverpool OH case study)	0.097 (annual value for all sites in Liverpool OH case study)	250	High	SHEDS-HT[l], Process information[e] Modeling[h]
Dishwasher Detergent	Land (unknown landfill)	Unknown	1.08E−03 (annual value for all sites in Liverpool, OH case study)	2.17E−03 (annual value for all sites in Liverpool, OH case study)	250	High	SHEDS-HT[l], Process information[e] Modeling[h]
Laundry Detergent (Institutional) – liquid detergents	Land (unknown landfill), incineration,	95,533	4.05E−12	3.95E−05	250 to 365	Medium	ESD[i] and Modeling[h]

OES	Type of Land Release	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
	or POTW (unknown partitioning)						
Laundry Detergent (Institutional) – powder detergents	Land (unknown landfill), incineration, or POTW (unknown partitioning)	95,533	5.36E−08	0.0018	250 to 365	Medium	ESD[i] and Modeling[h]
Laundry Detergent (Industrial) – liquid detergents	Land (unknown landfill), incineration, or POTW (unknown partitioning)	2,453	4.78E−12	1.46E−04	20 to 365	Medium	ESD[i] and Modeling[h]
Laundry Detergent (Industrial) – powder detergents	Land (unknown landfill), incineration, or POTW (unknown partitioning)	2,453	2.92E−11	3.92E−04	20 to 365	Medium	ESD[i] and Modeling[h]
Paints and Floor Lacquer	Land (unknown landfill)	33,648	3.04E−06 (annually)	0.010 (annually)	250	Medium	ESD[j]
Polyethylene Terephthalate (PET) Byproduct	Land (Land treatment)	13	0	45.4 (annually)	250	Medium	TRI
	Land (Non-RCRA landfills)	13	0	0.10 (annually)		Medium	TRI
	Land (all other types)	13	0	0		Medium	TRI
Ethoxylation Process Byproduct	Land (underground injection)	8	0	197,714 (annually)	250	Medium	TRI
	Land (all other types)	8	0	0		Medium	TRI

OES	Type of Land Release	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Hydraulic Fracturing	Surface water, incineration, or landfill (unknown partitioning)	411	3.61E−10	4.59	1 to 72	Medium	ESD[k] and Modeling[h]
	Land (underground injection)	411	5.35E−09	108		Medium	ESD[k] and Modeling[h]
	Recycle/Reuse (48%), underground injection (43%), Surface water (6%), or land (3%)	411	1.85E−10	1.12		Medium	ESD[k] and Modeling[h]

[a] Where available, EPA used 2013–2019 TRI (U.S. EPA, 2022g) and 2013–2019 DMR (U.S. EPA, 2022c) data to provide a basis to estimate the number of sites using 1,4-dioxane within a COU.
[b] Where available, EPA used the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), generic scenarios, and emission scenario documents to provide a basis to estimate the number of release days of 1,4-dioxane within a COU.
[c] Narrative descriptions of all release estimate sources are provided in Appendix E.4.2.
[d] The generic scenario used for this COU is the GS on Use of Laboratory Chemicals (U.S. EPA, 2022h).
[e] For this COU, EPA used process information, which is further described in Appendix E.4.2.
[f] The generic scenario used for this COU is the GS on Application of Spray Polyurethane Foam Insulation (U.S. EPA, 2018b).
[g] The emission scenario document used for this COU is the ESD on Textile Dyes (OECD, 2017).
[h] For this COU, EPA used various models and literature for model input parameters as described in Appendix E.4.2.
[I] The emission scenario document used for this COU is the ESD on Industrial and Institutional Laundries (OECD, 2011b).
[j] The emission scenario document used for this COU is the ESD on Coating Application via Spray Painting in the Automotive Refinishing Industry (OECD, 2011a).
[k] The emission scenario document used for this COU is the Draft ESD on Hydraulic Fracturing (U.S. EPA, 2022d).
[l] EPA used the down the drain water release estimates from the SHEDs-HT model for the Liverpool OH case study (see Section 2.1.1.2) to estimate air and land releases by back calculating 1,4-dioxane use rates and applying loss fractions for air and land releases using literature and standard models described in Appendix E.4.2.
[m] A single annual value was provided for all sites in the Liverpool, OH case study.

E.4.4 Summary of Weight of the Scientific Evidence Conclusions in Land Release Estimates

Table_Apx E-6 provides a summary of EPA's weight of the scientific evidence conclusions in its land release estimates for each of the Occupational Exposure Scenarios assessed. Detailed descriptions of non-OES specific strengths, limitations, assumptions, and uncertainties (*e.g.*, general limitations for TRI, DMR, etc.) are provided in Appendix E.6.

Table_Apx E-6. Summary of Weight of the Scientific Evidence Conclusions in Land Release Estimates by OES

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Manufacturing	Land releases are assessed using reported releases from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include the low number of data points, lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Additionally, EPA could not estimate the number of release days per year associated with land releases. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Import and Repackaging	Land releases are assessed using reported releases from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include the low number of data points, lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Additionally, the land release assessment is based on one reporting site that reported no land releases and EPA did not have additional sources to estimate land releases for other sites in this OES. Additionally, EPA could not estimate the number of release days per year associated with land releases. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Industrial Uses	Land releases are assessed using reported releases from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include the lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, uncertainty in EPA's use of Form A submissions, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Some facilities within this OES reported to TRI using a Form A, which does not include any details on chemical release quantities. When a facility has submitted a Form A, there is no way to discern the quantity released. Therefore, where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media; however, there is uncertainty in this because the actual release quantity is unknown. Furthermore, the threshold represents an upper limit on total releases from the facility; therefore, assessing releases at the threshold value may overestimate actual releases from the facility. Additionally, EPA could not estimate the number of release days per year associated with land releases. Based on this information, EPA has concluded

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Functional Fluids (Open-System)	Land releases are assessed using reported releases from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include the low number of data points, lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. The land release assessment is based on two reporting sites that both reported no land releases and EPA did not have additional sources to estimate land releases for sites in this OES. Additionally, EPA could not estimate the number of release days per year associated with land releases. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Functional Fluids (Closed-System)	No data was available to estimate releases for this OES. For the land release assessment, EPA grouped this OES with the Industrial Uses OES because the sources of release are expected to be similar between these OESs. Factors that increase the strength of evidence for this OES are that TRI has a medium overall data quality determination and consistency within the dataset (all reporters use the same or similar reporting forms). Factors that decrease the strength of evidence for this OES are that the Industrial Releases OES release data are use as surrogate for this OES, uncertainty in the accuracy of reported releases, limitations in representativeness to all sites because TRI may not capture all relevant sites, and lack of variability (only one year of data used). Refer to the Industrial Uses OES discussion for additional discussion. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Laboratory Chemicals	Land releases are assessed using the Draft GS on Use of Laboratory Chemicals. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the Draft GS on Use of Laboratory Chemicals has a high overall data quality determination, and the low level of uncertainty in the data. Factors that decrease the strength of the evidence for this OES include the that the GS has not been peer-reviewed, uncertainty in the representativeness of the GS towards all sites in this OES, and a lack of variability in the analysis. Specifically, because the default values in the ESD are generic, there is uncertainty in the representativeness of generic site estimates of actual releases from real-world sites that use 1,4-dioxane. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods, contributing to uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Film Cement	Land releases are assessed using process information from the *Final Risk Evaluation for 1,4-Dioxane* and EPA/OPPT models. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	OES (as opposed to surrogate), the underlying data sources for the process information have a high overall data quality determination, and the low level of uncertainty in the data because the process information comes directly from actual users of 1,4-dioxane in film cement. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of evidence to all sites in this OES and a lack of variability in the input parameters for the used models. Specifically, the process information for the production and use of film cement is based on information from three use sites, one from Australia and two from the U.S. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Spray Foam Application	Land releases are assessed using the GS on Application of Spray Polyurethane Foam Insulation. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the underlying data sources for the process information have a medium overall data quality determination, and a low level of uncertainty in the data. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of the GS to all sites since it is generic and not specific to sites that use 1,4-dioxane and a lack of variability. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Printing Inks (3D)	No data was available to estimate releases for this OES. For the land release assessment, EPA grouped this OES with the Industrial Uses OES because the sources of release are expected to be similar between these OESs. Factors that increase the strength of evidence for this OES are that TRI has a medium overall data quality determination and consistency within the dataset (all reporters use the same or similar reporting forms). Factors that decrease the strength of evidence for this OES are that the Industrial Releases OES release data are use as surrogate for this OES, uncertainty in the accuracy of reported releases, limitations in representativeness to all sites because TRI may not capture all relevant sites or smaller commercial 3D printing uses, and lack of variability (only one year of data used). Refer to the Industrial Uses OES discussion for additional discussion. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Dry Film Lubricant	Land releases are assessed using process information from the *Final Risk Evaluation for 1,4-Dioxane*. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), that the underlying data sources for the process information have a high overall data quality determination, and a low level of uncertainty in the data because the process information comes directly from an actual user of 1,4-dioxane in dry film lubricants. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of evidence to all sites and a lack of variability. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Disposal	Land releases are assessed using reported discharges from 2013–2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, high number of data points, and consistency within the dataset (all reporters use the same or similar reporting forms). Additionally, EPA included seven years of TRI data in the analysis, which increases the variability of the dataset. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include uncertainty in the accuracy of reported releases, uncertainty in EPA's use of Form A submissions, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Some facilities within this OES reported to TRI using a Form A, which does not include any details on chemical release quantities. When a facility has submitted a Form A, there is no way to discern the quantity released. Therefore, where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media; however, there is uncertainty in this because the actual release quantity is unknown. Furthermore, the threshold represents an upper limit on total releases from the facility; therefore, assessing releases at the threshold value may overestimate actual releases from the facility. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Textile Dye	Land releases are assessed using Monte Carlo modeling with information from the ESD on Textile Dyes. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the ESD on Textile Dyes has a medium overall data quality determination and was peer reviewed, the high number of data points (simulation runs), consistency within the dataset, and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential release values that represents a larger proportion of sites than a discrete value. Factors that decrease the strength of the evidence for this OES include uncertainties and limitations in the representativeness of the estimates for sites that specifically use 1,4-dioxane because the default values in the ESD are generic. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods, contributing to uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Antifreeze	Land releases are assessed using the OECD ESD on Chemical Additives used in Automotive Lubricants, the EPA MRD on Commercial Use of Automotive Detailing Products, and EPA/OPPT models. Factors that increase the strength of evidence for this OES are that the ESD and MRD used have high overall data quality determinations and consistency within the sources used. Factors that decrease the strength of the evidence for this OES include that the ESD and MRD are not directly applicable to antifreeze uses (used as surrogate), uncertainty in the representativeness of the ESD and MRD to all sites and sites that specifically use 1,4-dioxane since these documents contain generic values, and a lack of variability. Additionally, EPA scaled up a consumer antifreeze use rate to a commercial use rate based on information in the ESD and MRD, which increases uncertainty. Based on this information, EPA has concluded that the weight of the scientific

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Surface Cleaner	Land releases are assessed using SHEDS-HT modeled water releases in conjunction with the Draft GS on Furnishing Cleaning. Factors that increase the strength of evidence for this OES include that the release estimates are directly relevant to the OES (as opposed to surrogate), that the Draft GS used has a high overall data quality determination, and variability in the model input parameters. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness to all sites because the SHEDS-HT estimate is based on one case study for Liverpool, OH and because the estimate is not site-specific (the release estimate is a total for all sites in Liverpool, OH). Additionally, the Draft GS describes potential release points for this OES, identifying releases that may be to either water or land depending on site practices (*e.g.*, surface cleaner may be rinsed down drains or wiped off with rags that are disposed of as trash). Because there is no information to determine the quantity released specifically to land, EPA assumed that the entire quantity modeled to water with the SHEDS-HT model may also be released to land, which introduces uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Dish Soap	Land releases are assessed using the SHEDS-HT modeled water releases, the expected sources of release for this OES, and EPA/OPPT models. Factors that increase the strength of evidence for this OES include that the release estimates are directly relevant to the OES (as opposed to surrogate) and variability in the model input parameters. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness to all sites because the SHEDS-HT estimate is based on one case study for Liverpool, OH and because the estimate is not site-specific (the release estimate is a total for all sites in Liverpool, OH). Additionally, based on the quantity released to water and the expected loss fraction for water releases, EPA back-calculated a 1,4-dioxane use rate and applied the expected land release loss fraction to estimate land releases, which introduces uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Dishwasher Detergent	EPA used the same approach as described above for the dish soap OES to estimate land releases for the dishwasher detergent OES.
Laundry Detergent	Land releases are assessed using Monte Carlo modeling with information from the ESD on Industrial and Institutional Laundries. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), that the ESD on Industrial and Institutional Laundries has a medium overall data quality determination and was peer reviewed, there are high number of data points (simulation runs), consistency within the dataset, and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential release values that represents a larger proportion of sites than a discrete value. Additionally, EPA was able to separately estimate releases for industrial and institutional laundry settings. Factors that decrease the strength of the evidence for this OES include uncertainties and limitations in the representativeness of the estimates for sites that specifically use 1,4-dioxane because the default values in the ESD are

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	generic. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Paint and Floor Lacquer	Land releases are assessed using OECD ESD on Coating Application via Spray-Painting in the Automotive Refinishing Industry. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the ESD has a medium overall data quality determination and has been peer reviewed, consistency within the sources used, and a low amount of uncertainties. Factors that decrease the strength of the evidence for this OES include a lack of variability and uncertainty in the representativeness of the ESD to all sites and sites that specifically use 1,4-dioxane since the ESD is generic. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
PET Byproduct	Land releases are assessed using reported discharges from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include uncertainty in the accuracy of reported releases, lack of variability (only one year of data used), and the limitations in representativeness to all sites because TRI may not capture all relevant sites. The land release assessment is based on 13 reporting sites, 11 of which reported no land releases. EPA did not have additional sources to estimate land releases for site in this OES that may not report to TRI. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Ethoxylation Process Byproduct	Land releases are assessed using reported discharges from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include uncertainty in the accuracy of reported releases, lack of variability (only one year of data used), and the limitations in representativeness to all sites because TRI may not capture all relevant sites. The land release assessment is based on eight reporting sites, seven of which reported no land releases. EPA did not have additional sources to estimate land releases for site in this OES that may not report to TRI. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Hydraulic Fracturing	Land releases are assessed using Monte Carlo modeling with information from the Draft ESD on Hydraulic Fracturing and FracFocus 3.0. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), that the Draft ESD on Hydraulic Fracturing and FracFocus 3.0 have medium overall

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	data quality determinations, high number of data points (simulation runs), consistency within the dataset, and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential release values that represents a larger proportion of sites than a discrete value. Factors that decrease the strength of the evidence for this OES include that the Draft ESD has not been peer reviewed and the uncertainties and limitations in the representativeness of the estimates for sites that specifically use 1,4-dioxane because the default values from the Draft ESD on Hydraulic Fracturing. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods, contributing to uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

E.5 Air Release Assessment

This section describes EPA's methodology for estimating daily fugitive and stack air emissions from industrial and commercial facilities manufacturing, processing, or using 1,4-dioxane. Facilities report air emissions to the TRI. EPA used 2019 TRI (U.S. EPA, 2022g) data to estimate daily air emissions for each OES where available; however, EPA did not have these data for every OES. For OES without TRI data, EPA used alternate assessment approaches to estimate air emissions. These approaches are described below.

E.5.1 Assessment Using TRI

EPA found 2019 TRI data for facilities within the following OESs:

- Manufacturing
- Import and repackaging
- Industrial uses
- Functional fluids (open-system)
- Disposal
- PET byproduct
- Ethoxylation byproduct

The 2019 TRI data were rated "medium" in EPA's systematic review process. EPA estimated daily air emissions using TRI data for these OESs, with the following general steps as described in the rest of this section.

1. Collect air emission data from 2019 TRI data,
2. Map air emission data to occupational exposure scenarios,
3. Estimate the number of facility operating days per year, and
4. Estimate daily air emissions and prepare a summary of the air emissions for each OES.

Note that EPA compared the TRI data used to estimate air releases for the PET byproduct OES in this risk evaluation to information from a life cycle analysis on the PET manufacturing process in Appendix E.6.

Step 1: Collect Air Emission Data TRI

The first step in the methodology for estimating air emissions was to obtain 2019 TRI data for the chemical from EPA's Basic Plus Data Files. TRI requires U.S. facilities in various industry sectors to report the annual release volumes to the environment through air emissions, water discharges, and land disposal, and/or managed through recycling, energy recovery, and treatment, including by off-site transfers. TRI reporters may report either with a Form R or a Form A. Facilities must report with a Form R, which requires reporting of release quantities and uses/sub-uses of the chemical, among other information, unless they meet the alternate threshold requirements for submitting a Form A. Specifically, facilities may submit a Form A if the volume of chemical manufactured, processed, or otherwise used does not exceed 1,000,000 lb per year (lb/year) and the total annual reportable releases do not exceed 500 lb/year. Facilities do not need to report release quantities or uses/sub-uses on Form A. EPA included both TRI reporting Form R and TRI reporting Form A submissions in the air release assessment.

Air emissions in TRI are reported separately for stack air and fugitive air and always occur on-site at the facility. Where sites reported to 2019 TRI with Form A, EPA used the Form A threshold for total releases of 500 lb/year. EPA used the entire 500 lb/year for both the fugitive and stack air release estimates; however, since this threshold is for total site releases, these 500 lb/year are attributed either to fugitive air or stack air for this analysis—not both (to avoid double counting the releases and exceeding the total release threshold for Form A).

Step 2: Map Air Emission Data to Occupational Exposure Scenarios

In the next step of air release assessment, EPA mapped the chemical's 2019 TRI data to the 1,4-dioxane OES. EPA used the following procedure to map 2019 TRI data to OES:

1. *Compile TRI uses/sub-uses:* EPA first compiled all the reported TRI uses/sub-uses for each facility into one column.
2. *Map TRI uses/sub-uses to Chemical Data Reporting (CDR) IFC codes:* EPA then mapped the TRI uses/sub-uses for each facility to one or more 2016 CDR Industrial Function Category (IFC) codes using the TRI-to-CDR Use Mapping crosswalk (see Appendix E.9).
3. *Map OES to CDR IFC codes:* EPA then mapped each COU/OES combination to a 2016 CDR IFC code and description. The basis for this mapping was generally the COU category and subcategory.
4. *Map TRI facilities to an OES:* Using the CDR IFC codes from Step 2 and the COU-CDR Mapping from Step 3, EPA mapped each TRI facility to an OES. EPA's rationale for the OES determination is generally described below.
    - In some cases, the facility mapped to only one OES and the mapping appeared to be correct given the facility name and NAICS code. For these, the OES as mapped from Steps 2 and 3 was used without adjustment.
    - There were instances where a facility mapped to multiple OESs which required some engineering judgement to identify a primary OES. EPA documented the rationale for these determinations for each facility in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j). In summary, these determinations were made with the following considerations:
        - Industry and NAICS codes reported in TRI (*e.g.*, for a facility that reported TRI uses for both waste treatment and ancillary use, EPA assigned the Disposal OES if the NAICS code was 562211, Hazardous Waste Treatment and Disposal);
        - Internet research of the types of products manufactured at the facility (*e.g.*, if a facility's website indicates the facility manufactures PET, the facility is likely to produce 1,4-dioxane as a byproduct in PET manufacturing); and
        - Grouping of similar OES (*e.g.*, for facilities that reported the sub-use of chemical processing aid, process solvent, or processing as a reactant), EPA assigned the Industrial Uses OES because this includes multiple processes as described in the 2020 RE (U.S. EPA, 2020c)).
    - In some cases, EPA identified that there were instances where the preliminary OES mapping from the TRI use/sub-use – CDR IFC code required re-mapping. This re-mapping is a result of limitations of the TRI-to-CDR Use Mapping crosswalk. For example, the crosswalk maps the TRI use/sub-use of "Otherwise Use as Manufacturing Aid (Other)" to only CDR IFC codes U013 (closed-system functional fluids) and U023 (plating agents and surface treating agents); however, this TRI use/sub-use may encompass multiple other uses that are not captured in these CDR IFC codes. In these cases, EPA reviewed the reported NAICS codes and conducted internet research on the types of products manufactured at the facility to determine the likely OES.
    - Additionally, EPA reviewed 2016 CDR (U.S. EPA, 2016b) for sites that reported manufacturing (including importing) of 1,4-dioxane. If the sites that reported to 2016 CDR also reported in 2019 TRI, EPA assigned the OES according to 2016 CDR. Note

that some sites that reported to 2019 TRI may not be in 2016 CDR (*e.g.*, sites that manufacture the chemical as a byproduct). In these cases, EPA determined the OES using only the above bulleted steps.

5. *Form A's:* For Form A submissions, there were no reported TRI uses/sub-uses. To determine the COU for these facilities, EPA used 2016 CDR as described above, the NAICS codes, and internet searches to determine the type of products and operations at the facility.

The specific rationale for the OES mapping for each facility is described in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

Step 3: Estimate the Number of Facility Operating Days per Year

EPA then estimated the number of operating days (days/year) for each facility reporting air emissions to TRI. For the OES that were included in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), EPA used the number of operating days from that risk evaluation. For the additional OES included in this supplemental risk evaluation, EPA estimated the number of operating days using the methodology described in Appendix E.2.

Step 4: Estimate Daily Air Emissions and Summarize Air Emissions for each OES

The final step was to prepare a summary of the fugitive and stack releases. For each OES and facility mapped to that OES, EPA summarized the annual fugitive and stack air emissions reported in 2019 TRI and daily fugitive and stack air emissions that EPA estimated by dividing the annual emissions by the number of operating days determined for the OES in Step 3. EPA also summarized site information, including site identity, city, state, zip code, TRI facility ID, and Facility Registry Service (FRS) ID because the subsequent exposure modeling is site and location specific. Latitude and longitude coordinates are included in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j) but not in the summary tables.

To accompany the summary table for each OES, EPA also provided any reasonably available information on the release duration and pattern, which are needed for the exposure modeling. Release duration is the expected amount of time per day during which the air releases may occur. Release pattern is the temporal variation of the air release, such as over consecutive days throughout the year, over cycles that occur intermittently throughout the year, or in a puff/instantaneous release that occurs over a short duration. The TRI dataset does not include release pattern or duration; therefore, EPA used information from models or literature, where available. For release pattern, EPA provided the number of release days with the associated basis as described in Step 3. However, for most OES, no information was found on release duration and pattern. In such cases, EPA listed the release duration and pattern as "unknown."

EPA's summary of air releases for each OES is included in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

E.5.2 Assessment for OESs without TRI

EPA did not find TRI data for any of the years included in this analysis for the following OESs:

- Functional fluids (closed-systems)
- Laboratory chemicals
- Film cement
- Spray polyurethane foam
- 3D printing
- Dry film lubricant

- Textile dye
- Antifreeze
- Surface cleaner
- Dish soap
- Dishwasher detergent
- Laundry detergent
- Paints and floor lacquer
- Hydraulic fracturing

For these OESs, EPA estimated air emissions by using various modeling approaches, including the use of surrogate TRI data and data from literature, GSs, and ESDs. EPA's assessment of air emissions for each of these OESs is described below.

Functional Fluids (Closed-Systems)

Air emission data were not available for this OES and EPA did not find any information to model air emissions for this OES using literature, GSs, or ESDs. EPA expects that the sources of release for this OES to be similar to those for the Industrial Uses OES, based on the process information in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). Therefore, EPA grouped the air release assessment for Functional Fluids (Closed-Systems) into the OES for Industrial Uses. However, there is uncertainty in the assumption of similar release sources between these OESs.

Laboratory Chemicals

EPA estimated air emissions for facilities within the Laboratory Chemicals OES using the Draft GS on Use of Laboratory Chemicals (U.S. EPA, 2022h). The Draft GS on Use of Laboratory Chemicals was rated "high" during EPA's systematic review process.

The GS indicates that there are eight release points:

1. Release to air from transferring volatile chemicals from transport containers
2. Release to air, water, incineration, or landfill from transferring solid powders
3. Release to water, incineration, or land from cleaning or disposal of transport containers
4. Release to air from cleaning containers used for volatile chemicals
5. Labware equipment cleaning residuals released to water, incineration, or landfill
6. Release to air during labware equipment cleaning for volatile chemicals
7. Release to air from laboratory analyses for volatile chemicals
8. Release to water, incineration, or landfill from laboratory waste disposal

Based on the GS, release points #1, 2, 4, 6, and 7 have the potential for air emissions; however, release point #2 is not applicable because 1,4-dioxane is not a solid powder. To estimate the remaining air releases, EPA used the equations specified in the Draft GS (U.S. EPA, 2022h). EPA did not find any data specific to 1,4-dioxane in this OES. Therefore, the calculation of releases using this GS are for a "generic site," using the default input parameter values from the GS.

Using this methodology, EPA calculated the "typical" and "worst-case" air emissions for this OES. These characterizations are based on the GS, which provides default "typical" and "worst-case" input parameters for the release calculations. EPA's calculation of air emissions for this OES, including all calculation inputs, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

Film Cement

EPA estimated air emissions for facilities within the Film Cement OES using process information from the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). The underlying process information for this assessment was rated "high" during EPA's systematic review process.

The process of using film cement involves applying the cement onto edges of photographic films by hand using a small brush, then joining the pieces of film by pressing and heating to dry the cement. Based on this process information, EPA assumes that the 1,4-dioxane within film cement is volatilized to air during the drying process and that 1,4-dioxane residual within empty film cement bottles may also be volatilized to air. EPA estimated these air releases for this OES as a range, using a film cement use rate of 2.5 to 12 L/site-year and a concentration of 1,4-dioxane in the film cement of 45 to 50 percent, from the process information in the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). These releases may be to fugitive air or stack air depending on site-specific engineering controls.

EPA's calculation of air emissions for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

Spray Polyurethane Foam

EPA estimated air emissions for facilities within the spray polyurethane foam OES using the GS on Application of Spray Polyurethane Foam Insulation (U.S. EPA, 2020c). The GS on the Application of Spray Polyurethane Foam Insulation was rated "medium" during EPA's systematic review process.

The GS indicates that there are six release points:

1. Releases to fugitive air for volatile chemicals during unloading of raw materials from transport containers
2. Releases to water, incineration, or landfill from cleaning or disposal of transport containers
3. Releases to fugitive air for volatile chemicals during transport container cleaning
4. Releases to incineration or landfill from spray polyurethane foam application equipment cleaning
5. Releases to fugitive air for volatile chemicals during equipment cleaning
6. Releases to landfill of scrap foam from trimming applied foam

Based on the GS, release points #1, 3, and 5 have the potential for air emissions. To estimate these releases, EPA used the equations specified in the GS (U.S. EPA, 2020c). Apart from weight fraction in spray polyurethan foam, EPA did not find any data specific to 1,4-dioxane in this OES. Therefore, the calculation of releases using this GS are for a "generic site," using the default input parameter values from the GS. Specifically, EPA used the input parameter values that were presented in the original risk evaluation for estimates of occupational exposure; see Appendix G of the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Using this methodology, EPA calculated the "typical" and "worst-case" air emissions for this OES. These characterizations are based on the GS, which provides default "typical" and "worst-case" input parameters for the release calculations. EPA's calculation of air emissions for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

3D Printing

Air emission data were not available for this OES and EPA did not find any information to model air emissions for this OES using literature, GSs, or ESDs. EPA expects that industrial applications of this OES to be accounted for in the Industrial Uses TRI data. Per the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), 3D printing ink containing 1,4-dioxane is used in research labs to print biomedical products. Because the 2019 TRI data for the Industrial Uses OES include medicinal and pharmaceutical manufacturing NAICS codes, medical research labs that conduct 3D printing with 1,4-dioxane inks may be captured in that OES. Therefore, EPA grouped the air release assessment for 3D Printing into that for Industrial Uses. However, there is uncertainty in whether 3D printing sites are truly captured in the Industrial Uses TRI data.

Dry Film Lubricant

EPA estimated air emissions for facilities within the Dry Film Lubricant OES using process information from the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). The underlying process information for this assessment was rated “high” during EPA’s systematic review process.

The process for the production and use of dry film lubricant is described in the 2020 RE and is based on information provided to EPA by the one known user. In summary, the process entails producing the concentrated dry film lubricant by mixing 1,4-dioxane with other additives, followed by the dilution of the concentrated dry film lubricant with additional 1,4-dioxane and finally the use of the dry film lubricant. The use involves spray application onto substrates in a vented paint booth and the subsequent curing in a vented oven and cleaning of the dried parts in a 1,4-dioxane bath (U.S. EPA, 2020c). Based on this process description, EPA assumes that 100 percent of the 1,4-dioxane in the applied dry film lubricant is released to stack air from the paint booth and the oven. EPA estimated this release quantity using batch parameters from the process description, including 5 percent 1,4-dioxane in the dry film lubricant, 48 dry film lubricant applications per year, 0.5-pints of concentrated dry film lubricant, and 1.5-pints of pure 1,4-dioxane per application (U.S. EPA, 2020c).

EPA’s calculation of air emissions for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

Textile Dye

EPA used the OECD ESD on Textile Dyes (OECD, 2017) to estimate water and land releases for this OES; however, this ESD does not include approaches for estimating air releases. EPA did not find any other GS/ESD, literature sources, or process information to model air releases for this OES. In addition, EPA does not expect this OES to be sufficiently similar to other OES such that surrogate TRI data can be used to estimate air emissions for this OES. Therefore, EPA was not able to estimate air releases for these OESs.

Antifreeze

EPA did not find any directly applicable GS/ESD or literature sources for this OES; however, EPA evaluated the potential for releases using the OECD ESD on Chemical Additives used in Automotive Lubricants (OECD, 2020) and the EPA MRD on Commercial Use of Automotive Detailing Products (U.S. EPA, 2022b). The ESD and MRD were both rated “high” during EPA’s systematic review process.

For the use of antifreeze, EPA expects releases may occur from volatilizations of 1,4-dioxane during unloading/ pouring antifreeze into vehicles, disposal, or cleaning of empty antifreeze containers, and spent antifreeze. Both the ESD and MRD indicate that containers of automotive maintenance fluids are typically small and are not rinsed, but rather disposed of as solid waste (U.S. EPA, 2022b; OECD, 2020). Additionally, the ESD on Chemical Additives used in Automotive Lubricants indicates that spent lubricants are disposed of via incineration, which EPA expects is similarly done for spent antifreeze (OECD, 2020). Therefore, EPA expects the main source of air emissions to be from volatilizations of 1,4-dioxane during unloading/ pouring antifreeze into vehicles. EPA estimated this release using the EPA/OAQPS AP-42 Loading Model and batch parameters from the ESD, MRD, and other sources.

Specifically, EPA used the consumer use rate of antifreeze (0.15 kg antifreeze/job) from the *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c) and scaled this value up to a commercial use rate based on a range of the number of cars serviced at auto shops from the Near-Field/Far-Field Brake Model and Automotive Detailing MRD (1 to 9 jobs/day). EPA used a range of concentrations of 1,4-dioxane in antifreeze from the process description in Appendix F.4.2 and assumed antifreeze container sizes ranging from 16 ounces to 5 gallons per the default container sizes in the MRD and ESD, respectively (U.S. EPA, 2022b; OECD, 2020). Using these batch parameters and the default parameters for the EPA/OAQPS AP-42 Loading Model, EPA estimated low-end and high-end air emissions. EPA expects these air emissions to be to fugitive air based on the use setting (*e.g.*, outdoors, maintenance garages).

EPA's calculation of air emissions for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

Surface Cleaner

EPA did not find any directly applicable GS/ESD or literature sources for this OES; however, EPA estimated air releases using the SHEDs-HT modeling conducted for the one case study location (Liverpool OH) and the assumptions described herein. EPA expects that the main release points from the use of surface cleaners are from

1. Disposal of empty containers containing residual cleaning solution,
2. Application of the cleaning solution, and
3. Disposal of cleaning solution by rinsing or wiping.

Because EPA did not find any directly applicable GSs or ESDs, EPA used the Draft GS on Furnishing Cleaning (U.S. EPA, 2022a) to inform these releases due to the similarities in surface cleaning and furnishing cleaning. The Draft GS on Furnishing Cleaning was rated "high" during EPA's systematic review process. Per this Draft GS, empty containers may be rinsed out in sinks or disposed of without rinsing, such that releases may be to wastewater or landfill; the GS uses the EPA/OPPT Small Container Residual Model to estimate this release. Application losses are to fugitive air from spray application; the GS uses literature data to estimate this release. Once applied, the cleaner may be rinsed off or wiped off with rags or towels, such that releases may be to wastewater or landfill; the GS assumes 100 percent release scenario, estimating this release by subtracting the upstream losses from the cleaner use rate (U.S. EPA, 2022a).

The SHEDs-HT modeling estimated wastewater discharges of 72 g of 1,4-dioxane per day for commercial uses of surface cleaners containing 1,4-dioxane in Liverpool OH. EPA used this quantity and the above release information and models from the Draft GS on Furnishing Cleaning to back-calculate a 1,4-dioxane use rate. EPA then applied the loss fraction to fugitive air from release point #2 to estimate air releases for this OES. EPA's calculation of air releases for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

Dish Soap

EPA did not find relevant or surrogate TRI data, literature sources, sufficient process information, nor ESD or GS with air release estimation approaches to estimate air releases for these OESs. Therefore, EPA was not able to estimate air releases for these OESs.

Dishwasher Detergent

EPA did not find relevant or surrogate TRI data, literature sources, sufficient process information, nor ESD or GS with air release estimation approaches to estimate air releases for these OESs. Therefore, EPA was not able to estimate air releases for these OESs.

Laundry Detergent

EPA estimated air emissions for facilities within the Laundry Detergent OES using the OECD ESD on Industrial and Institutional Laundries (OECD, 2011b) and Monte Carlo modeling. The ESD on Industrial and Institutional Laundries was rated "medium" during EPA's systematic review process. The use of Monte Carlo modeling allows for variation of calculation input parameters such that a distribution of environmental releases can be calculated, from which EPA can estimate the 50th and 95th percentile releases. An explanation of this modeling approach is included in Appendix E.12.

Paints and Floor Lacquer

EPA estimated air emissions for facilities within the Paints and Floor Lacquers OES using the OECD ESD on Coating Application via Spray-Painting in the Automotive Refinishing Industry (OECD, 2011a). The ESD was rated "medium" during EPA's systematic review process.

As described in the process description in Appendix F.4.7, 1,4-dioxane was identified by a public comment as present in automotive refinishing products, thereby allowing EPA to identify the above ESD as the most applicable GS/ESD available. This ESD indicates that releases are expected from

1. Releases to incineration or landfill from container cleaning/disposal,
2. Releases to incineration or landfill from equipment cleaning,
3. Releases to incineration or landfill from over sprayed coating that is captured by emission controls, and
4. Releases to stack air from over sprayed coating that is not captured by emission controls.

Based on the GS, release point #4 has the potential for air emissions. To estimate this release, EPA used the equations specified in the ESD (OECD, 2011a). Apart from weight fraction in coatings (see Appendix F.4.7), EPA did not find any data specific to 1,4-dioxane in this OES. Therefore, the calculation of releases using this GS are for a "generic site," using the default input parameter values from the ESD.

Using this methodology, EPA calculated the low-end and high-end air emissions for this OES, which are expected to be to stack air per the ESD (OECD, 2011a). The low- and high-end estimates are based on the low- and high-end calculation input parameter defaults from the ESD. EPA's calculation of air emissions for this OES, including all calculation inputs and assumptions, can be found in *1,4-Dioxane Supplemental Information File: Environmental Releases to Air* (U.S. EPA, 2023j).

Hydraulic Fracturing

EPA estimated air emissions for facilities within the Hydraulic Fracturing OES using the Draft OECD ESD on Hydraulic Fracturing (U.S. EPA, 2022d) and Monte Carlo modeling. The Draft ESD on Hydraulic Fracturing was rated "high" during EPA's systematic review process. The use of Monte Carlo modeling allows for variation of calculation input parameters such that a distribution of environmental releases can be calculated, from which EPA can estimate the 50th and 95th percentile releases. An explanation of this modeling approach is included in Appendix E.13.

E.5.3 Air Release Estimates Summary

A summary of industrial and commercial air releases estimated using the above methods is presented in Table_Apx E-7 below.

Table_Apx E-7 Summary of Daily Industrial and Commercial Air Release Estimates for 1,4-Dioxane

OES	Type of Air Release	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Manufacturing	Fugitive Air	1	2.62		250	Medium	TRI
	Stack Air	1	0.0018			Medium	TRI
Import and Repackaging	Fugitive Air	1	0		250	Medium	TRI
	Stack Air	1	0.091			Medium	TRI
Industrial Uses	Fugitive Air	12	0	0.91	250	Medium	TRI
	Stack Air	12	0	8.14		Medium	TRI
Functional Fluids (Open-System)	Fugitive Air	2	0	0.009	247	Medium	TRI
	Stack Air	2	0.19	1.38		Medium	TRI
Functional Fluids (Closed-System)	All	Assessed as a part of Industrial Uses OES				N/A	N/A
Laboratory Chemical	Fugitive Air or Stack Air (-Unknown)	132	0.11 (typical)	0.41 (worst-case)	250	High	GS[d]
Film Cement	Fugitive Air or Stack Air (Unknown)	211	0.0046	0.025	250	High	Process information[e]
Spray Foam Application	Fugitive Air	1,553,559	0.0024 (typical)	0.012 (worst-case)	3	Medium	GS[f]
	Stack Air	1,553,559	0 (all air releases assessed to fugitive)			Medium	GS[f]
Printing Inks (3D)	Fugitive Air, Stack Air, and Land (all types)	Assessed as a part of Industrial Uses OES			250	N/A	N/A
Dry Film Lubricant	Fugitive Air	8	0 (no fugitive releases per process information)		48	High	Process information[e]
	Stack Air	8	0.75 (single value estimated from process information)			High	Process information[e]

OES	**Type of Air Release**	**Number of Facilities with Releases**[a]	**Estimated Daily Release Range across Sites (kg/site-day)**		**Estimated Release Frequency Range (days)**[b]	**Overall Data Quality Determination**	**Sources**[c]
			Min	**Max**			
Disposal	Fugitive Air	15	0	0.91	250	Medium	TRI
	Stack Air	15	0	0.91		Medium	TRI
Textile Dye	Fugitive Air and Stack Air	Not assessed			31 to 295	Medium	ESD[g] and Modeling[h]
Antifreeze	Fugitive Air and Stack Air	84,383	7.26E−16	1.80E−07	250	High	Process information[e] and Modeling[h]
Surface Cleaner	Fugitive Air	Unknown	0.0071 (typical; for all sites in Liverpool OH, case study)	0.013 (worst case; for all sites in Liverpool, OH, case study)	250	High	SHEDS-HT,[l] Process information[e] Modeling[h]
Dish Soap	Fugitive air and stack air	Not assessed			250	N/A	N/A
Dishwasher Detergent	Fugitive air and stack air	Not assessed			250	N/A	N/A
Laundry Detergent (Institutional) – liquid detergents	Fugitive air	95,533	1.83E−10	6.52E−07	250 to 365	Medium	ESD[i] and Modeling[h]
	Fugitive air, stack air, or POTW (unknown partitioning)	95,533	1.51E−10	0.00714		Medium	ESD[i] and Modeling[h]
	Land (unknown landfill), incineration, or POTW (unknown partitioning)	95,533	4.05E−12	3.95E−05		Medium	ESD[i] and Modeling[h]

OES	Type of Air Release	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Laundry Detergent (Institutional) – powder detergents	Fugitive air	95,533	3.42E−12	2.77E−07	250 to 365	Medium	ESD[i] and Modeling[h]
	Stack air	95,533	1.40E−11	3.75E−06		Medium	ESD[i] and Modeling[h]
	Fugitive air, stack air, or POTW (unknown partitioning)	95,533	3.05E−08	2.10E−04		Medium	ESD[i] and Modeling[h]
	Land (unknown landfill), incineration, or POTW (unknown partitioning)	95,533	5.36E−08	0.0018		Medium	ESD[i] and Modeling[h]
Laundry Detergent (Industrial) – liquid detergents	Fugitive air	2,453	6.25E−10	1.93E−06	20 to 365	Medium	ESD[i] and Modeling[h]
	Fugitive air, stack air, or POTW (unknown partitioning)	2,453	5.48E−12	0.011		Medium	ESD[i] and Modeling[h]
	Land (unknown landfill), incineration, or POTW (unknown partitioning)	2,453	4.78E−12	1.46E−04		Medium	ESD[i] and Modeling[h]

OES	Type of Air Release	Number of Facilities with Releases[a]	Estimated Daily Release Range across Sites (kg/site-day)		Estimated Release Frequency Range (days)[b]	Overall Data Quality Determination	Sources[c]
			Min	Max			
Laundry Detergent (Industrial) – powder detergents	Fugitive air	2,453	3.13E−13	1.47E−05	20 to 365	Medium	ESD[i] and Modeling[h]
	Stack air	2,453	1.68E−12	1.82E−04		Medium	ESD[i] and Modeling[h]
	Fugitive air, stack air, or POTW (unknown partitioning)	2,453	1.76E−09	0.0112		Medium	ESD[i] and Modeling[h]
	Land (unknown landfill), incineration, or POTW (unknown partitioning)	2,453	2.92E−11	3.92E−04		Medium	ESD[i] and Modeling[h]
Paints and Floor Lacquer	Stack air	33,648	4.68E−10	1.60E−06	250	Medium	ESD[j]
Polyethylene Terephthalate (PET) Byproduct	Fugitive Air	13	0	1.57	250	Medium	TRI
	Stack Air	13	0.0049	13.8		Medium	TRI
Ethoxylation Process Byproduct	Fugitive Air	8	0	7.4	250	Medium	TRI
	Stack Air	8	0	32		Medium	TRI
Hydraulic Fracturing	Fugitive air	411	1.99E−07	5482	1 to 72	Medium	ESD[k] and Modeling[h]
	Stack air	411	0 (all air releases assessed to fugitive)			Medium	ESD[n] and Modeling[k]
	Surface water, incineration, or landfill (unknown partitioning)	411	3.61E−10	4.59		Medium	ESD[k] and Modeling[h]

OES	**Type of Air Release**	**Number of Facilities with Releases**[a]	**Estimated Daily Release Range across Sites (kg/site-day)**		**Estimated Release Frequency Range (days)**[b]	**Overall Data Quality Determination**	**Sources**[c]
			Min	**Max**			

[a] Where available, EPA used 2013–2019 TRI (U.S. EPA, 2022g) and 2013–2019 DMR (U.S. EPA, 2022c) data to provide a basis to estimate the number of sites using 1,4-dioxane within a COU.

[b] Where available, EPA used the December 2020 Final Risk Evaluation for 1,4-Dioxane (U.S. EPA, 2020c), generic scenarios, and emission scenario documents to provide a basis to estimate the number of release days of 1,4-dioxane within a COU.

[c] Narrative descriptions of all release estimate sources are provided in Appendix E.5.

[d] The generic scenario used for this COU is the GS on Use of Laboratory Chemicals (U.S. EPA, 2022h).

[e] For this COU, EPA used process information, which is further described in Appendix E.5.2.

[f] The generic scenario used for this COU is the GS on Application of Spray Polyurethane Foam Insulation (U.S. EPA, 2018b).

[g] The emission scenario document used for this COU is the ESD on Textile Dyes (OECD, 2017).

[h] For this COU, EPA used various models and literature for model input parameters as described in Appendix E.5.2.

[i] The emission scenario document used for this COU is the ESD on Industrial and Institutional Laundries (OECD, 2011b).

[j] The emission scenario document used for this COU is the ESD on Coating Application via Spray Painting in the Automotive Refinishing Industry (OECD, 2011a).

[k] The emission scenario document used for this COU is the Draft ESD on Hydraulic Fracturing (U.S. EPA, 2022d).

[l] EPA used the down the drain water release estimates from the SHEDs-HT model for the Liverpool OH case study (see Section 2.1.1.2) to estimate air releases by back calculating 1,4-dioxane use rates and applying loss fractions for air releases using literature and standard models described in Appendix E.5.2.

E.5.4 Summary of Weight of the Scientific Evidence Conclusions in Air Release Estimates

Table_Apx E-8 provides a summary of EPA's weight of the scientific evidence conclusions in its air release estimates for each of the Occupational Exposure Scenarios assessed. Detailed descriptions of non-OES specific strengths, limitations, assumptions, and uncertainties (*e.g.*, general limitations for TRI, DMR, etc.) are provided in Appendix E.6.

Table_Apx E-8 Summary of Weight of the Scientific Evidence Conclusions in Air Release Estimates by OES

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Manufacturing	Air releases are assessed using reported releases from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include the low number of data points, lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Additionally, EPA made assumptions on the number of operating days to estimate daily releases. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Import and Repackaging	Air releases are assessed using reported releases from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include the low number of data points, lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, uncertainty in EPA's use of Form A submissions, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Some facilities within this OES reported to TRI using a Form A, which does not include any details on chemical release quantities. When a facility has submitted a Form A, there is no way to discern the quantity released. Therefore, where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media; however, there is uncertainty in this because the actual release quantity is unknown. Furthermore, the threshold represents an upper limit on total releases from the facility; therefore, assessing releases at the threshold value may overestimate actual releases from the facility. Additionally, EPA made assumptions on the number of operating days to estimate daily releases. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Industrial Uses	Air releases are assessed using reported releases from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	strength of the evidence for this OES include a lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, uncertainty in EPA's use of Form A submissions, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Some facilities within this OES reported to TRI using a Form A, which does not include any details on chemical release quantities. When a facility has submitted a Form A, there is no way to discern the quantity released. Therefore, where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media; however, there is uncertainty in this because the actual release quantity is unknown. Furthermore, the threshold represents an upper limit on total releases from the facility; therefore, assessing releases at the threshold value may overestimate actual releases from the facility. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Functional Fluids (Open-System)	Air releases are assessed using reported releases from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include the low number of data points (only two reporting sites), lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Additionally, EPA made assumptions on the number of operating days to estimate daily releases. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Functional Fluids (Closed-System)	No data was available to estimate releases for this OES. For the air release assessment, EPA grouped this OES with the Industrial Uses OES because the sources of release are expected to be similar between these OESs. Factors that increase the strength of evidence for this OES are that TRI has a medium overall data quality determination and consistency within the dataset (all reporters use the same or similar reporting forms). Factors that decrease the strength of evidence for this OES are that the Industrial Releases OES release data are use as surrogate for this OES, uncertainty in the accuracy of reported releases, limitations in representativeness to all sites because TRI may not capture all relevant sites, and lack of variability (only one year of data used). Refer to the Industrial Uses OES discussion for additional discussion. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Laboratory Chemicals	Air releases are assessed using the Draft GS on Use of Laboratory Chemicals. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the Draft GS on Use of Laboratory Chemicals has a high overall data quality determination, and the low level of uncertainty in the data. Factors that decrease the strength of the evidence for this OES include the that the GS has not been peer-reviewed, uncertainty in the representativeness of the GS towards all sites in this OES, and a lack of variability in the analysis. Specifically, because the default values in the ESD are generic, there is uncertainty in the representativeness of generic site estimates of actual releases from real-world sites that use 1,4-dioxane. Another uncertainty is lack of consideration for

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods, contributing to uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Film Cement	Air releases are assessed using process information from the *Final Risk Evaluation for 1,4-Dioxane*. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the underlying data sources for the process information have a high overall data quality determination, and the low level of uncertainty in the data because the process information comes directly from actual users of 1,4-dioxane in film cement. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of evidence to all sites in this OES and a lack of variability. Specifically, the process information for the production and use of film cement is based on information from three use sites, one from Australia and two from the U.S. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Spray Foam Application	Air releases are assessed using the GS on Application of Spray Polyurethane Foam Insulation. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the underlying data sources for the process information have a medium overall data quality determination, and the low level of uncertainty in the data. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of the GS to all sites since it is generic and not specific to sites that use 1,4-dioxane and a lack of variability. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Printing Inks (3D)	No data was available to estimate releases for this OES. For the air release assessment, EPA grouped this OES with the Industrial Uses OES because the sources of release are expected to be similar between these OESs. Factors that increase the strength of evidence for this OES are that TRI has a medium overall data quality determination and consistency within the dataset (all reporters use the same or similar reporting forms). Factors that decrease the strength of evidence for this OES are that the Industrial Releases OES release data are use as surrogate for this OES, uncertainty in the accuracy of reported releases, limitations in representativeness to all sites because TRI may not capture all relevant sites or smaller commercial 3D printing uses, and lack of variability (only one year of data used). Refer to the Industrial Uses OES discussion for additional discussion. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Dry Film Lubricant	Air releases are assessed using process information from the *Final Risk Evaluation for 1,4-Dioxane*. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), that the underlying data sources for the process information have a high overall data quality determination, and a low level of uncertainty in the data because the process information comes directly from an actual user of 1,4-dioxane in

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	dry film lubricants. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of evidence to all sites and a lack of variability. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Disposal	Air releases are assessed using reported discharges from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, high number of data points, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include lack of variability (only one year of data used), uncertainty in the accuracy of reported releases, uncertainty in EPA’s use of Form A submissions, and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Some facilities within this OES reported to TRI using a Form A, which does not include any details on chemical release quantities. When a facility has submitted a Form A, there is no way to discern the quantity released. Therefore, where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media; however, there is uncertainty in this because the actual release quantity is unknown. Furthermore, the threshold represents an upper limit on total releases from the facility; therefore, assessing releases at the threshold value may overestimate actual releases from the facility. Additionally, uncertainty is introduced from EPA’s assumptions on the number of operating days to estimate daily releases. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Textile Dye	EPA used the ESD on Textile Dyes to estimate land and water releases; however, this ESD does not include approaches for estimating air releases. EPA did not find any other GS/ESD, literature sources, or process information to model air releases for this OES. Furthermore, EPA does not expect this OES to be sufficiently similar to other OES such that surrogate TRI data can be used to estimate air emissions for this OES. Therefore, EPA was not able to estimate air releases for this OES and concluded that the weight of the scientific evidence is indeterminant.
Antifreeze	Air releases are assessed using the OECD ESD on Chemical Additives used in Automotive Lubricants, the EPA MRD on Commercial Use of Automotive Detailing Products, and EPA/OPPT models. Factors that increase the strength of evidence for this OES are that the ESD and MRD used have high overall data quality determinations and consistency within the sources used. Factors that decrease the strength of the evidence for this OES include that the ESD and MRD are not directly applicable to antifreeze uses (used as surrogate), uncertainty in the representativeness of the ESD and MRD to all sites and sites that specifically use 1,4-dioxane since these documents contain generic values, and a lack of variability. Additionally, EPA scaled up a consumer antifreeze use rate to a commercial use rate based on information in the ESD and MRD, which increases uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
Surface Cleaner	Air releases are assessed using the SHEDS-HT model and the Draft GS on Furnishing Cleaning. To estimate air releases, EPA used loss fractions for water releases from the GS and the modeled water release from SHEDS-HT to back-calculate a 1,4-dioxane use rate. EPA then applied loss fractions for air releases from the GS to estimate air releases for this OES. Factors that increase the strength of evidence for this OES include that the release estimates are directly relevant to the OES (as opposed to surrogate), that the Draft GS used has a high overall data quality determination, and variability in the model input parameters. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness to all sites because the SHEDS-HT estimate is based on one case study for Liverpool, OH and because the estimate is not site-specific (the release estimate is a total for all sites in Liverpool, OH). Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Dish Soap	EPA did not find relevant or surrogate TRI data, literature sources, sufficient process information, nor ESD or GS with air release estimation approaches to estimate air releases for these OESs. Therefore, EPA was not able to estimate air releases for this OES and concluded that the weight of the scientific evidence is indeterminant.
Dishwasher Detergent	EPA did not find relevant or surrogate TRI data, literature sources, sufficient process information, nor ESD or GS with air release estimation approaches to estimate air releases for these OESs. Therefore, EPA was not able to estimate air releases for this OES and concluded that the weight of the scientific evidence is indeterminant.
Laundry Detergent	Air releases are assessed using Monte Carlo modeling with information from the ESD on Industrial and Institutional Laundries. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), that the ESD on Industrial and Institutional Laundries has a medium overall data quality determination and was peer reviewed, high number of data points (simulation runs), consistency within the dataset, and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential release values that represents a larger proportion of sites than a discrete value. Additionally, EPA was able to separately estimate releases for industrial and institutional laundry settings. Factors that decrease the strength of the evidence for this OES include uncertainties and limitations in the representativeness of the estimates for sites that specifically use 1,4-dioxane because the default values in the ESD are generic. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Paint and Floor Lacquer	Air releases are assessed using OECD ESD on Coating Application via Spray-Painting in the Automotive Refinishing Industry. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), the ESD has a medium overall data quality determination, consistency within the sources used, and a low amount of uncertainties. Factors that decrease the strength of the evidence for this OES include a lack of variability and uncertainty in the representativeness of the ESD to all sites and sites that specifically use 1,4-dioxane since the ESD is generic. Based on this information, EPA has concluded that the weight of the scientific evidence

OES	Weight of the Scientific Evidence Conclusion in Release Estimates
	for this assessment is slight to moderate and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Polyethylene Terephthalate (PET) Byproduct	Air releases are assessed using reported discharges from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, consistency within the dataset (all reporters use the same or similar reporting forms), and consistency with the emission data from the related life cycle analysis discussed in Appendix E.6. A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include uncertainty in the accuracy of reported releases, lack of variability (only one year of data used), and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Additionally, EPA made assumptions on the number of operating days to estimate daily releases, which introduces additional uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Ethoxylation Process Byproduct	Air releases are assessed using reported discharges from 2019 TRI. Factors that increase the strength of evidence for this OES are that the release data are directly relevant to the OES (as opposed to surrogate), that TRI has a medium overall data quality determination, and consistency within the dataset (all reporters use the same or similar reporting forms). A strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. Factors that decrease the strength of the evidence for this OES include the uncertainty in the accuracy of reported releases, lack of variability (only one year of data used), and the limitations in representativeness to all sites because TRI may not capture all relevant sites. Additionally, EPA made assumptions on the number of operating days to estimate daily releases, which introduces additional uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.
Hydraulic Fracturing	Air releases are assessed using Monte Carlo modeling with information from the Draft ESD on Hydraulic Fracturing and FracFocus 3.0. Factors that increase the strength of evidence for this OES are that the release estimates are directly relevant to the OES (as opposed to surrogate), that the Draft ESD on Hydraulic Fracturing and FracFocus 3.0 have medium overall data quality determinations, high number of data points (simulation runs), consistency within the dataset, and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential release values that represents a larger proportion of sites than a discrete value. Factors that decrease the strength of the evidence for this OES include that the Draft ESD has not been peer reviewed and the uncertainties and limitations in the representativeness of the estimates for sites that specifically use 1,4-dioxane because the default values from the Draft ESD on Hydraulic Fracturing. Another uncertainty is lack of consideration for release controls. The ESD assumes that all activities occur without any release controls. Actual releases may be less than estimated if facilities utilize pollution control methods, contributing to uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of releases in consideration of the strengths and limitations of reasonably available data.

E.6 Comparison to PET Life Cycle Analysis

EPA identified a relevant life cycle analysis (LCA) published by the National Association for PET Container Resources (NAPCOR) that included 1,4-dioxane emission factors from PET resin production (Franklin Associates, 2020). EPA did not use these emission factors to estimate releases in this Risk Evaluation because there were site-specific releases reported in TRI and DMR. This LCA only provided generic emission factors for air and surface water releases aggregated across seven unspecified sites; the LCA did not provide 1,4-dioxane emission factors for land releases. The emission factors in the LCA were reported by three producers (seven sites) that account for 50% of the 2015 U.S. PET production in a survey, and the basis of the emission factors is not provided. However, the survey states that the release data is primary data (i.e., the data were provided by directly by the surveyed PET producers). As opposed to conventional emission factors, the report only provides the order of magnitude of the average amount of 1,4-dioxane released per amount of PET produced. Discrete, site-specific emission factors are not provided. As a result, the variability of 1,4-dioxane releases from site to site is unknown. EPA prefers the use of site-specific release data as opposed to generic emission factors. Therefore, a comparison between total annual air and water releases from the LCA and from the TRI and DMR data used in this Risk Evaluation is provided below for context.

The LCA estimated that 4.7 million tons of PET capacity was available in 2015 in North America (Franklin Associates, 2020). To obtain total annual air and water release estimates from the LCA, EPA multiplied this production volume by the reported 1,4-dioxane emission factors of 0.001 kg 1,4-dioxane emitted per 1000 kg PET for air releases, and 0.01 kg 1,4-dioxane emitted per 1000 kg PET for surface water releases. To obtain the total annual air and water releases from the TRI and DMR used in this Risk Evaluation, EPA summed all reported annual site-specific air emissions and surface water discharges that were mapped to the “PET Manufacturing” OES (see Appendix E.3 and E.5 for additional information on the use of TRI and DMR). The total annual releases from the LCA and from TRI and DMR is compiled in Table_Apx E-9. EPA did this comparison with 2019 TRI/DMR because EPA’s Risk Evaluation largely uses 2019 data, as well as 2015 TRI/DMR data because the releases estimated with the LCA data are based on 2015 PET manufacturing data.

For air emissions, the LCA estimate and EPA’s estimates from the 2019 and 2015 TRI are comparable, being within an order of magnitude. Differences in the estimates likely arise since EPA’s analysis accounted for emissions from 13 PET manufacturing facilities compared to the seven facilities in the LCA. Additionally, the LCA is an aggregate of releases across sites whereas EPA’s analysis accounts for variability by using data from individual sites.

For surface water discharges, the LCA estimate and EPA’s estimates from the 2019 TRI and DMR show a larger discrepancy, with EPA’s estimate being two orders of magnitude larger than the LCA estimate. However, over 2.51 million kg of the approximately 2.53 million kg (99.2%) of surface water discharges in EPA’s estimate comes from a single facility’s 2019 DMR report (DAK Americas LLC, Columbia). The other 12 facilities with non-zero surface water discharges in EPA’s analysis had site-specific annual discharges ranging between 5.37 and 8,922 kgs for 2019. EPA reviewed the DAK Americas LLC, Columbia DMR reports from other years for comparison, which indicated approximately 14,000 kg of 1,4-dioxane were discharged in 2022, 8,800 kg in 2021, 6.8 million kg in 2020, and 2,300 kg in 2018. DAK Americas LLC, Columbia did not include 1,4-dioxane in their DMRs in 2016 or 2017 (the two earliest reporting years EPA looked at for this analysis). It is unclear why DAK Americas LLC, Columbia’s discharges were significantly higher in 2019 and 2020 or why these discharges were different than other PET manufacturers in EPA’s analysis. However, it is more likely that the facilities analyzed in the LCA were more similar to the other PET manufacturing facilities in EPA’s analysis, with the total surface water emissions estimated from the LCA data being within one order of magnitude of EPA's estimate when excluding the discharges from the DAK Americas LLC, Columbia facility, as shown in Table_Apx E-9. The LCA estimate and EPA's estimates from the 2015 TRI are comparable.

As indicated above, estimates from the LCA analysis and EPA's analysis with TRI and DMR data may differ for several reasons. First, the LCA provides a single emission factor aggregated from data across seven sites, reducing the impact of site-specific variability in releases. Whereas EPA's analysis uses site-specific release data from 13 sites (for air emissions) and 19 sites (for surface water discharges, six of which reported zero surface water discharges). EPA also does not have access to site identities, or the underlying data/methodologies used to estimate emission factors in the LCA, which limits EPA's ability to do a direct site-to-site comparison of results between the two analyses. Additionally, the LCA study states that some emissions are reported only by the order of magnitude of the average to protect the confidentiality of individual companies, introducing further uncertainty in the emission factors presented in the study. Lastly, the LCA data is from 2015 whereas EPA used data from 2019.

Table_Apx E-9. Comparison of TRI/DMR Release Data to LCA Study for PET Byproduct

Data Source	Total Release for All Sites (kg/yr)
Air emissions	
EPA Estimate in this Risk Evaluation – Based on 2019 TRI (including DAK Americas LLC, Columbia)	10,695
EPA Estimate in this Risk Evaluation – Based on 2015 TRI (including DAK Americas LLC, Columbia)	12,407
LCA Estimate (Franklin Associates, 2020) – Based on 2015 data	4,264
Surface water discharges	
EPA Estimate in this Risk Evaluation – Based on 2019 DMR and TRI (including DAK Americas LLC, Columbia)	2,531,730
EPA Estimate in this Risk Evaluation – Based on 2019 DMR and TRI (excluding DAK Americas LLC, Columbia)	19,296
EPA Estimate in this Risk Evaluation – Based on 2015 DMR and TRI	20,511
LCA Estimate (Franklin Associates, 2020) – Based on 2015 data	42,638

E.7 Detailed Strengths, Limitations, Assumptions and Key Sources of Uncertainties for the Environmental Release Assessment

This section includes detailed strengths, limitations, assumptions, and uncertainties associated with EPA's approaches for estimating air, water, and land releases in this supplemental risk evaluation. This section is intended to supplement the summary of strengths, limitations, assumptions, and uncertainties discussed in Section 2.2.1.3 with additional details.

Facilities Not Captured by TRI or DMR Reporting

When using TRI data to analyze chemical releases, it is important to acknowledge that TRI reporting does not include all releases of the chemical and therefore, the number of sites for a given OES may be underestimated. For each OES that had TRI or DMR data, the analysis of releases for those OES was limited to the facilities that reported releases to TRI and DMR. Therefore, it is uncertain the extent to which sites not captured in these databases have air, water, or land releases of 1,4-dioxane and what the exact media of release for those releases would be (*e.g.*, stack vs. fugitive air, surface water vs POTW, RCRA or another type of landfill). To the extent additional sites are not captured, releases may be underestimated; however, the magnitude of this underestimation is unknown. TRI data do not include

- Releases from any facility that used the chemical in quantities below the applicable annual chemical activity threshold (*e.g.*, 25,000 lb manufactured or processed, or 10,000 lb otherwise used, for most chemicals);
- Releases from any facility that is not in a TRI covered sector; and
- Releases from any facility that does not meet the TRI employment threshold of greater than 10 full-time employee equivalents (20,000 labor hours) for the year.

DMR Release Data

For facilities that reported having zero pollutant loads to DMR, the ECHO Pollutant Loading Tool Advanced Search applies a hybrid method to analyze nondetects. The EZ Search Load Module uses a combination of setting nondetects equal to zero and as one half the detection limit to calculate the annual pollutant loadings. Specifically, if the pollutant was measured as nondetect for all monitoring periods in a reporting year, then the EZ Search Load Module sets the annual pollutant load to zero. If the pollutant was detected for at least one monitoring period in a reporting year, then the EZ Search Load Module calculates the annual pollutant load by setting the nondetects equal to one half the detection limit. This method could cause overestimation or underestimation of annual and daily pollutant loads.

TRI Release Data

EPCRA section 313 states that facilities may estimate their release quantities using “readily available data,” including monitoring data, collected for other purposes. When data are not readily available, EPCRA section 313 states that “reasonable estimates” may be used. The facility is not required to monitor or measure the quantities, concentration, or frequency of any toxic chemical release for TRI reporting. TRI guidance states that not using readily available information, such as relevant monitoring data collected for compliance with other regulations, could result in enforcement and penalties.

For each release quantity reported, TRI facilities select a “Basis of estimate” code indicating the principal method used to determine the amount of the release. TRI provides six basis of estimate codes to choose from: continuous monitoring, periodic monitoring, mass balance, published emissions factors, site-specific emissions factors, or engineering calculations/best engineering judgment. In facilities where a chemical is used in multiple operations, the facility may use a combination of methods to calculate the release reported. In such cases, TRI instructs the facility to enter the basis of estimate code of the method that applies to the largest portion of the release quantity. Additional details on the basis of estimate, such as any calculations and underlying assumptions, are not reported. Depending on the inputs and/or monitoring methods used by each facility, any of the methods used to estimate releases may over or underestimate releases. The magnitude of this uncertainty is unknown.

For any release quantity that is less than 1,000 lb, facilities may report either the estimated quantity or a range code. The 1,000-pound limit for range code reporting applies to each type of release reported to TRI – fugitive air emissions, stack air emissions, water discharges, each type of land disposal, and each type of off-site transfer. There are three TRI range codes: 1 to 10; 11 to 499; and 500 to 999 lb. TRI data tools display the approximate midpoint of the range (*i.e.*, 5, 250, or 750 lb). Using this midpoint value may be either an over or an underestimate of the true value, depending where on the range the true value lies. Although analyses using data that was reported as a range code may add uncertainty, it is not clear that the uncertainty associated with a range code is greater than that associated with any other estimated release value. Range code reporting is not permitted for chemicals of special concern.

TRI guidance states that release estimates need not be reported to more than two significant figures. However, the guidance also states that facilities should report release quantities at a level of precision supported by the accuracy of the underlying data and the estimation techniques on which the estimate was based. If a facility's release calculations support reporting an amount that is more precise than two significant digits, then the facility should report that more precise amount. The facility makes the determination of the accuracy of their estimate and the appropriate significant digits to use.

For chemicals that meet certain criteria, facilities have the option of submitting a TRI Form A Certification Statement instead of a TRI Form R. The Form A does not include any details on the chemical release or waste management quantities. The criteria for a Form A are that during the reporting year, the chemical (1) did not exceed 500 lb for the total annual reportable amount (including the sum of on- and off-site quantities released, treated, recycled, and used for energy recovery); (2) amounts manufactured, processed, or otherwise used do not exceed 1 million lb; and (3) the chemical is not a chemical of special concern. When conducting analyses of chemical releases and a facility has submitted a Form A for the chemical, there is no way to discern the quantity released to each medium or even if there were any releases. Where facilities reported to TRI with a Form A, EPA used the Form A threshold for total releases of 500 lb/year for each release media (*e.g.*, fugitive air, stack air, surface water, POTW, non-POTW WWT, RCRA landfill, other landfill). EPA used the entire 500 lb/year for each release media; however, since this threshold is for total site releases, these 500 lb/year are to only one of these media at a time (since assessing 500 lb/year for all media at once would double count the releases and exceed the total release threshold for Form A). Furthermore, the threshold represents an upper limit on total releases to all environmental media from the facility; therefore, assessing releases at the threshold value likely overestimates actual releases from the facility.

Mapping TRI and DMR Facilities to OES

EPA used a crosswalk between TRI uses/sub-uses and CDR Industrial Function Category (IFC) codes (see Appendix E.9) along with a mapping of CDR IFC codes to OES to assign the OES for each facility that reported to TRI. However, there are limitations to this approach. For example, this approach may result in the mapping of multiple OES for one facility. Additionally, there are limitations to the TRI – CDR crosswalk. For example, a TRI use/sub-use may encompass multiple uses that are not captured in the crosswalked CDR IFC codes. In these instances, EPA determined the primary OES using the NAICS codes that the facility reported in TRI, information on types of products that the facility manufactures found from internet searches, and which OES is most likely to result in release. OES are related to COUs as defined in the crosswalk in Table 2-1 and Appendix D.2.This approach requires some level of engineering judgment to determine which OES is the most applicable to the facility, which introduces uncertainty in the OES mapping. Additionally, this approach assumes only one OES is applicable to the facility, which may be incorrect if the facility uses 1,4-dioxane for multiple purposes. If facilities were categorized under a different OES, the annual releases for each site would remain unchanged; however, average daily releases may change depending on the release days expected for the different OES.

Additional uncertainty is present in the OES mapping for TRI sites that reported using a Form A and DMR sites because there is no reported use/sub-use information. EPA used a similar procedure as described above to map these sites to an OES, involving the use of NAICS and Standard Industrial Classification (SIC) codes reported to TRI and DMR, internet searches on the types of products made at the facility, and which OES is most likely to result in release. Since this approach involves engineering judgment to determine which OES is the most applicable to the facility, there is uncertainty in the OES mapping.

There is also uncertainty in the NAICS codes and SIC codes reported in TRI and DMR. TRI facilities enter the facility's primary NAICS code indicating the primary economic activity at the facility. Facilities can also enter secondary NAICS codes. When using TRI chemical release data for a facility that also reported secondary NAICS codes, there may be uncertainty as to which NAICS is associated with the use of the chemical. Additionally, NAICS codes and SIC codes are reported for the facility as a whole and are not chemical specific.

Estimating Daily Releases from Annual TRI and DMR Release Data

Facilities reporting to TRI and DMR only report annual releases; to assess daily air and water releases, EPA estimated the release days and averaged the annual releases over these days. There is some uncertainty that all facilities for a given OES operate for the assumed duration; therefore, the average daily release may be higher if sites have fewer release days or lower if they have greater release days. Furthermore, chemical concentrations in air emissions and wastewater streams at each facility may vary from day to day such that on any given day the actual daily releases may be higher or lower than the estimated average daily discharge. Thus, this approach minimizes spikes and drops in emissions and discharges from day to day.

Pollutant Load values of "0" in the ECHO Pollutant Loading Tool Advanced Search output file are by default reported as below the detection limit in the monitoring reports used by the Loading Calculator Module.

EPA did not estimate daily land releases due to the high level of uncertainty in the number of release days associated with land releases; instead, EPA estimated annual land releases.

Representativeness of TRI and DMR for an OES as a Whole

The representativeness of TRI and DMR data for an OES as a whole is dependent on (1) the extent to which these reporting mechanisms capture all potential sites within the OES and, (2) the extent to which the release quantities provided by reporting sites reflect releases from non-reporting sites.

For some OES, the total number of sites was determined from TRI. For these OESs, there is uncertainty in if there may be additional sites using the chemical within the OES that did not report to TRI (*e.g.*, due to being below reporting thresholds). For some OES, such as manufacturing and other OES involving larger industrial sites, TRI is more likely to capture the majority of potential sites because these sites typically meet the reporting threshold. For other OES, such as functional fluids (open-systems), 3D printing, and other OES that may be performed at a range of different scales, the extent to which TRI captures all potential sites is more uncertain because not all sites may meet the reporting threshold. This uncertainty may result in an underestimate in the total number of sites using the chemical within the OES. To the extent additional sites are not captured, releases may be underestimated; however, the magnitude of this underestimation is unknown for each OES. In some cases, the total number of facilities for a given OES was estimated using data from the U.S. Census. This may result in an overestimate of the actual number of sites that use the chemical for that OES.

Additionally, it is unknown how representative release estimates from TRI and DMR reporting sites accurately reflect all releases from within an OES since releases from non-reporting sites cannot be quantified. Specifically, where the number of sites was estimated from U.S. Census data, the average daily release calculated from sites reporting to TRI or DMR was applied to the total number of sites reported in (U.S. Census Bureau, 2015). It is uncertain how accurate this average release is to actual releases at these sites; therefore, releases may be higher or lower than the calculated amount.

Estimating Emissions for OES without TRI Data

For release estimates developed for an OES when directly applicable TRI or DMR data were not available, there are uncertainties related to the use of surrogate TRI or DMR data or, in their absence, the use of modeling.

Use of surrogate TRI or DMR data may introduce uncertainties related to the extent to which the surrogate OES and the OES being assessed are similar. Thus, the representativeness of the surrogate release data towards the actual releases for the OES being assessed is uncertain.

Although no new models were developed for this release assessment, the adaptations made to and uses of these models as part of the analysis (*e.g.*, varying input parameters, Monte Carlo simulation) may result in release estimates higher or lower than the actual amount. EPA presented central tendency (50th percentile) and high-end (95th percentile) modeled release values to capture a range of potential releases and reduce the uncertainty associated with using a single release estimate. Additionally, the approaches used for estimating releases based on modeling or literature differs from the facility-specific approach used for OES for which TRI or DMR data were available (as described previously, the modeled estimates are for a "generic site"). This may introduce uncertainties that differ from those of the scenarios using TRI or DMR data, described above.

E.8 Weight of the Scientific Evidence Conclusions for Environmental Releases

As discussed in Section 2.2.1.2, Table_Apx E-10 presents a summary of EPA's overall weight of the scientific evidence conclusions for its release estimates for each of the assessed OES.

Table_Apx E-10. Summary of Overall Weight of the Scientific Evidence Conclusions for Environmental Release Estimates by OES

OES	Monitoring[a]				Modeling				Weight of the Scientific Evidence Conclusion			Notes
	Air	Water	Land	Data Quality Rating	Air	Water	Land	Data Quality Rating	Air	Water	Land	
Manufacturing	✓	✓	✓	M/H	x	x	x	N/A	Moderate to Robust	Moderate to Robust	Moderate to Robust	Based on TRI and DMR which have "medium" data quality ratings. Information on the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited, but uncertainties are known.
Import and Repackaging	✓	✓	✓	M/H	x	x	x	N/A	Moderate to Robust	Moderate to Robust	Moderate to Robust	Based on TRI and DMR which have "medium" data quality ratings. Information on the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited, but uncertainties are known.
Industrial Uses	✓	✓	✓	M/H	x	x	x	N/A	Moderate to Robust	Moderate to Robust	Moderate to Robust	Based on TRI and DMR which have "medium" data quality ratings. Information on the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited, but uncertainties are known.
Functional Fluids (Open-System)	✓	✓	✓	M/H	x	x	x	N/A	Moderate to Robust	Moderate to Robust	Moderate to Robust	Based on TRI and DMR which have "medium" data quality ratings. Information on the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited, but uncertainties are known.
Functional Fluids (Closed-System)	Assessed as a part of Industrial Uses OES								Slight	Slight	Slight	No data were available to estimate releases for this OES, so it was grouped with Industrial Uses OES. There is uncertainty in the representativeness of the Industrial Uses data for this OES.
Laboratory Chemicals	x	x	x	N/A	✓	✓	✓	H	Slight to Moderate	Slight to Moderate	Slight to Moderate	Assessed using Laboratory Chemicals GS which has a "high" data quality rating. Activities could vary drastically on a site-by-site basis due to uncertainties and limitations in the model.
Film Cement	x	x	x	N/A	✓	Not expected	✓	H	Slight to Moderate	Slight to Moderate	Slight to Moderate	The underlying data sources for model parameters have a "high" data quality rating. Modeling may not be sufficiently representative of all the sites for this OES.
Spray Foam Application	x	x	x	N/A	✓	✓	✓	M	Slight to Moderate	Slight to Moderate	Slight to Moderate	Assessed using SPF GS which has a "medium" data quality rating. Activities could vary drastically on a site-by-site basis due to uncertainties and limitations in the model.

OES	Monitoring[a]				Modeling				Weight of the Scientific Evidence Conclusion			Notes
	Air	**Water**	**Land**	**Data Quality Rating**	**Air**	**Water**	**Land**	**Data Quality Rating**	**Air**	**Water**	**Land**	
Printing Inks (3D)	Assumed included in Industrial Uses OES	✓	Assumed included in Industrial Uses OES	M/H	x	x	x	N/A	Slight	Moderate to Robust	Slight	Based on TRI and DMR which have “medium” data quality ratings. Information on the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited. There is uncertainty in the representativeness of the Industrial Uses data for this OES.
Dry Film Lubricant	x	x	x	N/A	✓	Not expected	✓	H	Slight to Moderate	Slight to Moderate	Slight to Moderate	The underlying data sources for model parameters have a “high” data quality rating. Modeling may not be sufficiently representative of all the sites for this OES.
Disposal	✓	✓	✓	M/H	x	x	x	N/A	Moderate to Robust	Moderate to Robust	Moderate to Robust	Based on TRI and DMR which have “medium” data quality ratings. Information on the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited, but uncertainties are known.
Textile Dye	x	x	x	N/A	Not assessed	✓	✓	M	Indeterminate	Moderate	Moderate	Assessed using ESD on Textile Dyes, which has a “medium” data quality rating. Monte Carlo modeling allows for parameter variation; however, the modeling may not be sufficiently representative of all the sites for this OES.
Antifreeze	x	x	x	N/A	✓	Not expected	✓	H	Slight to Moderate	Slight to Moderate	Slight to Moderate	Assessed using process information from GSs with “high” data quality ratings. Activities could vary drastically on a site-by-site basis due to uncertainties and limitations in the model.
Surface Cleaner	x	x	x	N/A	✓	✓	✓	H	Slight	Slight	Slight	Assessed using SHEDs-HT data for the Liverpool OH case study and the Furnishing Cleaning GS, which has a data quality rating of “high.” There is uncertainty in the application of this modeling for a commercial setting, and this case study does not represent all sites in this OES.
Dish Soap	x	x	x	N/A	Not assessed	✓	✓	N/A	Indeterminate	Slight	Slight	Assessed using SHEDs-HT model for the Liverpool OH case study and standard EPA/OPPT models. There is uncertainty in the application of this modeling for a

OES	Monitoring[a]				Modeling				Weight of the Scientific Evidence Conclusion			Notes
	Air	Water	Land	Data Quality Rating	Air	Water	Land	Data Quality Rating	Air	Water	Land	
												commercial setting, and this case study does not represent all sites in this OES.
Dishwasher Detergent	x	x	x	N/A	Not assessed	✓	✓	N/A	Indeterminate	Slight	Slight	Assessed using SHEDs-HT data for the Liverpool OH case study and standard EPA/OPPT models. There is uncertainty in the application of this modeling for a commercial setting, and this case study does not represent all sites in this OES.
Laundry Detergent (Industrial)	x	x	x	N/A	✓	✓	✓	M	Moderate	Moderate	Moderate	Assessed using Laundries ESD, which has a “medium” data quality rating. Monte Carlo modeling allows for parameter variation; however, the modeling may not be sufficiently representative of all the sites for this OES.
Laundry Detergent (Institutional)	x	x	x	N/A	✓	✓	✓	M	Moderate	Moderate	Moderate	Assessed using Laundries ESD, which has a “medium” data quality rating. Monte Carlo modeling allows for parameter variation; however, the modeling may not be sufficiently representative of all the sites for this OES.
Paints and Floor Lacquer	x	x	x	N/A	✓	Not expected	✓	M	Slight to Moderate	Slight to Moderate	Slight to Moderate	Assessed using Automotive Spray Painting ESD, which has a “medium” data quality rating. Modeling may not be sufficiently representative of all the sites for this exposure scenario.
PET Byproduct	✓	✓	✓	M/H	x	x	x	N/A	Moderate to Robust	Moderate to Robust	Moderate to Robust	Based on TRI and DMR which have “medium” data quality ratings. Information on the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited, but uncertainties are known.
Ethoxylation Process Byproduct	✓	✓	✓	M/H	x	x	x	N/A	Moderate to Robust	Moderate to Robust	Moderate to Robust	Based on TRI and DMR which have “medium” data quality ratings. Information on the conditions of use of 1,4-dioxane at facilities in TRI and DMR is limited, but uncertainties are known.
Hydraulic Fracturing	✓	✓	✓	M	✓	✓	✓	M	Moderate to Robust	Moderate to Robust	Moderate to Robust	Based on FracFocus 3.0 and the Hydraulic Fracturing ESD, which has a “medium” data quality rating. Monte Carlo modeling allows for parameter variation; however,

OES	Monitoring[a]				Modeling				Weight of the Scientific Evidence Conclusion			Notes
	Air	Water	Land	Data Quality Rating	Air	Water	Land	Data Quality Rating	Air	Water	Land	
												the modeling may not be sufficiently representative of all the sites for this OES.

E.9 TRI to CDR Crosswalk

Table_Apx E-11 presents the TRI-CDR Crosswalk used to map facilities to the OES for each chemical. Blanks in the 2016 CDR code column indicate there is no corresponding CDR code that matches the TRI code.

Table_Apx E-11. TRI-CDR Use Code Crosswalk

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.1.a	Manufacture: Produce	N/A	N/A	N/A	N/A	N/A
3.1.b	Manufacture: Import	N/A	N/A	N/A	N/A	N/A
3.1.c	Manufacture: For on-site use/processing	N/A	N/A	N/A	N/A	N/A
3.1.d	Manufacture: For sale/distribution	N/A	N/A	N/A	N/A	N/A
3.1.e	Manufacture: As a byproduct	N/A	N/A	N/A	N/A	N/A
3.1.f	Manufacture: As an impurity	N/A	N/A	N/A	N/A	N/A
3.2.a	Processing: As a reactant	N/A	N/A	PC	Processing as a reactant	Chemical substance is used in chemical reactions for the manufacturing of another chemical substance or product.
3.2.a	Processing: As a reactant	P101	Feedstocks	N/A	N/A	N/A
3.2.a	Processing: As a reactant	P102	Raw Materials	N/A	N/A	N/A

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.2.a	Processing: As a reactant	P103	Intermediates	U015	Intermediates	Chemical substances consumed in a reaction to produce other chemical substances for commercial advantage. A residual of the intermediate chemical substance which has no separate function may remain in the reaction product.
3.2.a	Processing: As a reactant	P104	Initiators	U024	Process regulators	Chemical substances used to change the rate of a chemical reaction, start or stop the reaction, or otherwise influence the course of the reaction. Process regulators may be consumed or become part of the reaction product.
3.2.a	Processing: As a reactant	P199	Other	U016	Ion exchange agents	Chemical substances, usually in the form of a solid matrix, that are used to selectively remove targeted ions from a solution. Examples generally consist of an inert hydrophobic matrix such as styrenedivinylbenzene or phenol-formaldehyde, cross-linking polymer such as divinylbenzene, and ionic functional groups including sulfonic, carboxylic or phosphonic acids. This code also includes aluminosilicate zeolites.
3.2.a	Processing: As a reactant	P199	Other	U019	Oxidizing/ reducing agent	Chemical substances used to alter the valence state of another substance by donating or accepting electrons or by the addition or removal of hydrogen to a substance. Examples of oxidizing agents include nitric acid, perchlorates, hexavalent chromium compounds, and peroxydisulfuric acid salts. Examples of reducing agents include hydrazine, sodium thiosulfate, and coke produced from coal.
3.2.a	Processing: As a reactant	P199	Other	U999	Other (specify)	Chemical substances used in a way other than those described by other codes.
3.2.b	Processing: As a formulation component	N/A	N/A	PF	Processing-incorporation into formulation, mixture, or reaction product	Chemical substance is added to a product (or product mixture) prior to further distribution of the product.
3.2.b	Processing: As a formulation component	P201	Additives	U007	Corrosion inhibitors and anti-scaling agents	Chemical substances used to prevent or retard corrosion or the formation of scale. Examples include phenylenediamine, chromates, nitrates, phosphates, and hydrazine.

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.2.b	Processing: As a formulation component	P201	Additives	U009	Fillers	Chemical substances used to provide bulk, increase strength, increase hardness, or improve resistance to impact. Fillers incorporated in a matrix reduce production costs by minimizing the amount of more expensive substances used in the production of articles. Examples include calcium carbonate, barium sulfate, silicates, clays, zinc oxide and aluminum oxide.
3.2.b	Processing: As a formulation component	P201	Additives	U010	Finishing agents	Chemical substances used to impart such functions as softening, static proofing, wrinkle resistance, and water repellence. Substances may be applied to textiles, paper, and leather. Examples include quaternary ammonium compounds, ethoxylated amines, and silicone compounds.
3.2.b	Processing: As a formulation component	P201	Additives	U017	Lubricants and lubricant additives	Chemical substances used to reduce friction, heat, or wear between moving parts or adjacent solid surfaces, or that enhance the lubricity of other substances. Examples of lubricants include mineral oils, silicate and phosphate esters, silicone oil, greases, and solid film lubricants such as graphite and PTFE. Examples of lubricant additives include molybdenum disulphide and tungsten disulphide.
3.2.b	Processing: As a formulation component	P201	Additives	U034	Paint additives and coating additives not described by other codes	Chemical substances used in a paint or coating formulation to enhance properties such as water repellence, increased gloss, improved fade resistance, ease of application, foam prevention, etc. Examples of paint additives and coating additives include polyols, amines, vinyl acetate ethylene emulsions, and aliphatic polyisocyanates.
3.2.b	Processing: As a formulation component	P202	Dyes	U008	Dyes	Chemical substances used to impart color to other materials or mixtures (*i.e.*, substrates) by penetrating the surface of the substrate. Example types include azo, anthraquinone, amino azo, aniline, eosin, stilbene, acid, basic or cationic, reactive, dispersive, and natural dyes.
3.2.b	Processing: As a formulation component	P202	Dyes	U021	Pigments	Chemical substances used to impart color to other materials or mixtures (*i.e.*, substrates) by attaching themselves to the surface of the substrate through binding

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
						or adhesion. This code includes fluorescent agents, luminescent agents, whitening agents, pearlizing agents, and opacifiers. Examples include metallic oxides of iron, titanium, zinc, cobalt, and chromium; metal powder suspensions; lead chromates; vegetable and animal products; and synthetic organic pigments.
3.2.b	Processing: As a formulation component	P203	Reaction Diluents	U030	Solvents (which become part of product formulation or mixture)	Chemical substances used to dissolve another substance (solute) to form a uniformly dispersed mixture (solution) at the molecular level. Examples include diluents used to reduce the concentration of an active material to achieve a specified effect and low gravity materials added to reduce cost.
3.2.b	Processing: As a formulation component	P203	Reaction Diluents	U032	Viscosity adjustors	Chemical substances used to alter the viscosity of another substance. Examples include viscosity index (VI) improvers, pour point depressants, and thickeners.
3.2.b	Processing: As a formulation component	P204	Initiators	U024	Process regulators	Chemical substances used to change the rate of a chemical reaction, start or stop the reaction, or otherwise influence the course of the reaction. Process regulators may be consumed or become part of the reaction product.
3.2.b	Processing: As a formulation component	P205	Solvents	U030	Solvents (which become part of product formulation or mixture)	Chemical substances used to dissolve another substance (solute) to form a uniformly dispersed mixture (solution) at the molecular level. Examples include diluents used to reduce the concentration of an active material to achieve a specified effect and low gravity materials added to reduce cost.
3.2.b	Processing: As a formulation component	P206	Inhibitors	U024	Process regulators	Chemical substances used to change the rate of a chemical reaction, start or stop the reaction, or otherwise influence the course of the reaction. Process regulators may be consumed or become part of the reaction product.
3.2.b	Processing: As a formulation component	P207	Emulsifiers	U003	Adsorbents and absorbents	Chemical substances used to retain other substances by accumulation on their surface or by assimilation. Examples of adsorbents include silica gel, activated alumina, and activated carbon. Examples of absorbents include straw oil, alkaline solutions, and kerosene.

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.2.b	Processing: As a formulation component	P208	Surfactants	U002	Adhesives and sealant chemicals	Chemical substances used to promote bonding between other substances, promote adhesion of surfaces, or prevent seepage of moisture or air. Examples include epoxides, isocyanates, acrylamides, phenol, urea, melamine, and formaldehyde.
3.2.b	Processing: As a formulation component	P208	Surfactants	U023	Plating agents and surface treating agents	Chemical substances applied to metal, plastic, or other surfaces to alter physical or chemical properties of the surface. Examples include metal surface treating agents, strippers, etchants, rust and tarnish removers, and descaling agents.
3.2.b	Processing: As a formulation component	P208	Surfactants	U031	Surface active agents	Chemical substances used to modify surface tension when dissolved in water or water solutions or reduce interfacial tension between two liquids or between a liquid and a solid or between liquid and air. Examples include carboxylates, sulfonates, phosphates, carboxylic acid, esters, and quaternary ammonium salts.
3.2.b	Processing: As a formulation component	P209	Lubricants	U017	Lubricants and lubricant additives	Chemical substances used to reduce friction, heat, or wear between moving parts or adjacent solid surfaces, or that enhance the lubricity of other substances. Examples of lubricants include mineral oils, silicate and phosphate esters, silicone oil, greases, and solid film lubricants such as graphite and PTFE. Examples of lubricant additives include molybdenum disulphide and tungsten disulphide.
3.2.b	Processing: As a formulation component	P210	Flame Retardants	U011	Flame retardants	Chemical substances used on the surface of or incorporated into combustible materials to reduce or eliminate their tendency to ignite when exposed to heat or a flame for a short period of time. Examples include inorganic salts, chlorinated or brominated organic compounds, and organic phosphates/phosphonates.
3.2.b	Processing: As a formulation component	P211	Rheological Modifiers	U022	Plasticizers	Chemical substances used in plastics, cement, concrete, wallboard, clay bodies, or other materials to increase their plasticity or fluidity. Examples include phthalates, trimellitates, adipates, maleates, and lignosulphonates.

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.2.b	Processing: As a formulation component	P211	Rheological Modifiers	U032	Viscosity adjustors	Chemical substances used to alter the viscosity of another substance. Examples include viscosity index (VI) improvers, pour point depressants, and thickeners.
3.2.b	Processing: As a formulation component	P299	Other	U003	Adsorbents and absorbents	Chemical substances used to retain other substances by accumulation on their surface or by assimilation. Examples of adsorbents include silica gel, activated alumina, and activated carbon. Examples of absorbents include straw oil, alkaline solutions, and kerosene.
3.2.b	Processing: As a formulation component	P299	Other	U016	Ion exchange agents	Chemical substances, usually in the form of a solid matrix, that are used to selectively remove targeted ions from a solution. Examples generally consist of an inert hydrophobic matrix such as styrenedivinylbenzene or phenol-formaldehyde, cross-linking polymer such as divinylbenzene, and ionic functional groups including sulfonic, carboxylic or phosphonic acids. This code also includes aluminosilicate zeolites.
3.2.b	Processing: As a formulation component	P299	Other	U018	Odor agents	Chemical substances used to control odors, remove odors, mask odors, or impart odors. Examples include benzenoids, terpenes and terpenoids, musk chemicals, aliphatic aldehydes, aliphatic cyanides, and mercaptans.
3.2.b	Processing: As a formulation component	P299	Other	U019	Oxidizing/ reducing agent	Chemical substances used to alter the valence state of another substance by donating or accepting electrons or by the addition or removal of hydrogen to a substance. Examples of oxidizing agents include nitric acid, perchlorates, hexavalent chromium compounds, and peroxydisulfuric acid salts. Examples of reducing agents include hydrazine, sodium thiosulfate, and coke produced from coal.
3.2.b	Processing: As a formulation component	P299	Other	U020	Photosensitive chemicals	Chemical substances used for their ability to alter their physical or chemical structure through absorption of light, resulting in the emission of light, dissociation, discoloration, or other chemical reaction. Examples include sensitizers, fluorescents, photovoltaic agents, ultraviolet absorbers, and ultraviolet stabilizers.

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.2.b	Processing: As a formulation component	P299	Other	U027	Propellants and blowing agents	Chemical substances used to dissolve or suspend other substances and either to expel those substances from a container in the form of an aerosol or to impart a cellular structure to plastics, rubber, or thermo set resins. Examples include compressed gasses and liquids and substances which release ammonia, carbon dioxide, or nitrogen.
3.2.b	Processing: As a formulation component	P299	Other	U028	Solid separation agents	Chemical substances used to promote the separation of suspended solids from a liquid. Examples include flotation aids, flocculants, coagulants, dewatering aids, and drainage aids.
3.2.b	Processing: As a formulation component	P299	Other	U999	Other (specify)	Chemical substances used in a way other than those described by other codes.
3.2.c	Processing: As an article component	N/A	N/A	PA	Processing-incorporation into article	Chemical substance becomes an integral component of an article distributed for industrial, trade, or consumer use.
3.2.c	Processing: As an article component	N/A	N/A	U008	Dyes	Chemical substances used to impart color to other materials or mixtures (*i.e.*, substrates) by penetrating into the surface of the substrate. Examples types include azo, anthraquinone, amino azo, aniline, eosin, stilbene, acid, basic or cationic, reactive, dispersive, and natural dyes.
3.2.c	Processing: As an article component	N/A	N/A	U009	Fillers	Chemical substances used to provide bulk, increase strength, increase hardness, or improve resistance to impact. Fillers incorporated in a matrix reduce production costs by minimizing the amount of more expensive substances used in the production of articles. Examples include calcium carbonate, barium sulfate, silicates, clays, zinc oxide and aluminum oxide.
3.2.c	Processing: As an article component	N/A	N/A	U021	Pigments	Chemical substances used to impart color to other materials or mixtures (*i.e.*, substrates) by attaching themselves to the surface of the substrate through binding or adhesion. This code includes fluorescent agents, luminescent agents, whitening agents, pearlizing agents,

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
						and opacifiers. Examples include metallic oxides of iron, titanium, zinc, cobalt, and chromium; metal powder suspensions; lead chromates; vegetable and animal products; and synthetic organic pigments.
3.2.c	Processing: As an article component	N/A	N/A	U034	Paint additives and coating additives not described by other codes	Chemical substances used in a paint or coating formulation to enhance properties such as water repellence, increased gloss, improved fade resistance, ease of application, foam prevention, etc. Examples of paint additives and coating additives include polyols, amines, vinyl acetate ethylene emulsions, and aliphatic polyisocyanates.
3.2.c	Processing: As an article component	N/A	N/A	U999	Other (specify)	Chemical substances used in a way other than those described by other codes.
3.2.d	Processing: Repackaging	N/A	N/A	PK	Processing-repackaging	Preparation of a chemical substance for distribution in commerce in a different form, state, or quantity. This includes transferring the chemical substance from a bulk container into smaller containers. This definition does not apply to sites that only relabel or redistribute the reportable chemical substance without removing the chemical substance from the container in which it is received or purchased.
3.2.e	Processing: As an impurity	N/A	N/A	N/A	N/A	N/A
3.2.f	Processing: Recycling	N/A	N/A	N/A	N/A	N/A
3.3.a	Otherwise Use: As a chemical processing aid	N/A	N/A	U	Use-non incorporative Activities	Chemical substance is otherwise used (*e.g.*, as a chemical processing or manufacturing aid).
3.3.a	Otherwise Use: As a chemical processing aid	Z101	Process Solvents	U029	Solvents (for cleaning or degreasing)	Chemical substances used to dissolve oils, greases, and similar materials from textiles, glassware, metal surfaces, and other articles. Examples include trichloroethylene, perchloroethylene, methylene chloride, liquid carbon dioxide, and n-propyl bromide.

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.3.a	Otherwise Use: As a chemical processing aid	Z102	Catalysts	U020	Photosensitive chemicals	Chemical substances used for their ability to alter their physical or chemical structure through absorption of light, resulting in the emission of light, dissociation, discoloration, or other chemical reaction. Examples include sensitizers, fluorescents, photovoltaic agents, ultraviolet absorbers, and ultraviolet stabilizers.
3.3.a	Otherwise Use: As a chemical processing aid	Z102	Catalysts	U025	Processing aids, specific to petroleum production	Chemical substances added to water-, oil-, or synthetic drilling muds or other petroleum production fluids to control viscosity, foaming, corrosion, alkalinity and pH, microbiological growth, hydrate formation, etc., during the production of oil, gas, and other products from beneath the earth's surface.
3.3.a	Otherwise Use: As a chemical processing aid	Z102	Catalysts	U026	Processing aids, not otherwise listed	Chemical substances used to improve the processing characteristics or the operation of process equipment or to alter or buffer the pH of the substance or mixture, when added to a process or to a substance or mixture to be processed. Processing agents do not become a part of the reaction product and are not intended to affect the function of a substance or article created. Examples include buffers, dehumidifiers, dehydrating agents, sequestering agents, and chelators.
3.3.a	Otherwise Use: As a chemical processing aid	Z103	Inhibitors	U024	Process regulators	Chemical substances used to change the rate of a chemical reaction, start or stop the reaction, or otherwise influence the course of the reaction. Process regulators may be consumed or become part of the reaction product.
3.3.a	Otherwise Use: As a chemical processing aid	Z103	Inhibitors	U025	Processing aids, specific to petroleum production	Chemical substances added to water-, oil-, or synthetic drilling muds or other petroleum production fluids to control viscosity, foaming, corrosion, alkalinity and pH, microbiological growth, hydrate formation, etc., during the production of oil, gas, and other products from beneath the earth's surface.
3.3.a	Otherwise Use: As a chemical processing aid	Z103	Inhibitors	U026	Processing aids, not otherwise listed	Chemical substances used to improve the processing characteristics or the operation of process equipment or to alter or buffer the pH of the substance or mixture, when

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
						added to a process or to a substance or mixture to be processed. Processing agents do not become a part of the reaction product and are not intended to affect the function of a substance or article created. Examples include buffers, dehumidifiers, dehydrating agents, sequestering agents, and chelators.
3.3.a	Otherwise Use: As a chemical processing aid	Z104	Initiators	U024	Process regulators	Chemical substances used to change the rate of a chemical reaction, start or stop the reaction, or otherwise influence the course of the reaction. Process regulators may be consumed or become part of the reaction product.
3.3.a	Otherwise Use: As a chemical processing aid	Z104	Initiators	U025	Processing aids, specific to petroleum production	Chemical substances added to water-, oil-, or synthetic drilling muds or other petroleum production fluids to control viscosity, foaming, corrosion, alkalinity and pH, microbiological growth, hydrate formation, etc., during the production of oil, gas, and other products from beneath the earth's surface.
3.3.a	Otherwise Use: As a chemical processing aid	Z104	Initiators	U026	Processing aids, not otherwise listed	Chemical substances used to improve the processing characteristics or the operation of process equipment or to alter or buffer the pH of the substance or mixture, when added to a process or to a substance or mixture to be processed. Processing agents do not become a part of the reaction product and are not intended to affect the function of a substance or article created. Examples include buffers, dehumidifiers, dehydrating agents, sequestering agents, and chelators.
3.3.a	Otherwise Use: As a chemical processing aid	Z105	Reaction Terminators	U024	Process regulators	Chemical substances used to change the rate of a chemical reaction, start or stop the reaction, or otherwise influence the course of the reaction. Process regulators may be consumed or become part of the reaction product.
3.3.a	Otherwise Use: As a chemical processing aid	Z105	Reaction Terminators	U025	Processing aids, specific to petroleum production	Chemical substances added to water-, oil-, or synthetic drilling muds or other petroleum production fluids to control viscosity, foaming, corrosion, alkalinity and pH, microbiological growth, hydrate formation, etc., during

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
						the production of oil, gas, and other products from beneath the earth's surface.
3.3.a	Otherwise Use: As a chemical processing aid	Z105	Reaction Terminators	U026	Processing aids, not otherwise listed	Chemical substances used to improve the processing characteristics or the operation of process equipment or to alter or buffer the pH of the substance or mixture, when added to a process or to a substance or mixture to be processed. Processing agents do not become a part of the reaction product and are not intended to affect the function of a substance or article created. Examples include buffers, dehumidifiers, dehydrating agents, sequestering agents, and chelators.
3.3.a	Otherwise Use: As a chemical processing aid	Z106	Solution Buffers	U026	Processing aids, not otherwise listed	Chemical substances used to improve the processing characteristics or the operation of process equipment or to alter or buffer the pH of the substance or mixture, when added to a process or to a substance or mixture to be processed. Processing agents do not become a part of the reaction product and are not intended to affect the function of a substance or article created. Examples include buffers, dehumidifiers, dehydrating agents, sequestering agents, and chelators.
3.3.a	Otherwise Use: As a chemical processing aid	Z199	Other	U002	Adhesives and sealant chemicals	Chemical substances used to promote bonding between other substances, promote adhesion of surfaces, or prevent seepage of moisture or air. Examples include epoxides, isocyanates, acrylamides, phenol, urea, melamine, and formaldehyde.
3.3.a	Otherwise Use: As a chemical processing aid	Z199	Other	U006	Bleaching agents	Chemical substances used to lighten or whiten a substrate through chemical reaction, usually an oxidative process which degrades the color system. Examples generally fall into one of two groups: chlorine containing bleaching agents (*e.g.*, chlorine, hypochlorites, N-chloro compounds and chlorine dioxide); and peroxygen bleaching agents (*e.g.*, hydrogen peroxide, potassium permanganate, and sodium perborate).

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.3.a	Otherwise Use: As a chemical processing aid	Z199	Other	U018	Odor agents	Chemical substances used to control odors, remove odors, mask odors, or impart odors. Examples include benzenoids, terpenes and terpenoids, musk chemicals, aliphatic aldehydes, aliphatic cyanides, and mercaptans.
3.3.a	Otherwise Use: As a chemical processing aid	Z199	Other	U023	Plating agents and surface treating agents	Chemical substances applied to metal, plastic, or other surfaces to alter physical or chemical properties of the surface. Examples include metal surface treating agents, strippers, etchants, rust and tarnish removers, and descaling agents.
3.3.a	Otherwise Use: As a chemical processing aid	Z199	Other	U025	Processing aids, specific to petroleum production	Chemical substances added to water-, oil-, or synthetic drilling muds or other petroleum production fluids to control viscosity, foaming, corrosion, alkalinity and pH, microbiological growth, hydrate formation, etc., during the production of oil, gas, and other products from beneath the earth's surface.
3.3.a	Otherwise Use: As a chemical processing aid	Z199	Other	U026	Processing aids, not otherwise listed	Chemical substances used to improve the processing characteristics or the operation of process equipment or to alter or buffer the pH of the substance or mixture, when added to a process or to a substance or mixture to be processed. Processing agents do not become a part of the reaction product and are not intended to affect the function of a substance or article created. Examples include buffers, dehumidifiers, dehydrating agents, sequestering agents, and chelators.
3.3.a	Otherwise Use: As a chemical processing aid	Z199	Other	U028	Solid separation agents	Chemical substances used to promote the separation of suspended solids from a liquid. Examples include flotation aids, flocculants, coagulants, dewatering aids, and drainage aids.
3.3.b	Otherwise Use: As a manufacturing aid	N/A	N/A	U	Use−non incorporative Activities	Chemical substance is otherwise used (*e.g.*, as a chemical processing or manufacturing aid).
3.3.b	Otherwise Use: As a	Z201	Process Lubricants	U017	Lubricants and lubricant additives	Chemical substances used to reduce friction, heat, or wear between moving parts or adjacent solid surfaces, or that

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
	manufacturing aid					enhance the lubricity of other substances. Examples of lubricants include mineral oils, silicate and phosphate esters, silicone oil, greases, and solid film lubricants such as graphite and PTFE. Examples of lubricant additives include molybdenum disulphide and tungsten disulphide.
3.3.b	Otherwise Use: As a manufacturing aid	Z202	Metalworking Fluids	U007	Corrosion inhibitors and antiscaling agents	Chemical substances used to prevent or retard corrosion or the formation of scale. Examples include phenylenediamine, chromates, nitrates, phosphates, and hydrazine.
3.3.b	Otherwise Use: As a manufacturing aid	Z202	Metalworking Fluids	U014	Functional fluids (open systems)	Liquid or gaseous chemical substances used for one or more operational properties in an open system. Examples include antifreezes and de−icing fluids such as ethylene and propylene glycol, sodium formate, potassium acetate, and sodium acetate. This code also includes substances incorporated into metal working fluids.
3.3.b	Otherwise Use: As a manufacturing aid	Z203	Coolants	U013	Functional fluids (closed systems)	Liquid or gaseous chemical substances used for one or more operational properties in a closed system. Examples include: heat transfer agents (*e.g.*, coolants and refrigerants) such as polyalkylene glycols, silicone oils, liquified propane, and carbon dioxide; hydraulic/transmission fluids such as mineral oils, organophosphate esters, silicone, and propylene glycol; and dielectric fluids such as mineral insulating oil and high flash point kerosene. This code does not include fluids used as lubricants.
3.3.b	Otherwise Use: As a manufacturing aid	Z204	Refrigerants	U013	Functional fluids (closed systems)	Liquid or gaseous chemical substances used for one or more operational properties in a closed system. Examples include: heat transfer agents (*e.g.*, coolants and refrigerants) such as polyalkylene glycols, silicone oils, liquified propane, and carbon dioxide; hydraulic/transmission fluids such as mineral oils, organophosphate esters, silicone, and propylene glycol; and dielectric fluids such as mineral insulating oil and

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
						high flash point kerosene. This code does not include fluids used as lubricants.
3.3.b	Otherwise Use: As a manufacturing aid	Z205	Hydraulic Fluids	U013	Functional fluids (closed systems)	Liquid or gaseous chemical substances used for one or more operational properties in a closed system. Examples include: heat transfer agents (*e.g.*, coolants and refrigerants) such as polyalkylene glycols, silicone oils, liquified propane, and carbon dioxide; hydraulic/transmission fluids such as mineral oils, organophosphate esters, silicone, and propylene glycol; and dielectric fluids such as mineral insulating oil and high flash point kerosene. This code does not include fluids used as lubricants.
3.3.b	Otherwise Use: As a manufacturing aid	Z299	Other	U013	Functional fluids (closed systems)	Liquid or gaseous chemical substances used for one or more operational properties in a closed system. Examples include: heat transfer agents (*e.g.*, coolants and refrigerants) such as polyalkylene glycols, silicone oils, liquified propane, and carbon dioxide; hydraulic/transmission fluids such as mineral oils, organophosphate esters, silicone, and propylene glycol; and dielectric fluids such as mineral insulating oil and high flash point kerosene. This code does not include fluids used as lubricants.
3.3.b	Otherwise Use: As a manufacturing aid	Z299	Other	U023	Plating agents and surface treating agents	Chemical substances applied to metal, plastic, or other surfaces to alter physical or chemical properties of the surface. Examples include metal surface treating agents, strippers, etchants, rust and tarnish removers, and descaling agents.
3.3.c	Otherwise Use: Ancillary or other use	N/A	N/A	U	Use−non incorporative Activities	Chemical substance is otherwise used (*e.g.*, as a chemical processing or manufacturing aid).
3.3.c	Otherwise Use: Ancillary or other use	Z301	Cleaner	U007	Corrosion inhibitors and antiscaling agents	Chemical substances used to prevent or retard corrosion or the formation of scale. Examples include phenylenediamine, chromates, nitrates, phosphates, and hydrazine.

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.3.c	Otherwise Use: Ancillary or other use	Z301	Cleaner	U029	Solvents (for cleaning or degreasing)	Chemical substances used to dissolve oils, greases, and similar materials from textiles, glassware, metal surfaces, and other articles. Examples include trichloroethylene, perchloroethylene, methylene chloride, liquid carbon dioxide, and n-propyl bromide.
3.3.c	Otherwise Use: Ancillary or other use	Z302	Degreaser	U003	Adsorbents and Absorbents	Chemical substances used to retain other substances by accumulation on their surface or by assimilation. Examples of adsorbents include silica gel, activated alumina, and activated carbon. Examples of absorbents include straw oil, alkaline solutions, and kerosene.
3.3.c	Otherwise Use: Ancillary or other use	Z302	Degreaser	U029	Solvents (for cleaning or degreasing)	Chemical substances used to dissolve oils, greases, and similar materials from textiles, glassware, metal surfaces, and other articles. Examples include trichloroethylene, perchloroethylene, methylene chloride, liquid carbon dioxide, and n-propyl bromide.
3.3.c	Otherwise Use: Ancillary or other use	Z303	Lubricant	U017	Lubricants and lubricant additives	Chemical substances used to reduce friction, heat, or wear between moving parts or adjacent solid surfaces, or that enhance the lubricity of other substances. Examples of lubricants include mineral oils, silicate and phosphate esters, silicone oil, greases, and solid film lubricants such as graphite and PTFE. Examples of lubricant additives include molybdenum disulphide and tungsten disulphide.
3.3.c	Otherwise Use: Ancillary or other use	Z304	Fuel	U012	Fuels and fuel additives	Chemical substances used to create mechanical or thermal energy through chemical reactions, or which are added to a fuel for the purpose of controlling the rate of reaction or limiting the production of undesirable combustion products, or which provide other benefits such as corrosion inhibition, lubrication, or detergency. Examples of fuels include coal, oil, gasoline, and various grades of diesel fuel. Examples of fuel additives include oxygenated compound such as ethers and alcohols, antioxidants such as phenylenediamines and hindered phenols, corrosion inhibitors such as carboxylic acids,

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
						amines, and amine salts, and blending agents such as ethanol.
3.3.c	Otherwise Use: Ancillary or other use	Z305	Flame Retardant	U011	Flame retardants	Chemical substances used on the surface of or incorporated into combustible materials to reduce or eliminate their tendency to ignite when exposed to heat or a flame for a short period of time. Examples include inorganic salts, chlorinated or brominated organic compounds, and organic phosphates/phosphonates.
3.3.c	Otherwise Use: Ancillary or other use	Z306	Waste Treatment	U006	Bleaching agents	Chemical substances used to lighten or whiten a substrate through chemical reaction, usually an oxidative process which degrades the color system. Examples generally fall into one of two groups: chlorine containing bleaching agents (*e.g.*, chlorine, hypochlorites, N-chloro compounds and chlorine dioxide); and peroxygen bleaching agents (*e.g.*, hydrogen peroxide, potassium permanganate, and sodium perborate).
3.3.c	Otherwise Use: Ancillary or other use	Z306	Waste Treatment	U018	Odor agents	Chemical substances used to control odors, remove odors, mask odors, or impart odors. Examples include benzenoids, terpenes and terpenoids, musk chemicals, aliphatic aldehydes, aliphatic cyanides, and mercaptans.
3.3.c	Otherwise Use: Ancillary or other use	Z306	Waste Treatment	U019	Oxidizing/reducing agent	Chemical substances used to alter the valence state of another substance by donating or accepting electrons or by the addition or removal of hydrogen to a substance. Examples of oxidizing agents include nitric acid, perchlorates, hexavalent chromium compounds, and peroxydisulfuric acid salts. Examples of reducing agents include hydrazine, sodium thiosulfate, and coke produced from coal.
3.3.c	Otherwise Use: Ancillary or other use	Z306	Waste Treatment	U028	Solid separation agents	Chemical substances used to promote the separation of suspended solids from a liquid. Examples include flotation aids, flocculants, coagulants, dewatering aids, and drainage aids.

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
3.3.c	Otherwise Use: Ancillary or other use	Z307	Water Treatment	U006	Bleaching agents	Chemical substances used to lighten or whiten a substrate through chemical reaction, usually an oxidative process which degrades the color system. Examples generally fall into one of two groups: chlorine containing bleaching agents (*e.g.*, chlorine, hypochlorites, N-chloro compounds and chlorine dioxide); and, peroxygen bleaching agents (*e.g.*, hydrogen peroxide, potassium permanganate, and sodium perborate).
3.3.c	Otherwise Use: Ancillary or other use	Z307	Water Treatment	U018	Odor agents	Chemical substances used to control odors, remove odors, mask odors, or impart odors. Examples include benzenoids, terpenes and terpenoids, musk chemicals, aliphatic aldehydes, aliphatic cyanides, and mercaptans.
3.3.c	Otherwise Use: Ancillary or other use	Z307	Water Treatment	U019	Oxidizing/reducing agent	Chemical substances used to alter the valence state of another substance by donating or accepting electrons or by the addition or removal of hydrogen to a substance. Examples of oxidizing agents include nitric acid, perchlorates, hexavalent chromium compounds, and peroxydisulfuric acid salts. Examples of reducing agents include hydrazine, sodium thiosulfate, and coke produced from coal.
3.3.c	Otherwise Use: Ancillary or other use	Z307	Water Treatment	U028	Solid separation agents	Chemical substances used to promote the separation of suspended solids from a liquid. Examples include flotation aids, flocculants, coagulants, dewatering aids, and drainage aids.
3.3.c	Otherwise Use: Ancillary or other use	Z308	Construction Materials	N/A	N/A	N/A
3.3.c	Otherwise Use: Ancillary or other use	Z399	Other	U001	Abrasives	Chemical substances used to wear down or polish surfaces by rubbing against the surface. Examples include sandstones, pumice, silex, quartz, silicates, aluminum oxides, and glass.
3.3.c	Otherwise Use: Ancillary or other use	Z399	Other	U013	Functional fluids (closed systems)	Liquid or gaseous chemical substances used for one or more operational properties in a closed system. Examples include: heat transfer agents (*e.g.*, coolants and

TRI Section	TRI Description	TRI Sub-use Code	TRI Sub-use Code Name	2016 CDR Code	2016 CDR Code Name	2016 CDR Functional Use Definition
						refrigerants) such as polyalkylene glycols, silicone oils, liquified propane, and carbon dioxide; hydraulic/transmission fluids such as mineral oils, organophosphate esters, silicone, and propylene glycol; and dielectric fluids such as mineral insulating oil and high flash point kerosene. This code does not include fluids used as lubricants.
3.3.c	Otherwise Use: Ancillary or other use	Z399	Other	U014	Functional fluids (open systems)	Liquid or gaseous chemical substances used for one or more operational properties in an open system. Examples include antifreezes and de-icing fluids such as ethylene and propylene glycol, sodium formate, potassium acetate, and, sodium acetate. This code also includes substances incorporated into metal working fluids.
3.3.c	Otherwise Use: Ancillary or other use	Z399	Other	U018	Odor agents	Chemical substances used to control odors, remove odors, mask odors, or impart odors. Examples include benzenoids, terpenes and terpenoids, musk chemicals, aliphatic aldehydes, aliphatic cyanides, and mercaptans.
3.3.c	Otherwise Use: Ancillary or other use	Z399	Other	U020	Photosensitive chemicals	Chemical substances used for their ability to alter their physical or chemical structure through absorption of light, resulting in the emission of light, dissociation, discoloration, or other chemical reaction. Examples include sensitizers, fluorescents, photovoltaic agents, ultraviolet absorbers, and ultraviolet stabilizers.
3.3.c	Otherwise Use: Ancillary or other use	Z399	Other	U023	Plating agents and surface treating agents	Chemical substances applied to metal, plastic, or other surfaces to alter physical or chemical properties of the surface. Examples include metal surface treating agents, strippers, etchants, rust and tarnish removers, and descaling agents.

E.10 Developing Models that Use Monte Carlo Methods

This appendix provides background information on Monte Carlo methods, including an overview of deterministic and stochastic processes, an overview of the implementation of Monte Carlo methods, and a discussion of EPA's approach for building models that utilized Monte Carlo methods.

This appendix is only intended to provide general background information; information related to the specific models for which EPA implemented Monte Carlo methods is included in Appendices E.11 through E.13 and Appendices F.5 through F.9.

E.10.1 Background on Monte Carlo Methods

A *deterministic* process has a single output (or set of outputs) for a given input (or set of inputs). The process does not involve randomness and the direction of the process is known.

In contrast, *stochastic* processes are non-deterministic. The output is based on random trials and can proceed via multiple, or even infinite, directions.

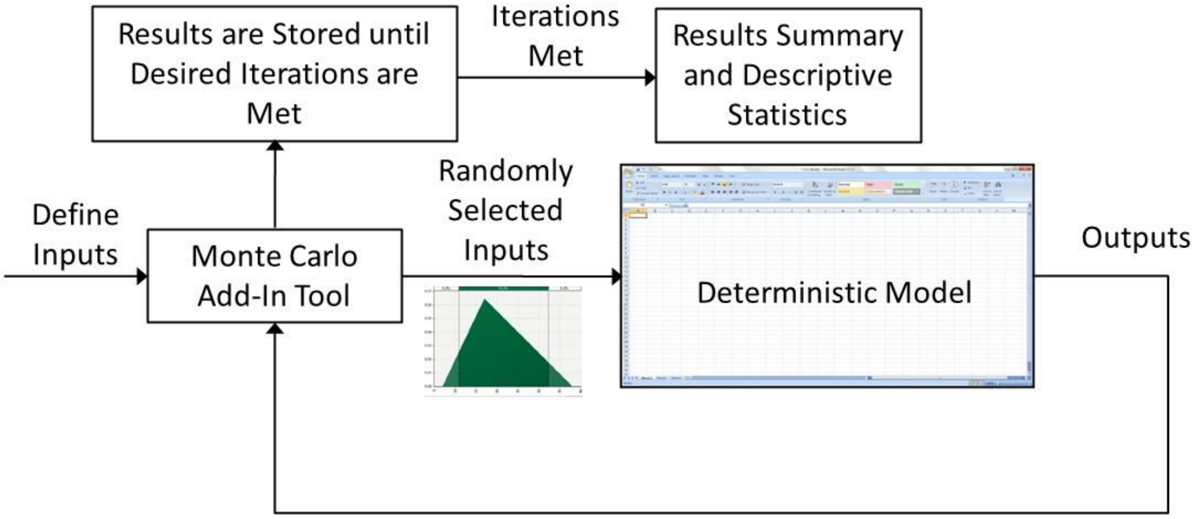
Monte Carlo methods fall under the umbrella of stochastic modeling. Monte Carlo methods are a replication technique for propagating uncertainty through a model. The model is run multiple times, and each run uses different input values and generates different output values: each run is independent of each other. The sample of output values is used to estimate the properties of the actual probability distribution of the outputs.

E.10.2 Implementation of Monte Carlo Methods

The implementation of Monte Carlo methods generally follows the following steps:

1. Define probability distributions for input parameters.
2. Generate a set of input values by randomly drawing a sample from each probability distribution.
3. Execute the deterministic model calculations.
4. Save the output results.
5. Repeat steps 2 through 4 through the desired number of iterations.
6. Aggregate the saved output results and calculate statistics.

Figure_Apx E-1 illustrates a flowchart of a Monte Carlo method implemented in a Microsoft Excel-based model using a Monte Carlo add-in tool, such as the Palisade @Risk software.



Figure_Apx E-1. Flowchart of a Monte Carlo Method Implemented in a Microsoft Excel-Based Model Using a Monte Carlo Add-In Tool

E.10.3 Building the Model

The steps for building a release or exposure model that incorporates Monte Carlo methods are as follows:

1. Build the deterministic model.
2. Define probability distributions for input parameters.
3. Select model outputs for aggregation of simulation results.
4. Select simulation settings and run model.
5. Aggregate the simulation results and calculate output statistics.

Each of these steps is discussed in the subsections below.

E.10.3.1 Build the Deterministic Model

First, the model is built as a deterministic model. EPA uses Microsoft Excel in order to use Palisade's @Risk software that is used for probabilistic analyses in Excel. The model parameters and equations are programmed into the spreadsheet. Model parameters are programmed in a summary table format for transparency and to aid in the assignment of probability distributions. Such summary tables are included in the model-specific write-ups in Appendices E.11 through E.13 and Appendices F.5 through F.9.

E.10.3.2 Define Probability Distributions for Input Parameters

Defining a probability distribution for an input parameter generally involves three steps:

1. Select the model input parameters for which probability distributions will be developed.
2. Determine a probability distribution from the available data.
3. Investigate if any parameters are statistically correlated. Define a statistical correlation among parameters if a correlation is desired.

Step 1: Select Input Parameters for Probability Distribution Development

When selecting parameters for which probability distributions will be developed, the following factors are considered:

- The availability of data to inform a distribution.
- The dependency of the input parameters on one another.
- The sensitivity of the model results to each input parameter.

Availability of Data to Inform a Distribution: Data sources to investigate for available data to inform probability distributions of model inputs include but are not limited to the following:

- EPA Generic Scenarios,
- OECD Emission Scenario Documents,
- Peer reviewed literature,
- Published chemical assessments, and
- Other gray literature.[20]

Model parameters may vary greatly in their available data. There may be a single study that provides detailed measurements or observation data. There may be multiple studies that provide limited measurements or observations. There may be only overall statistics available for a parameter. For a given model development, the available data goes through a systematic review process to evaluate the data quality, integrate the data, and decide how to use the data.

Dependency of Input Parameters on One Another: The model parameters are evaluated for any dependency on each other. When each varied parameter is sampled according to its defined probability distribution, they are sampled independently of each other. Therefore, the value of a sampled parameter should be independent of the other sampled parameters. An exception is if a statistical correlation is desired among two or more parameters. Correlating sampled parameters is discussed below in Step 3.

An example of dependency is the relationship between a facility's number of operating days, annual production volume (PV), and daily PV. These three parameters are not all independent of each other. The annual PV may be calculated from the daily PV and the operating days. Alternatively, the daily PV may be calculated from the annual PV and the operating days. Additionally, operating days may be calculated from the annual PV and daily PV. It is necessary to first understand the mathematical relationship among these parameters before selecting parameters for which probability distributions will be developed.

Sensitivity of the Model Results to Each Input Parameter: One consideration in selecting model parameters for probability distribution development is the sensitivity of the model outputs to each parameter. A sensitivity analysis can inform how sensitive each model output is to each model input parameter. EPA may choose to prioritize probability distribution development for parameters to which model outputs are more sensitive. Since the model outputs are more sensitive to these parameters, it would be more important to capture variability and/or uncertainty for these parameters compared to parameters to which model outputs are less sensitive.

A sensitivity analysis is conducted by varying each desired parameter and performing a Monte Carlo simulation. The varied range for each parameter should be consistent with the expected range in values for the parameter. The @Risk software (Palisade, 2022b) can perform sensitivity analyses. The statistic of the outputs for which sensitivity is measured, such as mean, mode, or a percentile, can be selected. As the simulation is run, the software tracks how each output changes with respect to each varied input.

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[20] Gray literature is defined as the broad category of data/information sources not found in standard, peer-reviewed literature databases. Gray literature includes data/information sources such as white papers, conference proceedings, technical reports, reference books, dissertations, information on various stakeholder websites, and various databases.

Step 2: Determine a Probability Distribution

To determine a probability distribution, first, all the information known about the parameter is evaluated (Oracle, 2017). The following considerations can help guide summarizing important information about the parameter (Analytica, 2015):

- Discrete or continuous
  - Consider whether the parameter is discrete or continuous. Does the parameter have a finite or countable number of possible values? Is the parameter logical or Boolean such as having possible values of “yes or no” or “true or false”? Can the parameter be represented by all real numbers within a domain?
- Bounds
  - Consider whether the parameter has bounds. A parameter may have a lower bound and/or an upper bound. Alternatively, a parameter may be unbounded and can range to negative and/or positive infinity.
- Modes
  - Consider whether the parameter has one or more modes. Does the parameter have no mode (such as represented by a uniform distribution)? If it has a mode, is it unimodal or multimodal? If multimodal, is the parameter a combination of two or more populations? In which case, the parameter may be best separated into its separate components and then develop probability distributions for the individual components.
- Symmetric or skewed
  - Consider whether the parameter is symmetric or skewed. If skewed, consider whether the parameter is positively skewed (thicker upper tail) or negatively skewed (thicker lower tail).

Second, review standard probability distributions and identify possible candidates that meet the considerations identified in the first step (Oracle, 2017). The following are common probability distributions:

- Uniform distribution
  - A uniform distribution has finite upper and lower bounds and all values between the bounds have equal probability.
- Triangular distribution
  - A triangular distribution has finite upper and lower bounds and a modal value. The modal value is the value that occurs most frequently. If the most frequent value is not known another statistic, such as the mean or a percentile, could be used to define the triangular distribution.
- Normal distribution
  - The parameters of a normal distribution are its mean and standard deviation. A normal distribution is unbounded, and values range from negative to positive infinity. If desired, the range of values of a normal distribution may be truncated to finite bounds to prevent unrealistic values from being sampled.
- Lognormal distribution
  - If a variable is lognormally distributed, it means that the logarithm of that variable is normally distributed. The parameters of a lognormal distribution are its mean and standard deviation. A lognormal distribution is bounded from zero to positive infinity. A lognormal distribution may be shifted and its upper bound truncated to fit the observed data and prevent unrealistic values from being sampled.

Lastly, select the best suited probability distribution (Oracle, 2017). Review the available data for the parameter to determine how to define the distribution's parameters. For example, if the only available data are an overall range (with a minimum and a maximum), then a uniform distribution is the appropriate distribution to use. If the only available data are an overall range and a mode, then a triangular distribution is the appropriate distribution to use. If historical data for the parameter are available, consider data fitting to determine the appropriate distribution and regress the distribution parameter values.

Step 3: Check for and Define Statistical Correlations

When developing a Monte Carlo model and setting statistical distributions for parameters, EPA evaluates possible correlations among parameters. When distributions are defined for the parameters, each parameter is independently sampled on each iteration of the model. This may result in combinations of parameter values that are not logical for the scenario. In the example of a model that uses annual PV, daily PV, and operating days as parameters, there are set distributions for annual PV and operating days, with the daily production volume calculated from the other two parameters. But annual PV and operating days may be correlated. For example, if a site has a fixed manufacturing capacity (as determined by the equipment size and production lines), then annual PV is a function of the number of operating days. A facility is more likely to scale-up or scale-down their annual PV by varying the operating days rather than varying their daily PV. Varying annual PV and operating days independently in the model may arrive at value combinations that are not logical. For example, one iteration may sample a high annual PV value with a low number of operating days that may result in a high daily production rate that is not logical. In this example, a different probability distribution strategy may be appropriate, such as defining probability distributions for daily PV and operating days since those two parameters are likely more independent of each other than annual PV and operating days.

When developing distributions from observed data, there are statistical tests that can be performed to indicate a statistical correlation. Two common ones are: 1) the Pearson product-moment correlation coefficient, which measures the linear correlation between two data sets; and 2) Spearman's rank correlation coefficient, which is a measure of rank correlation and how well a relationship between two data sets can be described using a monotonic function. A monotonic relationship is one where the two variables change together but not necessarily at a constant rate (Minitab, 2022). A linear correlation is necessarily monotonic. But a monotonic correlation is not necessarily linear.

Both the Pearson and Spearman coefficients range from −1 to +1. A value close to +/−1 indicates a strong correlation (either positive or negative). A positive correlation means as one variable increases, the other also increases. A negative correlation means as one variable increases, the other decreases. A value close to 0 means a weak or no correlation exists between the variables. The Pearson correlation only measures linear relationships, and the Spearman correlation only measures monotonic relationships. If two variables are correlated by a relationship that is neither linear nor monotonic, then the Pearson and Spearman coefficients would not be informative of the nature of the correlation (Minitab, 2022).

After testing for statistical correlations, statistical correlations can be defined for input parameters using @Risk. @Risk only uses Spearman coefficients to define statistical correlations among input parameters. Spearman coefficients to correlate two or more input parameters are defined through a correlation matrix. The correlation matrix allows the Spearman coefficient to be defined for each pair of correlated input parameters (Palisade, 2022a).

E.10.3.3 Select Model Outputs for Aggregation of Simulation Results

The last step before running the model is to select the model outputs for which statistical results are desired. Defining these outputs in @Risk will allow the software to save the output results from each iteration and aggregate the simulation results over all iterations together.

E.10.3.4 Select Simulation Settings and Run Model

Simulation settings must be defined before running the model. Important simulation settings include the number of iterations, the sampling method, and the random number generator.

- **Number of Iterations.** Generally speaking, a larger number of iterations is desired to ensure adequate sampling and representation of lower probability events. The number of iterations to achieve a desired margin of error for a given confidence interval for an output can be calculated using the Central Limit Theorem (Oberle, 2015; Palisade, 2015a). The equation shows that the margin of error is inversely proportional to the square root of the number of iterations. Therefore, the greater the number of iterations, the smaller the margin of error. Calculating the number of iterations can be difficult as the sample standard deviation is not known beforehand. EPA typically uses 100,000 iterations to ensure convergence and have minimal cost to the simulation time.

- **Sampling Method.** The sampling method is the method used to draw random samples from the input parameter probability distributions. @Risk uses two methods: Latin Hypercube (the default) and Monte Carlo. Monte Carlo sampling is a purely random sampling method. This can lead to clustering and under-representing low probability events. Latin Hypercube sampling is a stratified sampling method. This ensures the sampled input parameter distribution matches the assigned probability distribution closely. EPA typically uses Latin Hypercube sampling because it is efficient and can achieve convergence with fewer iterations than Monte Carlo sampling (Palisade, 2018).

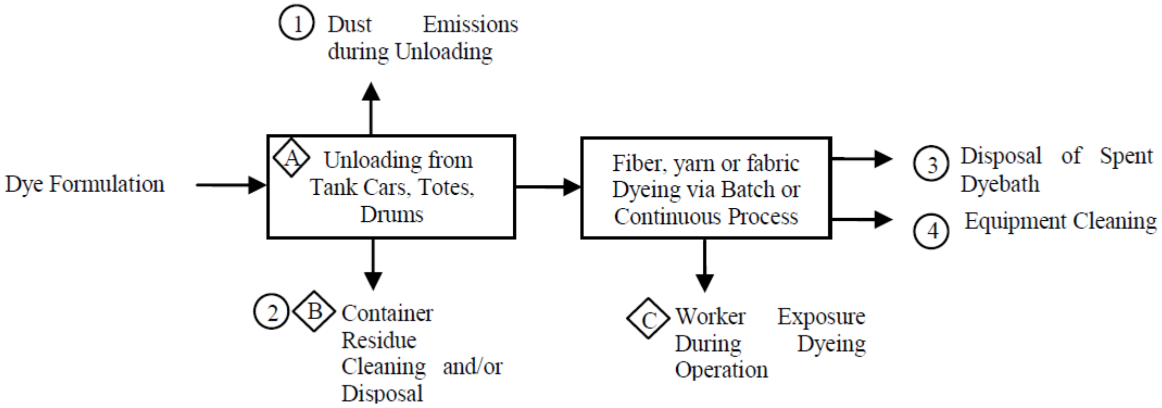
- **Random Number Generator.** The random number generator is used to generate pseudorandom numbers that are used in an algorithm to draw random samples from the probability distributions. The @Risk default is Mersenne Twister, which is a robust and efficient random number generator (Palisade, 2015b).

E.10.3.5 Aggregate the Simulation Results and Produce Output Statistics

During the simulation, @Risk will save the defined model outputs for aggregation on each iteration. After the simulation is completed, EPA can generate desired statistical results and distributions of the defined outputs. EPA typically uses the 50th percentile and 95th percentile of the output as the central tendency and high-end estimates, respectively.

E.11 Textile Dye Modeling Approach and Parameters for Estimating Environmental Releases

This appendix presents the modeling approach and equations used to estimate environmental releases of 1,4-dioxane during the commercial use of textile dyes. This approach utilizes the OECD ESD on Textile Dyes (OECD, 2017) combined with Monte Carlo simulation (a type of stochastic simulation). This ESD includes a diagram of release and exposure points during textile dying, as shown in Figure_Apx E-2.



Figure_Apx E-2. Environmental Release and Occupational Exposure Points during Textile Dying

Based on Figure_Apx E-2, EPA identified the following release points (note that diagram points 3 and 4 were combined for ease of analysis):

- Release point 1 (RP1): Dust emissions of during unloading of solid powders to air, landfill, POTW, or incineration;
- Release point 2 (RP2): Container residual losses to POTW, landfill, or incineration; and
- Release point 3 (RP3): Disposal of spent dye bath and equipment cleaning losses to POTW.

Environmental releases of textile dyes are a function of the chemical's physical properties, container size, mass fractions, and other model parameters. Although physical properties are fixed, some model parameters are expected to vary from one facility to another. An individual model input parameter could either have a discrete value or a distribution of values. EPA assigned statistical distributions based on available literature data or engineering judgment to address the variability in mass fraction of dye formulation in the bath ($F_{dye\_dyebath}$), container size ($V_{container}$), textile production rate (V_{fabric}), operating days (OD), and container residue fractions ($F_{container\_residue}$).

A Monte Carlo simulation was then conducted to capture variability in the model input parameters described above. The simulation was conducted using the Latin hypercube sampling method in @Risk (Palisade, Ithaca, NY). The Latin hypercube sampling method is a statistical method for generating a sample of possible values from a multi-dimensional distribution. Latin hypercube sampling is a stratified method, meaning it guarantees that its generated samples are representative of the probability density function (variability) defined in the model. EPA performed 100,000 iterations of the model to capture the range of possible input values, including values with low probability of occurrence.

From the distribution resulting from the Monte Carlo simulation, EPA selected the 95th and 50th percentile values to represent a high-end release and central tendency release level, respectively. The statistics were calculated directly in @Risk. The following subsections detail the model design equations and parameters used for environmental release estimates.

E.11.1 Model Equations

Daily use rate of dye formulation is calculated using the following equation:

Equation_Apx E-1

$$Q_{dye\_formulation\_day} = V_{fabric} * F_{fabric} * F_{dye\_fabric}$$

Where:

$Q_{dye\_formulation\_day}$ = Daily use rate of dye formulation [kg/site-day]
V_{fabric} = Textile production rate [kg/site-day]
F_{fabric} = Mass fraction of textiles treated with dye [kg/kg]
F_{dye_fabric} = Mass fraction of dye used per mass of textile dyed [kg/kg]

Daily use rate of 1,4-dioxane formulation is calculated using the following equation:

Equation_Apx E-2

$$Q_{dioxane\_site\_day} = Q_{dye\_formulation\_day} * F_{dioxane\_dye} * F_{dye}$$

Where:

$Q_{dioxane\_site\_day}$ = Daily use rate of 1,4-dioxane [kg/site-day]
$Q_{dye\_formulation\_day}$ = Daily use rate of dye formulation [kg/site-day]
$F_{dioxane\_dye}$ = Mass fraction of 1,4-dioxane in dye formulation [kg/kg]
F_{dye} = Fraction of dye containing 1,4-dioxane [kg/kg]

Containers emptied per facility is calculated using the following equation:

Equation_Apx E-3

$$N_{container\_unload\_site\_yr} = \frac{Q_{dioxane\_site\_day} * OD}{F_{dioxane\_dye} * V_{container} * 3.79\frac{L}{gal} * RHO_{form}}$$

Where:

$N_{container\_unload\_site\_yr}$ = Containers emptied per facility [containers/site-year]
$Q_{dioxane\_site\_day}$ = Daily use rate of 1,4-dioxane [kg/site-day]
OD = Operating days [days/year]
$F_{dioxane\_dye}$ = Mass fraction of 1,4-dioxane in dye formulation [kg/kg]
$V_{container}$ = Container size [gal/container]
RHO_{form} = Dye density [kg/L]

Container residual fraction is calculated using the below equations. To make the simulation more realistic, EPA assessed container size based on the dye formulation use rate. This avoids situations where a small container size is associated with a large use rate, such that an unrealistic number of containers are used each year, and vice-versa.

Equation_Apx E-4

If $Q_{dye\_formulation\_day}$ >700 kg/site-day:

$$F_{container\_residue} = F_{container\_residue\_tote}$$

If $Q_{dye\_formulation\_day}$ is 200-700 kg/site-day:

$$F_{container\_residue} = F_{container\_residue\_drum}$$

If $Q_{dye\_formulation\_day}$ < 200 kg/site-day

$$F_{container\_residue} = F_{container\_residue\_pail}$$

Where:

$Q_{dye\_formulation\_day}$ = Daily use rate of dye formulation [kg/site-day]
$F_{container\_residue}$ = Container residual fraction [kg/kg]
$F_{container\_residue\_tote}$ = Container residual fraction for totes [kg/kg]
$F_{container\_residue\_drum}$ = Container residual fraction for drums [kg/kg]
$F_{container\_residue\_pail}$ = Container residual fraction for pails [kg/kg]

Mass fraction of 1,4-dioxane in dye bath is calculated using the following equation:

Equation_Apx E-5

$$F_{dioxane\_dyebath} = F_{dioxane\_dye} * F_{dye\_dyebath}$$

Where:

$F_{dioxane\_dyebath}$ = Mass fraction of 1,4-dioxane in dye bath [kg/kg]
$F_{dioxane\_dye}$ = Mass fraction of 1,4-dioxane in dye formulation [kg/kg]
$F_{dye\_dyebath}$ = Mass fraction of the dye formulation in the dyebath [kg/kg]

Release point 2 (container residual) release per day is calculated using the following equation:

Equation_Apx E-6

$$Release\_perDay_{RP2} = Q_{dioxane\_site\_day} * F_{container\_residue}$$

Where:

$Release\_perDay_{RP2}$ = Container residual release from release point 2 [kg/site-day]
$Q_{dioxane\_site\_day}$ = Daily use rate of 1,4-dioxane [kg/site-day]
$F_{container\_residue}$ = Container residual fraction [kg/kg]

Release point 3 (spent dye bath and equipment cleaning) release per release day is calculated using the following equation:

Equation_Apx E-7

$$Release\_perDay_{RP3} = Q_{dioxane\_site\_day} * (1 - F_{fixation})$$

Where:

$Release\_perDay_{RP3}$ = Dye bath and equipment cleaning release from point 3 [kg/site-day]
$Q_{dioxane\_site\_day}$ = Daily use rate of 1,4-dioxane [kg/site-day]
$F_{fixation}$ = Fraction of dye affixed to textile during dye process [kg/kg]

E.11.2 Model Input Parameters

Table_Apx E-12 summarizes the model parameters and their values for the Monte Carlo simulation. Additional explanations of EPA's selection of the distributions for each parameter are provided after this table. High-end and central tendency releases are estimated by selecting the 50th and 95th percentile values from the output distribution.

Table_Apx E-12. Summary of Parameter Values and Distributions Used in the Textile Release Model

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational/Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Number of Sites	Ns	sites	783	–	–	–	–	See Section E.11.3
Fraction of dye containing 1,4-dioxane	F_{dye}	kg/kg	1	–	–	–	–	See Section E.11.4
1,4-dioxane vapor pressure	VP	Torr	40	–	–	–	–	Physical property
1,4-dioxane molecular weight	MW	g/mol	88.1	–	–	–	–	Physical property
Operating days	OD	days/year	157	30	296	157	Triangular	See Section E.11.5
Mass fraction of 1,4-dioxane in dye formulation	$F_{dioxane\_dye}$	kg/kg	0.0000047	–	–	–	–	See Section E.11.6
Textile production rate	V_{fabric}	kg/day	9,100	3,250	50,000	9,100	Triangular	See Section E.11.7
Mass fraction of textiles treated with dye	F_{fabric}	kg/kg	0.3	–	–	–	–	See Section E.11.8
Mass fraction of dye used per mass of textile dyed	F_{dye_fabric}	kg/kg	0.1	–	–	–	–	See Section E.11.9
Mass fraction of the dye formulation in the dyebath	$F_{dye\_dyebath}$	kg/kg	0.06	0.002	0.06	0.02	Triangular	See Section E.11.10
Container size for dye formulation	$V_{container}$	gal	35	7	264	35	Triangular	See Section E.11.11
Dye density	RHO_{form}	kg/L	1	–	–	–	–	ESD assumes a density equal to that of water
Container residual fraction for totes	$F_{container\_residue\_totes}$	kg/kg	0.002	0.0007	0.002	0.0007	Triangular	See Section E.11.12
Container residual fraction for drums	$F_{container\_residue\_drums}$	kg/kg	0.03	0.0003	0.03	0.025	Triangular	See Section E.11.13

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational/Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Container residual fraction for pails	$F_{container\_residue\_pails}$	kg/kg	0.006	0.0003	0.006	0.003	Triangular	See Section E.11.14
Fraction of dye product affixed to textile during dyeing process substrate	$F_{fixation}$	kg/kg	Multiple Triangular Distributions					See Section E.11.15

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E.11.3 Number of Sites

EPA did not find data on the number of textile dye sites that specifically use dyes containing 1,4-dioxane. As a bounding estimate, EPA used U.S. Census and BLS data for the NAICS code 313310, Textiles and Fabric Finishing Mills, to estimate a total of 783 sites within the industry (U.S. BLS, 2016).

E.11.4 Mass Fraction of Dye Containing 1,4-Dioxane

The ESD on the Use of Textile Dyes provided a single value for the mass fraction of dyes containing the chemical of interest, which is 1,4-dioxane. The ESD assumes that 100 percent of dyes contain the chemical of interest (OECD, 2017). Therefore, EPA could not develop a distribution of values for this parameter and used the single value of 1 kg dye with 1,4-dioxane/kg dye used from the ESD.

E.11.5 Operating Days

EPA modeled the number of operating days per year using a triangular distribution with a lower bound of 31 days per year, and upper bound of 295 days per year, and a mode of 157 days per year. This is based on the ESD on the Use of Textile Dyes (OECD, 2017). The ESD cites the basis of these data as past new chemical submissions that were submitted to EPA from 2006 through 2014. EPA used the 5th percentile, average, and 95th percentile in the ESD as the lower bound, mode, and upper bound of this distribution, respectively.

E.11.6 Mass Fraction of 1,4-Dioxane in Dye Formulation

The December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c) indicates that 1,4-dioxane is a byproduct in dye formulations and provided a single value of 0.0000047 kg 1,4-dioxane/kg dye. Therefore, EPA could not develop a distribution of values for this parameter and used the single value of 0.0000047 kg 1,4-dioxane/kg dye from the risk evaluation.

E.11.7 Textile Production Rate

EPA modeled textile production rate using a triangular distribution with a lower bound of 3,520 kg/site-day, and upper bound of 50,000 kg/site-day, and a mode of 9,100 kg/site-day. This is based on the ESD on the Use of Textile Dyes (OECD, 2017). The ESD cites data provided in fabric finishing new chemical submissions during 1993 and 1994. Note that the ESD uses a "typical" value as default and does not say what the typical is based on (*e.g.*, average, median). EPA used the range of textile production rates and the default typical value provided in the ESD as the lower and upper bounds and mode of the triangular distribution for this parameter, respectively.

E.11.8 Mass Fraction of Textiles Treated with Dye

The ESD on the Use of Textile Dyes provided a single value for the mass fraction of all textiles treated with dyes. The ESD states that the median share of textiles processed per day using the primary dyestuff is 30 percent (OECD, 2017). Therefore, EPA could not develop a distribution of values for this parameter and used the single value of 30 percent from the ESD.

E.11.9 Mass Fraction of Dye Used per Mass of Textile Dyed

The ESD on the Use of Textile Dyes provided a single value for the mass fraction of dye used per mass of textile dyed. The ESD states that as a "realistic worst case," liquid dye formulations are used in an amount of 10 percent (OECD, 2017). Therefore, EPA could not develop a distribution of values for this parameter and used the single value of 0.10 kg dye/kg textiles from the ESD.

E.11.10 Mass Fraction of the Dye Formulation in the Dyebath

EPA modeled mass fraction of dye formulation in the dyebath using a triangular distribution with a lower bound of 0.002 kg dye/kg bath, an upper bound of 0.06 kg dye/kg bath, and a mode of 0.02 kg dye/kg bath. This is based on the ESD on the Use of Textile Dyes. The ESD states that typical dye concentrations may range from 1.5 to 2.5 percent, lighter shades may be as low as 0.2 to 0.3 percent, and heavier shades may be between 4 to 6 percent (OECD, 2017). EPA used the overall range of dye concentrations (0.2 to 6 percent) and the mid-range of the typical concentration (2 percent) provided in the ESD as the lower and upper bounds and mode of the triangular distribution for this parameter, respectively.

E.11.11 Container Size for Dye Formulation

EPA modeled container size using a triangular distribution with a lower bound of 7 gallons, an upper bound of 264 gallons, and a mode of 35 gallons. This is based on the ESD on the Use of Textile Dyes. The ESD states that dyes can be transported in containers ranging from 25 kg through 1,000 kg, but most are shipped in 35-gallon drums (OECD, 2017). EPA converted this range from kilograms to gallons using an assumed dye density of 1 kg/L and a conversion factor of 3.785 L/gal.

E.11.12 Container Residual Fraction for Totes

EPA modeled container residual fraction for totes using a triangular distribution with a lower bound of 0.0007 kg residual/kg dye, and upper bound of 0.002 kg residual/kg dye, and a mode of 0.0007 kg residual/kg dye. The lower and upper bounds of this distribution are based on the central tendency and high-end values listed in the *EPA/OPPT Bulk Transport Residual Model* from the ChemSTEER User Guide (U.S. EPA, 2015a). EPA used the central tendency value as the mode of the triangular distribution.

E.11.13 Container Residual Fraction for Drums

EPA modeled container residual fraction for drums using a triangular distribution with a lower bound of 0.0003 kg residual/kg dye, an upper bound of 0.03 kg residual/kg dye, and a mode of 0.025 kg residual/kg dye. The lower bound is based on the minimum value for pouring and the upper bound is based on the default high-end value in the *EPA/OPPT Drum Residual Model* from the ChemSTEER User Guide (U.S. EPA, 2015a). EPA used the central tendency value for pumping as the mode of the triangular distribution.

E.11.14 Container Residual Fraction for Pails

EPA modeled container residual fraction for small containers using a triangular distribution with a lower bound of 0.0003 kg residual/kg dye, an upper bound of 0.006 kg residual/kg dye, and a mode of 0.003 kg residual/kg dye. The lower bound is based on the minimum value for pouring and the upper bound is based on the default high-end value listed in the *EPA/OPPT Small Container Residual Model* from the ChemSTEER User Guide (U.S. EPA, 2015a). EPA used the central tendency value for pouring as the mode of the triangular distribution.

E.11.15 Fraction of Dye Product Affixed to Textile during Dyeing Process Substrate

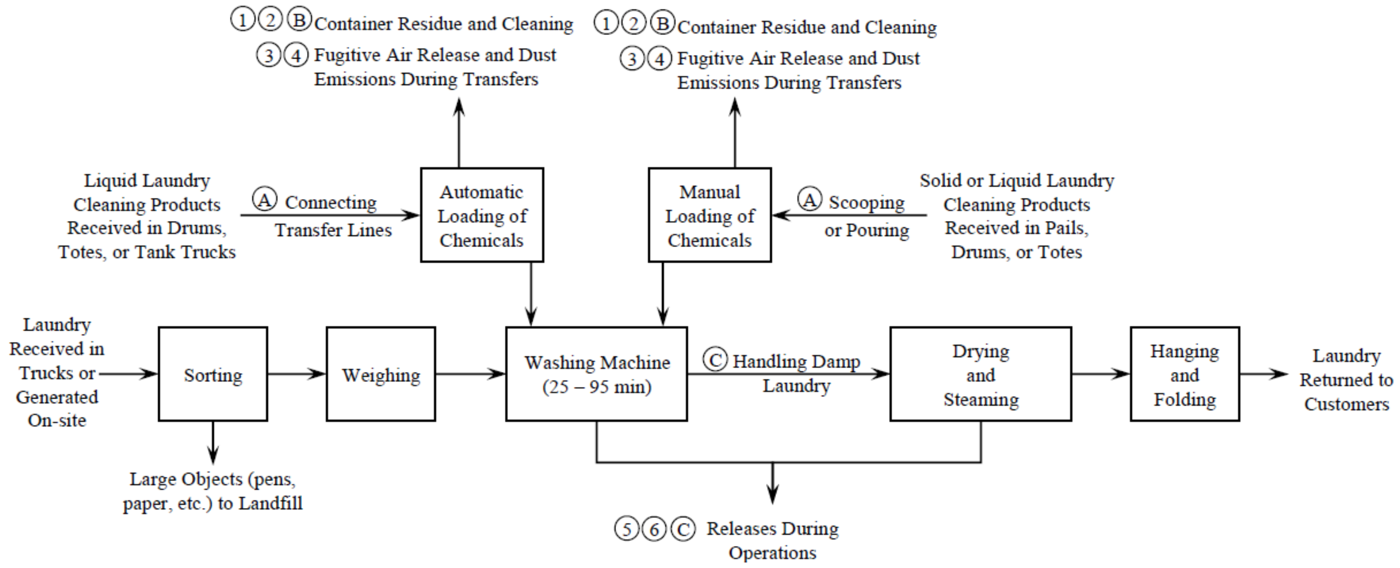
EPA modeled the fraction of dye product affixed to textiles during dyeing process substrate using multiple triangular distributions. The ESD on the Use of Textile Dyes provides a table containing ranges and averages for dye fixation percentages based on the nine different classes of dyes (OECD, 2017). EPA used the low-end of the range of dye fixation as the lower bound, the high-end of the range of dye fixation as the upper bound, and the average dye fixation as the mode for each of the nine triangular distributions. In the Monte Carlo simulation, each of the nine triangular distributions from the ESD has an equal probability of being selected and used for the parameter's output. The distribution selection probabilities and values are shown in Table_Apx E-13.

Table_Apx E-13. Triangular Distributions $F_{fixation}$

Dye Type	Dye Fixation (%) Triangular Distribution		
	Lower Bound	Upper Bound	Mode
Acid	85	98	93
Azoic	76	95	84
Basic	95	100	99
Direct	64	96	88
Disperse	80	100	96
Metal-Containing	82	98	94
Reactive	50	97	85
Sulfur	60	95	70
Vat	70	95	85
Source: (OECD, 2017)			

E.12 Laundry Detergent Modeling Approach and Parameters for Estimating Environmental Releases

This appendix presents the modeling approach and equations used to estimate environmental releases of 1,4-dioxane during the industrial and institutional use of laundry detergents. This approach utilizes the OECD ESD on the Chemicals Used in Water Based Washing Operations at Industrial and Institutional Laundries (OECD, 2011b) combined with Monte Carlo simulation (a type of stochastic simulation). This ESD categorizes laundry facilities into either industrial or institutional facilities and includes different process parameters for each. Therefore, EPA modeled the two types of laundry facilities separately. In addition, laundry detergents can be in liquid or powder physical forms. Because the difference in physical form results in different parameter distributions, EPA modeled liquid and powder detergents separately. This ESD includes a diagram of release and exposure points during textile dying, as shown in Figure_Apx E-3.



Figure_Apx E-3. Environmental Release and Occupational Exposure Points during Industrial/Institutional Laundering Operations

Based on Figure_Apx E-3, EPA identified the following release points:

- Release point 1 (RP1): Container residual losses to POTW, landfill, or incineration;
- Release point 2 (RP2): Fugitive air releases during container cleaning;
- Release point 3 (RP3): Fugitive air releases during container unloading;
- Release point 4 (RP4): Dust releases during container unloading;
    - 4a: Uncaptured dust releases;
    - 4b: Captured, uncontrolled dust releases;
    - 4c: Captured and controlled dust releases;
- Release point 5 (RP5): Fugitive air releases during washing; and
- Release point 6 (RP6): Release from washing and drying operations to fugitive air, stack air, or POTW.

Environmental releases of laundry detergent are a function of the chemical's physical properties, container size, mass fractions, and other model parameters. Although physical properties are fixed, some model parameters are expected to vary from one facility to another. An individual model input parameter could either have a discrete value or a distribution of values. EPA assigned statistical distributions based on available literature data or engineering judgment to address the variability in mass fraction of 1,4-dioxane in the detergent ($F_{dioxane\_laundry}$), container size ($V_{container}$), daily use rate of detergent ($Q_{facility\_day}$), air speed ($RATE_{air\_speed}$), duration of release (OH_{cont_unload}), operating days (OD), container residue fractions ($F_{container\_residue}$), and dust capture/control efficiency (F_{dust}).

A Monte Carlo simulation was conducted to capture variability in the model input parameters described above. The simulation was conducted using the Latin hypercube sampling method in @Risk (Palisade, Ithaca, New York). The Latin hypercube sampling method is a statistical method for generating a sample of possible values from a multi-dimensional distribution. Latin hypercube sampling is a stratified method, meaning it guarantees that its generated samples are representative of the probability density function (variability) defined in the model. EPA performed 100,000 iterations of the model to capture the range of possible input values, including values with low probability of occurrence.

From the distribution resulting from the Monte Carlo simulation, EPA selected the 95th and 50th percentile values to represent a high-end release and central tendency release level respectively. The statistics were calculated directly in @Risk. The following subsections detail the model design equations and parameters used for environmental release estimates.

E.12.1 Model Equations

Daily use rate selection based on physical form of detergent is based on the following two equations, the first being for liquid detergent and the second being for powder detergent:

Equation_Apx E-8

$$Q_{facility\_day} = Q_{facility\_day\_liquid}$$

or

$$Q_{facility\_day} = Q_{facility\_day\_powder}$$

Where:

$Q_{facility\_day}$ = Daily use rate based on physical form of detergent [kg/site-day]
$Q_{facility\_day\_liquid}$ = Daily use for liquid form detergent [kg/site-day]
$Q_{facility\_day\_powder}$ = Daily use for powder form detergent [kg/site-day]

Daily use rate of laundry detergents containing 1,4-dioxane is calculated using the equation below:

Equation_Apx E-9

$$Q_{facility\_day\_adjusted} = Q_{facility\_day} * F_{formulations\_dioxane}$$

Where:

$Q_{facility\_day\_adjusted}$ = Daily use rate of detergent containing 1,4-dioxane selected based on the physical form of the detergent [kg/site-day]
$Q_{facility\_day}$ = Daily use rate based on physical form of detergent [kg/site-day]
$F_{formulations\_dioxane}$ = Fraction of laundry detergents containing 1,4-dioxane [kg/kg]

Daily use rate of 1,4-dioxane is calculated using the equation below:

Equation_Apx E-10

$$Q_{dioxane\_day} = Q_{facility\_day\_adjusted} * F_{dioxane\_laundry}$$

Where:

$Q_{dioxane\_day}$ = Daily usage rate of 1,4-dioxane [kg/site-day]
$Q_{facility\_day\_adjusted}$ = Daily use rate of detergent with 1,4-dioxane [kg/site-day]
$F_{dioxane\_laundry}$ = Mass fraction of 1,4-dioxane in laundry detergent [kg/kg]

Number of containers used per year is calculated using the equation below:

Equation_Apx E-11

$$N_{cont\_site\_yr} = \frac{Q_{facility\_day\_adjusted} * OD}{V_{container} * 3.79\frac{L}{gal} * RHO_{detergent}}$$

Where:

$N_{cont\_site\_yr}$ = Number of containers used per site per year [containers/site-year]
$Q_{facility\_day\_adjusted}$ = Daily use rate of detergent with 1,4-dioxane [kg/site-day]
OD = Operating days [days/year]
$V_{container}$ = Container volume [gal/container]
$RHO_{detergent}$ = Detergent density [kg/L]

Vapor pressure correction factor for release points 2 and 3 is calculated using the equation below:

Equation_Apx E-12

$$X_{clean\_unload} = \frac{F_{dioxane\_laundry}/_{MW}}{\frac{F_{dioxane\_laundry}}{MW} + \frac{1 - F_{dioxane\_laundry}}{18}}$$

Where:

X_{clean_unload} = Vapor pressure correction factor for release points 2 and 3 [mol 1,4-dioxane/mol water]
$F_{dioxane\_laundry}$ = Mass fraction of 1,4-dioxane in detergent [kg/kg]
MW = 1,4-dioxane molecular weight [g/mol]

Fraction of 1,4-dioxane in wash water is calculated using the equation below:

Equation_Apx E-13

$$F_{dioxane\_wash} = F_{dilution} * F_{dioxane\_laundry}$$

Where:

$F_{dioxane\_wash}$ = Fraction of 1,4-dioxane in wash water [kg 1,4-dioxane/kg water]
$F_{dilution}$ = Dilution factor for detergent in the wash [unitless]
$F_{dioxane\_laundry}$ = Mass fraction of 1,4-dioxane in detergent [kg/kg]

Vapor pressure correction factor for release point 5 is calculated using the equation below:

Equation_Apx E-14

$$X_{washing} = \frac{F_{dioxane\_wash}/_{MW}}{\frac{F_{dioxane\_wash}}{MW} + \frac{1 - F_{dioxane\_wash}}{18}}$$

Where:

$X_{washing}$ = Vapor pressure correction factor for release point 5 [mol 1,4-dioxane/mol water]
$F_{dioxane\_wash}$ = Fraction of 1,4-dioxane in wash water [kg 1,4-dioxane/kg water]
MW = 1,4-dioxane molecular weight [g/mol]

Container residual fraction is calculated using the following equations. To make the simulation more realistic, EPA assessed container size based on the detergent use rate. This avoids situations where a small container size is associated with a large use rate, such that an unrealistic number of containers are used each year, and vice-versa:

Equation_Apx E-15

If $Q_{facility\_day} > 600 \frac{kg}{site-day}$:

$$F_{container\_residue} = F_{container\_residue\_tote}$$

If $Q_{facility\_day} = 200 - 600 \frac{kg}{site-day}$:

$$F_{container\_residue} = F_{container\_residue\_drum}$$

If $Q_{facility\_day} < 200 \frac{kg}{site-day}$:

$$F_{container\_residue} = F_{container\_residue\_pail}$$

If physical form of detergent is powder:

$$F_{container\_residue} = F_{container\_residue\_powder}$$

Where:

$Q_{facility\_day}$ = Daily use rate based on physical form of detergent [kg/site-day]
$F_{container\_residue}$ = Container residual fraction [kg/kg]
$F_{container\_residue\_tote}$ = Container residual fraction for totes [kg/kg]
$F_{container\_residue\_drum}$ = Container residual fraction for drums [kg/kg]
$F_{container\_residue\_pail}$ = Container residual fraction for pails [kg/kg]
$F_{container\_residue\_powder}$ = Container residual fraction for solid detergents [kg/kg]

Release Point 1 site release per day is calculated using the equation below:

Equation_Apx E-16

$$Release\_perDay_{RP1} = Q_{dioxane\_day} * F_{container\_residue}$$

Where:

$Release\_perDay_{RP1}$ = Daily 1,4-dioxane release at release point 1 [kg/site-day]
$Q_{dioxane\_day}$ = Daily usage rate of 1,4-dioxane [kg/site-day]
$F_{container\_residue}$ = Container residual fraction [kg/kg]

Release Point 2 fugitive emissions from container cleaning for pails and drums per day is calculated using the *Penetration Model* equation below (air speed ≤ 100 ft/min):

Equation_Apx E-17

$$Release\_perDay_{RP2} =$$

$$3600\frac{s}{hr} * 0.001\frac{kg}{g} * \frac{(8.24 \times 10^{-8}) * (MW^{0.835}) * X_{clean_{unload}} * VP * \sqrt{Rate_{air_{speed}}} * (0.25\pi D^2_{container\_opening})\sqrt[4]{\frac{1}{29} + \frac{1}{MW}}}{T^{0.05} * \sqrt{D_{container\_opening}}\sqrt{P}}$$

Where:

$Release\_perDay_{RP2}$ = Release point 2 fugitive emissions from pail/drum cleaning per day [kg/site-day]

MW = 1,4-dioxane molecular weight [g/mol]

X_{clean_unload} = Vapor pressure correction factor release point 2 [mol 1,4-dioxane/mol water]

VP = 1,4-dioxane vapor pressure [torr]

T = Ambient temperature [K]

$Rate_{air\_speed}$ = Air speed [cm/s]

$D_{container\_opening}$ = Diameter of container opening [cm]

P = Atmospheric pressure [atm]

Release Point 2 fugitive emissions from container cleaning per day for totes is calculated using the *Mass Transfer Coefficient Model* equation below (air speed >100 ft/min):

Equation_Apx E-18

$$Release\_perDay_{RP2} =$$

$$3600\frac{s}{hr} * 0.001\frac{kg}{g} * \frac{(1.93 \times 10^{-7}) * (MW^{0.78}) * X_{clean_{unload}} * VP * Rate_{air\_speed}^{0.78} * (0.25\pi D_{container\_opening}^{2})\sqrt[3]{\frac{1}{29} + \frac{1}{MW}}}{T^{0.4} D_{container\_opening}^{0.11} (\sqrt{T} - 5.87)^{2/3}}$$

Where:

$Release\_perDay_{RP2}$ = Release point 2 fugitive emissions from tote cleaning per day [kg/site-day]

X_{clean_unload} = Vapor pressure correction factor release point 5 [mol 1,4-dioxane/mol water]

MW = 1,4-dioxane molecular weight [g/mol]

VP = Vapor Pressure [torr]

T = Ambient Temperature [K]

$Rate_{air\_speed}$ = Air speed [cm/s]

$D_{container\_opening}$ = Diameter of container opening [cm]

P = Atmospheric pressure [atm]

Release Point 3 fugitive emissions from unloading of pails and drums during the day is calculated using the *Penetration Model* equation below (air speed ≤ 100 ft/min):

Equation_Apx E-19

$$Release\_perDay_{RP3} =$$

$$OH_{cont\_unload} * 3600\frac{s}{hr} * 0.001\frac{kg}{g} * \frac{(8.24 \times 10^{-8}) * (MW^{0.835}) * X_{clean_{unload}} * VP * \sqrt{Rate_{air_{speed}}} * (0.25\pi D_{container\_opening}^{2})\sqrt[4]{\frac{1}{29} + \frac{1}{MW}}}{T^{0.05} * \sqrt{D_{container\_opening}}\sqrt{P}}$$

Where:

$Release\_perDay_{RP3}$ = Point 3 fugitive emissions from unloading during the day [kg/site-day]

X_{clean_unload} = Vapor pressure correction factor release point 5 [mol 1,4-dioxane/mol water]

MW = 1,4-dioxane molecular weight [g/mol]

VP = Vapor pressure [torr]

T = Ambient temperature [K]

$Rate_{air\_speed}$ = Air speed from EPA model [cm/s]
$D_{container\_opening}$ = Diameter of the opening for containers [cm]
P = Atmospheric pressure [atm]
OH_{cont_unload} = Duration of container unloading [hrs/day]

Release Point 3 fugitive emissions from unloading totes during the day is calculated using the *Mass Transfer Coefficient Model* equation below (air speed >100 ft/min):

Equation_Apx E-20

$$Release\_perDay_{RP3} =$$

$$OH_{cont\_unload} 3600 \frac{s}{hr} * 0.001 \frac{kg}{g} * \frac{(1.93 \times 10^{-7}) * (MW^{0.78}) * X_{clean_{unload}} * VP * Rate_{air\_speed}^{0.78} * (0.25\pi D_{container\_opening}^{2}) \sqrt[3]{\frac{1}{29} + \frac{1}{MW}}}{T^{0.4} D_{container\_opening}^{0.11} (\sqrt{T} - 5.87)^{2/3}}$$

Where:

$Release\_perDay_{RP3}$ = Point 3 fugitive emissions from unloading during the day [kg/site-day]
X_{clean_unload} = Vapor pressure correction factor release point 5 [mol 1,4-dioxane/mol water]
MW = 1,4-dioxane molecular weight [g/mol]
VP = Vapor pressure [torr]
T = Ambient temperature [K]
$Rate_{air\_speed}$ = Air speed from EPA model [cm/s]
$D_{container\_opening}$ = Diameter of the opening for containers [cm]
P = Atmospheric pressure [atm]
OH_{cont_unload} = Duration of container unloading [hours/day]

Release Point 4a dust not captured to fugitive air, water, incineration, or landfill is calculated using the following equation:

Equation_Apx E-21

$$Release\_perDay_{RP4a} = Q_{dioxane\_day} * F_{dust_{generation}} * (1 - F_{dust_{capture}})$$

Where:

$Release\_perDay_{RP4a}$= Dust not captured to fugitive air, water, incineration, or landfill [kg/site-day]
$F_{dust_{generation}}$ = Fraction of chemical lost during transfer of solid powders [kg/kg]
$F_{dust_{capture}}$ = Capture efficiency for dust capture methods [kg/kg]

Release Point 4b dust captured but not controlled to stack air is calculated using the following equation:

Equation_Apx E-22

$$Release\_perDay_{RP4b} = Q_{dioxane\_day} * F_{dust_{generation}} * F_{dust_{capture}} * (1 - F_{dust_{control}})$$

Where:

$Release\_perDay_{RP4b}$= Dust captured but not controlled to stack air [kg/site-day]
$F_{dust_{generation}}$ = Fraction of chemical lost during transfer of solid powders [kg/kg]
$F_{dust_{capture}}$ = Capture efficiency for dust capture methods [kg/kg]

$F_{dust_{control}}$ = Control efficiency for dust control methods [kg/kg]

Release Point 4c dust captured and controlled to incineration of landfill is calculated using the following equation:

Equation_Apx E-23

$$Release\_perDay_{RP4b} = Q_{dioxane\_day} * F_{dust_{generation}} * F_{dust_{capture}} * F_{dust_{control}}$$

Where:

$Release\_perDay_{RP4b}$ = Dust captured but not controlled to stack air [kg/site-day]
$F_{dust_{generation}}$ = Fraction of chemical lost during transfer of solid powders [kg/kg]
$F_{dust_{capture}}$ = Capture efficiency for dust capture methods [kg/kg]
$F_{dust_{control}}$ = Control efficiency for dust control methods [kg/kg]

Release Point 5 fugitive emissions during washing per day is calculated when air speed ≤ 100 ft/min using the *Penetration Model* in the equation shown below:

Equation_Apx E-24

$$Release\_perDay_{RP5} =$$

$$OH * 3600\frac{s}{hr} * 0.001\frac{kg}{g} * \frac{(8.24 \times 10^{-8}) * (MW^{0.835}) * X_{clean_{unload}} * VP * \sqrt{Rate_{air_{speed}}} * (0.25\pi D^2_{container\_opening})\sqrt[4]{\frac{1}{29} + \frac{1}{MW}}}{T^{0.05} * \sqrt{D_{container\_opening}}\sqrt{P}}$$

Where:

$Release\_perDay_{RP5}$ = Point 5 fugitive emissions from washing [kg/site-day]
X_{clean_unload} = Vapor pressure correction factor release point 5 [mol 1,4-dioxane/mol water]
MW = 1,4-dioxane molecular weight [g/mol]
VP = Vapor pressure [torr]
T = Ambient temperature [K]
$Rate_{air\_speed}$ = Air speed [cm/s]
$D_{container\_opening}$ = Diameter of the opening for containers [cm]
P = Atmospheric pressure [atm]
OH = Operating hours [hours/day]

Release Point 5 fugitive emissions during washing per day is calculated when air speed >100 ft/min using the *Mass Transfer Coefficient Model* shown below:

Equation_Apx E-25

$$Release\_perDay_{RP5} =$$

$$OH_{cont\_unload}3600\frac{s}{hr} * 0.001\frac{kg}{g} * \frac{(1.93 \times 10^{-7}) * (MW^{0.78}) * X_{clean_{unload}} * VP * Rate^{0.78}_{air\_speed} * (0.25\pi D^2_{container\_opening})\sqrt[3]{\frac{1}{29} + \frac{1}{MW}}}{T^{0.4} D^{0.11}_{container\_opening}(\sqrt{T} - 5.87)^{2/3}}$$

Where:

$Release\_perDay_{RP5}$ = Point 5 fugitive emissions from washing [kg/site-day]
X_{clean_unload} = Vapor pressure correction factor release point 5 [mol 1,4-dioxane/mol water]
MW = 1,4-dioxane molecular weight [g/mol]

VP = Vapor pressure [torr]
T = Ambient temperature [K]
$Rate_{air\_speed}$ = Air speed [cm/s]
$D_{container\_opening}$ = Diameter of the opening for containers [cm]
P = Atmospheric pressure [atm]
OH = Operating hours [hours/day]

Release Point 6 site release per day (washing and drying) is calculated using the equations and criteria below:

Equation_Apx E-26

If $\sum_{i=1}^{5} Release\_perDay_{RPi}$ < $Q_{dioxane\_day}$:

$$Release\_perDay_{RP6} = Q_{dioxane\_day} - \sum_{i=1}^{5} Release\_perDay_{RPi}$$

If $\sum_{i=1}^{5} Release\_perDay_{RPi}$ > $Q_{dioxane\_day}$:

Liquid detergent:

$$Release\_perDay_{RP6} = Q_{dioxane\_day} - Release\_perDay_{RP1}$$

Powder detergent:

$$Release\_perDay_{RP6} = Q_{dioxane\_day} - Release_{perDay_{RP1}} - Release\_perDay_{RP4}$$

Where:

$Release\_perDay_{RP1}$ = Point 1 container residual releases [kg/site-day]
$Release\_perDay_{RP2}$ = Point 2 fugitive emissions from container cleaning [kg/site-day]
$Release\_perDay_{RP3}$ = Point 3 fugitive emissions from unloading [kg/site-day]
$Release\_perDay_{RP4}$ = Point 4 fugitive dust emissions [kg/site-day]
$Release\_perDay_{RP5}$ = Point 5 fugitive emissions from washing [kg/site-day]
$Release\_perDay_{RP6}$ = Point 6 daily site releases (washing and drying) [kg/site-day]
$Q_{dioxane\_day}$ = Daily usage rate of 1,4-dioxane [kg/site-day]
$\sum_{i=1}^{5} Release\_perDay_{RPi}$ = The sum of release points 1-5 emissions [kg/site-day]

E.12.2 Model Input Parameters

Table_Apx E-14 summarizes the model parameters and their values for the Monte Carlo simulation. Additional explanations of EPA's selection of the distributions for each parameter are provided after this table. High-end and central tendency releases are estimated by selecting the 50th and 95th percentile values from the output distribution.

Table_Apx E-14. Summary of Parameter Values and Distributions Used in the Industrial and Institutional Laundry Release Model

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational/ Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Operating Days	OD	days/year	260	Industrial: 19 Institutional: 249	366	260	Triangular	See Section E.12.3
Mass Fraction of 1,4-dioxane in Laundry Detergent	$F_{dioxane\_laundry}$	kg/kg	0.000014	5.00E−08	0.000014	—	Uniform	See Section E.12.4.
Daily Use Rate of Liquid Laundry Detergents	$Q_{facility\_day\_liquid}$	kg/day	Industrial: 35.7 Institutional: 16	Industrial: 0.116 Institutional: 0.124	Industrial: 814 Institutional: 513	—	Discrete	See Section E.12.5
Daily Use Rate of Powder Laundry Detergents	$Q_{facility\_day\_powder}$	kg/day	Industrial: 110.45 Institutional: 8.63	Industrial: 1.33 Institutional: 3.71	Industrial: 1,917.44 Institutional: 15	—	Discrete	See Section E.12.5
Container Size	$V_{container}$	gal	55	5	550	55	Triangular	See Section E.12.6
Air Speed	$RATE_{air\_speed}$	cm/s	10	1.3	202.2	—	Lognormal	See Section E.12.7
Container residual fraction for Totes	$F_{container\_residue\_totes}$	kg/kg	0.002	0.0002	0.002	0.0007	Triangular	See Section E.12.8
Container residual fraction for Drums	$F_{container\_residue\_drums}$	kg/kg	0.03	0.017	0.03	0.025	Triangular	See Section E.12.9
Container residual fraction for Pails	$F_{container\_residue\_pails}$	kg/kg	0.006	0.0003	0.006	0.003	Triangular	See Section E.12.10
Container residual fraction for Powders	$F_{container\_residue\_powders}$	kg/kg	0.01	—	—	—	—	See Section E.12.11
Fraction of Laundry Detergents Containing 1,4-dioxane	$F_{formulations\_dioxane}$	unitless	0.5	0.111	1	—	Industrial: Discrete Institutional:	See Section E.12.12

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational/ Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
							Uniform	
Duration of Release for Container Unloading	OH_{cont_unload}	h/day	Industrial: 0.0043 Institutional: 0.0114	Industrial: 0.0043 Institutional: 0.0114	Industrial: 12 Institutional: 8	—	Uniform	See Section E.12.13
Fraction of chemical lost during transfer of solid powders	$F_{dust\_generation}$	kg/kg	0.0050	0.0010	0.03	0.005	Triangular	See Section E.12.14
Control efficiency for dust control methods	$F_{dust\_control}$	kg/kg	0.7900	0.0000	1	0.79	Triangular	See Section E.12.15
Capture efficiency for dust capture methods	$F_{dust\_capture}$	kg/kg	0.9633	0.9310	1	0.9633	Triangular	See Section E.12.16
Number of Sites	Ns	sites	Industrial: 2,453 Institutional: 95,533	—	—	—	—	See Section E.12.17
Vapor Pressure of 1,4-dioxane	VP	Torr	40	—	—	—	—	Physical property
Molecular Weight of 1,4-dioxane	MW	g/mol	88.1	—	—	—	—	Physical property
Diameter of Container Opening	$D_{container\_opening}$	cm	5.08	—	—	—	—	See Section E.12.18
Diameter of Wash Opening	$D_{wash\_opening}$	cm	73	—	—	—	—	See Section E.12.19
Ambient Temperature	T	K	298	—	—	—	—	Process parameter
Ambient Pressure	P	atm	1	—	—	—	—	Process parameter

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational/ Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Dilution Factor	$F_{dilution}$	unitless	0.016	—	—	—	—	See Section E.12.20
Density of Laundry Detergent	RHO_{form}	kg/L	1	—	—	—	—	ESD assumes a density equal to that of water
Container Fill Rate	$RATE_{fill}$	containers / hour	20	—	—	—	—	See Section E.12.21

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E.12.3 Operating Days

EPA modeled the operating days per year using a triangular distribution with a lower bound of 20 days per year, an upper bound of 365 days per year, and a mode of 260 days per year for industrial laundries. EPA used a triangular distribution with a lower bound of 250 days per, an upper bound of 365 days per year, and a mode of 260 days per year for institutional laundries. This is based on the ESD on the Chemicals Used in Water Based Washing Operations at Industrial and Institutional Laundries (OECD, 2011b). The ESD provides the range and average of operating days for six separate years, which EPA took the minimum, maximum, and average of the 6 years to form the distributions.

E.12.4 Mass Fraction of 1,4-Dioxane in Laundry Detergent

EPA modeled the mass fraction of 1,4-dioxane in laundry detergent using a uniform distribution with a lower bound of 5.00×10^{-8} kg of 1,4-dioxane/kg detergent and an upper bound of 1.4×10^{-5} kg of 1,4-dioxane/kg detergent for both industrial and institutional laundries. This is based on the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). The risk evaluation indicates that 1,4-dioxane is a byproduct in the laundry detergents and provides a concentration range of 0.05 to 14 ppm of 1,4-dioxane in the detergent.

E.12.5 Daily Use Rate of Detergent

EPA modeled the daily use rate of detergent using a discrete distribution. For industrial laundries, the distribution ranged from 0.116 kg/day to 814 kg/day for liquid detergents and 1.33 kg/day to 1917.44 kg/day for powder detergents. For institutional laundries, the distribution ranged from 0.124 kg/day to 513 kg/day for liquid detergents and 3.71 kg/day to 15 kg/day for powder detergents. This discrete data was pulled from survey data from laundries sites used in the ESD on the Chemicals Used in Water Based Washing Operations at Industrial and Institutional Laundries (OECD, 2011b). Equal probability was given to each discrete survey value.

E.12.6 Container Size

EPA modeled container size using a triangular distribution with a lower bound of 5 gallons, an upper bound of 550 gallons, and a mode of 55 gallons for industrial laundries. Because EPA expects industrial laundries to have variation in the sizes of containers, EPA used values of 5, 55, and 550 gallons for the triangular distribution based on the default values from the *EPA/OPPT Small Container Residual Model*, *Drum Residual Model*, and *Bulk Transport Residual Model*, respectively.

EPA used a single value of 5 gallons for institutional laundries. This is based on the ESD on the default value for institutional laundries from the Chemicals Used in Water Based Washing Operations at Industrial and Institutional Laundries ESD (OECD, 2011b).

E.12.7 Indoor Air Speed

Baldwin (1998) measured indoor air speeds across a variety of occupational settings in the United Kingdom. Fifty-five work areas were surveyed across a variety of workplaces. EPA analyzed the air speed data from Baldwin (1998) and categorized the air speed surveys into settings representative of industrial facilities and representative of commercial facilities. EPA fit separate distributions for these industrial and commercial settings and used the industrial distribution for laundry facilities.

EPA fit a lognormal distribution for both data sets as consistent with the authors observations that the air speed measurements within a surveyed location were lognormally distributed and the population of the mean air speeds among all surveys were lognormally distributed. Since lognormal distributions are bound by zero and positive infinity, EPA truncated the distribution at the largest observed value among all of the survey mean air speeds from Baldwin (1998). EPA fit the air speed surveys representative of industrial facilities to a lognormal distribution with the following parameter values: mean of 22.414 cm/s and standard deviation of 19.958 cm/s. In the model, the lognormal distribution is truncated at a maximum allowed value of 202.2 cm/s (largest surveyed mean air speed observed in Baldwin (1998)) to prevent the model from sampling values that approach infinity or are otherwise unrealistically large.

Baldwin (1998) only presented the mean air speed of each survey. The authors did not present the individual measurements within each survey. Therefore, these distributions represent a distribution of mean air speeds and not a distribution of spatially variable air speeds within a single workplace setting.

E.12.8 Container Residual Fraction for Totes

EPA modeled container residual fraction for totes using a triangular distribution with a lower bound of 0.0007 kg residual/kg detergent, and upper bound of 0.002 kg residual/kg detergent, and a mode of 0.0007 kg residual/kg detergent. The lower and upper bounds of this distribution are based on the central tendency and high-end values listed in the *EPA/OPPT Bulk Transport Residual Model* from the ChemSTEER User Guide (U.S. EPA, 2015a). EPA used the central tendency value as the mode of the triangular distribution.

E.12.9 Container Residual Fraction for Drums

EPA modeled container residual fraction for drums using a triangular distribution with a lower bound of 0.0003 kg residual/kg detergent, an upper bound of 0.03 kg residual/kg detergent, and a mode of 0.025 kg residual/kg detergent. The lower bound is based on the minimum value for pouring and the upper bound is based on the default high-end value in the *EPA/OPPT Drum Residual Model* from the ChemSTEER User Guide (U.S. EPA, 2015a). EPA used the central tendency value for pumping as the mode of the triangular distribution.

E.12.10 Container Residual Fraction for Pails

EPA modeled container residual fraction for small containers using a triangular distribution with a lower bound of 0.0003 kg residual/kg detergent, an upper bound of 0.006 kg residual/kg detergent, and a mode of 0.003 kg residual/kg detergent. The lower bound is based on the minimum value for pouring and the upper bound is based on the default high-end value listed in the *EPA/OPPT Small Container Residual Model* from the ChemSTEER User Guide (U.S. EPA, 2015a). EPA used the central tendency value for pouring as the mode of the triangular distribution.

E.12.11 Container Residual Fraction for Powders

The *EPA/OPPT Solid Residuals in Transport Containers Model* provides a loss fraction 0.01 kg of solid chemicals remaining in a container per kg transported. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 0.01 kg/kg from the model (U.S. EPA, 2015a).

E.12.12 Fraction of Laundry Detergents Containing 1,4-Dioxane

EPA modeled the fraction of laundry detergents containing 1,4-dioxane using a discrete distribution. For industrial and institutional laundries, the distribution ranged from 0.111 to 1 kg detergents containing 1,4-dioxane/kg all detergents. This discrete data was pulled from survey data from laundries sites used in the ESD on the Chemicals Used in Water Based Washing Operations at Industrial and Institutional Laundries (OECD, 2011b). Equal probability was given to each discrete survey value.

E.12.13 Duration of Release for Container Unloading

EPA modeled the duration of release for container unloading using a uniform distribution. For industrial and institutional laundries, EPA assumed the distribution had a maximum of 12 hours/day. The lower bound was based on the length of time to unload detergent containers each day, calculated using the number of containers used per day and the container fill rate (see Section E.12.21). This means that each iteration of the simulation would calculate a new lower bound based on the parameters for that iteration.

E.12.14 Fraction of Chemical Lost during Transfer of Solid Powders

EPA modeled the fraction of chemical lost during transfer of solid powders using a triangular distribution with a lower bound of 0.001 kg dust lost/kg transferred, an upper bound of 0.03 kg dust lost/kg transferred, and a mode of 0.005 kg dust lost/kg transferred for both industrial and institutional laundries. These values were taken from the *EPA/OPPT Dust Emissions from Transferring Solids Model* from the ChemSTEER User Guide (U.S. EPA, 2015a).

E.12.15 Control Efficiency for Dust Control Methods

EPA modeled the control efficiency for dust control methods using a triangular distribution with a lower bound of 0 kg controlled/kg transferred, an upper bound of 1 kg controlled/kg transferred, and a mode of 0.79 kg controlled/kg transferred for both industrial and institutional laundries. These values were taken from the *EPA/OPPT Dust Emissions from Transferring Solids Model* from the ChemSTEER User Guide (U.S. EPA, 2015a).

E.12.16 Capture Efficiency for Dust Capture Methods

EPA modeled the capture efficiency for dust capture methods using a triangular distribution with a lower bound of 0.9310 kg captured/kg transferred, an upper bound of 1 kg captured/kg transferred, and a mode of 0.9633 kg captured/kg transferred for both industrial and institutional laundries. These values were taken from the *EPA/OPPT Dust Emissions from Transferring Solids Model* from the ChemSTEER User Guide (U.S. EPA, 2015a).

E.12.17 Number of Sites

EPA did not find data on the number of laundry sites that specifically use detergents containing 1,4-dioxane. As a bounding estimate for the number of industrial laundries, EPA used U.S. Census and BLS data for the NAICS code 812330, Linen and Uniform Supply, to estimate a total of 2,453 industrial laundry sites within the industry (U.S. BLS, 2016). As a bounding estimate for the number of institutional sites, EPA used industry information as described in the ESD to estimate a total of 95,533 institutional laundries (OECD, 2011b).

E.12.18 Diameter of Container Opening

The ChemSTEER User Guide (U.S. EPA, 2015a) provides a typical diameter of container openings as 5.08 cm. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 5.08 cm from the ChemSTEER User Guide.

E.12.19 Diameter of Wash Opening

The ESD on the Chemicals Used in Water Based Washing Operations at Industrial and Institutional Laundries provided a single value for the diameter of washer openings to estimate air releases during operation (OECD, 2011b). The ESD states that the wash opening is 73 cm (OECD, 2017). Therefore, EPA could not develop a distribution of values for this parameter and used the single value of 73 cm from the ESD.

E.12.20 Dilution Factor

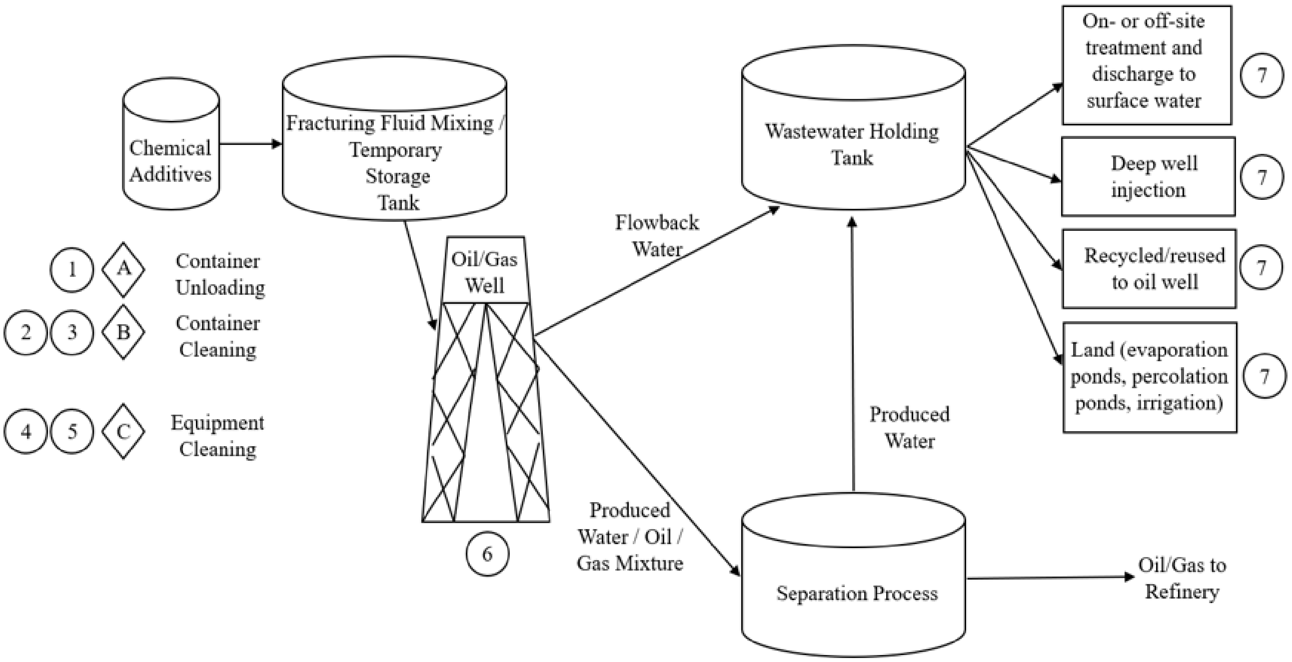
The December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c) provided a single value for the dilution factor of 1,4-dioxane in laundry detergents. The risk evaluation states that a dilution factor of 0.016 was estimated assuming a high-end mass of product used (60g) in one gallon of water (U.S. EPA, 2020c). Therefore, EPA could not develop a distribution of values for this parameter and used the single value of 0.016 from the 2020 RE.

E.12.21 Container Fill Rate

The ChemSTEER User Guide (U.S. EPA, 2015a) provides a typical fill rate of 20 containers per hour for containers with 20 to 100 gallons of liquid. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 20 containers/hour from the ChemSTEER User Guide.

E.13 Hydraulic Fracturing Modeling Approach and Parameters for Estimating Environmental Releases

This appendix presents the modeling approach and equations used to estimate environmental releases of 1,4-dioxane during hydraulic fracturing. This approach utilizes the Draft ESD on Chemicals Used in Hydraulic Fracturing (U.S. EPA, 2022d) combined with Monte Carlo simulation (a type of stochastic simulation). This ESD includes a diagram of release and exposure points during textile dying, as shown in Figure_Apx E-4.



Figure_Apx E-4. Environmental Release and Occupational Exposure Points during Hydraulic Fracturing

Based on Figure_Apx E-4, EPA identified the following release points:

- Release point 1 (RP1): Fugitive air releases during container unloading;
- Release point 2 (RP2): Container residue losses to surface water, incineration, or landfill;
- Release point 3 (RP3): Fugitive air releases during container cleaning;

- Release point 4 (RP4): Equipment and storage tank cleaning losses to surface water, incineration, or landfill;
- Release point 5 (RP5): Fugitive air releases during equipment and storage tank cleaning;
- Release point 6 (RP6): Release of hydraulic fracturing fluid that remains underground to deep well injection; and
- Release point 7 (RP7): Hydraulic fracturing fluid flowback and produced wastewater to recycle/reuse, deep well injection, surface water, or land.

Environmental releases of hydraulic fracturing are a function of the chemical's physical properties, container size, mass fractions, and other model parameters. Although physical properties are fixed for a chemical, some model parameters are expected to vary from one facility to another. An individual model input parameter could either have a discrete value or a distribution of values. EPA assigned statistical distributions based on available literature data or engineering judgment to address the variability in operating days (OD), mass fraction of 1,4-dioxane in fracturing fluid ($F_{dioxane\_fracturing\_fluid}$), mass fraction of 1,4-dioxane in additive ($F_{dioxane\_additive}$), container container size (V_{cont}), annual use rate of fracturing fluids (Q_{site_yr}), saturation factor ($F_{saturation}$), container cleaning losses ($F_{cont\_cleaning}$), and fraction of injected fracturing fluid that returns to the surface ($F_{recovered}$).

A Monte Carlo simulation (a type of stochastic simulation) was conducted to capture variability in the model input parameters. The simulation was conducted using the Latin hypercube sampling method in @Risk (Palisade, Ithaca, NY). The Latin hypercube sampling method is a statistical method for generating a sample of possible values from a multi-dimensional distribution. Latin hypercube sampling is a stratified method, meaning it guarantees that its generated samples are representative of the probability density function (variability) defined in the model. EPA performed 100,000 iterations of the model to capture the range of possible input values, including values with low probability of occurrence.

From the distribution resulting from the Monte Carlo simulation, EPA selected the 95th and 50th percentile values to represent a high-end release and central tendency release level respectively. The statistics were calculated directly in @Risk. The following subsections detail the model design equations and parameters used for environmental release estimates.

E.13.1 Model Equations

Daily use rate of fracturing fluids containing 1,4-dioxane is calculated using the equation below:

Equation_Apx E-27

$$Q_{site\_day} = Q_{site\_yr} * (3.79\frac{L}{gal}) * \frac{RHO_{fracturing\_fluid}}{OD}$$

Where:

Q_{site_day} = Daily use rate of fracturing fluids with 1,4-dioxane [kg/site-day]
Q_{site_yr} = Annual use rate of fracturing fluids with 1,4-dioxane [gal/site-year]
OD = Operating days [days/year]
$RHO_{fracturing\_fluid}$ = Density of fracturing fluid [kg/L]

Annual use rate of 1,4-dioxane is calculated using the equation below:

Equation_Apx E-28

$$Q_{dioxane\_site\_yr} = Q_{site\_yr} * \left(3.79\frac{L}{gal}\right) * RHO_{fracturing\_fluid} * F_{dioxane\_fracturing\_fluid}$$

Where:

$Q_{dioxane\_site\_yr}$ = Annual use rate of 1,4-dioxane [kg/site-year]
Q_{site_yr} = Annual use rate of fracturing fluids with 1,4-dioxane [gal/site-year]
$RHO_{fracturing\_fluid}$ = Density of fracturing fluid [kg/L]
$F_{dioxane\_fracturing\_fluid}$ = Mass fraction of 1,4-dioxane in hydraulic fracturing fluid [kg/kg]

Number of containers used per year is calculated using the equation below:

Equation_Apx E-29

$$N_{cont\_unlaod\_yr} = \frac{Q_{dioxane\_site\_yr}}{F_{dioxane\_additive} * V_{cont} * RHO_{fracturing\_fluid} * 3.79 \frac{L}{gal}}$$

Where:

$N_{cont\_unlaod\_yr}$ = Number of containers used yearly [containers/site-year]
$Q_{dioxane\_site\_yr}$ = Annual use rate of 1,4-dioxane [kg/site-year]
$F_{dioxane\_additive}$ = Mass fraction of 1,4-dioxane in hydraulic fracturing additive [kg/kg]
V_{cont} = Container size for fracturing fluids [gal]
$RHO_{fracturing\_fluid}$ = Density of fracturing fluid [kg/L]

The vapor pressure correction factor for release point 1 (unloading) and release point 3 (container cleaning) is calculated using the equation below:

Equation_Apx E-30

$$X_{clean\_unload} = \frac{F_{dioxane\_additive}/_{MW}}{\frac{F_{dioxane\_additive}}{MW} + \frac{1 - F_{dioxane\_additive}}{18}}$$

Where:

X_{clean_unload} = Vapor pressure correction factor for RP 1 and 3 [mol 1,4-dioxane/mol H_2O]
$F_{dioxane\_additive}$ = Mass fraction of 1,4-dioxane in hydraulic fracturing additive [kg/kg]
MW = 1,4-dioxane molecular weight [g/mol]

The vapor pressure correction factor for release point 5 (storage tank cleaning) is calculated using the equation below:

Equation_Apx E-31

$$X_{tank\_clean} = \frac{F_{dioxane\_fracturing\_fluid}/_{MW}}{\frac{F_{dioxane\_fracturing\_fluid}}{MW} + \frac{1 - F_{dioxane\_fracturing\_fluid}}{18}}$$

Where:

X_{tank_clean} = Vapor pressure correction factor for RP 5 [mol 1,4-dioxane/mol H_2O]
MW = 1,4-dioxane molecular weight [g/mol]

$F_{dioxane\_fracturing\_fluid}$ = Mass fraction of 1,4-dioxane in hydraulic fracturing fluid [kg/kg]

Container residual fraction is calculated using the following equations. To make the simulation more realistic, EPA assessed container size based on the fracturing fluid use rate. This avoids situations where a small container size is associated with a large use rate, such that an unrealistic number of containers are used each year, and vice-versa:

Equation_Apx E-32

If $Q_{site\text{-}day}$ > 1500 kg/site-day:

$$F_{container\_residue} = F_{cont\_cleaning\_tote}$$

If $Q_{site\text{-}day} \leq$ 1500 kg/site-day:

$$F_{container\_residue} = F_{cont\_cleaning\_drum}$$

Where:

$F_{container\_residue}$ = Container residual fraction [kg/kg]
$F_{container\_residue\_tote}$ = Container residual fraction for totes [kg/kg]
$F_{container\_residue\_drum}$ = Container residual fraction for drums [kg/kg]
Q_{site_day} = Daily use rate of fracturing fluids with 1,4-dioxane [kg/site-day]

Release Point 1 daily releases per site (unloading volatile chemicals) are calculated using the *AP-42 Loading Model* shown in the equation below:

Equation_Apx E-33

$$Release\_perDay_{RP1} = F_{saturation\_factor} * MW * 3785.4 * V_{cont} * Rate_{fill} * X_{clean_{unload}} * \frac{VP/_{760}}{T * R} * \frac{N_{cont\_unload\_yr}/(OD * RATE_{fill})}{1000\frac{g}{kg}}$$

Where:

$Release\_perDay_{RP1}$ = Release point 1 daily releases [kg/site-day]
$F_{saturation\_factor}$ = Saturation factor [unitless]
MW = 1,4-dioxane molecular weight [g/mol]
V_{cont} = Container size for fracturing fluids [gal]
X_{clean_unload} = Vapor pressure correction factor for RP 1 and 3 [mol 1,4-dioxane/mol H_2O]
VP = 1,4-dioxane vapor pressure [torr]
T = Ambient temperature [K]
R = Universal gas constant [atm-cm^3/gmol-K]
$N_{cont\_unlaod\_yr}$ = Number of containers used yearly [containers/site-year]
OD = Operating days [days/year]
$RATE_{fill}$ = Container fill rate [containers/hour]

Release Point 2 daily releases per site (container residuals) are calculated using the following equation:

Equation_Apx E-34

$$Release\_perDay_{RP2} = Q_{dioxane\_site\_day} * F_{container\_residue}$$

Where:

$Release\_perDay_{RP2}$ = Release point 2 daily releases [kg/site-day]

$Q_{dioxane\_site\_day}$ = Daily use rate of 1,4-dioxane [kg/site-day]
$F_{container\_residue}$ = Container residual fraction [kg/kg]

Release Point 3 daily releases per site (container cleaning) are calculated using the *Mass Transfer Coefficient Model* shown in the following equation:

Equation_Apx E-35

$$Release\_perDay_{RP3} = \frac{N_{cont\_unload\_yr}}{OD * RATE_{fill}} * 3600\frac{s}{hr} * 0.001\frac{kg}{g} * \frac{(1.93 \times 10^{-7}) * (MW^{0.78}) * X_{clean_{unload}} * VP * Rate_{air\_speed}^{0.78} * (0.25\pi D_{container\_opening}^{2})\sqrt[3]{\frac{1}{29} + \frac{1}{MW}}}{T^{0.4} D_{container\_opening}^{0.11} (\sqrt{T} - 5.87)^{2/3}}$$

Where:

$Release\_perDay_{RP3}$ = Release point 3 daily releases [kg/site-day]
X_{clean_unload} = Vapor pressure correction factor release point 3 [mol 1,4-dioxane/mol water]
MW = 1,4-dioxane molecular weight [g/mol]
VP = Vapor Pressure [torr]
T = Ambient Temperature [K]
$Rate_{air\_speed}$ = Air speed [cm/s]
$D_{container\_opening}$ = Diameter of the opening for containers [cm]
P = Atmospheric pressure [atm]
OD = Operating days [days/year]
$N_{cont\_unlaod\_yr}$ = Number of containers used yearly [containers/site-year]
$RATE_{fill}$ = Container fill rate [containers/hour]

Release Point 4 daily releases per site (equipment cleaning) are calculated using the following equation:

Equation_Apx E-36

$$Release\_perDay_{RP4} = Q_{dioxane\_site\_day} * F_{equip\_cleaning}$$

Where:

$Release\_perDay_{RP4}$ = Release point 4 daily releases [kg/site-day]
$Q_{dioxane\_site\_day}$ = Daily use rate of 1,4-dioxane [kg/site-day]
$F_{equip\_cleaning}$ = Equipment cleaning loss fraction [kg/kg]

Release point 5 daily releases per site (equipment and storage tank cleaning surface losses) are calculated using the *Mass Transfer Coefficient Model* shown in the equation below:

Equation_Apx E-37

$$Release\_perDay_{RP5} = OH_{equip\_clean} * 3600\frac{s}{hr} * 0.001\frac{kg}{g} * \frac{(1.93 \times 10^{-7}) * (MW^{0.78}) * X_{tank\_clean} * VP * Rate_{air\_speed}^{0.78} * (0.25\pi D_{container\_opening}^{2})\sqrt[3]{\frac{1}{29} + \frac{1}{MW}}}{T^{0.4} D_{container\_opening}^{0.11} (\sqrt{T} - 5.87)^{2/3}}$$

Where:

$Release\_perDay_{RP5}$ = Release point 5 daily releases [kg/site-day]
X_{tank_clean} = Vapor pressure correction factor release point 5 [mol 1,4-dioxane/mol water]

MW = 1,4-dioxane molecular weight [g/mol]
VP = Vapor Pressure [torr]
T = Ambient Temperature [K]
$Rate_{air\_speed}$ = Air speed [cm/s]
$D_{container\_opening}$ = Diameter of the opening for containers [cm]
P = Atmospheric pressure [atm]
OH_{equip_clean} = Equipment cleaning operating hours [hours/day]

Release point 6 daily releases per site (deep well injection of fracturing fluid) are calculated using the following equation:

Equation_Apx E-38

$$Release\_perDay_{RP6} = Q_{dioxane\_site\_day} * \left(1 - F_{container\_residue} - F_{equip\_cleaning}\right) * (1 - F_{recovered})$$

Where:

$Release\_perDay_{RP6}$ = Release point 6 daily releases [kg/site-day]
$Q_{dioxane\_site\_day}$ = Daily use rate of 1,4-dioxane [kg/site-day]
$F_{container\_residue}$ = Container residual fraction [kg/kg]
$F_{equip\_cleaning}$ = Equipment cleaning loss fraction [kg/kg]
$F_{recovered}$ = Fraction of injected fracturing fluid returning to surface [kg/kg]

Release point 7 daily releases per site (flowback and produced wastewater) are calculated using the following equation:

Equation_Apx E-39

$$Release\_perDay_{RP7} = \frac{Q_{dioxane\_site\_yr}}{350\,\frac{days}{yr}} * \left(1 - F_{container\_residue} - F_{equip\_cleaning}\right) * F_{recovered}$$

Where:

$Release\_perDay_{RP7}$ = Release point 7 daily releases [kg/site-day]
$Q_{dioxane\_site\_yr}$ = Annual use rate of 1,4-dioxane [kg/site-year]
$F_{container\_residue}$ = Container residual fraction [kg/kg]
$F_{equip\_cleaning}$ = Equipment cleaning loss fraction [kg/kg]
$F_{recovered}$ = Fraction of injected fracturing fluid returning to surface [kg/kg]

E.13.2 Model Input Parameters

Table_Apx E-15 summarizes the model parameters and their values for the Monte Carlo simulation. Additional explanations of EPA’s selection of the distributions for each parameter are provided after this table. High-end and central tendency releases are estimated by selecting the 50th and 95th percentile values from the output distribution.

8842 **Table_Apx E-15. Summary of Parameter Values and Distributions Used in the Hydraulic Fracturing Release Model**

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational / Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Molecular weight of 1,4-dioxane	MW	g/mol	88.1	—	—	—	—	Physical property
Vapor pressure of 1,4-dioxane	VP	torr	40	—	—	—	—	Physical property
Gas constant	R	atm-cm^3/mol-K	82.05	—	—	—	—	Universal constant
Ambient temperature	T	K	298	—	—	—	—	Process parameter
Ambient pressure	P	Atm	1	—	—	—	—	Process parameter
Number of sites	N_s	sites	411	—	—	—	—	See Section E.13.3
Operating days	OD	days/year	16	1	72	16	Discrete	See Section E.13.4
Container volume (fracturing fluid)	V_{cont}	gal	55	20	1,000	55	Triangular	See Section E.13.5
Density of fracturing fluid	$RHO_{fracturing\_fluid}$	kg/L	1	—	—	—	—	ESD assumes a density equal to that of water
Diameter of container opening	$D_{container\_opening}$	cm	5.08	—	—	—	—	See Section E.13.6
Diameter of equipment opening	$D_{equip\_opening}$	cm	92	—	—	—	—	See Section E.13.7
Air speed during equipment cleaning	$RATE_{air\_speed}$	ft/min	440	—	—	—	—	See Section E.13.8
Equipment cleaning loss fraction	$F_{equip\_cleaning}$	kg/kg	0.02	—	—	—	—	See Section E.13.9

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational / Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Container fill rate	$RATE_{fill}$	containers/h	20	—	—	—	—	See Section E.13.10
Equipment cleaning operating hours	OH_{equip_clean}	h/day	4	—	—	—	—	See Section E.13.11
Annual use rate of fracturing fluids containing 1,4-dioxane	Q_{site_yr}	gal/site-year	18013874.1	26,675.00	35,429,826.00	18,013,874.10	Discrete	See Section E.13.12
Mass fraction of 1,4-dioxane in hydraulic fracturing additive	$F_{dioxane\_additive}$	kg/kg	1.00E−04	2.3E−11	0.05	1.00E−04	Discrete	See Section E.13.13
Mass fraction of 1,4-dioxane in hydraulic fracturing fluid	$F_{dioxane\_fracturing\_fluid}$	kg/kg	7.56E−08	1.00E−12	4.30E−06	7.56E−08	Discrete	
Saturation factor	$F_{saturation\_factor}$	unitless	1	0.5	1.45	0.5	Triangular	See Section E.13.14
Container cleaning loss fraction for totes	$F_{cont\_cleaning\_totes}$	kg/kg	0.002	0.0002	0.002	0.0007	Triangular	See Section E.13.15
Container cleaning loss fraction for drums	$F_{cont\_cleaning\_drums}$	kg/kg	0.03	0.017	0.03	0.025	Triangular	See Section E.13.16
Fraction of injected fracturing fluid that returns to the surface	$F_{recovered}$	kg/kg	0.51	0.1	1	0.51	Triangular	See Section E.13.17

E.13.3 Number of Sites

EPA estimates 411 sites based on found the number of hydraulic fracturing sites that reported using fracturing fluids containing 1,4-dioxane to FracFocus 3.0 (GWPC and IOGCC, 2022).

E.13.4 Operating Days

EPA modeled the operating days per year using a discrete distribution with a lower bound of 1 day per year, an upper bound of 72 days per year, and a mode of 16 days per year. Discrete data points on the number of operating days were taken from FracFocus 3.0 for the 411 sites that reported using fracturing fluids containing 1,4-dioxane (GWPC and IOGCC, 2022). The discrete distribution uses an equal probability of each operating days from FracFocus 3.0 submissions.

E.13.5 Container Size

EPA modeled container size using a triangular distribution with a lower bound of 20 gallons, an upper bound of 1,000 gallons, and a mode of 55 gallons. The Draft ESD on Chemicals Used in Hydraulic Fracturing states that hydraulic fracturing chemicals are received in drums or bulk containers. Drums are defined as containing between 20 and 100 gallons of liquid, so EPA set the lower bound of the triangular distribution at 20 gallons. Bulk containers (totes) are defined as containing between 100 and 1,000 gallons of liquid, so EPA set the upper bound of the triangular distribution at 1,000 gallons. The ESD assumes 55-gallon as default for container size at wells (U.S. EPA, 2022d), which EPA used as the mode of the triangular distribution.

E.13.6 Diameter of Container Opening

The ChemSTEER User Guide (U.S. EPA, 2015a) provides a single diameter of container openings as 5.08 cm. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 5.08 cm from the ChemSTEER User Guide.

E.13.7 Diameter of Equipment Opening

The ChemSTEER User Guide (U.S. EPA, 2015a) provides a typical diameter of equipment openings as 92 cm. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 92 cm from the ChemSTEER User Guide.

E.13.8 Air Speed during Equipment Cleaning

The ChemSTEER User Guide (U.S. EPA, 2015a) provides a single air speed of 440 ft/min during equipment cleaning activities. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 440 ft/min from the ChemSTEER User Guide.

E.13.9 Equipment Cleaning Loss Fraction

The *EPA/OPPT Multiple Process Vessel Residual Model* provides a single loss fraction 0.02 kg of material remaining as equipment residual per kg of material processed. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 0.02 kg/kg from the model (U.S. EPA, 2015a).

E.13.10 Container Fill Rate

The ChemSTEER User Guide (U.S. EPA, 2015a) provides a typical fill rate of 20 containers per hour for drums and totes. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 20 containers/hour from the ChemSTEER User Guide.

E.13.11 Equipment Cleaning Operating Hours

The ChemSTEER User Guide (U.S. EPA, 2015a) provides a single duration of 4 hours/day for equipment cleaning of multiple vessels. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 4 hours/day from the ChemSTEER User Guide.

E.13.12 Annual Use Rate of Fracturing Fluids Containing 1,4-Dioxane

EPA modeled the annual use rate of fracturing fluids containing 1,4-dioxane using a discrete distribution with a lower bound of 26,675 gal/site-year, an upper bound of 35,429,826 gal/site-year, and a mode of 18,013,874.1 gal/site-year. This discrete data was obtained from FracFocus 3.0 for the 411 sites that reported using fracturing fluids containing 1,4-dioxane (GWPC and IOGCC, 2022). The distribution was calculated using an equal probability for each of the submissions from FracFocus 3.0.

E.13.13 Mass Fraction of 1,4-Dioxane in Hydraulic Fracturing Additive/Fluid

EPA modeled the mass fraction of 1,4-dioxane in the hydraulic fracturing additive using a discrete distribution with a lower bound of 2.3×10^{-11} kg of 1,4-dioxane/kg additive, an upper bound of 0.05 kg of 1,4-dioxane/kg additive, and a mode of 1.00×10^{-4} kg of 1,4-dioxane/kg additive. EPA modeled the mass fraction of 1,4-dioxane in the hydraulic fracturing fluid using a discrete distribution with a lower bound of 1.0×10^{-12} kg of 1,4-dioxane/kg fluid, an upper bound of 4.30×10^{-6} kg of 1,4-dioxane/kg fluid, and a mode of 7.56×10^{-8} kg of 1,4-dioxane/kg fluid.

Due to the correlation between these two parameters, EPA calculated the distributions for these parameters using equal probability of submitted pairs of the mass fraction of 1,4-dioxane in hydraulic fracturing additive and the mass fraction of 1,4-dioxane in hydraulic fracturing fluid from FracFocus 3.0 submissions (GWPC and IOGCC, 2022). Initial analysis of the mass fraction of 1,4-dioxane in hydraulic fracturing additive and the mass fraction of 1,4-dioxane in hydraulic fracturing fluid using a Pearson correlation resulted in a coefficient of 0.6, which indicates a moderately strong correlation between the two sets of data.

E.13.14 Saturation Factor

The Chemical Engineering Branch Manual for the Preparation of Engineering Assessments, Volume 1 [CEB Manual] (U.S. EPA, 1991) indicates that the saturation concentration was reached or exceeded by misting with a maximum saturation factor of 1.45 during splash filling. The CEB manual indicates that the saturation factor for bottom filling was expected to be about 0.5 (U.S. EPA, 1991). The underlying distribution of this parameter is not known; therefore, EPA assigned triangular distributions, since triangular distribution is completely defined by range and mode of a parameter. Because a mode was not provided for this parameter, EPA assigned a mode value of 0.5 for bottom filling as bottom filling minimizes volatilization (U.S. EPA, 1991). This value also corresponds to the typical value provided in the ChemSTEER User Guide (U.S. EPA, 2015a) for the *EPA/OAQPS AP-42 Loading Model* for drums.

E.13.15 Container Residual Fraction for Totes

EPA modeled container residual fraction for totes using a triangular distribution with a lower bound of 0.0007 kg residual/kg fracturing fluid additive, and upper bound of 0.002 kg residual/kg fracturing fluid additive, and a mode of 0.0007 kg residual/kg fracturing fluid additive. The lower and upper bounds of this distribution are based on the central tendency and high-end values listed in the *EPA/OPPT Bulk Transport Residual Model* from the ChemSTEER User Guide (U.S. EPA, 2015a). EPA used the central tendency value as the mode of the triangular distribution.

E.13.16 Container Residual Fraction for Drums

EPA modeled container residual fraction for drums using a triangular distribution with a lower bound of 0.017 kg residual/kg fracturing fluid additive, an upper bound of 0.03 kg residual/kg fracturing fluid additive, and a mode of 0.025 kg residual/kg fracturing fluid additive. The lower bound is based on the minimum value for pumping and the upper bound is based on the default high-end value in the *EPA/OPPT Drum Residual Model* from the ChemSTEER User Guide (U.S. EPA, 2015a). EPA used the central tendency value for pumping as the mode of the triangular distribution.

E.13.17 Fraction of Injected Fracturing Fluid that Returns to the Surface

EPA modeled the fraction of injected fracturing fluid that returns to the surface using a triangular distribution with a lower bound of 0.1 kg returned/kg injected, an upper bound of 1 kg returned/kg injected, and a mode of 0.51 kg returned/kg injected. The Draft ESD on Chemicals Used in Hydraulic Fracturing provides a range of fractions from three separate data sources, with a total range of 10 to 100 percent of fracturing fluid that is injected into the ground being recovered at the surface (U.S. EPA, 2022d). The ESD uses the median amount of 51 percent as the default value, which EPA uses as the mode of the triangular distribution. The remaining amount is assumed to stay underground as a source of release (release point 6).

Appendix F OCCUPATIONAL EXPOSURES

F.1 Calculating Acute and Chronic Inhalation Exposures and Dermal Doses

For inhalation exposures, this risk evaluation assessed 1,4-dioxane exposures to workers in occupational settings, presented as 8-hour time weighted average (TWA). The 8-hour TWA exposures were used to calculate average daily concentration (ADC) for chronic, non-cancer risks, and lifetime average daily concentration (LADC) for chronic, cancer risks. Refer to Appendix G.2 of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c) for the equations EPA used for these inhalation exposure calculations. Refer to Appendix G.3 of the December 2020 Risk Evaluation for sample calculations.

For dermal exposures, this risk evaluation assessed 1,4-dioxane exposures to worker in occupational settings, presented as daily dermal potential dose rates (mg/day). The potential dose rates were then used to calculate acute retained doses (ARD), and chronic retained doses (CRD) for non-cancer and cancer risks. Refer to Appendix G.7.6 of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c) for the equations EPA used for these dermal dose calculations. Refer to Appendix G.3 of the December 2020 Risk Evaluation for sample calculations.

F.2 Approach for Estimating Number of Workers and Occupational Non-users

EPA used the same approach for estimating the number of workers and occupational non-users (ONUs) potentially exposed to the OES (listed in Section 3.1.1) as presented in the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). Refer to Appendix G.5 of the December 2020 Risk Evaluation for explanation of this approach.

Table_Apx F-1 contains a summary of the total number of workers and ONUs for each supplemental OES corresponding to estimated exposures for this supplemental risk evaluation.

8967 **Table_Apx F-1. Summary of Total Number of Workers and ONUs Potentially Exposed to 1,4-Dioxane for Each Supplemental OES**[a]

OES	Total Exposed Workers	Total Exposed ONUs	Total Exposed	Number of Facilities	Notes
Textile Dye	5,353	2,634	7,987	783	Bounding estimate based on U.S. Census Bureau data for NAICS code 313310, Textiles and Fabric Finishing Mills.
Antifreeze	182,615	18,096	200,711	84,383	Bounding estimate based on U.S. Census Bureau data for NAICS codes 811111, General Automotive Repair, and 811198, All Other Automotive Repair and Maintenance.
Surface Cleaner	552,300	32,133	584,433	55,998	Bounding estimate for the industry is based on U.S. Census Bureau data for NAICS code 561720, Janitorial Services.
Dish Soap	465,270	881,870	1,347,140	773,851	Bounding estimate for the industry is based on U.S. Census Bureau data for NAICS codes 623300, 713900, 721100, 721300, 722300, 722400, and 722500.
Dishwasher Detergent	465,270	881,870	1,347,140	773,851	Bounding estimate for the industry is based on U.S. Census Bureau data for NAICS codes 623300, 713900, 721100, 721300, 722300, 722400, and 722500.
Laundry Detergent (Industrial)	66,231	7,359	73,590	2,453	Bounding estimate based on U.S. Census Bureau data for NAICS code 812330, Linen and Uniform Supply.
Laundry Detergent (Institutional)	573,198	Unknown	Unknown	95,533	Bounding estimate based on industry information as described in the ESD on Water Based Washing operations at Industrial and Institutional Laundries (OECD, 2011b).
Paint and Floor Lacquer	111,511	11,050	122,561	33,648	Bounding estimate based on U.S. Census Bureau data for NAICS code 811121, Automotive Body, Paint, and Interior Repair and Maintenance.
PET Byproduct	43,528	17,195	60,723	1,695	Bounding estimate based on U.S. Census Bureau data for NAICS codes 325211 and 326113.

OES	Total Exposed Workers	Total Exposed ONUs	Total Exposed	Number of Facilities	Notes
Ethoxylation Process Byproduct	64,926	24,835	89,761	2,730	Bounding estimate based on U.S. Census Bureau data for NAICS codes 325110, 325199, 325611, 325613, and 325998.
Hydraulic Fracturing	46,315	26,007	72,322	411	Estimate for the number of facilities is based on the number of fracking sites that reported using 1,4-dioxane to FracFocus 3.0 (GWPC and IOGCC, 2022). Estimates for number of workers and ONUs are based on per site estimates from U.S. Census Bureau data for NAICS codes 213111 and 213112, multiplied by the number of fracking sites from FracFocus 3.0.

[a] EPA's approach and methodology for using U.S. Census Bureau data to estimate the number of facilities using 1,4-dioxane and the number of workers and ONUs potentially exposed to 1,4-dioxane can be found in the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

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F.3 Occupational Dermal Exposure Assessment Method

To assess dermal exposure, EPA used the same modeling approach as that described in Appendix G.7 of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). Specifically, EPA used the *EPA Dermal Exposure to Volatile Liquids* to calculate the dermal retained dose for each COU included in this supplemental risk evaluation. The equation modifies the *EPA 2-Hand Dermal Exposure to Liquids Model* by incorporating a "fraction absorbed (f_{abs})" parameter to account for the evaporation of volatile chemicals and a "protection factor (PF)" to account for glove use. The ECETOC TRA v3 model represents the protection factor of gloves as a fixed, assigned protection factor equal to 5, 10, or 20 (Marquart et al., 2017). Given the limited state of knowledge about the protection afforded by gloves in the workplace, EPA utilize the PF values of the ECETOC TRA v3 model (Marquart et al., 2017) as shown in Table_Apx F-2 rather than attempt to derive new values.

The fraction absorbed (f_{abs}) for 1,4-dioxane is estimated to be 0.86 in commercial settings with lower indoor wind speeds and 0.78 in industrial settings with higher indoor wind flows based on a theoretical framework provided by Kasting and Miller (2006), indicating that 86 or 78 percent of the applied dose is retained by the stratum corneum, the outermost layer of the epidermis, and absorbed systemically. Additional details on this approach can be found in the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Table_Apx F-2. Glove Protection Factors for Different Dermal Protection Strategies from ECETOC TRA v3

Dermal Protection Characteristics	Setting	Protection Factor (PF)
a. No gloves used, or any glove/gauntlet without permeation data and without employee training	Industrial and Commercial Uses	1
b. Gloves with available permeation data indicating that the material of construction offers good protection for the substance		5
c. Chemically resistant gloves (*i.e.*, as *b* above) with "basic" employee training		10
d. Chemically resistant gloves in combination with specific activity training (*e.g.*, procedure for glove removal and disposal) for tasks where dermal exposure can be expected to occur	Industrial Uses Only	20
Source: (Marquart et al., 2017)		

Occupational Dermal Exposure Assessment Bins

The December 2020 *Final Risk Evaluation for 1,4-Dioxane* included six "bins" of OES (Bins 1 through 6) for the occupational dermal analysis (U.S. EPA, 2020c). This supplemental risk evaluation builds off that analysis with the inclusion of nine additional "bins" of OES, described below.

Bin 7: covers the use of 1,4-dioxane present in textile dyes, which EPA expects may involve both commercial and industrial facilities. Workers may be exposed to 1,4-dioxane during unloading and transferring of dye products, transport container cleaning, and textile dyeing machine operation (OECD, 2017).

No Gloves Used: Actual use of gloves at textile dyeing facilities in the United States is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

Gloves Used with a Protection Factor of 5, 10, and 20: According to the GS on the Use of Textile Dyes, workers may wear proper chemical-specific personal protective equipment (OECD, 2017). EPA assumes gloves may offer a range of protection, depending on the type of glove and employee training provided.

Bin 8: covers the use of 1,4-dioxane present in antifreeze. Workers may be exposed to 1,4-dioxane during container unloading and transferring, container cleaning, and filling of antifreeze into mechanical equipment (Stefl and George, 2014).

No Gloves Used: Actual use of gloves at facilities using antifreeze is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

Gloves Used with a Protection Factor of 5 and 10: Workers may wear chemical-resistant gloves in accordance with the associated safety data sheets. Gloves may offer a range of protection, depending on the type of glove and employee training provided. A glove protection factor of 20 is not applied to this bin because the use of antifreeze is expected to be commercial and a protection factor of 20 is only applicable to industrial settings, per Table_Apx F-2.

Bin 9: covers the use of 1,4-dioxane in surface cleaner. Workers may be exposed to 1,4-dioxane during dilution of cleaner (if needed), transferring the formulations into application equipment, applying the formulation to a surface, and wiping the cleaner off the surface (OECD, 2015).

No Gloves Used: Actual use of gloves at facilities using surface cleaner is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

Gloves Used with a Protection Factor of 5 and 10: Workers may wear chemical-resistant gloves in accordance with the associated safety data sheets. Gloves may offer a range of protection, depending on the type of glove and employee training provided. A glove protection factor of 20 is not applied to this bin because the use of surface cleaners is expected to be commercial and a protection factor of 20 is only applicable to industrial settings, per Table_Apx F-2.

Bin 10: covers the use of 1,4-dioxane in dish soap. EPA expects workers may be exposed to 1,4-dioxane during the use of dish soap from unloading the dish soap, rinsing empty dish soap containers (if performed), and dish washing operations.

No Gloves Used: Actual use of gloves at facilities using dish soap is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

Gloves Used with a Protection Factor of 5 and 10: Workers may wear chemical-resistant gloves in accordance with the associated safety data sheets. Gloves may offer a range of protection, depending on the type of glove and employee training provided. A glove protection factor of 20 is not applied to this bin because the use of dish soap is expected to be commercial and a protection factor of 20 is only applicable to industrial settings, per Table_Apx F-2.

Bin 11: covers the use of 1,4-dioxane in dishwasher detergent. EPA expects workers to be exposed to 1,4-dioxane during use of dishwasher detergent from unloading and transferring formulation into machine and rinsing empty dish detergent containers (if performed).

<u>No Gloves Used</u>: Actual use of gloves at facilities using dishwasher detergent is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

<u>Gloves used with a protection factor of 5 and 10</u>: Workers may wear chemical-resistant gloves in accordance with the associated safety data sheets. Gloves may offer a range of protection, depending on the type of glove and employee training provided. A glove protection factor of 20 is not applied to this bin because the use of dishwasher detergent is expected to be commercial and a protection factor of 20 is only applicable to industrial settings, per Table_Apx F-2.

Bin 12: covers the use of 1,4-dioxane in laundry detergent, which EPA expects may involve both commercial and industrial facilities. Workers may be exposed to 1,4-dioxane during use of laundry detergent from transfer operations, container cleaning, handling damp laundry, and other operational activities (OECD, 2011b).

<u>No Gloves Used</u>: Actual use of gloves at facilities using laundry detergent is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

<u>Gloves Used with a Protection Factor of 5, 10, and 20</u>: According to the ESD on Chemicals Used in Water-Based Washing Operations at Industrial and Institutional Laundries, workers may wear proper chemical-specific personal protective equipment (OECD, 2011b). Gloves may offer a range of protection, depending on the type of glove and employee training provided.

Bin 13: covers the use of 1,4-dioxane in paint and floor lacquer, which EPA expects may involve both commercial and industrial facilities. Workers may be exposed to 1,4-dioxane during use of paint and floor lacquer from quality testing of formulations, transferring the formulations into application equipment (if used), applying the formulation to a substrate, and maintenance and cleaning activities (OECD, 2009).

<u>No Gloves Used</u>: Actual use of gloves at facilities using paint and floor lacquer is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

<u>Gloves Used with a Protection Factor of 5, 10, and 20</u>: NIOSH recommends that workers wear gloves impervious to paints and floor lacquer to prevent skin contact and avoid possible dermal exposure route (Hills et al., 1989). Gloves may offer a range of protection, depending on the type of glove and employee training provided.

Bin 14: covers the presence of 1,4-dioxane as a byproduct in industrial facilities performing PET manufacturing and ethoxylation processes. Workers may be exposed to 1,4-dioxane during PET manufacture from transferring of produced PET containing 1,4-dioxane as a byproduct and equipment

cleaning (U.S. EPA, 2021b). EPA expects workers to may be exposed to 1,4-dioxane during ethoxylation processes from transferring ethoxylated products containing 1,4-dioxane as a byproduct and equipment cleaning.

No Gloves Used: Actual use of gloves at facilities conducting PET manufacture and ethoxylation processes is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

Gloves Used with a Protection Factor of 5, 10, and 20: According to the GS on Use of Additives in Plastic Compounding, workers typically wear suitable gloves (U.S. EPA, 2021b). Gloves may offer a range of protection, depending on the type of glove and employee training provided.

Bin 15: covers the use of 1,4-dioxane in hydraulic fracturing, which EPA expects may involve both commercial and industrial settings because workers may be part of a larger company with multiple industrial facilities or from commercial contractor companies hired to support the fracturing operations. Workers may be exposed to 1,4-dioxane during multiple activities involved in hydraulic fracturing operations, including container unloading and transferring, container cleaning, and equipment cleaning (U.S. EPA, 2022d).

No Gloves Used: Actual use of gloves at hydraulic fracturing facilities is uncertain. EPA assumes workers may not wear gloves or may only wear gloves for abrasion protection or gripping that are not chemical resistant during routine operations.

Gloves Used with a Protection Factor of 5, 10, and 20: The ESD on Chemicals Used in Hydraulic Fracturing indicates that workers may wear proper chemical-specific personal protective equipment (U.S. EPA, 2022d). Gloves may offer a range of protection, depending on the type of glove and employee training provided.

F.4 Occupational Exposure Scenarios

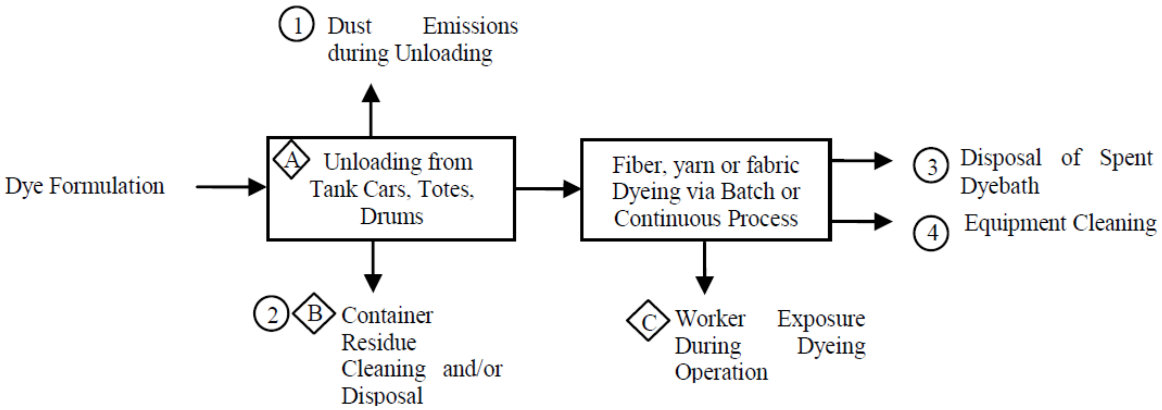
This appendix includes a process description, worker activities, estimates of the number of potentially exposed workers and ONUs, worker inhalation exposure assessment details, and key uncertainties in the exposure assessment for each OES. The process descriptions included in this appendix are applicable to the OES as a whole, including general information that is applicable to both the environmental release and occupational exposure assessments.

F.4.1 Textile Dye

Process Description

1,4-Dioxane is present in textile dyes as an unintentional byproduct in ethoxylated substances that may be used as a formulation component in textile dyes (U.S. EPA, 2020c). EPA has identified 1,4-dioxane in a textile dye formulation at a concentration of 4.7 ppm (U.S. EPA, 2020c). According to the ESD on the Use of Textile Dyes, liquid dye formulations arrive at facilities in containers ranging from 25 kg to 1,000 kg, with 35-gallon drums being the most common container size (OECD, 2017). Dyes are typically unloaded manually into equipment but may also be supplied to equipment via automated feed lines. Textile substrates are immersed in a bath in which the dye is dispersed, heated, and agitated in a batch process. Fibers in the textile substrates absorb a portion of the textile dye solution to produce the final desired product. The remaining spent dye bath is disposed of, typically to a POTW for treatment (OECD, 2017).

The volume of 1,4-dioxane present in textile dyes is unknown. Additionally, the number and location of sites that use textile dyes containing 1,4-dioxane are unknown. According to the ESD on the Use of Textile Dyes, textile dye facilities operate over a range of 31 to 295 days per year (OECD, 2017). EPA modeled the 1,4-dioxane use rate for a generic site using the ESD on the Use of Textile Dyes to estimate releases, resulting in a 50th and 95th percentile 1,4-dioxane use rate of 0.0027 and 0.0057 kg/site-day, respectively. The flow diagram with release and exposure points from the ESD on the Use of Textile Dyes is shown in Figure_Apx F-1 (OECD, 2017) below. For additional information on the modeling and associated input parameters used to estimate the daily use rate, refer to Appendix E.11.



○ = Environmental Releases:

1. Transfer operation losses of dust emissions (release to POTW, air, incineration, or landfill).
2. Container residues from dye transport container (release to POTW, incineration or landfill).
3. Disposal of spent dyebath (release to POTW).
4. Equipment cleaning (release to POTW)

◇ = Occupational Exposures:

A. Inhalation (solid particulate dyes only) and dermal exposure during equipment loading/container unloading.
B. Inhalation (solid particulate dyes only) and dermal exposure during container cleaning.
C. Dermal exposure during dyeing operation.

Figure_Apx F-1. Environmental Release and Occupational Exposure Points during Textile Dying

Worker Activities

Workers are potentially exposed to 1,4-dioxane during the use of textile dyes from unloading and transferring dye product, transport container cleaning, and machine operation (OECD, 2017). These activities are all potential sources of worker exposure through dermal contact and inhalation of 1,4-dioxane in liquid dye.

The ESD on the Use of Textile Dyes indicates that workers may connect transfer lines or manually unload chemicals from transport containers into dyeing equipment or storage (OECD, 2017). Dermal exposure is expected for both automated and manual unloading activities. Workers may experience inhalation and dermal exposure to 1,4-dioxane while rinsing containers used to transport textile dyes. Workers may be exposed to 1,4-dioxane in the liquid dyebath during removal of dyed goods after batch processes or during handling of dyed rolls of material (OECD, 2017).

According to the ESD on the Use of Textile Dyes, workers at sites that use textile dyes may wear proper chemical-specific personal protective equipment (OECD, 2017). Workers may wear safety glasses, goggles, aprons, respirators, and/or masks (OECD, 2017). EPA did not find information that indicates the extent that engineering controls and worker PPE are used at facilities that use textiles dyes in the United States.

ONUs include employees that work at the sites where textile dyes are used, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and are not expected to have dermal exposures through contact with liquids or solids. ONUs for this scenario include supervisors, managers, and other employees that may be in the dyeing area but do not perform tasks that result in the same level of exposure as those workers that engage in tasks related to the use of textile dyes.

Number of Potentially Exposed Workers and ONUs

EPA used U.S. Census and BLS data for the NAICS code 313310, Textiles and Fabric Finishing Mills, to estimate a total of 783 sites, 5,353 workers, and 2,634 ONUs, which corresponds to an estimated average of seven workers and three ONUs per site (U.S. BLS, 2016). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

1,4-Dioxane is present in textile dyes as an unintentional byproduct in ethoxylated substances that may be used as a formulation component in textile dyes (U.S. EPA, 2020c). The information and data quality evaluation to assess occupational exposures during use of textile dye is listed in Table_Apx F-3 and described in detail below.

Table_Apx F-3. Textile Dye Worker Exposure Data Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source Reference
Unknown	PBZ Monitoring	14	High	(OSHA, 2020)

EPA assessed occupational inhalation exposures for this OES using OSHA's Chemical Exposure Health Data (CEHD) (OSHA, 2020). EPA obtained CEHD for 1,4-dioxane from the OSHA webpage, including sampling data from 1984 to the present (data were pulled in mid-2022). EPA then edited the resulting data download by excluding all sample types except for personal and area samples (*e.g.*, excluding wipe samples, bulk samples) and excluding blank samples. EPA converted the CEHD from parts per million (ppm) to mg/m^3 by multiplying the values by the molecular weight of 1,4-dioxane and dividing by the molar volume. EPA then mapped the CEHD to 1,4-dioxane OES. To map the CEHD, EPA used the SIC codes reported in the CEHD and corresponding SIC descriptions to identify the most likely OES for the establishment at which the inhalation monitoring data was taken. In some cases, EPA searched the internet for the establishment name to identify the types of products manufactured at the facility to aid the OES mapping process. Due to the subjectivity of OES mapping and broadness of SIC codes, OES mapping is an uncertainty of the assessment.

For this OES, monitoring data were available in CEHD from four sites with SIC codes 2399 (All Other Misc. Textile Product Mills), 3111 (Leather and Hide Tanning and Finishing), 5136 (Men and Boy's Clothing and Furnishings), and 2326 (Men and Boy's Work Clothing). EPA determined these four sites to be clothing manufacturers, which may use textile dyes. Note that data were also available in CEHD for one site that reported the SIC code 3143 (Footwear Manufacturing); however, EPA determined that 1,4-dioxane may be used in a variety of ways within footwear manufacturing such that the potential for use in textile dyes was low. 1,4-Dioxane may be used as a functional fluid for automated footwear production machines, a detergent in washing footwear before distribution, or as a polymerization catalyst to make plastic components of shoes. In addition, footwear is often composed of leather or plastic, which would not use textile dyes. Therefore, EPA excluded the data for this one site from that used for assessing occupational inhalation exposures for this OES.

Table_Apx F-5 shows the discrete inhalation monitoring points from the CEHD set that EPA mapped to the textile dyes OES. The majority of data are from 1991 and 1992, with a smaller portion from 2010. The data include 14 inhalation monitoring data points, 12 of which are PBZ samples and two are area samples. Although EPA prefers PBZ data over area data to estimate worker exposures, EPA used both the PBZ and area samples in this analysis due to the limited amount of data available. CEHD does not include information on worker activities for PBZ samples or sampling locations for area samples, therefore EPA's assessment assumes that that all 14 samples are relevant to this assessment. However, it is uncertain the extent to which all potential worker activities are represented in these data.

The CEHD includes an inspection number, which corresponds to the OSHA visit at the facility, and a sampling number, which corresponds to the worker sampling event at the facility. EPA combined the exposure concentrations and sampling times for samples with the same inspection and sampling numbers because these correspond to the same worker and the same day. Therefore, combining these exposure results is more reflective of full-shift exposures for the worker than the individual short-term samples. As shown in Table_Apx F-5, sample durations ranged from 34 to 275 minutes. For these samples with detected values, EPA translated the sample results into 8-hour TWA concentrations by assuming that exposure concentration is zero for the time remaining in the 8-hour durations.

Where non-detect values were included in the dataset, EPA first calculated the LOD for the sample. EPA assumed the use of NIOSH method 1602, which has an estimated LOD of 0.01 mg/sample. To calculate LOD in terms of an air concentration, EPA divided the limit of 0.01 mg/sample by the sampled air volume provided in the CEHD, which converted from L to m^3. For the non-detect values, EPA then used the LOD divided by two in subsequent central tendency (50th percentile) and high-end (95th percentile) calculations. EPA used the LOD/2 for approximating an air concentration for non-detect samples because the geometric standard deviation of the dataset is greater than three (U.S. EPA, 1994a). Because greater than 50 percent of the monitoring data results are non-detect for 1,4-dioxane, this method for the calculation of statistics will result in potentially biased estimates.

EPA then used the air concentrations and LOD/2 as shown in Table_Apx F-5 to calculate full shift (8-hour TWA) central tendency (50th percentile) and high-end (95th percentile) inhalation exposures for workers. EPA used these central tendency and high-end values to calculate the ADC and LADC. The calculated values are summarized in Table_Apx F-4. Equations for calculating ADC and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Exposure data for ONUs were not available. The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-4. Inhalation Exposures of Workers for the Use of Textile Dye Based on Monitoring Data

Exposure Type	Central Tendency (50th Percentile) (mg/m^3) [a]	High-end (95th Percentile) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	0.07	74
Average Daily Concentration (ADC)	0.040	71.15
Lifetime Average Daily Concentration (LADC)	0.016	36.49

[a] See Table_Apx F-3 for corresponding references.

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Row #	Type of Sample	Worker Activity or Sample Location	No. of Samples	Sample Date	Sample Time (min)	1,4-Dioxane Exposure Concentration (mg/m^3)	EPA Determined 8-hour TWA (mg/m^3)	Source	Overall Data Quality Determination
1	Personal	N/A	1	1/17/1991	175	42.54	55.89	(OSHA, 2020)	High
2	Personal	N/A	1	1/17/1991	188	54.96	77.57	(OSHA, 2020)	High
3	Personal	N/A	1	6/3/1992	259	37.05	72.04	(OSHA, 2020)	High
4	Personal	N/A	1	6/3/1992	272	ND (LOD = 0.5 mg/m^3)	0.14	(OSHA, 2020)	High
5	Personal	N/A	1	6/3/1992	176	ND (LOD = 0.78 mg/m^3)	0.14	(OSHA, 2020)	High
6	Personal	N/A	1	6/3/1992	173	ND (LOD = 0.79 mg/m^3)	0.14	(OSHA, 2020)	High
7	Personal	N/A	1	12/10/1992	37	ND (LOD = 0.10 mg/m^3)	0.004	(OSHA, 2020)	High
8	Personal	N/A	1	12/9/1992	35	ND (LOD = 0.11 mg/m^3)	0.004	(OSHA, 2020)	High
9	Area	N/A	1	7/15/2010	69	ND (LOD = 0.86 mg/m^3)	0.06	(OSHA, 2020)	High
10	Area	N/A	1	7/15/2010	270	ND (LOD = 0.25 mg/m^3)	0.07	(OSHA, 2020)	High
11	Personal	N/A	1	7/15/2010	244	ND (LOD = 0.22 mg/m^3)	0.06	(OSHA, 2020)	High
12	Personal	N/A	1	7/15/2010	150	ND (LOD = 0.39 mg/m^3)	0.06	(OSHA, 2020)	High
13	Personal	N/A	1	7/15/2010	155	ND (LOD = 0.39 mg/m^3)	0.06	(OSHA, 2020)	High
14	Personal	N/A	1	7/15/2010	294	ND (LOD = 0.20 mg/m^3)	0.06	(OSHA, 2020)	High

ND = Non-detect for 1,4-dioxane; LOD = limit of detection; TWA = time-weighted average

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Key Uncertainties

The OSHA CEHD monitoring data does not include process information or worker activities; therefore, there is uncertainty as to which worker activities these data cover and whether all potential workers activities are represented in this data. Additionally, these data are from four facilities, and it is unclear how representative the data are for all sites and all workers across the United States. Approximately half of OSHA CEHD used for this assessment are from the 1990s and the other half are from 2010. Therefore, the age of the monitoring data can also introduce uncertainty.

As discussed above, EPA used half the detection limit for the non-detect values in the central tendency and high-end exposure calculations. Due to the high number of non-detects (11 of the 14 TWAs were non-detect), this method may result in bias (U.S. EPA, 1994a). Additional uncertainties are listed in Section 3.1.2.4.

F.4.2 Antifreeze

Process Description

1,4-Dioxane is present in antifreeze as an unintentional byproduct of certain ethoxylated substances that may be used as a formulation component in antifreeze (U.S. EPA, 2020c). EPA has identified 1,4-dioxane concentrations in antifreeze ranging from 0.01 to 86 ppm (U.S. EPA, 2020c). Antifreeze is formulated for use in motor vehicles and other mechanical equipment to prevent freezing of engine fluids (Stefl and George, 2014). EPA did not find any container specific information on 1,4-dioxane in antifreeze; however, EPA expects the antifreeze formulation to ship to automotive maintenance facilities as a liquid in drums or smaller containers. Antifreeze is manually added to engines and is typically replaced every two to three years. Upon completion of use, the spent antifreeze may be recycled or disposed to municipal waste treatment facilities (Stefl and George, 2014).

The volume of 1,4-dioxane present in antifreeze is unknown. Additionally, the number and location of sites that use antifreeze containing 1,4-dioxane are unknown. EPA modeled the 1,4-dioxane use rate using the consumer model from the published risk evaluation and the MRD on Automotive Detailing, resulting in a low-end and high-end 1,4-dioxane use rate of 0.15 and 1.35 kg/site-day, respectively (U.S. EPA, 2022b, 2020c). EPA assumes facilities use antifreeze 5 days/week, 50 weeks/year or 250 days/year. For additional information on the modeling and associated input parameters used to estimate the daily use rate, refer to Appendix F.5.

Worker Activities

Workers are potentially exposed to 1,4-dioxane during multiple activities involved in use of antifreeze, including container unloading and transferring, container cleaning, and filling of antifreeze into mechanical equipment (Stefl and George, 2014). These activities are all potential sources of worker exposure through dermal contact to liquid and inhalation of volatile chemical vapors.

Workers may don personal protective equipment (PPE) during the use of antifreeze in accordance with the associated safety data sheets. EPA did not find information that indicates the extent to which engineering controls are present or worker PPE are worn at U.S. facilities that use antifreeze.

ONUs include employees that work at the sites where antifreeze is used, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and are not expected to have dermal exposures through contact with liquids. ONUs for this scenario include supervisors, managers, and other employees that may be in the filling area but do not perform tasks that result in the same level of exposures as those workers that engage in tasks related to the use of antifreeze.

Number of Potentially Exposed Workers and ONUs

EPA estimated the number of workers and occupational non-users potentially exposed to 1,4-dioxane in antifreeze using 2016 BLS data for NAICS codes 811111, General Automotive Repair, and 811198, All Other Automotive Repair and Maintenance. Using BLS data, EPA estimated a total of 84,383 sites, two workers per site, and 0.2 ONUs per site (U.S. BLS, 2016). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

1,4-Dioxane is present in antifreeze as an unintentional byproduct of certain ethoxylated substances that may be used as formulation components in antifreeze (U.S. EPA, 2020c). The information and data quality evaluation to assess occupational exposures during use of antifreeze is listed in Table_Apx F-6 and described below.

Table_Apx F-6. Antifreeze Data Source Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source Reference
Unloading/transferring antifreeze from containers	Input parameters for Monte Carlo modeling	N/A	High[a]	(U.S. EPA, 2022b)
Unloading/transferring antifreeze from containers	Input parameters for Monte Carlo modeling	N/A	High[a]	(OECD, 2020)

[a] This is the rating for the underlying data used in the model, and not the Monte Carlo model itself.

EPA did not find relevant inhalation monitoring data for the use of antifreeze. Therefore, EPA modeled 1,4-dioxane air concentrations using a Monte Carlo modeling approach, which is described in Appendix F.7. This modeling approach utilizes the *EPA AP-42 Loading Model* and the *EPA Mass Balance Inhalation Model,* with variation in input parameters for container size, jobs per day, concentration of 1,4-dioxane in antifreeze, ventilation rate, mixing factor, and saturation factor based on available data. Table_Apx F-7 provides a summary of the modeled inhalation exposures.

Table_Apx F-7. Modeled Occupational Inhalation Exposures for Antifreeze

Statistic	1,4-Dioxane Exposure Concentration, 8h-TWA (mg/m^3)
Maximum	1.19E−06
99th Percentile	2.11E−07
95th Percentile	1.10E−07
50th Percentile	2.18E−08
5th Percentile	1.94E−09
Minimum	2.37E−12
Mean	3.49E−08

EPA used the 50th and 95th percentile modeled 8-hour TWA exposures values presented in Table_Apx F-7 to calculate the central tendency and high-end ADC and LADC for workers, respectively. The calculated values are summarized in Table_Apx F-8. Equations for calculating ADC and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Exposure data for ONUs were not available. The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-8. Inhalation Exposures of Workers for the Use of Antifreeze Based on Modeling

Exposure Type	Central Tendency (50th Percentile) (mg/m^3) [a]	High-end (95th Percentile) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	2.18E−08	1.10E−07
Average Daily Concentration (ADC)	2.10E−08	1.06E−07
Lifetime Average Daily Concentration (LADC)	8.34E−09	5.44E−08

[a] See Table_Apx F-6 for corresponding references.

Key Uncertainties

Due to a lack of data specific to 1,4-dioxane for this use, EPA used assumptions and values from the Automotive Detailing MRD, Automotive Lubricant ESD, *EPA AP-42 Loading Model*, *EPA Mass Balance Inhalation Model*, and *Near-Field/Far-Field Brake Model*. The uncertainties associated with this modeling approach are described in Section 3.1.2.4.

In addition, the commercial use rate of antifreeze was scaled up from the consumer use rate provided by the SHEDS-HT model, using the number of jobs per day from the Automotive Detailing MRD, Automotive Lubricant GS. These scaling factors may overestimate exposure if the actual number of jobs at commercial sites is lower or may underestimate exposure if the actual number of jobs at commercial sites is higher.

F.4.3 Surface Cleaner

Process Description

1,4-Dioxane is present an unintentional byproduct in ethoxylated substances that may be used as a formulation component in surface cleaners (U.S. EPA, 2020c). EPA has identified concentrations of 1,4-dioxane in surface cleaners ranging from 0.36 to 9.0 ppm (U.S. EPA, 2015b). Surface cleaners are used to disinfect and remove unwanted foreign matter from various types of surfaces (Nagy and Theiner, 2020). EPA did not find any container specific information on 1,4-dioxane in surface cleaners; however, EPA expects formulation to arrive as a liquid in small containers of various sizes. Surface cleaners may be aqueous, semi-aqueous, or non-aqueous. Aqueous and semi-aqueous cleaners may be diluted with water prior to use. The cleaner is typically spray applied to the surface and wiped off (OECD, 2015).

The volume of 1,4-dioxane present in surface cleaners is unknown. Additionally, the number and location of sites that use surface cleaners containing 1,4-dioxane are unknown. EPA modeled the 1,4-dioxane use rate the SHEDS-HT case study from Liverpool, OH, resulting in a central tendency and high end 1,4-dioxane use rate of 79 and 85 g/site-day, respectively. EPA assumes facilities use surface cleaners 5 days/week, 50 weeks/year or 250 days/year.

Worker Activities

During the use of surface cleaners, workers are potentially exposed during the dilution of cleaner (if needed), transferring the formulations into application equipment, applying the formulation to a surface, and wiping the cleaner off the surface. These activities are all potential sources of worker exposure through dermal contact to liquid and inhalation of vapors (OECD, 2015).

EPA did not find information that indicates the extent that engineering controls and worker PPE are used at facilities that use surface cleaners in the United States.

ONUs include employees that work at the site where surface cleaners are used, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and vapor-through-skin uptake. Additionally, dermal exposures through contact with liquids are not expected. ONUs include supervisors, managers, and other employees that may be in the cleaning area but do not perform tasks that result in the same level of exposures as workers that engage in tasks related to the use of surface cleaner.

Number of Potentially Exposed Workers and ONUs

EPA used U.S. Census and BLS data for the NAICS code 561720, Janitorial Services, to estimate a total of 55,998 sites, 552,300 workers, and 32,144 ONUs, which corresponds to an estimated average of 9.9 workers and 0.6 ONUs per site (U.S. BLS, 2016). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

1,4-Dioxane is present an unintentional byproduct in ethoxylated substances that may be used as a formulation component in surface cleaners (U.S. EPA, 2020c). The information and data quality evaluation to assess occupational exposures during use of surface cleaner is listed in Table_Apx F-9 and described below.

Table_Apx F-9. Surface Cleaner Worker Exposure Data Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source Reference
Use of surface cleaning products	PBZ Monitoring	49	High	(Harley et al., 2021)
Use of surface cleaning products	Activity duration	N/A	High	(U.S. EPA, 2022a)

Table_Apx F-10 shows the 1,4-dioxane inhalation monitoring data available in published literature related to the use of surface cleaners (Harley et al., 2021). This data is from a study in which 49 PBZ samples were taken in 2019 during the use of surface cleaners in domestic kitchens and bathrooms. The study does not provide the discrete values for the 49 samples but does provide the geometric mean and maximum of the 49 samples, which are 0.57 $\mu g/m^3$ and 7.38 $\mu g/m^3$, respectively. In this study, personal air monitoring was conducted on 50 consumers while they cleaned their homes with standard cleaning products for 30 minutes. The volunteers were asked to clean their own kitchen and bathroom using their regular cleaning products while wearing a small backpack containing personal air monitoring equipment. For this OES, EPA did not find air monitoring of workers or other occupational non-users; therefore, EPA uses the data from Harley (2021), which is for consumer use, as surrogate for occupational exposures. EPA expects that both consumers and workers utilize similar practices for surface cleaning such that the inhalation exposure potential is similar between the two. EPA recognizes, however, that workers are more likely to conduct surface cleaning at a higher frequency or for longer durations than consumers. Therefore, EPA used available information to determine the appropriate exposure durations for workers, which is described further below.

EPA converted the geometric mean and maximum 30-minute air concentration values into 8-hour TWAs by using information from the Draft GS on Furnishing Cleaning on expected occupational cleaning durations. The GS indicates a cleaning duration of up to 4 hours is expected for furnishing cleaning (U.S. EPA, 2022a). EPA did not find data specific to surface cleaning durations and assumes the exposure duration for surface cleaning is similar to that for furnishing cleaning operations; however, this assumption is a limitation of EPA’s analysis. Therefore, to convert the 30-minute geometric mean and maximum air concentrations from Harley (2021) to 8-hour TWAs, EPA assumed the air concentrations were representative of a 4-hour period and there was no exposure for the remaining 4 hours in the 8-hour shift. EPA then used these values to calculate the ADC and LADC. The calculated values are summarized in Table_Apx F-11. Equations for calculating ADC and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Exposure data for ONUs were not available. The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-10. Inhalation Exposures of Workers for the Use of Surface Cleaner Based on Monitoring Data

Exposure Type	Central Tendency (Geometric Mean) (mg/m^3) [a]	High-End (Maximum) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	2.9E−04	3.70E−03
Average Daily Concentration (ADC)	2.79E−04	3.56E−03
Lifetime Average Daily Concentration (LADC)	1.11E−04	1.82E−03

[a] See Table_Apx F-9 for corresponding references.

Table_Apx F-11. Occupational Inhalation Monitoring Data for Surface Cleaner

Row #	Type of Sample	Worker Activity or Sample Location	Number of Samples	Sample Date	Sample Time	1,4-Dioxane Exposure Concentration (mg/m^3)	EPA Determined 8-hour TWA (mg/m^3)	Source	Overall Data Quality Determination
1	Personal	Use of surface cleaning products	49[a]	2019	30 min	0.00057 (Geometric mean)	0.00029 (Geometric mean)	(Harley et al., 2021)	High
						0.00738 (Maximum)	0.0037 (Maximum)		

[a] Source did not include discrete values for each of the 49 samples but provided the geometric mean and maximum.
TWA = Time-weighted average.

Key Uncertainties

The literature source EPA used for this COU did not present discrete sampling values, so EPA used the geometric mean and maximum of the sample results as they were provided in the source (Harley et al., 2021). The representativeness of these values towards the central tendency and high-end exposures is uncertain due to the lack of discrete data provided and inability to verify summary statistics. Additionally, there is uncertainty in how the literature source accounted for non-detect values in the geometric mean calculation.

It is unknown whether the activities performed in this study accurately reflect all surface cleaning scenarios or the cleaning industry as whole. Also, EPA assumed that cleaning activities occur over four hours per day per the Draft Furnishing Cleaning GS (U.S. EPA, 2022a). Besides the Furnishing Cleaning GS, ERG didn't identify any other sources to estimate frequency and duration of cleaners. This assumption may result in an underestimate or overestimate of exposures if cleaning occurs over a different timeframe. Additional uncertainties are listed in Section 3.1.2.4.

F.4.4 Dish Soap

Process Description

1,4-Dioxane has been identified as an unintentional component in dish soaps (U.S. EPA, 2020c). Sources indicate 1,4-dioxane content in dish soaps ranges from 0.03 to 204 ppm (U.S. EPA, 2020c; Lin et al., 2017; Saraji and Shirvani, 2017; Makino et al., 2006; Wala-Jerzykiewicz and Szymanowski, 1998). Note that some sources identify “dishwashing liquids”; EPA assumed these products may be either dish soaps or dishwashing detergents. Additionally, some of these data are for 1,4-dioxane concentrations in consumer dish soaps; however, EPA expects similar formulations may be used commercially. EPA did not find any container specific information on 1,4-dioxane in dish soaps; however, EPA expects formulation containing 1,4-dioxane contaminant to arrive as a liquid in small containers of various sizes. In an occupational setting, EPA expects dish soaps are to be used when a worker handwashes commercial cookware, cutlery, tableware, and kitchen tools in sinks. Dirty water containing the used dish soap are rinsed down sink drains to POTWs (ATSDR, 2012).

The volume of 1,4-dioxane present in dish soaps is unknown. Additionally, the number and location of sites that use dish soaps containing 1,4-dioxane are unknown. EPA assumes facilities use dish soaps 5 days/week, 50 weeks/year or 250 days/year. EPA modeled the 1,4-dioxane use rate using the SHEDS-HT case study from Liverpool, OH to estimate releases, resulting in a central tendency and high end 1,4-dioxane use rate of 64.6 and 64.8 g/site-day, respectively.

Worker Activities

EPA expects workers to be potentially exposed to 1,4-dioxane during the use of dish soap from unloading and transferring formulation, transport container cleaning, and washing operations. These activities are all potential sources of worker exposure through dermal contact to liquid and inhalation of vapors.

EPA expects suitable personal protective equipment (PPE) to be worn in accordance with the safety data sheets. EPA did not find information that indicates the extent that engineering controls and worker PPE are used at facilities that use dish soap in the United States.

ONUs include employees that work at the sites where dish soaps are used, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and are not expected to have dermal exposures by contact with liquids. ONUs for this scenario include supervisors, managers, and other employees that may be in the washing area are but do not perform tasks that result in the same level of exposure as those workers that engage in tasks related to the use of dish soaps.

Number of Potentially Exposed Workers and ONUs

To estimate the number of workers, EPA used U.S. Census and BLS data for the following NAICS codes: 623300, 713900, 721100, 721300, 722300, 722400, and 722500. EPA estimated a total of 773,851 sites, 0.6 workers per site, and 1.1 ONUs per site (U.S. BLS, 2016). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

1,4-Dioxane has been identified as an unintentional component in dish soaps (U.S. EPA, 2020c). The information and data quality evaluation to assess occupational exposures during use of dish soap is listed in Table_Apx F-12 and described below.

Table_Apx F-12. Dish Soap Worker Exposure Data Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source Reference
Transfer of detergents to/from storage, liquid mixing, and detergent bottling	PBZ Monitoring Data	29	Medium	(Belanger, 1980)

Table_Apx F-14 shows the inhalation monitoring data that is available from a NIOSH HHE report (Belanger, 1980). The report provided 29 PBZ and area samples for 1,4-dioxane collected in 1980. The source does not provide discrete sample information, indicating that all samples were non-detect for 1,4-dioxane. Worker activities included transfers of detergents and mixing activities. Six detergents were analyzed for 1,4-dioxane, with bulk concentrations in these detergents ranging from 0–0.423 percent 1,4-dioxane.

In this study, personal and area air samples were collected over one 8-hour work shift. Because all samples were non-detect, but the study did not provide the limit of detection, EPA calculated the limit of detection. This sampling performed for this study used NIOSH method S360, which is a precursor to NIOSH method 1602. NIOSH method 1602 has a limit of detection of 0.01 mg/sample. Because this study does not include the volume of air sampled, EPA used the sampling flow rate from the method to convert this limit of detection into an air concentration (mg/m^3). Specifically, this method indicates the typical sampling flow rate is 0.01 to 0.2 L/min. EPA assumed a flow rate of 0.01 L/min to maximize the potential air concentration. Using the method detection limit of 0.01 mg/sample, sampling flow rate of 0.01 L/min, and a duration of 8-hours per the study, EPA calculated an LOD of 2.1 mg/m^3. Due to EPA's assumptions to maximize this LOD, EPA used this value of 2.1 mg/m^3 as the high-end inhalation exposure for this OES.

EPA used the LOD divided by two as the central tendency inhalation exposure for this OES because EPA guidance indicates to use this approach for calculating statistics in datasets with non-detect samples (U.S. EPA, 1994a). EPA used the above high-end and central tendency (LOD and LOD/2) values to calculate the ADC and LADC. The calculated values are summarized in Table_Apx F-13. Equations for calculating ADC and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

EPA (,1994, 5071455} cautions that when greater than 50 percent of the monitoring data results are non-detect, this method using the LOD/2 will result in potentially biased estimates. Therefore, EPA compared the estimated occupational inhalation exposure estimates with the inhalation exposure estimate for consumers included in the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). EPA estimated that consumer use of dish soap results in 1,4-dioxane air concentrations of up to 0.03 mg/m^3 (U.S. EPA, 2020c). This value is a factor of two lower than EPA's estimates of occupational inhalation exposure; however, EPA expects that occupational users of dish soap are potentially exposed to higher concentrations of 1,4-dioxane because they use these dish soaps at a higher frequency and for a longer duration than consumers.

Exposure data for ONUs were not available. The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-13. Inhalation Exposures of Workers for the Use of Dish Soap Based on Monitoring Data

Exposure Type	Central Tendency (LOD/2) (mg/m^3) [a]	High-end (LOD) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	1.0[b]	2.1[c]
Acute Exposure Concentration (AEC)	N/A	N/A
Average Daily Concentration (ADC)	1.0	2.0
Lifetime Average Daily Concentration (LADC)	0.398	1.03

[a] See Table_Apx F-12 for corresponding references.
[b] All data were non-detect; EPA presented the LOD/2 for the central tendency value.
[c] All data were non-detect; EPA presented the LOD for the high-end value.

9555 **Table_Apx F-14. Occupational Inhalation Monitoring Data for Dish Soap**

Row #	Type of Sample	Worker Activity or Sample Location	Number of Samples	Sample Date	Sample Time (min)	1,4-Dioxane Exposure Concentration (mg/m^3)	EPA Determined 8-hour TWA (mg/m^3)	Source	Overall Data Quality Determination
1	Personal	N/A	29[a]	November 1979	N/A	ND	N/A	(Belanger, 1980)	N/A

[a] The NIOSH HHE took 29 samples and indicated all were non-detect. Individual sample times or other sample information was not provided for each discrete data point.

ND = non-detect for 1,4-dioxane; TWA = time-weighted average

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Key Uncertainties

Because all of the PBZ and area samples from the NIOSH HHE (Belanger, 1980) were non-detect for 1,4-dioxane, EPA used the level of detection and half the level of detection as the high-end and central tendency inhalation exposure estimates, respectively, for this COU. This assumption may overestimate or underestimate exposures depending on the actual concentration of 1,4-dioxane relative to zero and the detection limit. Additionally, the exposure estimates in this section are only based on one literature source. It is unclear how representative the data from this one source are for all sites and all workers across the United States. Additionally, the data are from 1980, so the age of the monitoring data may also introduce uncertainty. Additional uncertainties are listed in Section 3.1.2.4.

F.4.5 Dishwasher Detergent

Process Description

1,4-Dioxane has been identified as an unintentional component in dishwasher detergent containing ethoxylated surfactants (U.S. EPA, 2020c). Sources indicate 1,4-dioxane content in dishwasher detergents ranges from 0.86 to 51 ppm (U.S. EPA, 2020c; Lin et al., 2017; Saraji and Shirvani, 2017; Davarani et al., 2012; Makino et al., 2006; Wala-Jerzykiewicz and Szymanowski, 1998). Note that some sources identify "dishwashing liquids"; EPA assumed these products may be either dish soaps or dishwashing detergents. Additionally, some of these data are for 1,4-dioxane concentrations in consumer dishwashing detergents; however, EPA expects similar formulations may be used commercially. EPA did not find any container specific information on 1,4-dioxane in dishwasher detergents; however, EPA expects any formulation containing 1,4-dioxane to be transported as a liquid in small containers of various sizes. In an occupational setting, EPA expects dishwasher detergent to be used in a machine to clean dishware, cookware, and cutlery. Dirty water containing the used dishwasher detergent and 1,4-dioxane are rinsed down machine drains to POTWs (ATSDR, 2012).

The volume of 1,4-dioxane present in dishwasher detergents is unknown. Additionally, the number and location of sites that use dishwasher detergents containing 1,4- are unknown. EPA did not identify data on facility operating schedules. EPA assumes facilities use 1,4-dioxane 5 days/week, 50 weeks/year or 250 days/year. EPA modeled the 1,4-dioxane use rate using the SHEDS-HT case study from Liverpool, OH to estimate releases, resulting in a 50th and 95th percentile 1,4-dioxane use rate of 1.44 g/site-day.

Worker Activities

EPA expects workers to be potentially exposed to 1,4-dioxane during the use of dishwasher detergent from unloading and transferring formulation into machine and transport container cleaning. These activities are all potential sources of worker exposure through dermal contact to liquid and inhalation of vapors.

ONUs include employees that work at the sites where dishwasher detergents are used, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and are not expected to have dermal exposures through contact with liquids. ONUs for this scenario include supervisors, managers, and other employees that may be in the dishwashing area are but do not perform tasks that result in the same level of exposure as those workers that engage in tasks related to the use of dishwasher detergent.

Number of Potentially Exposed Workers and ONUs

To estimate the number of workers, EPA used U.S. Census and BLS data for the following NAICS codes: 623300, 713900, 721100, 721300, 722300, 722400, and 722500. EPA estimated a total of 773,851 sites, 0.6 workers per site, and 1.1 ONUs per site (U.S. BLS, 2016). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

EPA used the same approach as discussed for dish soap in Appendix F.4.4 to estimate inhalation exposures to 1,4-dioxane during the use of dishwasher detergent. This approach was taken due to the lack of available data for dishwasher detergents. One limitation of this assumption is that the use of dish soap may involve more hands-on activities (scrubbing, rinsing, etc.) compared to the automated washing operations in the use of dishwasher detergent. Therefore, estimates for inhalation exposures during use of dish soap may potentially overestimate inhalation exposures during the use of dishwasher detergents.

EPA estimated that consumer use of dish soap results in 1,4-dioxane air concentrations of up to 0.00069 mg/m^3 (U.S. EPA, 2020c). This value is a factor of four lower than EPA’s estimates of occupational inhalation exposure, likely for the reason stated above. However, EPA expects that occupational users of dishwasher detergent are potentially exposed to higher concentrations of 1,4-dioxane because they use these detergents at a higher frequency and for a longer duration than consumers.

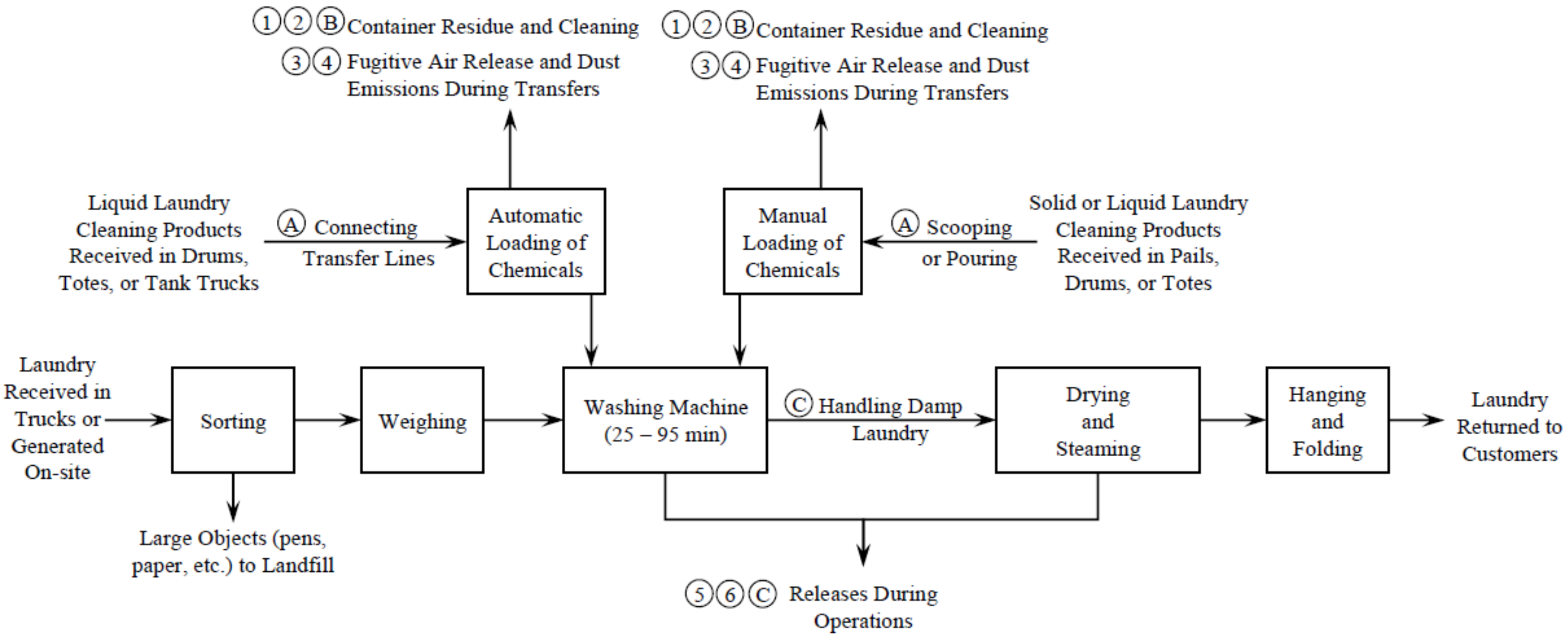
Key Uncertainties

Since EPA used the same approach as discussed for dish soap in Appendix F.4.4, the same key uncertainties in that appendix apply.

F.4.6 Laundry Detergent (Industrial and Institutional)

Process Description

1,4-Dioxane is found in laundry detergents due to its presence as an unintentional byproduct in certain ethoxylated substances that may be used as formulation components (U.S. EPA, 2020c). Laundries can be classified into two main categories in the United States: industrial and institutional (OECD, 2011b). For both categories, the laundered items are loaded into the mechanical washers and the laundry is washed using water and a detergent appropriate for the item type and soil loading. Washing is completed in a continuous process composed of a series of cycles. The wash cycle is typically followed by a rinse cycle to remove the of the detergent chemicals. While many facilities may have on-site wastewater treatment, most of these treatment technologies are designed to remove dirt and oil, not detergent chemicals. Subsequently, the wastewater is transferred down drains to a POTW. A flow diagram including release and exposure points from the ESD on Water Based Washing Operations at Industrial and Institutional Laundries is presented in Figure_Apx F-2 (OECD, 2011b).



Exposures:

Ⓐ Dermal and inhalation exposure from connecting transfer lines or from scooping and pouring.

Ⓑ Dermal and inhalation exposure during container cleaning (if containers cleaned on-site).

Ⓒ Dermal exposure from handling damp laundry and inhalation exposure to vaporized chemicals during operations.

Releases:

① Transport container residue released to water, incineration, or landfill.

② Open surface losses to air during transport container cleaning (if containers cleaned on-site; volatile chemicals only).

③ Transfer operation losses to air from unloading and transferring laundry cleaning product (volatile chemicals only).

④ Dust losses during unloading and transferring solids (powdered laundry products only).

⑤ Releases to air within the workers' breathing zone from operations.

⑥ Washing water discharge to POTW (non-volatile and volatile chemicals) and evaporation losses to air during washing and drying operations (volatile chemicals only).

Figure_Apx F-2. Environmental Release and Occupational Exposure Points during Industrial/Institutional Laundering Operation

Industrial Laundries: Industrial laundries wash soiled garments, linens, etc., received from hospitals, repair shops, doctor's offices, industrial sites, as well as other customers (OECD, 2011b). EPA did not find specific container information for 1,4-dioxane in industrial laundry detergents; however, the ESD on Water Based Washing Operations at Industrial and Institutional Laundries indicates that industrial laundry detergents are typically transported as a liquid or powder in drums, totes, or bulk tanker trucks (OECD, 2011b). 1,4-Dioxane can be present in institutional laundry detergents at concentrations from 0.05 to 14 ppm (U.S. EPA, 2020c).

The volume of 1,4-dioxane present in industrial laundry detergents is unknown. Additionally, the number and location of sites that use industrial laundry detergents containing 1,4-dioxane as a are unknown. According to the ESD on Water Based Washing Operations at Industrial and Institutional Laundries, industrial laundry facilities operate over a range of 20 to 365 days per year (OECD, 2011b). EPA modeled the 1,4-dioxane use rate for a generic site using the ESD on Water Based Washing Operations at Industrial and Institutional Laundries to estimate releases, resulting in 50th and 95th percentile 1,4-dioxane use rates of approximately 7×10^{-5} and 0.0013 kg/site-day in both industrial power and liquid laundry detergents, respectively (OECD, 2011b). For additional information on the modeling and associated input parameters used to estimate the daily use rate, refer to Appendix E.12.

Institutional Laundries: Institutional laundries are typically located within a hospital, nursing home, hotel, or other institutional facility (OECD, 2011b). EPA did not find specific container information for 1,4-dioxane in institutional laundry detergents; however, the ESD on Water Based Washing Operations at Industrial and Institutional Laundries indicates that institutional laundry detergents are typically transported as a liquid or powder in 5-gallon pails (OECD, 2011b). 1,4-Dioxane can be present in institutional laundry detergents at concentrations from 0.05 to 14 ppm (U.S. EPA, 2020c).

The volume of 1,4-dioxane present in institutional laundry detergents is unknown. Additionally, the number and location of sites that use institutional laundry detergents containing 1,4-dioxane as a contaminant are unknown. According to the ESD on Water Based Washing Operations at Industrial and Institutional Laundries, institutional laundry facilities operate over a range of 250 to 365 days per year (OECD, 2011b). EPA modeled the 1,4-dioxane use rate for a generic site using the ESD on Water Based Washing Operations at Industrial and Institutional Laundries to estimate releases, resulting in 50th and 95th percentile 1,4-dioxane use rates of approximately 2.2×10^{-5} and 1×10^{-4} kg/site-day in power detergents and 3.4×10^{-5} and 0.0014 kg/site-day in liquid detergents, respectively (OECD, 2011b). For additional information on the modeling and associated input parameters used to estimate the daily use rate, refer to Appendix E.12.

Worker Activities

Workers are potentially exposed to 1,4-dioxane in laundry detergents during transfer operations, container cleaning, handling damp laundry, and other operational activities, which are expected for both industrial and institutional laundries (OECD, 2011b). These activities are all potential sources of worker exposure through dermal contact or inhalation exposure to solid or liquid chemicals.

During the use of laundry detergents, workers may be exposed during manual loading of solid or liquid detergent chemicals into the washing machine (OECD, 2011b). Automatic liquid injection systems may be employed which reduce worker exposure; however, workers may still be exposed when connecting transfer lines or transferring the liquid chemicals from the transport container to storage tanks. Solid detergents are less frequently used than liquid detergents due to their increased risk of exposure from dusts and inability to be automatically loaded into machines (OECD, 2011b).

The 2011 ESD on The Chemicals Used in Water-Based Washing Operations at Industrial and Institutional Laundries indicates that PPE may be required in both industrial and institutional laundry settings in the case of handling substances that may be corrosive or produce dust or vapors that can be inhaled, or if the worker's hands are constantly immersed in water or wash solutions containing detergents (OECD, 2011b). However, these situations are not typical for most activities at industrial and institutional laundries.

ONUs include employees that work at the sites where laundry detergent is used, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and are not expected to have dermal exposures through contact with liquids. ONUs for this scenario include supervisors, managers, and other employees that may be in the laundry areas but do not perform tasks that result in the same level of exposures as those workers that engage in tasks related to the use of laundry detergents.

Number of Potentially Exposed Workers and ONUs

For industrial laundries, EPA used U.S. Census and BLS data for the NAICS code 812330, Linen and Uniform Supply, to estimate a total of 2,453 sites, 27 workers per site, and three ONUs per site (U.S. BLS, 2016). EPA estimated the number of institutional laundries based on industry information as described in the ESD on Water Based Washing operations at Industrial and Institutional Laundries, resulting in a total of 95,533 sites and six workers per site. The number of ONUs per institutional laundry site is unknown (OECD, 2011b). For additional information regarding the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

1,4-Dioxane is found in laundry detergents due to its presence as an unintentional byproduct in certain ethoxylated substances that may be used as formulation components (U.S. EPA, 2020c). The information and data quality evaluation to assess occupational exposures during use of laundry detergent is listed in Table_Apx F-15 and described below.

Table_Apx F-15. Laundry Detergent Worker Exposure Data Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source/ Reference
Unloading detergent into machines, cleaning empty detergent containers, laundry operations	Input parameters for Monte Carlo modeling	N/A	Medium[a]	(OECD, 2011b)

[a] This is the rating for the underlying data used in the model, and not the Monte Carlo model itself.

EPA did not find relevant inhalation monitoring data for the use of laundry detergent. Therefore, EPA modeled 1,4-dioxane air concentrations using a Monte Carlo modeling approach, which is described in Appendix F.8. This modeling approach utilizes the *EPA/OPPT Penetration Model*, *EPA/OPPT Mass Transfer Coefficient Model*, *EPA Mass Balance Inhalation Model*, and *Generic Model for Central Tendency and High-End Inhalation Exposure to Total and Respirable Particulates Not Otherwise Regulated (PNOR)*, with variation in input parameters for mass fraction of 1,4-dioxane in detergent, ventilation rate, mixing factor, and total/respirable PNOR concentrations based on available data. To compile a full-shift estimate, EPA combined exposure estimates for all activities, ensuring that the total exposure duration for all activities combined did not exceed the shift length, which could be 8, 10, or 12 hours per the OECD ESD on the Chemicals Used in Water Based Washing Operations at Industrial and Institutional Laundries (OECD, 2011b). Container unloading and cleaning duration was calculated by taking the number of containers unloaded and dividing by fill rate and operating days. Laundry operation duration was calculated by taking the total work shift duration and subtracting the duration of container unloading and cleaning. Table_Apx F-16 and Table_Apx F-17 present the modeled 8-hour, 10-hour, and 12-hour TWA exposures for industrial and institutional laundries, respectively.

Table_Apx F-16. Modeled Occupational Inhalation Exposures for Industrial Laundries

Statistic	1,4-Dioxane Exposure, 8h-TWA Vapor (mg/m^3)	1,4-Dioxane Exposure, 10h-TWA Vapor (mg/m^3)	1,4-Dioxane Exposure, 12h-TWA Vapor (mg/m^3)	1,4-Dioxane Exposure, 8h-TWA Total Particulate (mg/m^3)	1,4-Dioxane Exposure, 8h-TWA Respirable Particulate (mg/m^3)
Maximum	4.07E−03	5.03E−03	6.04E−03	2.10E−04	7.00E−05
99th Percentile	2.61E−03	3.21E−03	3.83E−03	2.08E−04	6.93E−05
95th Percentile	1.87E−03	2.30E−03	2.75E−03	2.00E−04	6.65E−05
50th Percentile	5.15E−04	6.09E−04	7.10E−04	1.05E−04	3.51E−05
5th Percentile	5.02E−05	5.84E−05	6.43E−05	1.12E−05	3.74E−06
Minimum	7.93E−07	9.07E−07	9.79E−07	7.51E−07	2.50E−07
Mean	6.79E−04	8.22E−04	9.67E−04	1.05E−04	3.51E−05

Table_Apx F-17. Modeled Occupational Inhalation Exposures for Institutional Laundries

Statistic	1,4-Dioxane Exposure, 8h-TWA Vapor (mg/m^3)	1,4-Dioxane Exposure, 10h-TWA Vapor (mg/m^3)	1,4-Dioxane Exposure, 12h-TWA Vapor (mg/m^3)	1,4-Dioxane Exposure, 8h-TWA Total Particulate (mg/m^3)	1,4-Dioxane Exposure, 8h-TWA Respirable Particulate (mg/m^3)
Maximum	3.73E−03	4.84E−03	5.97E−03	2.10E−04	7.00E−05
99th Percentile	2.04E−03	2.52E−03	3.04E−03	2.08E−04	6.93E−05
95th Percentile	1.45E−03	1.78E−03	2.12E−03	2.00E−04	6.65E−05
50th Percentile	4.10E−04	4.85E−04	5.60E−04	1.05E−04	3.51E−05
5th Percentile	3.94E−05	4.62E−05	5.18E−05	1.12E−05	3.74E−06
Minimum	5.27E−07	8.58E−07	7.02E−07	7.52E−07	2.51E−07
Mean	5.34E−04	6.46E−04	7.61E−04	1.05E−04	3.51E−05

EPA used the 50th and 95th percentile modeled 8-hour TWA exposures from Table_Apx F-16 and Table_Apx F-17 to calculate the central tendency and high-end ADC and LADC for laundry detergents, based on the timeframe for the available health hazard data. The calculated values are summarized in

Table_Apx F-18 and Table_Apx F-18, respectively. Equations for calculating ADC and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Exposure data for ONUs were not available. The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-18. Inhalation Exposures of Workers for the Use of Laundry Detergent in Industrial Laundries Based on Modeling

Exposure Type	Physical Form	Central Tendency (50th Percentile) (mg/m^3) [a]	High-end (95th Percentile) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	Liquid detergents: vapor	5.2E−04	1.9E−03
	Solid detergents: total particulate	1.1E−04	2.0E−04
	Solid detergents: respirable particulate	3.5E−05	6.7E−05
Average Daily Concentration (ADC)	Liquid detergents: vapor	4.96E−04	1.80E−03
	Solid detergents: total particulate	1.01E−04	1.92E−04
	Solid detergents: respirable particulate	3.38E−05	6.40E−05
Lifetime Average Daily Concentration (LADC)	Liquid detergents: vapor	1.97E−04	9.22E−04
	Solid detergents: total particulate	4.03E−05	9.84E−05
	Solid detergents: respirable particulate	1.34E−05	3.28E−05

[a] See Table_Apx F-15 for corresponding references.

Table_Apx F-19. Acute and Chronic Inhalation Exposures of Workers for the Use of Laundry Detergent in Institutional Laundries Based on Modeling

Exposure Type	Physical Form	Central Tendency (50th Percentile) (mg/m^3) [a]	High-End (95th Percentile) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	Liquid detergents: vapor	4.10E−04	1.45E−03
	Solid detergents: total particulate	1.05E−04	2.00E−04
	Solid detergents: respirable particulate	3.51E−05	6.65E−05
Average Daily Concentration (ADC)	Liquid detergents: vapor	3.94E−04	1.39E−03
	Solid detergents: total particulate	1.01E−04	1.92E−04
	Solid detergents: respirable particulate	3.38E−05	6.40E−05
Lifetime Average Daily Concentration (LADC)	Liquid detergents: vapor	1.57E−04	7.14E−04
	Solid detergents: total particulate	4.03E−05	9.84E−04

Exposure Type	Physical Form	Central Tendency (50th Percentile) (mg/m^3) [a]	High-End (95th Percentile) (mg/m^3) [a]
	Solid detergents: respirable particulate	1.34E−05	3.28E−05

[a] See Table_Apx F-15 for corresponding references.

Key Uncertainties

Due to a lack of data specific to 1,4-dioxane for this use, EPA used assumptions and values from the ESD on Water Based Washing Operations at Industrial and Institutional Laundries and EPA models to estimate inhalation exposures during container transfers, container cleaning, and laundry operations (see Appendix F.8). The uncertainties associated with this modeling approach are described in Section 3.1.2.4.

F.4.7 Paint and Floor Lacquer

Process Description

EPA identified 1,4-dioxane present in commercial paints and floor lacquer as an unintentional byproduct in formulation components (U.S. EPA, 2020c). Concentrations of 1,4-dioxane in paints and floor lacquer range from 0.02 to 30 ppm (U.S. EPA, 2020c). Similarly, 1,4-dioxane is present as an unintentional component of automotive refinishing coatings, architectural paints/coatings, and industrial coatings (Franz et al., 2015). Paint and coating formulations are typically transported as a liquid in drums and are loaded into the reservoir of application equipment (OECD, 2009). The application procedure depends on the type of paint or floor lacquer and the type of substrate. The paint or lacquer may be applied to the substrate via spray, brush, or roller application. Following application, the paint or lacquer is allowed to cure or dry. The curing process may involve air drying, baking, or radiation curing, depending on the substrate being painted or coated (OECD, 2009).

The volume of 1,4-dioxane present in paints and floor lacquer is unknown. Information from the CDR indicate that 1,4-dioxane is imported and present in paint and coatings as a formulation component (U.S. EPA, 2020a). Additionally, the number and location sites that use paints and floor lacquer containing 1,4-dioxane are unknown. The ESD on Coating Application via Spray Painting in the Automotive Refinishing Industry (referenced due to identification of 1,4-dioxane in automotive refinishing coatings) indicates a default of 250 days/year of operation (OECD, 2011a). Using the default values from the ESD and the concentration of 1,4-dioxane above (0.02 to 30 ppm), EPA calculates a daily use rate of 1,4-dioxane at an automotive refinishing site of 3.2×10^{-8} to 4.8×10^{-5} kg/site-day.

Worker Activities

Workers are potentially exposed to 1,4-dioxane in paint and floor lacquer formulations during multiple activities, including quality testing of formulations, transferring the formulations into application equipment (if used), applying the formulation to a substrate, and maintenance and cleaning activities (OECD, 2009). These activities are all potential sources of worker exposure through dermal contact to liquid and inhalation of 1,4-dioxane vapors.

During application of paint or floor lacquer, workers may manually apply the formulation with a variety of application techniques, including spray application, brush application, dipping, or rolling (OECD, 2009). All application methods have potential exposure points for workers. Some application methods may be automated, which reduces the potential for worker exposures. For example, if the dip coating apparatus has an enclosed reservoir, this reduces the potential for 1,4-dioxane vapors to escape and become available for worker inhalation and vapor-through-skin exposure (OECD, 2009). The extent of automated application processes and use of open versus closed systems in the various industries that conduct paint or floor lacquer applications is unknown.

A NIOSH evaluation of a small parts and vehicle painting facility revealed that half-face respirators with organic vapor cartridges were available to workers at the identified site (Hills et al., 1989). The workers mainly used brushes for paint application but occasionally used spray gun applicators for brief periods of time. NIOSH suggests implementing a respiratory protection program for the painters; details of which can be found in the NIOSH publication, Guide to Industrial Respiratory Protection, DHHS (NIOSH) publication number 87-116 (NIOSH, 1987). NIOSH also recommends wearing gloves impervious to the paints and solvents to prevent skin contact and avoid possible dermal exposure route (Hills et al., 1989). EPA did not find any additional information regarding PPE used at facilities that apply paints and floor lacquer.

ONUs include employees that work at the sites where paint and floor lacquer is used, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and vapor-through-skin uptake and are not expected to have dermal exposures through contact with liquids. ONUs for this scenario include supervisors, managers, and other employees that may be in the application areas but do not perform tasks that result in the same level of exposures as those workers that engage in tasks related to the use of paint and floor lacquer.

Number of Potentially Exposed Workers and ONUs

EPA used U.S. Census and BLS data for the NAICS code 811121, Automotive Body, Paint, and Interior Repair and Maintenance, to estimate a total of 33,648 sites, 111,511 workers, and 11,050 ONUs, which corresponds to an estimated average of three workers and 0.3 ONUs per site (U.S. BLS, 2016). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

EPA identified 1,4-dioxane present in commercial paints and floor lacquer as an unintentional byproduct in formulation components (U.S. EPA, 2020c). The information and data quality evaluation to assess occupational exposures during use of paints and floor lacquer is listed in Table_Apx F-20 and described below.

Table_Apx F-20. Paint and Floor Lacquer Worker Exposure Data Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source Reference
Applying tape to parts that are not to be painted, painting, touch-ups	PBZ Monitoring	17	High	(Hills et al., 1989)

Table_Apx F-22 shows the 17 discrete inhalation monitoring data points available in published literature for the use of paint and floor lacquer containing 1,4-dioxane (Hills et al., 1989). This data is from a NIOSH study in which PBZ samples were taken at a military vehicle manufacturing site in 1987. The study was conducted in the final processing where approximately 47 workers touch-up vehicles and perform quality checks. The worker activities captured in this sampling include taping vehicles prior to painting, painting vehicles, and performing paint touch-ups. The study does not identify where 1,4-dioxane is present at the site, which is a limitation of this data.

The NIOSH report provided 17 PBZ sample results, three of which are 8-hour TWAs and the remaining 14 of which were taken over a shorter period of time. Many of these, however, are still close to a full shift duration of 8 hours. EPA converted these 14 samples into 8-hour TWAs by assuming no exposure for the remainder of the eight hours. EPA made this assumption because the site analyzed in the study was not strictly a vehicle painting site. As such, workers may spend time doing other jobs that did not involve formulations containing 1,4-dioxane. Therefore, EPA assumed that sampling occurred for the duration of the employee's painting tasks where there was potential exposure to 1,4-dioxane.

Four of the 17 samples were non-detect for 1,4-dioxane. The study indicated that the LOD for all samples was 0.1 mg/m^3 of 1,4-dioxane. For the non-detect samples, EPA used the LOD divided by two for subsequent central tendency and high-end calculations. EPA used this method for approximating a concentration for non-detect samples because the geometric standard deviation of the dataset is greater than three (U.S. EPA, 1994a).

EPA used the 8-hour TWA air concentration measurements and LOD/2 (for the non-detects) to calculate central tendency (50th percentile) and high-end exposures (95th percentile). EPA used these values to calculate the ADC and LADC. The calculated values are summarized in Table_Apx F-21. Equations for calculating ADC and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Exposure data for ONUs were not available. The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-21. Inhalation Exposures of Workers for the Use of Paint and Floor Lacquer Based on Monitoring Data

Exposure Type	Central Tendency (50th Percentile) (mg/m^3) [a]	High-end (95th Percentile) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	0.210	1.20
Average Daily Concentration (ADC)	0.202	1.15
Lifetime Average Daily Concentration (LADC)	0.080	0.592

[a] See Table_Apx F-20 for corresponding references.

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Row #	Type of Sample	Worker Activity or Sample Location	Number of Samples	Sample Date	Sample Time	1,4-Dioxane Exposure Concentration (mg/m^3)	EPA Determined 8-hour TWA (mg/m^3)	Source	Overall Data Quality Determination
1	Personal	N/A	1	10/27/1987	480	ND (LOD = 0.1 mg/m^3)	0.05	(Hills et al., 1989)	High
2	Personal	N/A	1	10/27/1987	480	ND (LOD = 0.1 mg/m^3)	0.05	(Hills et al., 1989)	High
3	Personal	N/A	1	10/27/1987	480	ND (LOD = 0.1 mg/m^3)	0.05	(Hills et al., 1989)	High
4	Personal	N/A	1	10/27/1987	463	0.1	0.10	(Hills et al., 1989)	High
5	Personal	N/A	1	10/27/1987	457	0.2	0.19	(Hills et al., 1989)	High
6	Personal	N/A	1	10/27/1987	456	0.5	0.48	(Hills et al., 1989)	High
7	Personal	N/A	1	10/27/1987	439	0.1	0.09	(Hills et al., 1989)	High
8	Personal	N/A	1	10/27/1987	441	0.7	0.64	(Hills et al., 1989)	High
9	Personal	N/A	1	10/27/1987	428	1.3	1.7	(Hills et al., 1989)	High
10	Personal	N/A	1	10/27/1987	251	1.7	0.89	(Hills et al., 1989)	High
11	Personal	N/A	1	10/27/1987	148	0.7	0.22	(Hills et al., 1989)	High
12	Personal	N/A	1	10/27/1987	456	1.3	1.24	(Hills et al., 1989)	High

Row #	Type of Sample	Worker Activity or Sample Location	Number of Samples	Sample Date	Sample Time	1,4-Dioxane Exposure Concentration (mg/m^3)	EPA Determined 8-hour TWA (mg/m^3)	Source	Overall Data Quality Determination
13	Personal	N/A	1	10/27/1987	229	0.4	0.19	(Hills et al., 1989)	High
14	Personal	N/A	1	10/27/1987	145	0.7	0.21	(Hills et al., 1989)	High
15	Personal	N/A	1	10/27/1987	347	1.0	0.72	(Hills et al., 1989)	High
16	Personal	N/A	1	10/27/1987	410	1.4	1.2	(Hills et al., 1989)	High
17	Personal	N/A	1	10/27/1987	400	ND (LOD = 0.1 mg/m^3)	0.040	(Hills et al., 1989)	High

ND = non-detect for 1,4-dioxane; LOD = limit of detection; TWA = time-weighted average

Key Uncertainties

As discussed above, EPA translated short-term samples from the NIOSH HHE report (Hills et al., 1989) into 8-hour TWAs by assuming no exposure for the remainder of the eight hours after the sampling duration. This assumption may result in underestimation of exposures if workers perform additional activities that may result in exposures to 1,4-dioxane that were not captured in the monitoring performed in the NIOSH HHE report. However, the data set did include full-shift monitoring, which EPA included in this analysis. Additionally, these data are from one facility, and it is unclear how representative the data are for all sites and all workers across the United States. The monitoring performed for the NIOSH HHE was completed in the 1980s; therefore, the age of the monitoring data can also introduce uncertainty.

As discussed above, EPA used half the detection limit for the non-detect values in the central tendency and high-end exposure calculations. Due to the high number of non-detects (13 of the 17 TWAs were non-detect), this method may result in bias (U.S. EPA, 1994a). Additional uncertainties are listed in Section 3.1.2.4.

F.4.8 Spray Foam Application

Process Description

There are three main types of spray polyurethane foam (SPF): two-component high-pressure, two-component low-pressure, and one-component foam (OCF) (U.S. EPA, 2017a). The low-pressure and OCF types are available for DIY-use, but the high-pressure type is only available for professional use. A safety data sheet (SDS) identified in the *Preliminary Information on Manufacturing, Processing, Distribution, Use, and Disposal: 1,4-Dioxane* indicate that 1,4-dioxane is present in open- and closed-cell SPFs, which are subsets of two-component high-pressure SPFs (U.S. EPA, 2017a, b). While one SDS has been identified where 1,4-dioxane was listed as an ingredient, it could also be a byproduct and the concentration could vary by the type of SPF.

High-pressure SPF is used for larger insulation applications, as an air sealant in hybrid insulations, and in roofing applications. The components are typically stored in 55-gallon drums. The operator pumps both components (sides A and B) through heated tubes from the supply tanks into a nozzle. 1,4-Dioxane is a component in side B with concentrations typically around 0.1 percent U.S. EPA (U.S. EPA, 2018a, 2017a). Sides A and B begin to react in the nozzle and are sprayed at elevated pressures and temperatures (>150 °F and 1,200 psi). The formulation may be applied via hand-held spray gun or automated spray system. Closed-cell foam could be applied in layers. As the foam cures, it expands up to 120 times its original size. After curing, the foam may be trimmed or cut. Trimmings and waste foam are collected and disposed.

The volume of 1,4-dioxane present in spray polyurethane foams is unknown. In 2008, U.S. production of two-component spray foams reached 365 million in 2008 (U.S. EPA, 2018a). The GS on Application of Spray Foam Insulation indicates a default of 260 days/year of operation (U.S. EPA, 2018a).

Worker Activities

Workers are potentially exposed to 1,4-dioxane during the application of spray polyurethane foam while unloading SPF chemicals into spray rig equipment, transport container cleaning, SPF application, and trimming of the applied and hardened SPF insulation (U.S. EPA, 2018a). These activities are all potential sources of worker exposure through dermal contact to liquid and the inhalation of mist or vapors. Exposure during equipment cleaning is not expected, as the spray equipment is a closed system that is flushed with solvent; workers do not come into contact with the inside of the equipment.

During application of spray foam insulation, workers may manually apply the formulation via hand-held spray gun or employ an automated spray system (U.S. EPA, 2018a). Both types of application are potential exposure points for workers. Typically, the main engineering controls used by SPF applicators are containment and ventilation. A containment system is often made up of plastic sheeting or cardboard secured to walls to isolate the work zone, thus reducing the potential for airborne chemicals to enter the building ventilation systems. Ventilation systems, including active exhaust and air supply systems, are typically used to avoid accumulation of chemical vapors and particulate emissions near the application area (U.S. EPA, 2018a).

According to the GS on Application of Spray Polyurethane Foam Insulation, workers at sites that apply SPF insulation are expected to wear proper chemical-specific personal protective equipment (U.S. EPA, 2018a). Workers may wear chemical-resistant gloves, protective clothing (*e.g.*, long sleeves, body suit, coveralls), eye and face protection (*e.g.*, safety glasses, chemical goggles), and respiratory protection. Additionally, an SPF sprayer may wear a full-face, air-supplied respirator with chemical protective coveralls and chemical protective gloves (U.S. EPA, 2018a). The appropriate PPE may vary for the specific application.

ONUs include employees that work at the sites where spray polyurethane foam is applied, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and vapor-through-skin uptake and are not expected to have dermal exposures through contact with liquids. ONUs for this scenario include supervisors, managers, and other employees that may be in the application areas but do not perform tasks that result in the same level of exposures as those workers that engage in the tasks related to the use of spray polyurethane foam.

Number of Potentially Exposed Workers and ONUs

EPA estimated the number of potentially exposed workers and ONUs in Appendix G.6.7 of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

EPA estimated occupational inhalation exposures during the use of spray polyurethane foam containing 1,4-dioxane in Section 2.4.1.1.9 of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c). EPA did not conduct additional analyses of occupational inhalation exposures for spray polyurethane foam for this supplemental risk evaluation. Refer to the December 2020 *Final Risk Evaluation for 1,4-Dioxane* for additional details.

Key Uncertainties

Key uncertainties are listed in Section 2.4.1.1.9 of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

F.4.9 Polyethylene Terephthalate Byproduct

Process Description

1,4-Dioxane has been identified as a byproduct in the manufacture of PET plastics (U.S. EPA, 2017c). EPA does not have information on the byproduct concentration of 1,4-dioxane in PET. PET is produced by the esterification of terephthalic acid to form bishydroxyethyl terephthalate (BHET) (Forkner et al., 2004). BHET polymerizes in a transesterification reaction catalyzed by antimony oxide to form PET (Forkner et al., 2004).

In 2014, 20.6 million metric tons of PET were used in the United States (McDaniel and DesLauriers, 2015). The volume of 1,4-dioxane produced as a byproduct of PET manufacturing is unknown. Due to lack of information, EPA does not present annual or daily site throughputs. EPA assumes facilities that produce 1,4-dioxane as a byproduct during PET manufacturing operate 5 days/week, 50 weeks/year or 250 days/year.

Worker Activities

Workers are potentially exposed to 1,4-dioxane during transferring of produced PET and equipment cleaning (U.S. EPA, 2021b). These activities are potential sources of worker exposure through dermal contact to liquid and inhalation of volatile chemical vapors.

According to the GS on Use of Additives in Plastic Compounding, workers may wear suitable gloves, hearing protection, and eye protection (U.S. EPA, 2021b). Facilities may use forced ventilation techniques to reduce worker exposure to vapors. Local exhaust ventilation may be used in areas where there is potential for the formation of particulates or vapors (U.S. EPA, 2021b). EPA did not find information that indicates the extent that and worker PPE is used at facilities that manufacture PET in the United States.

ONUs include employees that work at the sites where PET is manufactured, but they do not directly handle the chemical and are therefore expected to have lower inhalation exposures and are not expected to have dermal exposures through contact with liquids or solids. ONUs for this scenario include supervisors, managers, and other employees that may be in the manufacturing areas but do not perform tasks that result in the same level of exposures as those workers that engage in tasks related to the manufacture of PET.

Number of Potentially Exposed Workers and ONUs

To estimate the number of workers, EPA used U.S. Census and BLS data for NAICS codes 325211 and 326113. EPA estimated a total of 1,695 sites, 43,528 workers, and 17,195 ONUs (U.S. BLS, 2016). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

1,4-Dioxane has been identified as a byproduct in the manufacture of polyethylene terephthalate (PET) (U.S. EPA, 2017c). Occupational exposure to 1,4-dioxane in PET was determined using monitoring data provided by Chemical Exposure Health Data (OSHA, 2020). The information and data quality evaluation to assess occupational exposures during manufacture of PET is listed in Table_Apx F-23 and summarized below.

Table_Apx F-23. Polyethylene Terephthalate (PET) Byproduct Worker Exposure Data Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source Reference
Unknown	PBZ Monitoring	11	High	(OSHA, 2020)

EPA assessed occupational inhalation exposures for this OES using OSHA's CEHD (OSHA, 2020). For detailed information on where/how CEHD was obtained and mapped to OES, see Appendix F.4.1. For this OES, monitoring data were available in CEHD from five sites with SIC codes 3089 (All Other Plastics Product Manufacturing), 2653 (Corrugated and Solid Fiber Box Manufacturing), 3052 (Rubber and Plastics Hoses and Belting Manufacturing), and 3069 (All Other Rubber Product Manufacturing). All sites were determined to be manufacturers of plastic products (foams, packaging, etc.).

Table_Apx F-25 shows the 11 discrete inhalation monitoring data points from CEHD for this OES, all of which are PBZ samples. The data are from 1985 to 1994; however, EPA did not find more recent data. CEHD does not include information on the worker activities included in the PBZ sampling, therefore EPA's assessment assumed that all 11 samples are relevant to this assessment. Furthermore, it is uncertain the extent to which all potential worker activities are represented in these data.

As discussed in Appendix F.4.1, EPA combined sample results with the same inspection number and sampling number to attempt to construct a full-shift exposure concentration. Sample durations ranged from 74 to 477 minutes. For the samples with detected values, EPA translated the sample results into 8-hour TWA concentrations by assuming that the exposure concentration is zero for the time remaining in the 8-hour durations. Where non-detect values were included in the dataset, EPA calculated the LOD for each sample and used the LOD/2 for subsequent central tendency (50th percentile) and high-end (95th percentile) calculations, as discussed in Appendix F.4.1.

EPA then used the air concentrations and LOD/2 as shown in Table_Apx F-25 to calculate full shift (8-hour TWA) central tendency (50th percentile) and high-end (95th percentile) inhalation exposures. EPA used these values to calculate the ADC and LADC. The calculated values are summarized in Table_Apx F-24. Equations for calculating ADC and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-24. Inhalation Exposures of Workers for PET Byproduct Based on Monitoring Data

Exposure Type	Central Tendency (50th Percentile) (mg/m^3) [a]	High-End (95th Percentile) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	4.7	47
Average Daily Concentration (ADC)	4.52	45.2
Lifetime Average Daily Concentration (LADC)	1.80	23.2

[a] See Table_Apx F-23 for corresponding references.

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Row #	Type of Sample	Worker Activity or Sample Location	Number of Samples	Sample Date	Sample Time (min)	1,4-Dioxane Exposure Concentration (mg/m³)	EPA Determined 8-hour TWA (mg/m³)	Source	Overall Data Quality Determination
1	Personal	N/A	1	4/23/1985	270	2.9	5.879	(OSHA, 2020)	High
2	Personal	N/A	1	4/23/1985	270	ND (LOD = 0.74 mg/m³)	0.2083	(OSHA, 2020)	High
3	Personal	N/A	1	4/23/1985	270	3.1	6.284	(OSHA, 2020)	High
4	Personal	N/A	1	11/20/1990	74	0.2513	0.1396	(OSHA, 2020)	High
5	Personal	N/A	1	3/15/1991	325	23.77	57.99	(OSHA, 2020)	High
6	Personal	N/A	1	3/15/1991	150	14.29	16.10	(OSHA, 2020)	High
7	Personal	N/A	1	8/12/1993	457	10.72	36.78	(OSHA, 2020)	High
8	Personal	N/A	1	8/12/1993	457	1.36	4.666	(OSHA, 2020)	High
9	Personal	N/A	1	1/10/1994	477	ND (LOD = 0.59 mg/m³)	0.2919	(OSHA, 2020)	High
10	Personal	N/A	1	1/10/1994	455	ND (LOD = 0.34 mg/m³)	0.1593	(OSHA, 2020)	High
11	Personal	N/A	1	1/10/1994	461	ND (LOD = 0.39 mg/m³)	0.1864	(OSHA, 2020)	High
ND = non-detect for 1,4-dioxane; LOD = limit of detection; TWA = time-weighted average									

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Key Uncertainties

The OSHA CEHD monitoring data does not include process information or worker activities; therefore, there is uncertainty as to which worker activities these data cover and whether all potential workers activities are represented in this data. Additionally, these data are from five facilities, and it is unclear how representative the data are for all sites and all workers across the United States. The OSHA CEHD used for this assessment is from the 1980s and 1990s. Therefore, the age of the monitoring data may also introduce uncertainty. EPA used half the detection limit for the non-detect values in the central tendency and high-end exposure calculations. This introduces uncertainty into the assessment because the true value of 1,4-dioxane is unknown (although expected to be between zero and the level of detection). Additional uncertainties are listed in Section 3.1.2.4.

F.4.10 Ethoxylation Process Byproduct

Process Description

1,4-Dioxane may be formed as a byproduct of reactions based on condensing ethylene oxide or ethylene glycol during manufacture of detergents, shampoos, surfactants, some food additives, and certain pharmaceuticals (HHS, 2016). In cosmetic ethoxylated raw materials and ethoxylated alkyl sulfates, 1,4-dioxane has been detected at concentrations of 0.48 to 1410 ppm (U.S. EPA, 2020c; Saraji and Shirvani, 2017; Davarani et al., 2012; Black et al., 2001). Polyethoxylated raw materials are widely used in cosmetic products as emulsifiers, foaming agents, and dispersants (Black et al., 2001). They are produced by polymerizing ethylene oxide, usually with a fatty alcohol, to form polyethoxylated alcohols which may be used to synthesize other products such as sulfated surface-active agent. During the ethoxylation process, 1,4-dioxane can be formed as a byproduct by the dimerization of ethylene oxide (Black et al., 2001).

The volume of 1,4-dioxane produced as a byproduct of ethoxylation reactions is unknown. Due to lack of information, EPA does not present annual or daily site throughputs. EPA assumes facilities that produce 1,4-dioxane as a byproduct during ethoxylation reactions operate 5 days/week, 50 weeks/year, or 250 days/year.

Worker Activities

Workers are potentially exposed to 1,4-dioxane during ethoxylation processes through transferring ethoxylation product and equipment cleaning. All of these activities are potential sources of worker exposure through dermal contact to liquid and inhalation of volatile chemical vapors.

Suitable PPE may be worn in accordance with safety data sheets. EPA did not find information that indicates the extent that and worker PPE is used at facilities that conduct ethoxylation processes in the United States.

ONUs include employees that work at the sites where ethoxylation processes occur, but they do not directly handle the chemicals and are therefore expected to have lower inhalation exposures and are not expected to have dermal exposures through contact with liquids. ONUs for this scenario include supervisors, managers, and other employees that may be in the process areas but do not perform tasks that result in the same level of exposures as those workers that engage in tasks related to ethoxylation.

Number of Potentially Exposed Workers and ONUs

To estimate the number of workers, EPA used U.S. Census and BLS data for the following NAICS codes: 325110, 325199, 325611, 325613, and 325998. EPA estimated a total of 2,730 sites, 64,926 workers, and 24,835 ONUs (U.S. BLS, 2016). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

1,4-Dioxane may be formed as a byproduct of reactions based on condensing ethylene oxide or ethylene glycol during manufacture of detergents, shampoos, surfactants, some food additives, and certain pharmaceuticals (HHS, 2016). Occupational exposure to 1,4-dioxane in ethoxylation process byproduct was determined using monitoring data provided by Chemical Exposure Health Data (OSHA, 2020). The information and data quality evaluation to assess occupational exposures during the ethoxylation process is listed in Table_Apx F-26 and described below.

Table_Apx F-26. Ethoxylation Process Byproduct Worker Exposure Data Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source Reference
Unknown	PBZ Monitoring	1	High	(OSHA, 2020)

EPA assessed occupational inhalation exposures for this OES using OSHA's CEHD (OSHA, 2020). For detailed information on where/how CEHD was obtained and mapped to OES, see Appendix F.4.1. For this OES, monitoring data were available in CEHD from one site with SIC code 2841 (Soap and Other Detergent Manufacturing). This site was determined to be a detergent manufacturer.

Table_Apx F-28 shows the one inhalation monitoring data point from CEHD for this OES, which is a PBZ sample taken in 2000. EPA did not find additional or more recent data for this OES. CEHD does not include information on worker activities, so EPA's assessment assumes that that the sample is relevant to this assessment. However, it is uncertain the extent to which all potential worker activities are represented in this sample point.

This one sample point is comprised of multiple short-term samples with the same inspection number and sampling number. EPA's rationale and process for combining samples with the same inspection and sampling numbers is described in Appendix F.4.1. The combined sample duration was 381 minutes, which is close to the full-shift duration of 8 hours (480 minutes). EPA translated this into 8-hour TWA concentration by assuming that the exposure concentration is zero for the time remaining in the 8-hour shift. EPA used this value to calculate the ADC and LADC. The calculated values are summarized in Table_Apx F-27. Equations for calculating ADC and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-27. Inhalation Exposures of Workers for the Ethoxylation Process Byproduct Based on Monitoring Data

Exposure Type	Exposure Estimate (mg/m^3)[a]
8-hour TWA Exposure Concentrations	1.2 (single value)
Average Daily Concentration (ADC)	1.15 (single value)
Lifetime Average Daily Concentration (LADC)	0.459 (single value)

[a] See Table_Apx F-26 for corresponding references.

Table_Apx F-28. Occupational Inhalation Monitoring Data for Ethoxylation Process Byproduct

Row #	Type of Sample	Worker Activity or Sample Location	Number of Samples	Sample Date	Sample Time (min)	1,4-Dioxane Exposure Concentration (mg/m^3)	EPA Determined 8-hour TWA (mg/m^3)	Source	Overall Data Quality Determination
1	Personal	N/A	1	6/16/2000	381	0.4359	1.247	(OSHA, 2020)	High
TWA = Time-weighted average									

Key Uncertainties

The OSHA CEHD monitoring data does not include process information or worker activities; therefore, there is uncertainty as to which worker activities these data cover and whether all potential workers activities are represented in this data. Additionally, the OSHA CEHD only include one 8-hour TWA from one facility. Therefore, EPA cannot determine the statistical representativeness of this data point (*e.g.*, high-end, central tendency) towards potential exposures from this COU. Further, it is unclear how representative the data are for all sites and all workers across the United States. The OSHA CEHD point used for this assessment is from the year 2000. Therefore, the age of the monitoring data can also introduce uncertainty. Additional uncertainties are listed in Section 3.1.2.4.

F.4.11 Hydraulic Fracturing

Process Description

Facilities have self-reported to FracFocus 3.0 that 1,4-dioxane is present in hydraulic fracturing fluid additives, as scale inhibitors, additives, biocides, friction reducers, and surfactants (GWPC and IOGCC, 2022). EPA also expects that 1,4-dioxane is present as an unintentional component in hydraulic fracturing fluids, due to its presence as a byproduct in ethoxylated substances. According to the FracFocus 3.0 database, 1,4 dioxane is present in weight fractions ranging from 2.3×10^{-11} to 0.05 within hydraulic fracturing additives and 1.00×10^{-12} to 4.30×10^{-6} in hydraulic fracturing fluids (GWPC and IOGCC, 2022).

Hydraulic fracturing stimulates an existing oil or gas well by injecting a pressurized fluid containing chemical additives into the well (U.S. EPA, 2022d). EPA did not find specific container information for 1,4-dioxane in hydraulic fracturing; however, the Draft ESD on Hydraulic Fracturing indicates that hydraulic fracturing fluids are typically transported as a liquid in totes, drums, or bulk containers. Hydraulic fracturing fluid formulations are charged to a temporary storage tank, or they may be charged to a mixing tank with other additives to formulate the final fracturing fluid that is injected into the well (U.S. EPA, 2022d).

Multiple types of wastewaters are created by hydraulic fracturing: flowback water, produced water, and naturally occurring wastewater.

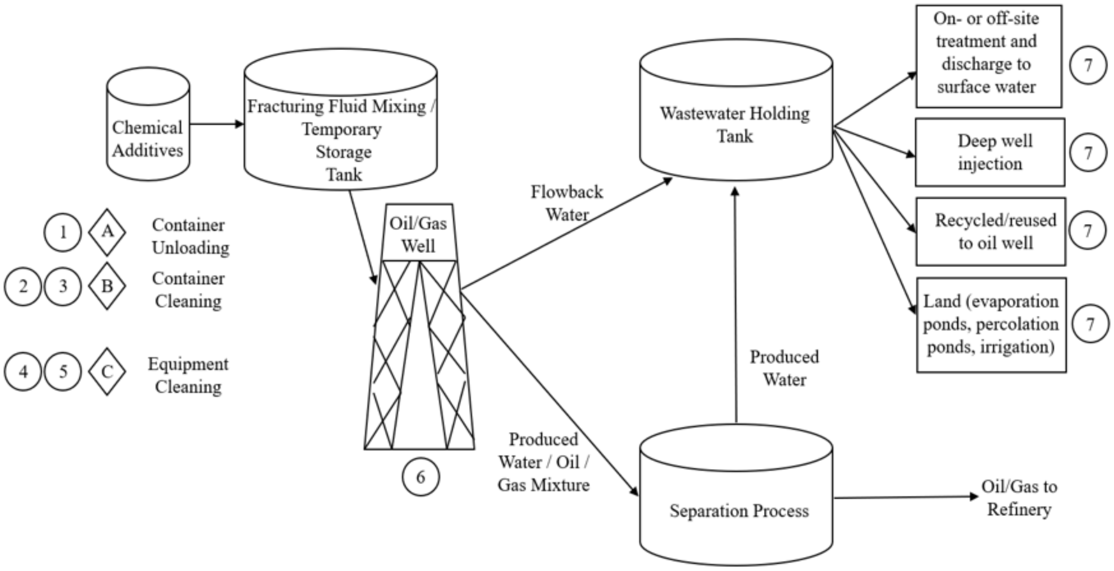
After formulation, the hydraulic fracturing fluid is pumped into a wellbore where it cracks and permeates the surrounding rock (U.S. EPA, 2022d). A portion of the fracturing fluid, including any chemical additives such as 1,4-dioxane, may remain in the underground shale formation. The remaining fluid will return to the surface as *flowback water* that flows back to the surface from the well. Flowback water is the first wastewater to return to the surface after hydraulic fracturing (U.S. EPA, 2022d).

Naturally Occurring Water: exists in the rock formation prior to hydraulic fracturing. Initially flowback water is mostly fracturing fluid, which includes 1,4-dioxane. However, over time, it becomes primarily composed of naturally existing water from the rock formation (U.S. EPA, 2022d).

Produced Water: is defined as “water trapped in underground formations that is brought to the surface along with oil or gas” (U.S. EPA, 2022d). Produced water returns to the surface of the well after flowback water. Produced water may contain many constituents, water and compounds from the rock formation, oil or gas from the rock formation, and smaller portions of hydraulic fracturing fluid including 1,4-dioxane (U.S. EPA, 2022d).

Wastewater containing chemical additives such as 1,4-dioxane is stored and accumulated at the surface for eventual reuse or disposal (U.S. EPA, 2022d). Typical storage facilities include open air impoundments and closed containers. This wastewater is collected and may be taken to disposal wells, recyclers, wastewater treatment plants (on- or off-site), or in some cases the water may be left in pits to evaporate or infiltrate (U.S. EPA, 2022d).

FracFocus 3.0 reports 411 sites that utilize 1,4-dioxane in hydraulic fracturing fluid. These sites are located throughout the United States (GWPC and IOGCC, 2022). FracFocus 3.0 also reports that a typical number of operating days per year is 1 to 72 days/year (GWPC and IOGCC, 2022). EPA modeled the 1,4-dioxane use rate for a generic site using data from FracFocus 3.0 and the Draft ESD on Hydraulic Fracturing to estimate releases, resulting in a 50th and 95th percentile 1,4-dioxane use rate of 0.3 and 5.18 kg/site-day, respectively. A flow diagram including release and exposure points from the Draft ESD on Hydraulic Fracturing is presented in Figure_Apx F-3 (U.S. EPA, 2022d). For additional information on the modeling and associated input parameters used to estimate the daily use rate, refer to Appendix E.13.



Environmental Releases:

1. Releases to fugitive air from unloading volatile chemicals
2. Release to uncertain media (water [on- or off-site treatment then discharge to surface water], incineration or landfill) from container residuals
3. Releases to fugitive air during container cleaning
4. Release to uncertain media (water [on- or off-site treatment then discharge to surface water], incineration or landfill) from equipment cleaning residuals
5. Releases to fugitive air during equipment cleaning
6. Release to deep well injection from fracturing fluid that remains underground
7. Flowback and produced wastewater release to recycle/reuse (48%), deep well injection (43%), on- or off-site treatment and discharge to surface water (6%), or land (3%)

Occupational Exposures:

A. Inhalation and dermal exposure to liquid raw materials during container unloading
B. Inhalation and dermal exposure to liquid raw materials/product during equipment cleaning
C. Inhalation and dermal exposure to liquid raw materials during container cleaning

Figure_Apx F-3. Environmental Release and Occupational Exposure Points during Hydraulic Fracturing

Worker Activities

Workers are potentially exposed to 1,4-dioxane during multiple activities involved in hydraulic fracturing operations, including container unloading and transferring, container cleaning, and equipment cleaning (U.S. EPA, 2022d). These activities are all potential sources of worker exposure through dermal contact to liquid and inhalation of volatile chemical vapors.

The ESD on the Use of Chemicals in Hydraulic Fracturing indicates that workers may connect transfer lines to pump chemical additives directly from transport containers, or manually unload chemicals from transport containers into mixing tanks or injection system (U.S. EPA, 2022d). Dermal exposure may occur during both automated and manual unloading activities. Container cleaning and equipment cleaning are typically manual activities. (U.S. EPA, 2022d).

The ESD on Chemicals Used in Hydraulic Fracturing suggests that workers consult the Safety Data Sheet (SDS) which may identify specific hazards and recommend the appropriate personal protective equipment (PPE) (U.S. EPA, 2022d). EPA did not find information that indicates the extent that engineering controls and worker PPE are used at facilities that use in the United States.

ONUs include employees that work at the sites where hydraulic fracturing chemicals are used, but they do not directly handle the chemicals and are therefore expected to have lower inhalation exposures and are not expected to have dermal exposures through contact with liquids. ONUs for this scenario include supervisors, managers, and other employees that may be in the oil/gas well area but do not perform tasks that result in the same level of exposures as those workers that engage in tasks related to the use of fracturing chemicals.

Number of Potentially Exposed Workers and ONUs

Use of hydraulic fracturing chemicals are expected to fall within NAICS codes 213111, Drilling Oil and Gas Wells, and 213112, Support Activities for Oil and Gas Operations. EPA estimated a total of 14,193 sites, 46,315 workers, and 26,007 ONUs (U.S. BLS, 2016). The number of sites conducting hydraulic fracturing using 1,4-dioxane is provided by FracFocus 3.0 data, with a total of 411 sites (GWPC and IOGCC, 2022). For additional information on the steps used to estimate the number of potentially exposed workers and ONUs, refer to Appendix G.5 of the 2020 *Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Worker Inhalation Exposure Assessment

Facilities have self-reported to FracFocus 3.0 that 1,4-dioxane is present in hydraulic fracturing fluid additives, such as scale inhibitors, additives, biocides, friction reducers, and surfactants (GWPC and IOGCC, 2022). The information and data quality evaluation to assess occupational exposures during hydraulic fracturing is listed in Table_Apx F-29 and described below.

Table_Apx F-29. Hydraulic Fracturing Worker Exposure Data Evaluation

Worker Activity or Sampling Location	Data Type	Number of Samples	Overall Data Quality Determination	Source Reference
Unloading hydraulic fracturing fluid additives, cleaning empty additive containers, equipment cleaning	Input parameters for Monte Carlo modeling	N/A	Medium[a]	(U.S. EPA, 2022d)
		N/A	Medium[a]	(GWPC and IOGCC, 2022)

[a] This is the rating for the underlying data used in the model, and not the Monte Carlo model itself.

EPA did not find relevant inhalation monitoring data for the use of hydraulic fracturing fluids. Therefore, EPA modeled 1,4-dioxane air concentrations using a Monte Carlo modeling approach, which is described in Appendix F.9. This modeling approach utilizes the *EPA AP-42 Loading Model*, *EPA/OPPT Mass Transfer Coefficient Model*, and *EPA Mass Balance Inhalation Model*, with variation in input parameters for mass fraction of 1,4-dioxane in hydraulic fracturing additive and fluid, saturation factor, container size, use rate of fracturing fluid, ventilation rate, and mixing factor based on available data. During modeling, EPA noted that if the durations for all individual hydraulic fracturing activities were summed, the total exposure time can exceed a full shift duration of eight hours. To avoid this, the time spent unloading containers and cleaning containers was capped at two hours each, since the other activity for equipment cleaning occurs over four hours (*i.e.*, 2 hours for container unloading + 2 hours for container cleaning + 4 hours for equipment cleaning = 8 hours). This is a limitation of the assessment because EPA is unsure the extent to which the assessed activity durations are representative of real-world conditions. A summary of the modeled exposures is presented in Table_Apx F-30.

Table_Apx F-30. Modeled Occupational Inhalation Exposures for Hydraulic Fracturing

Statistic	1,4-Dioxane Exposure Concentration, 8-hour TWA (mg/m^3)
Maximum	1.48E04
99th Percentile	239.331
95th Percentile	66.816
50th Percentile	2.873
5th Percentile	0.114
Minimum	1.54E−07
Mean	17.691

EPA used the 50th and 95th percentile modeled 8-hour TWA exposures values presented in Table_Apx F-31 to calculate the central tendency and high-end ADC and LADC, respectively. The calculated values are summarized in Table_Apx F-30. Equations for calculating ACD and LADC are presented in Appendix G of the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c).

Exposure data for ONUs were not available. The ONU exposures are anticipated to be lower than worker exposures since ONUs do not typically directly handle the chemical. Only inhalation exposures to vapors or incidental dermal exposures may be expected to ONUs.

Table_Apx F-31. Inhalation Exposures of Workers for Hydraulic Fracturing Based on Modeling

Exposure Type	Central Tendency (50th Percentile) (mg/m^3) [a]	High-end (95th Percentile) (mg/m^3) [a]
8-hour TWA Exposure Concentrations	2.87	66.8
Average Daily Concentration (ADC)	0.177	18.5
Lifetime Average Daily Concentration (LADC)	0.070	9.49

[a] See Table_Apx F-29 for corresponding references.

Key Uncertainties

EPA used assumptions and values from the ESD on Hydraulic Fracturing and various EPA models to estimate inhalation exposures during container transfers, container cleaning, and equipment cleaning within the hydraulic fracturing COU (see Appendix F.9). The uncertainties associated with this modeling approach are described in Section 3.1.2.4.

EPA also used data from FracFocus 3.0 (GWPC and IOGCC, 2022) to inform input parameters for the exposure calculations. FracFocus contains self-reported data; therefore, the extent to which these data represent operations across multiple sites throughout the United States is unclear.

F.5 Summary of Occupational Inhalation Exposures

A summary of the inhalation exposure estimates previously discussed is included in Table_Apx F-32. The table presents high-end and central tendency inhalation exposures by condition of use. The table also indicates whether the source data are monitoring values or modeled estimates. For more details on how each inhalation exposure was estimated, see Appendix F.4.

Table_Apx F-32. Estimated Inhalation Exposure (mg/m^3) for Workers during Various Conditions of Use

OES	Category	Exposure Time-frame	Exposure Frequency (day/year)		8-hour TWA Exposures		Chronic, Non-cancer Exposures		Chronic, Cancer Exposures		8-hour Data Points	Total Samples	Sources & Notes	Data Type
					$C_{8\text{-h TWA}}$ (mg/m^3)		$ADC_{8\text{-h TWA}}$ (mg/m^3)		$LADC_{8\text{-h TWA}}$ (mg/m^3)					
			HE	CT	HE	CT	HE	CT	HE	CT				
Textile Dye	Worker	8-hour	250	31	74	6.6E−02	84	7.9E−03	43	3.1E−03	14	51	OSHA CEHD from 1991–2010 (OSHA, 2020). 51 PBZ samples, from which 14 8-h TWAs were derived.	Monitoring Data
Antifreeze	Worker	8-hour	250	250	1.1E−07	2.2E−08	1.1E−07	2.1E−08	5.4E−08	8.3E−09	N/A	N/A	Monte Carlo Simulation results	Modeled Data
Surface Cleaner	Worker	8-hour	250	250	3.7E−03	2.9E−04	3.6E−03	2.8E−04	1.8E−03	1.1E−04	49	49	(Harley et al., 2021)	Monitoring Data
Dish Soap	Worker	8-hour	250	250	2.1	1.0	2.0	1.0	1.0	4.0E−01	29	29	(Belanger, 1980)	Monitoring Data
Dishwasher Detergent	Worker	8-hour	250	250	2.1	1.0	2.0	1.0	1.0	4.0E−01	29	29	(Belanger, 1980)	Monitoring Data
Laundry Detergent (Industrial)	Worker (vapor)	8-hour	250	250	1.9E−03	5.2E−04	1.8E−03	5.0E−04	9.2E−04	2.0E−04	N/A	N/A	Monte Carlo Simulation results	Modeled Data
Laundry Detergent (Industrial)	Worker (Total Particulates)	8-hour	250	250	2.0E−04	1.1E−04	1.9E−04	1.0E−04	9.8E−05	4.0E−05	N/A	N/A	Monte Carlo Simulation results	Modeled Data
Laundry Detergent (Industrial)	Worker (Respirable Particulates)	8-hour	250	250	6.7E−05	3.5E−05	6.4E−05	3.4E−05	3.3E−05	1.3E−05	N/A	N/A	Monte Carlo Simulation results	Modeled Data
Laundry Detergent (Institutional)	Worker (vapor)	8-hour	250	250	1.4E−03	4.1E−04	1.4E−03	3.9E−04	7.1E−04	1.6E−04	N/A	N/A	Monte Carlo Simulation results	Modeled Data

OES	Category	Exposure Time-frame	Exposure Frequency (day/year)		8-hour TWA Exposures		Chronic, Non-cancer Exposures		Chronic, Cancer Exposures		8-hour Data Points	Total Samples	Sources & Notes	Data Type
					$C_{8\text{-h TWA}}$ (mg/m^3)		$ADC_{8\text{-h TWA}}$ (mg/m^3)		$LADC_{8\text{-h TWA}}$ (mg/m^3)					
			HE	CT	HE	CT	HE	CT	HE	CT				
Laundry Detergent (Institutional)	Worker (Total Particulates)	8-hour	250	250	2.0E−04	1.1E−04	1.9E−04	1.0E−04	9.8E−05	4.0E−05	N/A	N/A	Monte Carlo Simulation results	Modeled Data
Laundry Detergent (Institutional)	Worker (Respirable Particulates)	8-hour	250	250	6.7E−05	3.5E−05	6.4E−05	3.4E−05	3.3E−05	1.3E−05	N/A	N/A	Monte Carlo Simulation results	Modeled Data
Paint and Floor Lacquer	Worker	8-hour	250	250	1.2	2.1E−01	1.2	2.0E−01	5.9E−01	8.0E−02	17	17	(Hills et al., 1989)	Monitoring Data
Polyethylene Terephthalate (PET) Byproduct	Worker	8-hour	250	250	47	4.7	45	4.5	23	1.8	11	35	OSHA CEHD from 1985–1994 (OSHA, 2020). 35 PBZ samples, from which 11 8-hour TWAs were derived.	Monitoring Data
Ethoxylation Process Byproduct	Worker	8-hour	250	250	1.2	1.2	1.2	1.2	5.9E−01	4.6E−01	1	7	OSHA CEHD from 1985–1994 (OSHA, 2020). 7 PBZ samples, from which 1 8-hour TWA was derived.	Monitoring Data
Hydraulic Fracturing	Worker	8-hour	72	1	67	2.9	19	1.1E−02	9.5	4.4E−03	N/A	N/A	Monte Carlo Simulation results	Modeled Data
CT = Central Tendency; HE = High End														

F.6 Summary of Weight of the Scientific Evidence Conclusions in Inhalation Exposure Estimates

Table_Apx F-33 provides a summary of EPA's weight of the scientific evidence conclusions in its inhalation exposure estimates for each of the Occupational Exposure Scenarios assessed.

Table_Apx F-33. Summary of Weight of the Scientific Evidence Conclusions in Inhalation Exposure Estimates by OES

OES	Weight of the Scientific Evidence Conclusion in Inhalation Exposure Estimates
Textile Dye	8-hour TWA inhalation exposure estimates are assessed using OSHA's CEHD. Factors that increase the strength of evidence for this OES are that the exposure data are directly relevant to the OES (as opposed to surrogate), that OSHA CEHD has a high overall data quality determination, and consistency within the dataset (all measurements are taken by OSHA through NIOSH method 1602). The data includes personal and area samples from multiple sites, which increases the variability of the dataset. Factors that decrease the strength of the evidence for this OES include the low number of data points, uncertainty in the representativeness of the monitoring data for all sites in this OES, and uncertainty in the representativeness of the older monitoring data towards more current operations (some data were from 1991-1992). Additionally, worker activity descriptions are not provided in the dataset and there was a high number of non-detects present. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.
Antifreeze	8-hour TWA inhalation exposure estimates are assessed using Monte Carlo modeling with information from the OECD ESD on Chemical Additives used in Automotive Lubricants, the EPA MRD on Commercial Use of Automotive Detailing Products, and EPA/OPPT models. Factors that increase the strength of evidence for this OES are that the ESD and MRD used have high overall data quality determinations, high number of data points (simulation runs), and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential exposure values that represents a larger proportion of sites than a discrete value. Factors that decrease the strength of the evidence for this OES include that the ESD and MRD are not directly applicable to antifreeze uses (used as surrogate), uncertainty in the representativeness of evidence to all sites, and uncertainty in the use of generic default values from the ESD and MRD for sites that specifically use 1,4-dioxane. Additionally, EPA scaled up a consumer antifreeze use rate to a commercial use rate based on information in the ESD and MRD, which increases uncertainty. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.
Surface Cleaner	8-hour TWA inhalation exposure estimates are assessed using monitoring data from published literature. Factors that increase the strength of evidence for this OES are that the exposure data are directly relevant to the OES (as opposed to surrogate), that the literature has a medium overall data quality determination, and consistency within the dataset (all measurements are taken via the same method). Additionally, the literature includes information on worker activities during sampling. Factors that decrease the strength of the evidence for this OES include the lack of variability (only one study), uncertainty in the representativeness of the monitoring data for all sites in this OES, uncertainty from using summary statistics from the study (discrete sample results not provided), and uncertainty in whether the activities performed in this study accurately reflect all

OES	Weight of the Scientific Evidence Conclusion in Inhalation Exposure Estimates
	surface cleaning scenarios or the cleaning industry as whole. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.
Dish Soap	8-hour TWA inhalation exposure estimates are assessed using monitoring data from a NIOSH HHE. Factors that increase the strength of evidence for this OES are that the exposure data are directly relevant to the OES (as opposed to surrogate), that the literature has a medium overall data quality determination, and consistency within the dataset (all measurements are taken via the same NIOSH method). Factors that decrease the strength of the evidence for this OES include the lack of variability (only one study), uncertainty in the representativeness of the monitoring data for all sites and worker activities in this OES, and uncertainty in the representativeness of the older monitoring data towards more current operations (data were from 1980). Additionally, all samples were non-detected for 1,4-dioxane and EPA used the sample method level of detection and half the level of detection as the high-end and central tendency inhalation exposure estimates. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.
Dishwasher Detergent	Since EPA used the same approach as discussed for dish soap, the same information and weight of the scientific evidence conclusion apply.
Laundry Detergent	8-hour TWA inhalation exposure estimates are assessed using Monte Carlo modeling with information from the ESD on Water Based Washing Operations at Industrial and Institutional Laundries and EPA/OPPT models. Factors that increase the strength of evidence for this OES are that the exposure estimates are directly relevant to the OES (as opposed to surrogate), that the ESD on Industrial and Institutional Laundries has a medium overall data quality determination and was peer reviewed, high number of data points (simulation runs), and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential exposure values that represents a larger proportion of sites than a discrete value. Also, EPA was able to separately estimate exposures for industrial and institutional laundry settings. Factors that decrease the strength of the evidence for this OES include uncertainty in the representativeness of evidence to all sites and uncertainty in the representativeness of generic values in the ESD towards real-world sites that use 1,4-dioxane. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.
Paint and Floor Lacquer	8-hour TWA inhalation exposure estimates are assessed using monitoring data from a NIOSH HHE. Factors that increase the strength of evidence for this OES are that the exposure data are directly relevant to the OES (as opposed to surrogate), that the literature has a high overall data quality determination, and consistency within the dataset (all measurements are taken via the same NIOSH method). Factors that decrease the strength of the evidence for this OES include the low number of data points, lack of variability (one study), uncertainty in the representativeness of the monitoring data for all sites and worker activities in this OES, and uncertainty in the representativeness of the older monitoring data towards more current operations (data were from 1989). Additionally, some of these data were short-term samples that EPA converted to 8-hour TWAs by assuming there was no exposure for the remainder of the 8 hours after the sampling duration, which adds uncertainty. This assumption may result in underestimation of exposures if workers perform additional activities that may result in exposures to 1,4-dioxane that

OES	Weight of the Scientific Evidence Conclusion in Inhalation Exposure Estimates
	were not captured in the monitoring performed. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.
Polyethylene Terephthalate (PET) Byproduct	8-hour TWA inhalation exposure estimates are assessed using OSHA's CEHD. Factors that increase the strength of evidence for this OES are that the exposure data are directly relevant to the OES (as opposed to surrogate), that OSHA CEHD has a high overall data quality determination, and consistency within the dataset (all measurements are taken by OSHA through NIOSH method 1602). OSHA includes personal samples from multiple sites, which increases the variability of the dataset. Factors that decrease the strength of the evidence for this OES include the low number of data points, uncertainty in the representativeness of the monitoring data for all sites and worker activities in this OES, and uncertainty in the representativeness of the older monitoring data towards more current operations (data were from 1985–1994). Additionally, worker activity descriptions are not provided in the dataset. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.
Ethoxylation Process Byproduct	8-hour TWA inhalation exposure estimates are assessed using OSHA's CEHD. Factors that increase the strength of evidence for this OES are that the exposure data are directly relevant to the OES (as opposed to surrogate), that OSHA CEHD has a high overall data quality determination, and consistency within the dataset (all measurements are taken by OSHA through NIOSH method 1602). Factors that decrease the strength of the evidence for this OES include the low number of data points, lack of variability (only one site was sampled), uncertainty in the representativeness of the monitoring data for all sites and worker activities in this OES, and uncertainty in the representativeness of the older monitoring data towards more current operations (data were from 2000). Additionally, worker activity descriptions are not provided in the dataset. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is slight to moderate and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.
Hydraulic Fracturing	8-hour TWA inhalation exposure estimates are assessed using Monte Carlo modeling with information from the Draft ESD on Hydraulic Fracturing and FracFocus 3.0. Factors that increase the strength of evidence for this OES are that the exposure estimates are directly relevant to the OES (as opposed to surrogate), that the Draft ESD on Hydraulic Fracturing and FracFocus 3.0 have medium overall data quality determinations, high number of data points (simulation runs), and full distributions of input parameters. The Monte Carlo modeling accounts for the entire distribution of input parameters, calculating a distribution of potential exposure values that represents a larger proportion of sites than a discrete value. Factors that decrease the strength of the evidence for this OES include that the Draft ESD has not been peer reviewed and the uncertainties and limitations in the representativeness of the estimates for sites that specifically use 1,4-dioxane because the default values from the Draft ESD on Hydraulic Fracturing. Additionally, the duration of exposure for container unloading and cleaning activities is uncertain. To avoid unrealistic output parameters, exposure duration was capped at 2 hours for each activity. This is a limitation of the assessment because there is uncertainty in the extent to which the assessed activity durations are representative of real-world conditions. Based on this information, EPA has concluded that the weight of the scientific evidence for this assessment is moderate to robust and provides a plausible estimate of exposures in consideration of the strengths and limitations of reasonably available data.

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F.7 Antifreeze Modeling Approach and Parameters for Estimating Occupational Inhalation Exposures

This appendix presents the modeling approach used to estimate occupational inhalation exposures to 1,4-dioxane during the commercial use of antifreeze. EPA expects that the main source of occupational inhalation exposure during the use of antifreeze is from the unloading of antifreeze from containers into vehicles. Therefore, this approach applies a stochastic modeling approach to the *EPA/OAQPS AP-42 Loading Model*, which estimates air releases during liquid transfer operations, and the *EPA/OPPT Mass Balance Model*, which estimates the corresponding inhalation exposures resulting from these air releases.

Inhalation exposure to chemical vapors is a function of the chemical's physical properties, ventilation rate of the container loading area, type of loading method, and other model parameters. While physical properties are fixed for a chemical, some model parameters are expected to vary from one facility to another. An individual model input parameter could either have a discrete value or a distribution of values. EPA assigned statistical distributions based on available literature data or engineering judgment to address the variability in parameters such as ventilation rate ($RATE_{ventilation}$), mixing factor (F_{mixing}), saturation factor ($F_{saturation}$), concentration of 1,4-dioxane in antifreeze ($F_{dioxane}$), container size (V_{cont}), and number of jobs per day (N_{jobs}).

A Monte Carlo simulation was conducted to capture variability in the model input parameters described above. The simulation was conducted using the Latin hypercube sampling method in @Risk (Palisade, Ithaca, NY). The Latin hypercube sampling method is a statistical method for generating a sample of possible values from a multi-dimensional distribution. Latin hypercube sampling is a stratified method, meaning it guarantees that its generated samples are representative of the probability density function (variability) defined in the model. EPA performed 100,000 iterations of the model to capture the range of possible input values, including values with low probability of occurrence.

From the distribution resulting from the Monte Carlo simulation, EPA selected the 95th and 50th percentile values to represent a high-end exposure and central tendency exposure level respectively. The statistics were calculated directly in @Risk. The following subsections detail the model design equations and parameters used for inhalation exposure estimates.

F.7.1 Model Equations

Daily use rate of antifreeze at commercial sites is calculated using the following equation:

Equation_Apx F-1

$$Q_{antifreeze\_day} = Q_{consumer} * N_{jobs}$$

Where:

$Q_{antifreeze\_day}$ = Commercial daily use rate of antifreeze [kg/site-day]
$Q_{consumer}$ = Consumer use rate of antifreeze [kg/job]
N_{jobs} = Commercial antifreeze jobs per day [jobs/day]

Annual use rate of antifreeze at commercial sites is calculated using the following equation:

Equation_Apx F-2

$$Q_{antifreeze\_yr} = OD * Q_{antifreeze\_day}$$

Where:

$Q_{antifreeze\_yr}$ = Commercial annual use rate of antifreeze [kg/site-year]
OD = Operating days [days/site-year]
$Q_{antifreeze\_day}$ = Commercial daily use rate of antifreeze [kg/site-day]

The number of antifreeze container used per year is calculated using the following equation:

Equation_Apx F-3

$$N_{cont\_site\_yr} = \frac{Q_{antifreeze\_yr}}{3.79\frac{L}{gal} * 1\frac{kg}{L} * V_{cont}}$$

Where:

$N_{cont\_site\_yr}$ = Number of antifreeze containers used per year [containers/site-year]
$Q_{antifreeze\_yr}$ = Commercial annual use rate of antifreeze [kg/site-year]
V_{cont} = Antifreeze container size [gal]

Duration of release for container activities is calculated using the following equation:

Equation_Apx F-4

$$OH_{cont\_unload} = \frac{N_{cont\_site\_yr}}{OD * RATE_{fill}}$$

Where:

OH_{cont_unload} = Duration of release for container activities [hours]
$N_{cont\_site\_yr}$ = Number of antifreeze containers used per year [containers/site-year]
OD = Operating days [days/site-year]
$RATE_{fill}$ = Container fill/unloading rate [containers/hour]

Vapor pressure correction factor is calculated using the following equation:

Equation_Apx F-5

$$X = \frac{F_{dioxane}/_{MW}}{\frac{F_{dioxane}}{MW} + \frac{1 - F_{dioxane}}{18}}$$

Where:

X = Vapor pressure correction factor [mol dioxane/mol water]
MW = Molecular weight of 1,4-dioxane [g/mol]
$F_{dioxane}$ = 1,4-dioxane concentration in antifreeze [kg/kg]

Vapor generation rate of 1,4-dioane during container unloading is calculated using the following equation:

Equation_Apx F-6

$$Q_{vapor\_generation} = F_{saturation} * MW * 3785.4 * V_{cont} * \frac{RATE_{fill}}{3600} * X * \frac{VP}{760} * \frac{1}{T * R}$$

Where:

$Q_{vapor\_generation}$ = Vapor generation rate of 1,4-dioxane[g/s]
$F_{saturation}$ = Saturation factor [unitless]
MW = Molecular weight of 1,4-dioxane [g/mol]
V_{cont} = Antifreeze container size [gal]
$RATE_{fill}$ = Container fill/unloading rate [containers/hour]
VP = Vapor pressure of 1,4-dioxane [torr]
T = Ambient temperature [K]
R = Universal gas constant [atm-cm^3/gmol-K]

Volumetric concentration of 1,4-dioxane in air during unloading is calculated using the following equation:

Equation_Apx F-7

$$C_v = \frac{170000 * T * Q_{vapor\_generation}}{MW * RATE_{ventilation} * F_{mixing}}$$

Where:

C_v = Volumetric concentration of 1,4-dioxane in air [ppm]
T = Ambient temperature [K]
$Q_{vapor\_generation}$ = Vapor generation rate of 1,4-dioxane [g/s]
MW = Molecular weight of 1,4-dioxane [g/mol]
$RATE_{ventilation}$ = Ventilation rate [ft^3/min]
F_{mixing} = Mixing factor [unitless]

8-hour TWA mass concentration of 1,4-dioxane in air is calculated using the following equation. Note that this equation assumes that no exposure occurs for the remainder of the 8-hour shift after container unloading takes place:

Equation_Apx F-8

$$Concentration_{EP1} = \frac{C_v * MW}{V_m} * \frac{OH_{cont\_unload}}{8}$$

Where:

$Concentration_{EP1}$ = 8-hour TWA mass concentration of 1,4-dioxane in air [mg/m^3]
C_v = Volumetric concentration of 1,4-dioxane in air [ppm]
MW = Molecular weight of 1,4-dioxane [g/mol]
OH_{cont_unload} = Duration of release for container activities [hours]
V_m = Molar volume [L/mol]

F.7.2 Modeling Input Parameters

Table_Apx F-34 summarizes the model parameters and their values for the Monte Carlo simulation. Additional explanations of EPA's selection of the distributions for each parameter are provided after this table. High-end and central tendency exposures are estimated by selecting the 50th and 95th percentile values from the output distribution.

Table_Apx F-34. Summary of Parameter Values and Distributions Used in the Antifreeze Exposure Modeling

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational/Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Container Size	V_{cont}	gal	0.125	0.03125	20	0.125	Triangular	See Section F.7.3
Jobs per Day	N_{jobs}	jobs/day	4	1	9	4	Triangular	See Section F.7.4
Concentration of 1,4-dioxane in Antifreeze	$F_{dioxane}$	kg/kg	0.000086	0.00000001	0.000086	—	Uniform	See Section F.7.5
Ventilation Rate	$RATE_{ventilation}$	ft^3/min	500	500	10,000	3,000	Triangular	See Section F.7.6
Mixing Factor	F_{mixing}	Dimensionless	0.1	0.1	1	0.5	Triangular	See Section F.7.7
Saturation Factor	$F_{saturation}$	Dimensionless	1	0.5	1	0.5	Triangular	See Section F.7.8
Vapor Pressure of 1,4-dioxane	VP	torr	40	—	—	—	—	Physical property
Molecular Weight of 1,4-dioxane	MW	g/mol	88.1	—	—	—	—	Physical property
Ambient Temperature	T	K	298	—	—	—	—	Process parameter
Universal Gas Constant	R	atm-cm3/gmol-K	82.05	—	—	—	—	Universal constant
Molar Volume	V_m	L/mol	24.45	—	—	—	—	Physical property
Consumer Use Rate of Antifreeze	$Q_{consumer}$	Kg/job	0.15	—	—	—	—	See Section F.7.9
Operating Days	OD	days/year	250	—	—	—	—	See Section F.4.2
Fill Rate of Containers	$RATE_{fill}$	containers/hour	60	—	—	—	—	See Section F.7.10
Operating Hours	OH_{cont_unload}	hours/day	8	—	—	—	—	Process parameter

F.7.3 Container Size

EPA modeled container size using a triangular distribution with a lower bound of 0.03125 gallons, an upper bound of 20 gallons, and a mode of 0.125 gallons. The lower bound and mode of the distribution were taken from the EPA MRD on Commercial Use of Automotive Detailing Products (U.S. EPA, 2022b), which indicates a minimum container size of 4 ounces (0.03125 gal) and a default container size of 16 ounces (0.125 gallons). The upper bound of the distribution comes from the OECD ESD on Chemical Additives used in Automotive Lubricants (OECD, 2020), which indicates a maximum container size of 20 gallons.

F.7.4 Jobs per Day

EPA modeled number of jobs per day (*e.g.*, number of cars serviced) using a triangular distribution with a lower bound of 1 job/day, an upper bound of 9 jobs/day, and a mode of 4 jobs/day. The lower bound and mode of the distribution come from EPA's *Brake Servicing Near-Field/Far-Field Inhalation Exposure Model*, which indicates one to four cars are serviced per day.

F.7.5 Concentration of 1,4-Dioxane in Antifreeze

EPA modeled concentration of 1,4-dioxane in antifreeze using a uniform distribution from a lower bound of 1.00×10^{-8} kg 1,4-dioxane/kg antifreeze to an upper bound of 8.60×10^{-5} kg 1,4-dioxane/kg antifreeze. This is based on the December 2020 *Final Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2020c), which indicates that 1,4-dioxane is a byproduct in antifreeze at concentrations ranging from 0.01 to 86 ppm.

F.7.6 Ventilation Rate

The CEB Manual (U.S. EPA, 1991) indicates general ventilation rates in industry range from 500 to 10,000 ft^3/min, with a typical value of 3,000 ft^3/min. The underlying distribution of this parameter is not known; therefore, EPA assigned a triangular distribution, which is completely defined by the range and mode of a parameter. EPA assumed the mode is equal to the typical value provided by the CEB Manual (U.S. EPA, 1991).

F.7.7 Mixing Factor

The CEB Manual (U.S. EPA, 1991) indicates mixing factors may range from 0.1 to 1, with 1 representing ideal mixing. The CEB Manual references the 1988 ACGIH Ventilation Handbook, which suggests the following factors and descriptions: 0.67 to 1 for best mixing; 0.5 to 0.67 for good mixing; 0.2 to 0.5 for fair mixing; and 0.1 to 0.2 for poor mixing. The underlying distribution of this parameter is not known; therefore, EPA assigned a triangular distribution, which is completely defined by the range and mode of a parameter. The mode for this distribution was not provided; therefore, EPA assigned a mode value of 0.5 based on the typical value provided in the ChemSTEER User Guide (U.S. EPA, 2015a) for the *EPA/OPPT Mass Balance Inhalation Model*.

F.7.8 Saturation Factor

The CEB Manual (U.S. EPA, 1991) indicates that the saturation concentration was reached or exceeded by misting with a maximum saturation factor of 1.45 during splash filling. The CEB manual indicates that the saturation factor for bottom filling was expected to be about 0.5 (U.S. EPA, 1991). The underlying distribution of this parameter is not known; therefore, EPA assigned a triangular distribution, which is completely defined by range and mode of a parameter. Because a mode was not provided for this parameter, EPA assigned a mode value of 0.5 for bottom filling as bottom filling minimizes volatilization (U.S. EPA, 1991). This value also corresponds to the typical value provided in the

ChemSTEER User Guide (U.S. EPA, 2015a) for the *EPA/OAQPS AP-42 Loading Model* for small containers.

F.7.9 Consumer Use Rate of Antifreeze

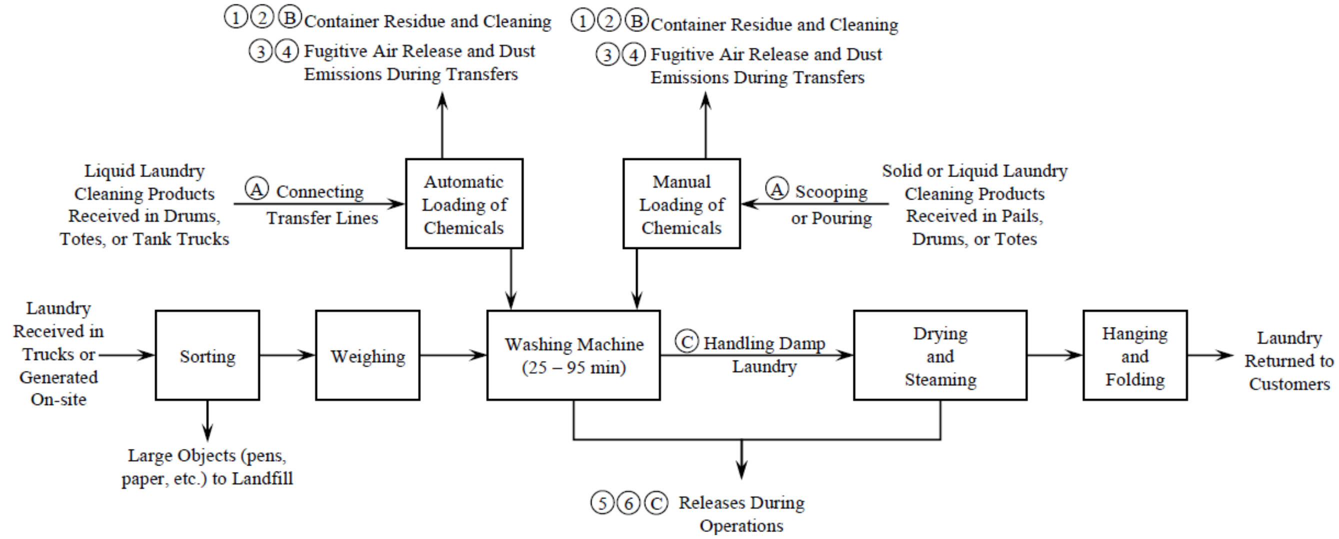
EPA did not find information on the commercial use rates of antifreeze. Therefore, EPA used a consumer use rate scaled up by the number of antifreeze jobs expected in a commercial setting (see Section F.7.4). The December 2020 *Final Risk Evaluation for 1,4-Dioxane* provided a single value of 0.15 kg/job for the consumer use rate of antifreeze (U.S. EPA, 2020c). Therefore, EPA could not develop a distribution of values for this parameter and used the single value of 0.15 kg/job from the 2020 RE.

F.7.10 Container Fill Rate

The ChemSTEER User Guide (U.S. EPA, 2015a) provides a typical fill rate of 60 containers per hour for small containers and bottles, which are anything less than 20 gallons in capacity. Therefore, EPA could not develop a distribution of values for this parameter and used the single value 60 containers/hour from the ChemSTEER User Guide.

F.8 Laundry Detergent Modeling Approach and Parameters for Estimating Occupational Inhalation Exposures

This appendix presents the modeling approach used to estimate occupational inhalation exposures to 1,4-dioxane during the industrial and institutional use of laundry detergents. This approach utilizes the OECD ESD on the Chemicals Used in Water Based Washing Operations at Industrial and Institutional Laundries (OECD, 2011b) combined with Monte Carlo simulation (a type of stochastic simulation). This ESD categorized laundry facilities into either industrial or institutional facilities and includes different process parameters for each. Therefore, EPA modeled the two types of laundry facilities separately. In addition, laundry detergents can be in liquid or powder physical forms. The difference in physical form results in different parameter distributions. Therefore, EPA modeled liquid and powder detergents separately. This ESD includes a diagram of release and exposure points during textile dying, as shown in Figure_Apx F-4.



Figure_Apx F-4. Environmental Release and Occupational Exposure Points during Industrial/Institutional Laundering Operations

Based on Figure_Apx F-4, EPA identified the following exposure points (note that exposure point 1 corresponds to diagram point A, point 2 to diagram point B, etc.):

- Exposure point 1 (EP1): Exposure to vapors during container transfers;
- Exposure point 2 (EP2): Exposure to vapors during container cleaning;
- Exposure point 3 (EP3): Exposure to vapors during laundry operations;
- Exposure point 4 (EP4): Exposure to total particulates over all activities; and
- Exposure point 5 (EP5): Exposure to respirable particulates over all activities.

To estimate inhalation exposures to vapors, this model utilizes the previously modeled vapor releases for each corresponding release point, as explained in Appendix E.12. To calculate a full-shift TWA, the 1,4-dioxane concentrations calculated for each exposure point above are multiplied by their respective exposure durations, then summed and divided by the total workday duration (8, 10, or 12 hours per the ESD).

Inhalation exposure to chemical vapors is a function of the chemical's physical properties, ventilation rate of the container loading area, type of loading method, and other model parameters. While physical properties are fixed for a chemical, some model parameters are expected to vary from one facility to another. An individual model input parameter could either have a discrete value or a distribution of values. EPA assigned statistical distributions based on available literature data or engineering judgment to address the variability in parameters such as ventilation rate ($RATE_{ventilation}$), mixing factor (F_{mixing}), total and respirable particulate concentration ($C_{particulate}$) and mass fraction of 1,4-dioxane ($F_{dioxane\_laundry}$).

A Monte Carlo simulation was then conducted to capture variability in the model input parameters described above. The simulation was conducted using the Latin hypercube sampling method in @Risk (Palisade, Ithaca, NY). The Latin hypercube sampling method is a statistical method for generating a sample of possible values from a multi-dimensional distribution. Latin hypercube sampling is a stratified method, meaning it guarantees that its generated samples are representative of the probability density function (variability) defined in the model. EPA performed 100,000 iterations of the model to capture the range of possible input values, including values with low probability of occurrence.

From the distribution resulting from the Monte Carlo simulation, EPA selected the 95th and 50th percentile values to represent a high-end exposure and central tendency exposure level respectively. The statistics were calculated directly in @Risk. The following subsections detail the model design equations and parameters used for inhalation exposure estimates.

F.8.1 Model Equations

Exposure point 1 (container transfers) volumetric concentration of 1,4-dioxane is calculated using the following equation:

Equation_Apx F-9

$$C_{v1} = (1.75 \times 10^5) * T * \frac{\left(\frac{Release\_perDay_{RP3}}{OH_{cont\_unload} * \left(3600 \frac{s}{hr}\right) * \left(0.001 \frac{kg}{g}\right)}\right)}{MW * RATE_{ventilation} * F_{mixing}}$$

Where:

C_{v1} = Volumetric concentration of 1,4-dioxane in air for exposure point 1 [ppm]

T = Ambient temperature [K]

$Release\_perDay_{RP3}$ = Daily vapor release for release point 3, Appendix E.12 [kg/site-day]

OH_{cont_unload} = Duration of release for container unloading, Appendix E.12 [hours/day]

MW = Molecular weight of 1,4-dioxane [g/mol]

$RATE_{ventilation}$ = Ventilation rate [ft^3/min]

F_{mixing} = Mixing factor [unitless]

Exposure point 1 (container transfers) mass concentration of 1,4-dioxane is calculated using the following equation:

Equation_Apx F-10

$$Concentration\_Vapor_{EP1} = \frac{C_{v1} * MW}{V_m}$$

Where:

$Concentration\_Vapor_{EP1}$ = Mass concentration of 1,4-dioxane in air for exposure point 1 [mg/m^3]

C_{v1} = Volumetric concentration of 1,4-dioxane in air for exposure point 1 [ppm]

MW = Molecular weight of 1,4-dioxane [g/mol]

V_m = Molar volume [L/mol]

Exposure point 2 (container cleaning) volumetric concentration of 1,4-dioxane is calculated using the following equation:

Equation_Apx F-11

$$C_{v2} = (1.75 \times 10^5) * T * \frac{\left(\dfrac{Release\_perDay_{RP2}}{\dfrac{N_{cont\_site\_yr}}{OD * RATE_{fill}} * \left(3600\,\dfrac{s}{hr}\right) * \left(0.001\,\dfrac{kg}{g}\right)}\right)}{MW * RATE_{ventilation} * F_{mixing}}$$

Where:

C_{v2} = Volumetric concentration of 1,4-dioxane in air for exposure point 2 [ppm]
T = Ambient temperature [K]
$Release\_perDay_{RP2}$ = Daily vapor release for release point 2, see Appendix E.12 [kg/site-day]
$N_{cont\_site\_yr}$ = Number of detergent containers used per year, see Appendix E.12 [containers/site-year]
OD = Operating days, see Appendix E.12 [days/year]
$RATE_{fill}$ = Container fill/unload rate, see Appendix E.12 [containers/hour]
MW = Molecular weight of 1,4-dioxane [g/mol]
$RATE_{ventilation}$ = Ventilation rate [ft^3/min]
F_{mixing} = Mixing factor [unitless]

Exposure point 2 (container cleaning) mass concentration of 1,4-dioxane is calculated using the following equation:

Equation_Apx F-12

$$Concentration\_Vapor_{EP2} = \frac{C_{v2} * MW}{V_m}$$

Where:

$Concentration\_Vapor_{EP2}$ = Mass concentration of 1,4-dioxane in air for exposure point 2 [mg/m^3]
C_{v2} = Volumetric concentration of 1,4-dioxane in air for exposure point 2 [ppm]
MW = Molecular weight of 1,4-dioxane [g/mol]
V_m = Molar volume [L/mol]

Exposure point 3 (laundry washing operations) volumetric concentration of 1,4-dioxane is calculated using the following equation:

Equation_Apx F-13

$$C_{v3} = (1.75 \times 10^5) * T * \frac{\left(\dfrac{Release\_perDay_{RP5}}{OH * \left(3600\,\dfrac{s}{hr}\right) * \left(0.001\,\dfrac{kg}{g}\right)}\right)}{MW * RATE_{ventilation} * F_{mixing}}$$

Where:

C_{v3} = Volumetric concentration of 1,4-dioxane in air for exposure

point 3 [ppm]
T = Ambient temperature [K]
$Release\_perDay_{RP5}$ = Daily vapor release for release point 5, see Appendix E.12 [kg/site-day]
OH = Operating hours, see Appendix E.12 [hours/day]
MW = Molecular weight of 1,4-dioxane [g/mol]
$RATE_{ventilation}$ = Ventilation rate [ft^3/min]
F_{mixing} = Mixing factor [unitless]

Exposure point 3 (laundry washing operations) mass concentration of 1,4-dioxane is calculated using the following equation:

Equation_Apx F-14

$$Concentration\_Vapor_{EP3} = \frac{C_{v3} * MW}{V_m}$$

Where:
$Concentration\_Vapor_{EP3}$ = Mass concentration of 1,4-dioxane in air for exposure point 3 [mg/m^3]
C_{v3} = Volumetric concentration of 1,4-dioxane in air for exposure point 3 [ppm]
MW = Molecular weight of 1,4-dioxane [g/mol]
V_m = Molar volume [L/mol]

The total full-shift vapor exposure (8-, 10-, and 12-hour TWAs) accounting for EP1 through EP3 is calculated using the following equation:

Equation_Apx F-15

$$Vapor\_Exposure\_TWA = \frac{\Bigg(Concentration\_Vapor_{EP1} * OH_{cont\_unload} + Concentration\_Vapor_{EP2} * \bigg(\frac{N_{cont\_site\_yr}}{OD * RATE_{fill}}\bigg) + Concentration\_Vapor_{EP3} * (OH - OH_{cont\_unload} - \bigg(\frac{N_{cont\_site\_yr}}{OD * RATE_{fill}}\bigg)\Bigg)}{OH}$$

Where:
$Vapor\_Exposure\_TWA$ = Full-shift TWA of 1,4-dioxane vapor exposure [mg/m^3]
$Concentration\_Vapor_{EP1}$ = Mass concentration of 1,4-dioxane in air for exposure point 1 [mg/m^3]
$Concentration\_Vapor_{EP2}$ = Mass concentration of 1,4-dioxane in air for exposure point 2 [mg/m^3]
$Concentration\_Vapor_{EP3}$ = Mass concentration of 1,4-dioxane in air for exposure point 3 [mg/m^3]
OH_{cont_unload} = Duration of release for container unloading, see Appendix E.12 [hours/day]
OD = Operating days, see Appendix E.12 [days/year]
$N_{cont\_site\_yr}$ = Number of detergent containers used per year, see Appendix E.12 [containers/site-year]
$RATE_{fill}$ = Container fill/unload rate, see Appendix E.12 [containers/hour]

OH = Operating hours, see Appendix E.12 [hours/day]

Exposure point 4 (total particulate exposure) mass concentration of 1,4-dioxane is calculated using the following equation:

Equation_Apx F-16

$$Concentration\_particulate\_total = C_{particulate\_total} * F_{dioxane\_laundry}$$

Where:

$Concentration\_particulate\_total$ = Air concentration of total 1,4-dioxane particles in the worker's breathing zone [mg/m^3]

$C_{particulate\_total}$ = Air concentration of all particles in the worker's breathing zone [mg/m^3]

$F_{dioxane\_laundry}$ = Mass fraction of 1,4-dioxane in laundry detergent, see Appendix E.12 [kg/kg]

Exposure point 5 (respirable particulate exposure) mass concentration of 1,4-dioxane is calculated using the following equation:

Equation_Apx F-17

$$Concentration\_particulate\_respirable = C_{particulate_{respirable}} * F_{dioxane\_laundry}$$

Where:

$Concentration\_particulate\_respirable$ = Air concentration of respirable1,4-dioxane particles in the worker's breathing zone [mg/m^3]

$C_{particulate\_respirable}$ = Air concentration of all respirable particles in the worker's breathing zone [mg/m3]

$F_{dioxane\_laundry}$ = Mass fraction of 1,4-dioxane in laundry detergent, see Appendix E.12 [kg/kg]

F.8.2 Model Input Parameters

Table_Apx F-35 summarizes the model parameters and their values for the Monte Carlo simulation. Additional explanations of EPA's selection of the distributions for each parameter are provided after this table. High-end and central tendency exposures are estimated by selecting the 50th and 95th percentile values from the output distribution.

10682 **Table_Apx F-35. Summary of Parameter Values and Distributions Used in the Laundry Detergent Exposure Modeling**

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rational/ Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Ventilation Rate	$RATE_{ventilation}$	ft^3/min	500	500	10,000	3,000	Triangular	See Section F.8.3
Mixing Factor	F_{mixing}	dimensionless	0.1	0.1	1	0.5	Triangular	See Section F.8.4
Total Particulate Concentration	$C_{particulate\_total}$	mg/m^3	15	0.01	15	9.5	Triangular	See Section F.8.5
Respirable Particulate Concentration	$C_{particulate\_respirable}$	mg/m^3	5	0.018	Institutional: 19 Industrial: 5	Institutional: 0.21 Industrial: 1.3	Triangular	See Section F.8.6
Molecular Weight of 1,4-dioxane	MW	g/mol	88.1	—	—	—	—	Physical property
Ambient Temperature	T	K	298	—	—	—	—	Process parameter
Molar Volume	V_m	L/mol	24.45	—	—	—	—	Physical property

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F.8.3 Ventilation Rate

The CEB Manual (U.S. EPA, 1991) indicates general ventilation rates in industry range from 500 to 10,000 ft^3/min, with a typical value of 3,000 ft^3/min. The underlying distribution of this parameter is not known; therefore, EPA assigned a triangular distribution, which is completely defined by the range and mode of a parameter. EPA assumed the mode is equal to the typical value provided by the CEB Manual (U.S. EPA, 1991).

F.8.4 Mixing Factor

The CEB Manual (U.S. EPA, 1991) indicates mixing factors may range from 0.1 to 1, with 1 representing ideal mixing. The CEB Manual references the 1988 *ACGIH Ventilation Handbook* that suggests the following factors and descriptions: 0.67 to 1 for best mixing; 0.5 to 0.67 for good mixing; 0.2 to 0.5 for fair mixing; and 0.1 to 0.2 for poor mixing. The underlying distribution of this parameter is not known; therefore, EPA assigned a triangular distribution, which is completely defined by the range and mode of a parameter. The mode for this distribution was not provided; therefore, EPA assigned a mode value of 0.5 based on the typical value provided in the ChemSTEER User Guide (U.S. EPA, 2015a) for the *EPA/OPPT Mass Balance Inhalation Model*.

F.8.5 Total Particulate Concentration

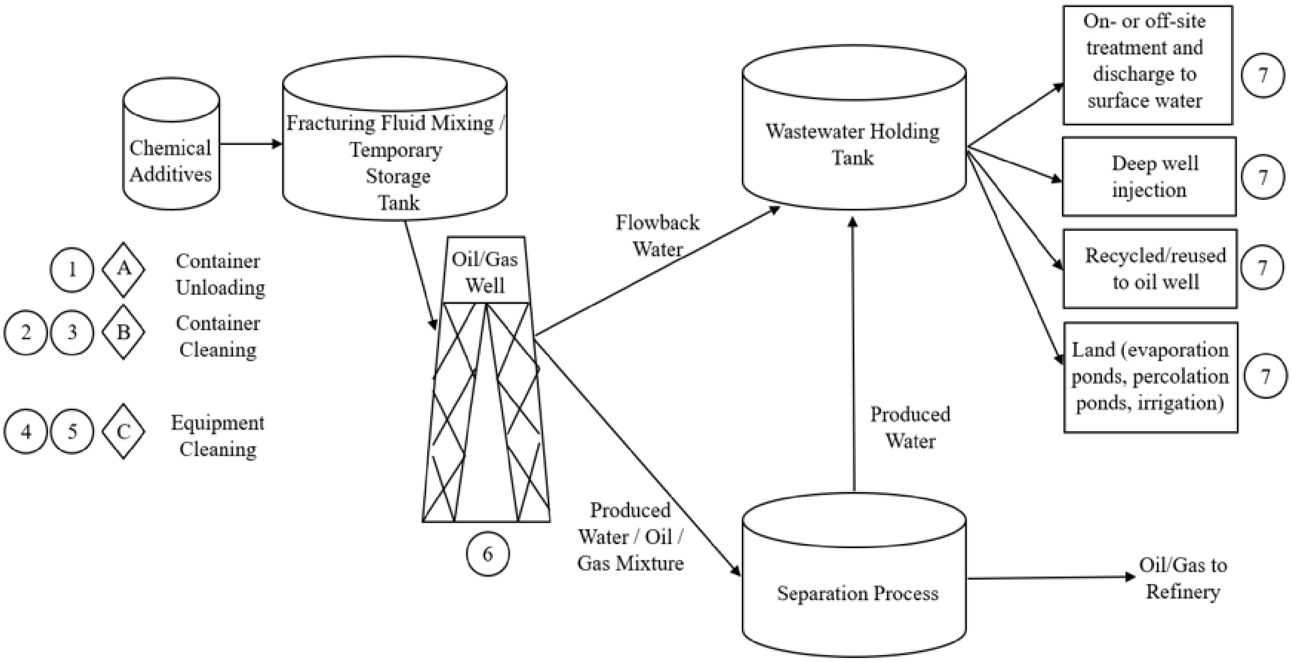
EPA modeled the total particulate concentration using a triangular distribution with a lower bound of 0.01 mg/m^3, an upper bound of 15 mg/m^3, and a mode of 9.5 mg/m^3 for both industrial and institutional laundries. These values were taken from EPA's *Generic Model for Central Tendency and High-End Inhalation Exposure to Total and Respirable Particulates Not Otherwise Regulated*. This model utilizes inhalation monitoring data from OSHA, which are analyzed by industry type (at the 2-digit or 3-digit NAICS code level). EPA specifically used the data for NAICS industry group 81 (Other Services, Except Public Administration) because this includes the NAICS code relevant to this OES, which is 812330, Linen and Uniform Supply. For this industry group, the *Generic Model for Central Tendency and High-End Inhalation Exposure to Total and Respirable Particulates Not Otherwise Regulated* indicates a total PNOR concentration ranging from 0.01 to 699 mg/m^3, with a mean of 9.5 mg/m^3. EPA used the low-end of this range and the mean as the lower bound and mode of the triangular distribution for this model. EPA used the OSHA permissible exposure limit (PEL) for total particulates of 15 mg/m^3 as the upper bound of the distribution per the *Generic Model for Central Tendency and High-End Inhalation Exposure to Total and Respirable Particulates Not Otherwise Regulated*, which indicates assessments should not assume that the PEL is exceeded without case-specific data.

F.8.6 Respirable Particulate Concentration

EPA modeled the respirable particulate concentration using a triangular distribution with lower bound of 0.018 mg/m^3, an upper bound of 5 mg/m^3, and a mode of 1.3 mg/m^3 for both industrial laundries and institutional laundries. These values were taken from EPA's *Generic Model for Central Tendency and High-End Inhalation Exposure to Total and Respirable Particulates Not Otherwise Regulated* for NAICS industry group 81 (Other Services, Except Public Administration) as described above in Section F.8.5. For this industry group, the *Generic Model for Central Tendency and High-End Inhalation Exposure to Total and Respirable Particulates Not Otherwise Regulated* indicates a respirable PNOR concentration ranging from 0.018 to 19 mg/m^3, with a mean of 1.3 mg/m^3. EPA used the low-end of this range and the mean as the lower bound and mode of the triangular distribution for this model. EPA used the OSHA permissible exposure limit (PEL) for respirable particulates of 5 mg/m^3 as the upper bound of the distribution per the *Generic Model for Central Tendency and High-End Inhalation Exposure to Total and Respirable Particulates Not Otherwise Regulated*, which indicates assessments should not assume that the PEL is exceeded without case-specific data.

F.9 Hydraulic Fracturing Modeling Approach and Parameters for Estimating Occupational Inhalation Exposures

This appendix presents the modeling approach used to estimate occupational inhalation exposures to 1,4-dioxane during hydraulic fracturing. This approach utilizes the Draft ESD on Chemicals Used in Hydraulic Fracturing (U.S. EPA, 2022d) combined with Monte Carlo simulation (a type of stochastic simulation). This ESD includes a diagram of release and exposure points during textile dying, as shown in Figure_Apx F-5.



Figure_Apx F-5. Environmental Release and Occupational Exposure Points during Hydraulic Fracturing

Based on Figure_Apx F-5, EPA identified the following release points. Note that exposure point 1 corresponds to diagram point A, etc.:

- Exposure point 1 (EP1): Exposure to vapors during container unloading and/or transferring;
- Exposure point 2 (EP2): Exposure to vapors during container cleaning; and
- Exposure point 3 (EP3): Exposure to vapors during equipment cleaning.

To calculate a full-shift TWA, the 1,4-dioxane concentrations calculated for each exposure point above are multiplied by their respective exposure durations, then summed and divided by the total workday duration (8 hours per the ESD).

Inhalation exposure to chemical vapors is a function of the chemical's physical properties, ventilation rate of the container loading area, type of loading method, and other model parameters. While physical properties are fixed for a chemical, some model parameters are expected to vary from one facility to another. An individual model input parameter could either have a discrete value or a distribution of values. EPA assigned statistical distributions based on available literature data or engineering judgment to address the variability in parameters such as ventilation rate ($RATE_{ventilation}$) and mixing factor (F_{mixing}).

A Monte Carlo simulation was then conducted to capture variability in the model input parameters described above. The simulation was conducted using the Latin hypercube sampling method in @Risk (Palisade, Ithaca, New York). The Latin hypercube sampling method is a statistical method for generating a sample of possible values from a multi-dimensional distribution. Latin hypercube sampling is a stratified method, meaning it guarantees that its generated samples are representative of the probability density function (variability) defined in the model. EPA performed 100,000 iterations of the model to capture the range of possible input values, including values with low probability of occurrence.

From the distribution resulting from the Monte Carlo simulation, EPA selected the 95th and 50th percentile values to represent a high-end exposure and central tendency exposure level respectively. The statistics were calculated directly in @Risk. The following subsections detail the model design equations and parameters used for inhalation exposure estimates.

F.9.1 Model Equations

Duration of exposure for container unloading is calculated using the following equation:

Equation_Apx F-18

$$OH_{cont\_exposures} = \frac{N_{cont\_unload\_yr}}{OD * RATE_{fill\_adjusted}}$$

Where:

$OH_{cont\_exposures}$ = Duration of exposure for container unloading [hours/day]

$N_{cont\_unlaod\_yr}$ = Number of containers unloaded annually, see Appendix E.13 [containers/site-year]

OD = Operating days in a year, see Appendix E.13 [days/year]

$RATE_{fill\_adjusted}$ = Container fill rate that is adjusted so that the release duration does not exceed 24 hours [containers/hour]

To make the simulation more realistic and account for subsequent exposure points 2 and 3, EPA set a maximum exposure duration for container unloading (exposure point 1) of 2 hours per day, assuming workers would not be unloading containers for a full shift. Therefore, the duration of exposure for container unloading is adjusted with the following equation:

Equation_Apx F-19

If $OH_{cont\_exposures} > 2$

$$OH_{cont\_exposures\_adjusted} = 2$$

If $OH_{cont\_exposures} \leq 2$

$$OH_{cont\_exposures\_adjusted} = OH_{cont\_exposures}$$

Where:

$OH_{cont\_exposures\_adjusted}$ = Duration of exposure for container unloading adjusted so that it is capped at 2 hours/day [hours/day]

$OH_{cont\_exposures}$ = Duration of exposure for container unloading [hours/day]

Exposure point 1 (container unloading) volumetric concentration in air for 1,4-dioxane is calculated using the *EPA Mass Balance Inhalation Model* shown in the following equation:

Equation_Apx F-20

$$C_{v1} = (1.75 \times 10^5) * \frac{F_{saturation\_factor} * (3785.4) * V_{cont} * \frac{RATE_{fill}}{3600} * X_{clean\_unload} * \frac{VP}{760}}{R * RATE_{ventilation} * F_{mixing}}$$

Where:

C_{v1} = Volumetric concentration of 1,4-dioxane in air for exposure point 1 [ppm]
$F_{saturation\_factor}$ = Saturation factor, see Appendix E.13 [unitless]
V_{cont} = Container size for fracturing fluids, see Appendix E.13 [gal/container]
$RATE_{fill}$ = Container fill/unloading rate, see Appendix E.13 [containers/hour]
X_{clean_unload} = Vapor pressure correction factor for container unloading and Cleaning, see Appendix E.13 [mol dioxane/mol water]
VP = Vapor pressure of 1,4-dioxane [torr]
R = universal gas constant [atm-cm^3/gmol-K]
$RATE_{ventilation}$ = Ventilation rate [ft^3/min]
F_{mixing} = Mixing factor [unitless]

Exposure point 1 (container unloading) mass concentration of 1,4-dioxane in air is calculated using the following equation:

Equation_Apx F-21

$$Concentration_{EP1} = C_{v1} * \frac{MW}{V_m}$$

Where:

$Concentration_{EP1}$ = Mass concentration of 1,4-dioxane in air for exposure point 1 [mg/m^3]
C_{v1} = Volumetric concentration of 1,4-dioxane in air for exposure point 1 [ppm]
MW = Molecular weight of 1,4-dioxane [g/mol]
V_m = Molar volume [L/mol]

Exposure point 2 (container cleaning) volumetric concentration in air for 1,4-dioxane is calculated using the *EPA Mass Balance Inhalation Model* shown in the following equation:

Equation_Apx F-22

$$C_{v2} = (1.75 \times 10^{5}) * T * \frac{(1.93 * 10^{-7}) * MW^{0.78} * X_{clean\_unload} * \left(\frac{1}{29} + \frac{1}{MW}\right)^{\frac{1}{3}} * RATE_{air\_speed}^{0.78} * (0.25\pi * D_{container\_opening}^{2})}{T^{0.4} * D_{container\_opening}^{0.11} * \left(\sqrt{T} - 5.87\right)^{\frac{2}{3}} * MW * RATE_{ventilation} * F_{mixing}}$$

Where:

C_{v2} = Volumetric concentration of 1,4-dioxane in air for exposure point 2 [ppm]
T = Ambient temperature [K]
MW = Molecular weight of 1,4-dioxane [g/mol]
X_{clean_unload} = Vapor pressure correction factor for container unloading and Cleaning, see Appendix E.13 [mol dioxane/mol water]
$RATE_{air\_speed}$ = Speed of air flow for evaporation from open surfaces, see Appendix E.13 [ft/min]
$D_{container\_opening}$ = Diameter of container opening, see Appendix E.13 [cm]
$RATE_{ventilation}$ = Ventilation rate [ft^3/min]
F_{mixing} = Mixing factor [unitless]

Exposure point 2 (container cleaning) mass concentration of 1,4-dioxane in air is calculated using the following equation:

Equation_Apx F-23

$$Concentration_{EP2} = C_{v2} * \frac{MW}{V_m}$$

Where:

$Concentration_{EP2}$ = Mass concentration of 1,4-dioxane in air for exposure point 2 [mg/m^3]
C_{v2} = Volumetric concentration of 1,4-dioxane in air for exposure point 2 [ppm]
MW = Molecular weight of 1,4-dioxane [g/mol]
V_m = Molar volume [L/mol]

Exposure point 3 (equipment cleaning) volumetric concentration in air for 1,4-dioxane is calculated using the *EPA Mass Balance Inhalation Model* shown in the following equation:

Equation_Apx F-24

$$C_{v3} = (1.75 \times 10^{5}) * T * \frac{(1.93 * 10^{-7}) * MW^{0.78} * X_{tank\_clean} * \left(\frac{1}{29} + \frac{1}{MW}\right)^{\frac{1}{3}} * RATE_{air\_speed}^{0.78} * (0.25\pi * D_{equip\_opening}^{2})}{T^{0.4} * D_{equip\_opening}^{0.11} * \left(\sqrt{T} - 5.87\right)^{\frac{2}{3}} * MW * RATE_{ventilation} * F_{mixing}}$$

Where:

C_{v3} = Volumetric concentration of 1,4-dioxane in air for exposure point 3 [ppm]
T = Ambient temperature [K]
MW = Molecular weight of 1,4-dioxane [g/mol]
X_{tank_clean} = Vapor pressure correction factor for equipment and storage tank cleaning [mol dioxane/mol water]
$RATE_{air\_speed}$ = Speed of air flow for evaporation from open surfaces, see Appendix E.13 [ft/min]
$D_{equip\_opening}$ = Diameter of equipment opening, see Appendix E.13 [cm]
$RATE_{ventilation}$ = Ventilation rate [ft^3/min]
F_{mixing} = Mixing factor [unitless]

Exposure point 3 (equipment cleaning) mass concentration of 1,4-dioxane in air is calculated using the following equation:

Equation_Apx F-25

$$Concentration_{EP3} = C_{v3} * \frac{MW}{V_m}$$

Where:

$Concentration_{EP3}$ = Mass concentration of 1,4-dioxane in air for exposure point 3 [mg/m^3]
C_{v3} = Volumetric concentration of 1,4-dioxane in air for exposure point 3 [ppm]
MW = Molecular weight of 1,4-dioxane [g/mol]
V_m = Molar volume [L/mol]

The total vapor 8-hour TWA based on the mass concentrations of 1,4-dioxane for exposure points 1 through 3 is calculated using the following equation:

Equation_Apx F-26

$$Total\_Vapor\_TWA = \frac{(Concentration_{EP1} * OH_{cont\_exposures\_adjusted} + Concentration_{EP2} * OH_{cont\_exposures\_adjusted} + Concentration_{EP3} * OH_{equip\_clean}))}{8}$$

Where:

$Total\_Vapor\_TWA$ = Full-shift 8-hour TWA of 1,4-dioxane vapor exposure [mg/m3]
$Concentration_{EP1}$ = Mass concentration of 1,4-dioxane in air for exposure point 1 [mg/m^3]
$Concentration_{EP2}$ = Mass concentration of 1,4-dioxane in air for exposure point 2 [mg/m^3]
$Concentration_{EP3}$ = Mass concentration of 1,4-dioxane in air for exposure point 3 [mg/m^3]
$OH_{cont\_exposures\_adjusted}$ = Duration of exposure for container unloading adjusted so that it is capped at 2 hours/day [hours/day]
OH_{equip_clean} = Duration equipment cleaning releases, see Appendix E.13 [hours/day]

F.9.2 Model Input Parameters

Table_Apx F-36 summarizes the model parameters and their values for the Monte Carlo simulation. Additional explanations of EPA's selection of the distributions for each parameter are provided after this table. High-end and central tendency exposures are estimated by selecting the 50th and 95th percentile values from the output distribution.

10934 **Table_Apx F-36. Summary of Parameter Values and Distributions Used in the Hydraulic Fracturing Exposure Modeling**

Input Parameter	Symbol	Unit	Constant Model Parameter Values	Variable Model Parameter Values				Rationale/ Basis
			Value	Lower Bound	Upper Bound	Mode	Distribution Type	
Ventilation Rate	$RATE_{ventilation}$	ft^3/min	132,000	132,000	237,600	—	Uniform	See Section F.9.3
Mixing Factor	F_{mixing}	none	0.1	0.1	1	0.5	Triangular	See Section F.9.4
Vapor Pressure of 1,4-dioxane	VP	Torr	40	—	—	—	—	Physical property
Molecular Weight of 1,4-dioxane	MW	g/mol	88.1	—	—	—	—	Physical property
Ambient Temperature	T	K	298	—	—	—	—	Process parameter
Universal Gas Constant	R	atm-cm^3/gmol-K	82.05	—	—	—	—	Universal constant
Molar Volume	V_m	L/mol	24.45	—	—	—	—	Physical property

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F.9.3 Ventilation Rate

The CEB Manual (U.S. EPA, 1991) indicates general outdoor ventilation rates in industry range from 132,000 to 237,600 ft^3/min in outdoor conditions. The underlying distribution of this parameter is not known; therefore, EPA assigned a uniform distribution, since a uniform distribution is completely defined by range of a parameter.

F.9.4 Mixing Factor

The CEB Manual (U.S. EPA, 1991) indicates mixing factors may range from 0.1 to 1, with 1 representing ideal mixing. The CEB Manual references the 1988 ACGIH Ventilation Handbook which suggests the following factors and descriptions: 0.67 to 1 for best mixing; 0.5 to 0.67 for good mixing; 0.2 to 0.5 for fair mixing; and 0.1 to 0.2 for poor mixing. The underlying distribution of this parameter is not known; therefore, EPA assigned a triangular distribution, which is completely defined by the range and mode of a parameter. The mode for this distribution was not provided; therefore, EPA assigned a mode value of 0.5 based on the typical value provided in the ChemSTEER User Guide (U.S. EPA, 2015a) for the *EPA/OPPT Mass Balance Inhalation Model*.

Appendix G SURFACE WATER CONCENTRATIONS

G.1 Surface Water Monitoring Data

G.1.1 Monitoring Data Retrieval and Processing

The complete set of 1,4-dioxane monitoring results stored in the WQP was retrieved in July 2022, with no filters applied other than the chemical name (NWQMC, 2022). This raw dataset included 12,471 samples. To filter down to only the desired surface water samples to include in this analysis, only samples with the "ActivityMediaSubdivisionName" attribute of "Surface Water" were kept, and among those, only samples with a "MonitoringLocationTypeName" that was one of the following:

- Spring
- Stream
- Wetland
- Lake
- Reservoir
- Impoundment
- Stream: Canal
- Stream: Ditch
- Facility Other
- Floodwater Urban
- River/Stream
- Great Lake
- Reservoir
- Lake
- River/Stream Intermittent
- River/Stream Perennial

After these steps, 1,449 surface water samples remained in the dataset. Samples flagged as QC blanks in the "ActivityTypeCode" column were then removed, leaving 1,359 surface water samples for analysis. Of these remaining samples, only 12 percent were results above the respective reported detection limit. This monitoring dataset is attached as *1,4-Dioxane Supplemental Information File: WQP Processed Surface Water Data* (U.S. EPA, 2023s).

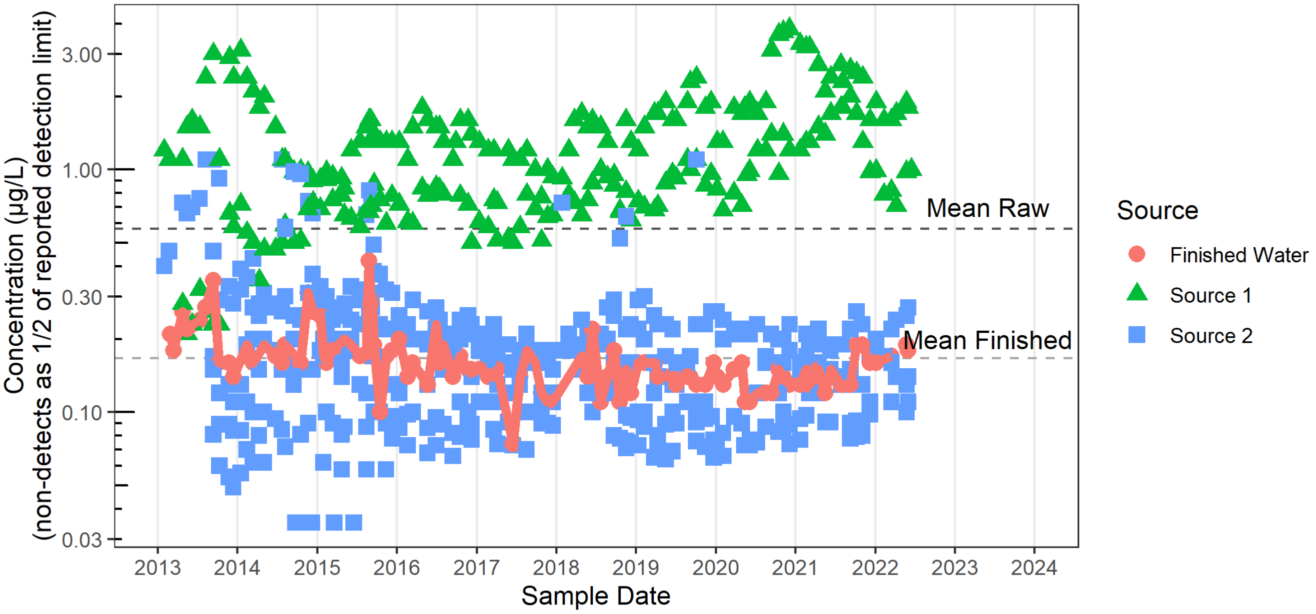
Monitoring data from drinking water systems were obtained from state drinking water databases (CA, MA, NY) and the Third Unregulated Contaminant Monitoring Rule (UCMR3) results database (CA Water Board, 2022; NY DOH, 2022; Commonwealth of Massachusetts, 2018; U.S. EPA, 2017d). For each, the complete history of 1,4-dioxane records was retrieved. The datasets were evaluated to ensure that duplicate samples were not included (*i.e.*, UCMR3 samples included in the state database results). UCMR3 records include a designation of source water at the facility level, and only samples denoted as being collected from facilities processing surface water were included in this analysis. For the sample data collected from state databases, water system information was retrieved from the Federal SDWIS database to identify water systems primarily drawing surface water as their source water (U.S. EPA, 2022f). Sample details were reviewed and screened to remove samples indicating they were collected from groundwater (*i.e.*, including "well" in the sampling point description). Samples collected from both raw water, and at the entry point to the distribution system were kept for the summary dataset, due to typical drinking water treatment processes not removing 1,4-dioxane. After these steps, 6,742 drinking water samples remained in the dataset, only 22 percent of which were results above the respective reported detection limit.

The sampling design of the UCMR3 dataset includes all PWSs serving more than 10,000 people and 800 representative PWSs serving 10,000 or fewer people around the country. In California, monitoring and reporting for 1,4-dioxane is currently voluntary. In Massachusetts, all community water systems (PWSs that serve at least 25 people at their primary residences or with at least 15 connections to primary residences) are required to monitor, while in New York all PWSs are required to monitor.

G.1.2 Raw and Finished Drinking Water

In analyzing drinking water monitoring data in Sections 2, 3, and 5, the conservative approach of treating both raw water and finished drinking water samples as representing 1,4-dioxane concentrations that could be served to PWS customers. The reason behind this is that the most common treatment processes utilized by PWS do not effectively remove 1,4-dioxane. EPA acknowledges that even without treatment to remove 1,4-dioxane, a PWS may utilize multiple sources of raw water, which could be combined to dilute concentrations of 1,4-dioxane. An example case is apparent in monitoring data retrieved from the state of Massachusetts.

Concurrent monitoring of raw and finished water at this PWS show that even with higher 1,4-dioxane concentrations at a particular intake or source water body, concentrations can be reduced by mixing (Figure_Apx G-1). Despite this treatment facility not utilizing advanced treatment that could remove 1,4-dioxane from the treated water, the finished water contains lower concentrations than what would be expected from the average concentration of raw water samples. This is due to multiple sources of water, and a greater portion of the water with a lower concentration being used.



Figure_Apx G-1. Example Raw and Finished Water Concentrations from a PWS without Processes to Remove 1,4-Dioxane

Some treatment processes can remove 1,4-dioxane from contaminated water sources (Broughton et al., 2019; Godri Pollitt et al., 2019; Otto and Nagaraja, 2007; U.S. EPA, 2006b). Advanced oxidation treatments (*e.g.*, hydrogen peroxide with ferrous iron, ozone treatment with ultraviolet light, etc.) have substantially lowered concentrations in treated water but may result in the formation of additional byproducts (bromates) and may increase the cost of water treatment. Granular activated carbon has also lowered 1,4-dioxane concentrations when contaminated water is in the 10 µg/L to 100 µg/L range. Due to the physical-chemical properties of the chemical substance (*e.g.*, water solubility, octanol-water partitioning coefficient) and the variable characteristics of granulated active carbon (*e.g.*, pore-size distribution, activation sites, and nonuniformity of lots), this treatment process does not consistently reduce 1,4-dioxane concentrations in contaminated water (Table_Apx G-1).

EPA assessed the prevalence of treatment processes that may more consistently remove 1,4-dioxane using treatment process information contained in the federal SDWIS database (see Table_Apx G-2). Less than one percent of community water systems (CWS) list oxidation processes which could more reliably reduce 1,4-dioxane concentrations, representing about 14 percent of the population served drinking water by CWSs.

Table_Apx G-1. Summary of Community Water Systems with Treatment Processes Capable of Removing 1,4-Dioxane

Process	Number of CWS	Percent of All CWS	Population Served Count	Percent of Population Served by CWS
Ozonation, Post	120	0.22	11,994,890	3.68
Ozonation, Pre	260	0.49	29,357,673	9.00
Peroxide	100	0.19	5,345,429	1.64
Activated Carbon, Granular	1,029	1.93	38,815,800	11.90

G.2 Surface Water Modeling

G.2.1 Hydrologic Flow Data

The NHDPlus V2.1 national seamless flowline network database was used as the source of stream or river (hereby referred to as stream) flow data for both the facility-specific and aggregate probabilistic modeling approaches. The NHD dataset is one of the largest national hydrologic datasets, containing delineated flowline networks, flow sequence data, and associated modeled flow values for >2.7 million stream segments (U.S. EPA; U.S. GS, 2016). The joint USGS-EPA data product represents one of the most comprehensive and functional datasets that can be applied for national-scale hydrologic modeling studies to date. The Enhanced Runoff Method (EROM) flow database, which includes modeled mean annual flows, as well as mean monthly flows, for each stream segment in the national flow network, is developed from multi-step process to estimate and calibrate hydrologic flows. This standard dataset is incorporated into recordkeeping and modeling across EPA programs, providing consistency and compatibility with projects such as EPA's ECHO database.

Lists of facilities discharging 1,4-dioxane directly and indirectly via transfers to disposal facilities were collected from EPA's TRI and DMR databases, as described in Appendix E. For each direct release facility, NPDES permit information associated with the facility's FRS Identification (FRS ID) was pulled from the ECHO database API, including the 14-digit NHDPlus reach code. When a facility-assigned reach code is missing in the ECHO database, the nearest neighboring NHD flowline and associated reach code within a 2 km radius was identified using GIS software. This process was repeated for the facilities reported as receiving indirect releases. The QE flow metrics from the EROM database were used, which represent modeled flows adjusted according to observed flows at USGS flow monitoring gages. QE values are reported by the user manual to be the "best EROM estimate of actual mean flow." These modeled flows are based on observed flows from the years 1971 to 2000. The mean annual and mean monthly modeled QE flows (QE) were extracted from the NHDPlus V2.1 database for the identified reaches. An individual reach code may be associated with multiple individual flowline segments in the NHDPlus V2.1 database, each with its own unique COMID identifier. Typically, there is very little variation in flow metrics between segments that share a reach code. When multiple segments were with the same reach code were returned during this process, the mean of each flow metric was calculated and applied to the associated facility. In two instances, the water body associated with a releasing facility was a lake or coastal water body, without a flow metric. In these cases, the facility flow (described below) was used, if available. For facilities with no available hydrologic or facility flow rates (or a modeled flow rate of zero), the lowest non-zero flow within the COU was used.

In addition to the receiving water body information, the Pollutant Loading Tool API was also queried for available records of water discharge rates from each facility, for each year of release records. The following facility flow data fields were acquired from the database: Facility Design Flow, Actual Average Facility Flow, Average Facility Flow. The Average Facility Flow record is most commonly available, and is preferentially selected to represent the facility flow, followed by the Actual Average Facility Flow, and finally the Facility Design Flow.

For both the facility-specific and probabilistic modeling approaches, the flow of the receiving water body is combined with a daily pollutant loading value to estimate a surface water concentration. For each modeled scenario, before calculating this concentration, the hydrologic flow value is checked against the best available facility flow. The modeled concentration is sensitive to the flow used in the calculation, particularly when that flow is very small. In reality, a small stream receiving a large volume of discharge would have its flow increased substantially by the facility flow rate and modeling the concentration using only the small stream's flow rate would result in erroneously high concentrations. When the facility flow is greater than the stream flow, the facility flow is used to calculate the resulting concentration instead of the stream flow. If a facility flow is not available, the modeled stream flow is used.

G.2.2 Facility-Specific Release Modeling

In previous TSCA risk evaluations, EPA applied the E-FAST 2014 tool to conduct facility-specific modeling. In an effort to make the calculations more flexible and rapidly repeatable, rather than using the E-FAST model directly, the formulas employed in E-FAST were written into an Excel workbook. This allowed for the incorporation of the NHDPlus V2.1 flow data as a refinement of the methodology, and for manual adjustments to parameters as needed. Therefore, facility-specific modeling was conducted using the methodology and logic of the E-FAST 2014 tool, but in a deconstructed form that provided an opportunity to update flow metrics to improve overall confidence in the resulting concentrations.

In the past, E-FAST modeling for risk evaluations have used several flow metrics: the arithmetic mean flow, the harmonic mean flow, the 30Q5 (lowest 30-day average flow that occurs in a 5-year period), and 7Q10 (lowest 7-day average flow that occurs in a 10-year period). Of these flow metrics, only a modeled arithmetic mean flow can be obtained from the EROM flow database. Without a national dataset of these additional flow statistics with the resolution and reliability of the EROM dataset, due to the challenges of modeling these values across the national dataset, an alternative method to estimate these metrics consistent with our application of the E-FAST methodology was adapted for this modeling effort. Regression equations from the E-FAST technical manual relating the arithmetic mean, harmonic mean, 30Q5, and 7Q10 flows were used to solve for the desired metrics. In addition to an annual arithmetic mean flow, the EROM database 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 arithmetic mean and substitute 30Q5 flow were then plugged into the regression equations used by E-FAST to convert between flow metrics and solved for the remaining terms:

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

Where:

$\boldsymbol{7Q10}$ = the modeled 7Q10 flow, in MLD
$\boldsymbol{30Q5}$ = the lowest monthly average flow from NHD, in MLD

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

Where:

$\boldsymbol{HM}$ = the modeled harmonic mean flow, in MLD
$\boldsymbol{AM}$ = the annual average flow from NHD, in MLD
$\boldsymbol{7Q10}$ = the modeled 7Q10 flow from the previous equation, in MLD

For each facility, the year of highest annual loading was chosen as a conservative screen for potential aquatic releases. Average daily loadings are calculated by dividing the annual loading by the number of days of operation per year. Three different scenarios for operating days were evaluated: one day, 30 days, and the maximum expected days of operation listed in Appendix E.2. The 1- and 30-day scenarios provide more conservative approaches to evaluating resulting stream concentrations and allow more confidence in screening out risk from facilities (*i.e.*, identifying which facilities have releases that do not exceed any thresholds for risk). Conversely, the maximum number of days of operation provides more confidence for identifying risk that exceeds a threshold.

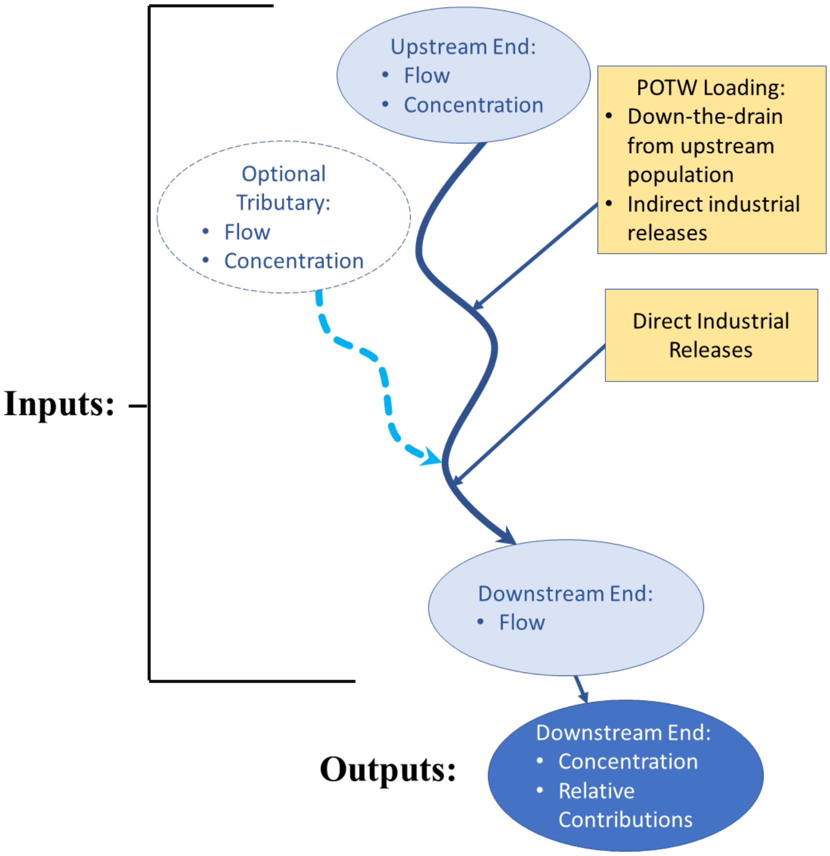
For each scenario, the in-stream concentration due to the release is calculated by:

$$Concentration\ \left(\frac{\mu g}{L}\right) = \frac{Daily\ Load\ \left(\frac{kg}{day}\right) *\ 10^{9}\ \left(\frac{\mu g}{kg}\right)}{Flow\ (MLD) *\ 10^{6}\left(\frac{L}{ML}\right)}$$

G.2.3 Aggregate and Probabilistic Modeling

G.2.3.1 The Fit-For-Purpose Aggregate Surface Water Model

The EWISRD-XL model was developed as a fit-for-purpose model to consider industrial 1,4-dioxane releases and DTD loading of 1,4-dioxane in aggregate. It applies a steady-state mass-balance approach to estimate surface water concentrations and relative contributions from different sources for a single point on a stream at a single point in time. A summary of the inputs and outputs used in this application of the model are presented in Figure_Apx G-2.



Figure_Apx G-2. Schematic of the General Fit-for-Purpose EWISRD-XL Model

The model produces an estimation of surface water concentrations at the downstream end of a stream segment, by combining the total upstream mass flux and dividing by the downstream flow rate:

$$Conc_{down} = \frac{\left(Flux_{up} + Flux_{trib} + Flux_{DTD} + Flux_{release}\right)}{Flow_{down}}$$

Where:

$Conc_{down}$ = the 1,4-dioxane concentration at the downstream end (µg/L)
$Flux_{up}$ = the mass flux into the stream at the upstream end (µg/day)
$Flux_{trib}$ = the mass flux into the stream from a tributary (µg/day)
$Flux_{DTD}$ = the mass flux into the stream from DTD loading (µg/day)
$Flux_{Release}$ = the mass flux into the stream from a direct release (µg/day)
$Flow_{down}$ = the stream flow at the downstream end (L/day)

The EWISRD-XL model assumes that 1,4-dioxane stays within the water column as it travels downstream, with no partitioning to sediment or air, and no biological uptake. The total mass flux into the modeled reach is conserved and assumed to be equal to the mass flux out at the downstream end. These assumptions are based on the physical chemistry properties (*e.g.*, water solubility, Henry's law constant) and fate characteristics (*e.g.*, biodegradability) and appear to represent the behavior of the chemical fairly well over the relatively small distances covered by most of the case studies.

The mass flux from the upstream end of the segment, or a tributary, is calculated from a known flow rate and concentration at that location:

$$Flux_{up,trib}\left(\frac{ug}{day}\right) = Conc_{up,trib}\left(\frac{ug}{L}\right) * Flow_{up,trib}\left(\frac{L}{day}\right)$$

The mass flux from DTD loading is calculated using a per capita estimation of loading and the estimated population contributing to DTD loading of the stream segment. Although the upstream flux incorporates all expected DTD loading from upstream of the segment being modeled, the DTD loading estimated by the model captures the expected loading between the upstream and downstream ends of the modeled segment. Per capita estimates of DTD loading are derived from SHEDS-HT model output, for nine product types (Table_Apx G-2). The product mass ratios described in Appendix E were used as inputs to the SHEDS-HT modeling, along with the default model parameters. The DTD component of the SHEDS-HT output was isolated and evaluated for use in the EWISRD model. SHEDS-HT models non-commercial consumer product use and reports a distribution of per capita DTD loading values. The mean DTD loading value was applied in the EWISRD model to represent general non-commercial uses, while the 90th percentile DTD loading value was applied to represent commercial uses of the same products.

Table_Apx G-2. Summary of per Capita DTD Loading Estimates from SHEDS-HT Modeling

Consumer Products	Non-commercial DTD Loading (g/day per Capita)	Commercial DTD Loading (g/day per Capita)
Antifreeze	0.0000	0.0000
Dish Soap	0.0235	22.0016
Dishwasher Detergent	0.0003	0.4906
Spray Polyurethane	0.0000	0.0000
Laundry Detergent	0.0004	0.2437
Surface Cleaner	0.0014	38.5925
Textile Dye	0.0000	0.0000
Floor Lacquer	0.0000	0.0000
Latex Wall Paint	0.0008	0.0000

For case study applications of the EWISRD-XL model, populations contributing to DTD loading within the case study area were estimated using the 2020 Census Designated Places polygons and accompanying population records (U.S. Census Bureau, 2015). By visual inspection, Census places alongside water bodies contributing flow to the stream segment of interest were identified, and the total population was summed and entered into the EWISRD-XL model. The entirety of the estimated population was assumed to be contributing to non-commercial DTD loading. The commercial DTD loading was calculated using average proportions of the population expected to have occupations resulting in commercial use of the consumer products, derived from the 2020 U.S. Bureau of Labor Statistics Current Population Survey (U.S. BLS, 2022).

G.2.3.2 Case Studies to Validate Aggregate Model

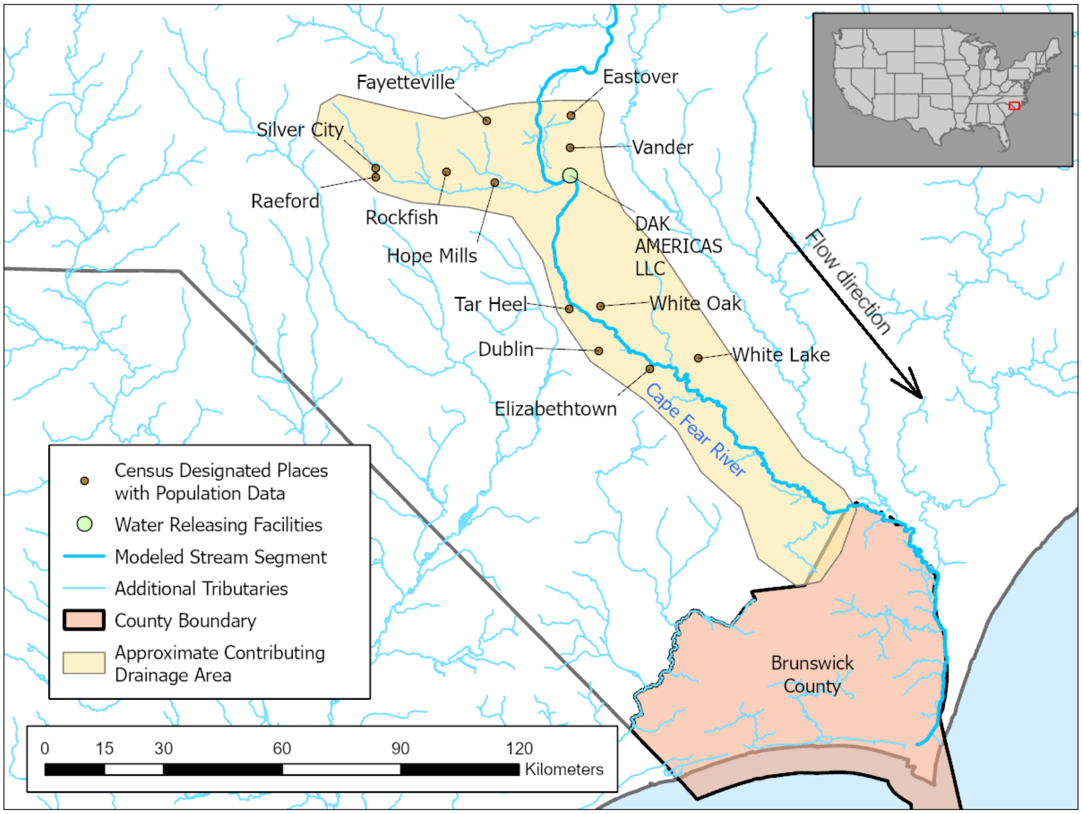
Case studies of locations with adequate 1,4-dioxane surface water monitoring data were conducted with the EWISRD-XL model, to validate the performance of the fit-for-purpose model (Table_Apx G-3). Rather than targeting a conservative estimate of release concentrations, the intention was to best reproduce the observed monitored concentrations. Therefore, the modeled concentrations within the case studies represent more average conditions for the time periods modeled. Overall, the application of the EWISRD-XL model, which incorporated facility releases combined with DTD loading estimations derived from SHEDS-HT, resulted in reasonable, if not conservative, estimates of average aggregate concentrations of 1,4-dioxane downstream of multiple sources.

Table_Apx G-3. Summary of Case Study Locations Including Modeled and Observed Surface Water Concentrations

Location	Modeled Water Body	Modeled Concentration (μg/L)	Observed Concentration (μg/L)	Inputs Included
Brunswick County, NC	Cape Fear River	(range, see below)	(range, see below)	Direct industrial release, DTD, and upstream concentration
Columbia, TN	Duck River	0.35	<0.07–0.22	Only DTD
East Liverpool, OH	Ohio River	0.61	<0.07	Direct industrial release, DTD, and upstream concentration

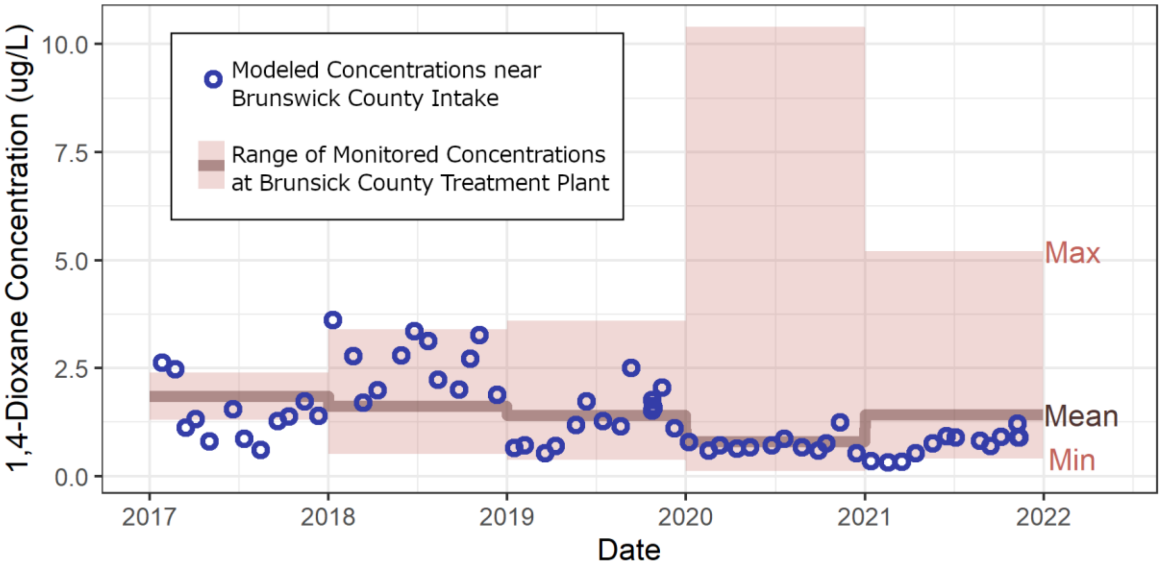
Brunswick County, NC – Cape Fear River

The Cape Fear River upstream of the Brunswick County, NC drinking water intake was selected as a case study to test the model due to abundant monitoring data in the region (Figure_Apx G-3). At the upstream boundary of the modeled reach, approximately monthly monitoring data from 2017 to 2021 at the Cape Fear River intake of the PWS in Fayetteville, NC was used to provide the concentration of 1,4-dioxane at the upstream end of the model. The direct release from the DAK Americas LLC plant in Fayetteville was included in the modeling (green dot in Figure_Apx G-3). The daily loading from this direct release was calculated as the average daily release from 250 days of operation, using the TRI annual release records from 2017 to 2021, which ranged from 173 to 7,965 kg/year. In this case study, the availability of concentrations of 1,4-dioxane in the Cape Fear River at Fayetteville, NC meant that any DTD contributions from further upstream were already accounted for in the modeling, and therefore only DTD loading between Fayetteville and Brunswick County needed to be quantified. The population contributing to DTD loading was calculated by visually approximating the drainage area contributing to the modeled segment using the NHD flowline network, from the upstream point near Fayetteville, NC to the downstream endpoint near the Brunswick County intake and summing the 2020 Census populations for the Census Designated Places within the boundary. At the downstream end, monitoring data, reported as a minimum, average, and maximum concentration, from the Brunswick County drinking water plant on the Cape Fear River were collected from Consumer Confidence Reports released by the county for 2017 to 2021 (Brunswick County, 2022).



Figure_Apx G-3. Map of Brunswick County, NC Model Case Study
Note: The downstream end of the modeled reach coincides with the location of the Brunswick County drinking water intake on the Cape Fear River, which is located near where the Cape Fear River enters Brunswick County.

A separate model run was conducted for each measurement of 1,4-dioxane concentration in the Cape Fear River near Fayetteville, NC (66 total), to incorporate more temporally-specific flow data and produce a corresponding downstream modeled concentration at the Brunswick County intake. For each year, the corresponding calculated average daily release from the DAK Americas LLC plant was included as an input. For each month, the average corresponding monthly flow from NHDPlus V2.1 was used for the upstream and downstream hydrologic flow inputs to the model. A static total contributing population of 191,201 for the DTD component was used. Results from the 66 model runs were compared with the values reported by Brunswick County (Figure_Apx G-4). The EWISRD-XL model file used for this case study included as *1,4-Dioxane Supplemental Information File: EWISRDXL BrunswickCountyNC Case Study* (U.S. EPA, 2023o).

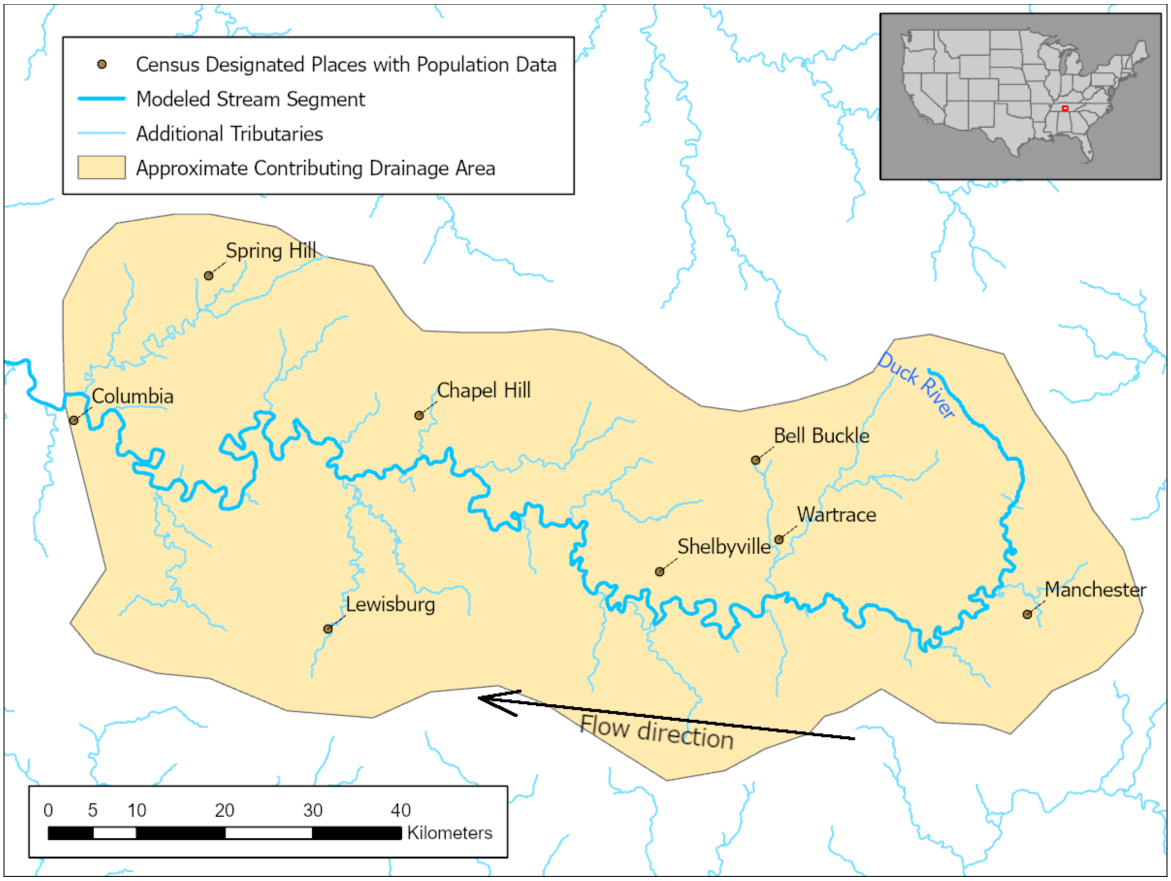


Figure_Apx G-4. Plot Comparing Results from Brunswick County Case Study Modeling with Observed Concentrations

Modeled surface water concentrations generally fell within the ranges reported from monitored concentrations. Wide ranges of both monitored and modeled values were noted, indicating variability among inputs to the system. While the direct discharge, DTD and flow components of the model represent average daily or monthly values, finer-scale variations in these values could account for the variability in monitored observations. In this case study, the upstream input concentration ranged from <0.07 to 5.9 µg/L, and the output was sensitive to this upstream concentration. Modeled downstream concentrations could only be produced for days with available upstream concentrations, so the full range of variability could not be captured in this approach. The overall modeled average concentration from 2017 to 2021 was 1.35 µg/L, and the annual averages for 2017 to 2021 reported by Brunswick County ranged from 0.8 to 1.85 µg/L. The general tendency of the model results to follow the mean observed values reported from Brunswick County indicate that the assumptions of the model and inputs effectively approximate resulting downstream concentrations of 1,4-dioxane resulting from aggregate down the drain and facility releases.

Columbia, TN – Duck River

The Columbia, TN case study was selected because of available monitoring data from the Columbia PWS located on the Duck River (Figure_Apx G-5) with monitored detections of 1,4-dioxane reported under UCMR3 (U.S. EPA, 2017d). Its location near the headwaters of the Duck River meant that there were no known upstream direct facility releases of 1,4-dioxane into this water body. Therefore, it was assumed that any 1,4-dioxane in surface water detected in the Duck River at Columbia, TN, would be due to the DTD contribution from the upstream population.

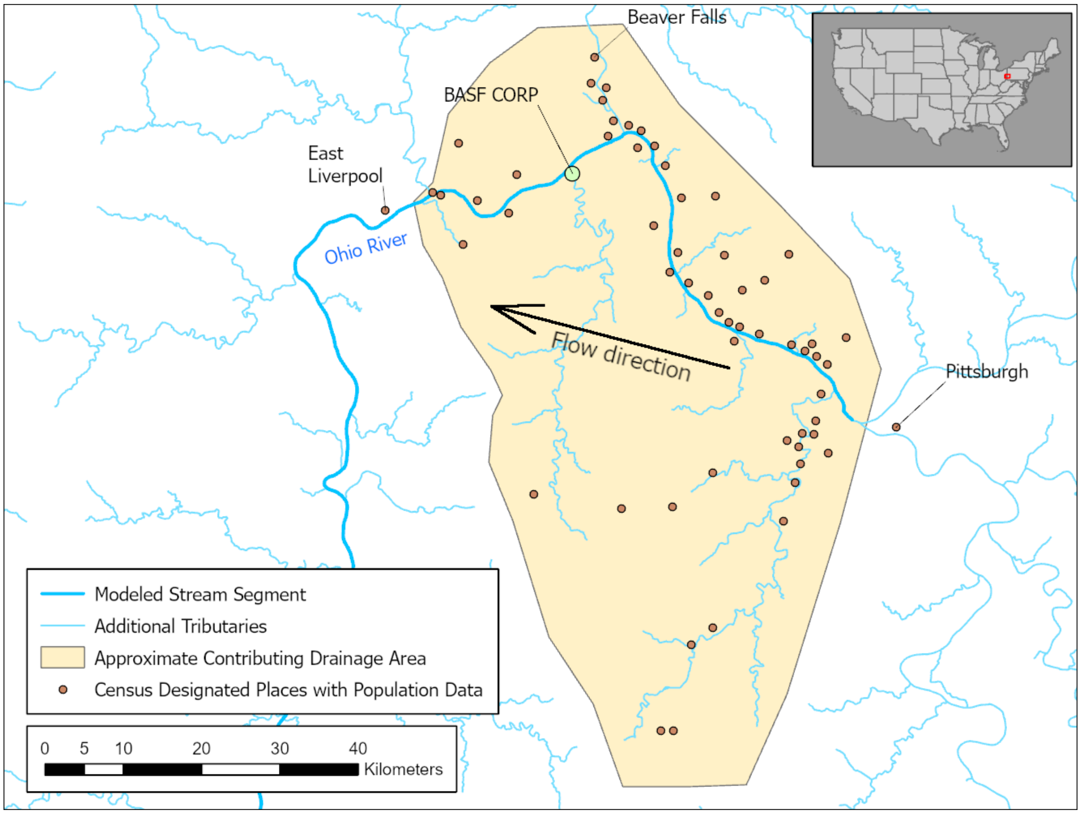


Figure_Apx G-5. Map of the Columbia, TN, Case Study

The upstream drainage area contributing flow to the Duck River at Columbia was visually approximated from the NHD flowline network, and the 2020 populations of the identified Census Designated Places were summed as inputs to the DTD component of the model (totaling 70,974 people). The mean annual flow at the downstream end from NHDPlus V2.1 was used. The four reported measurements of 1,4-dioxane at the Columbia PWS ranged from <0.07 (not detected) to 0.22 µg/L. Because of the static DTD inputs, a single model run was conducted using a mean annual flow rate, resulting in a modeled concentration at the downstream end of 0.35 µg/L. The intent of this case study was to target the effectiveness of the model to estimate the DTD contribution to instream concentrations, and the results suggest that the model assumptions for DTD loading are a reasonable but conservative estimate of downstream concentrations. The EWISRD-XL model file used for this case study included as *1,4-Dioxane Supplemental Information File: EWISRDXL ColumbiaTN Case Study* (U.S. EPA, 2023p).

East Liverpool, OH

The case study for the Ohio River at East Liverpool, OH, was selected due to the availability of coincident UCMR3 monitoring data (U.S. EPA, 2017d) and a known direct release from a facility (Figure_Apx G-6). For the sake of averaging reported monitoring measurements, half of the reported detection limit of 0.07 µg/L was applied for non-detects. At the upstream end of the model, the average concentration measured at the Pittsburgh, PA, PWS of eight samples from 2014 to 2015 (via UCMR3) was used (0.23 µg/L). An additional tributary, the Beaver River, was included in the model using UCMR3 monitoring data from Beaver Falls, PA. The average concentration of four samples from 2013 to 2014 reported from the Beaver Falls PWS was 2.66 µg/L. In this case study, the availability of concentrations of 1,4-dioxane in the Ohio River at Pittsburgh, PA, and the Beaver River at Beaver Falls meant that any DTD contributions from further upstream were already accounted for in the modeling, and therefore only DTD loading between these locations and East Liverpool, OH, needed to be quantified. These PWS locations can be approximated by the points representing the respective towns and cities in Figure_Apx G-5. The 2020 population from the Census Designated Places within the approximate drainage area were summed for the DTD component, totaling 559,505 people. Annual releases were only available for 2018 and 2019 from the BASF Corp facility, ranging from 2.98 to 3.66 kg. The average daily loading from this facility was calculated from the greater of these two numbers divided by 250 days of operation.



Figure_Apx G-6. Map of the East Liverpool, OH, Case Study

All four of the reported sample results at East Liverpool, OH, from 2013, were below the detection limit of 0.07 µg/L. The modeled concentration from all of the inputs resulted in 0.61 µg/L at the downstream end, which appears to be an overestimation for this system, based on the monitoring data. Due to the timing of samples at the upstream and downstream ends not aligning, average values were used in this case study, but some temporal variation may still be missed by these values. Additionally, results of this case study appeared to be sensitive to the high concentrations reported for the Beaver River tributary as well as the high population estimated to be contributing to the DTD component. The DTD component was found to result in a small overestimation in the second case study, where the contributing population was nearly an order of magnitude lower. The EWISRD-XL model file used for this case study included as *1,4-Dioxane Supplemental Information File: EWISRDXL LiverpoolOH Case Study* (U.S. EPA, 2023q).

G.2.3.3 The Probabilistic Model

The probabilistic EWISRD-XL-R model was developed by creating an R script that interfaces with the EWISRD-XL document (via the XLConnect R library (Mirai Solutions GmbH, 2021)). In this arrangement, the underlying modeling and calculation process is handled within an EWISRD-XL document. The accompanying R script handles the loading and arrangement of input data, then iteratively feeds inputs to the model and retrieves the outputs, which are then summarized and visualized. This allows the EWISRD-XL-R model to run thousands of iterations rapidly.

Although the individual facility modeling focused on only the highest releases per facility, using the EWISRD-XL-R model allowed the inclusion multiple years of release data available for each facility, and inclusion of multiple flow estimations, to produce a range of estimated concentrations resulting from releases. The ability to aggregate multiple inputs within the model also allowed the inclusion of background 1,4-dioxane concentrations expected to be present in waterways from DTD or other unregulated sources.

The EWISRD-XL-R model, as applied for the COU-specific probabilistic model, has four major components:

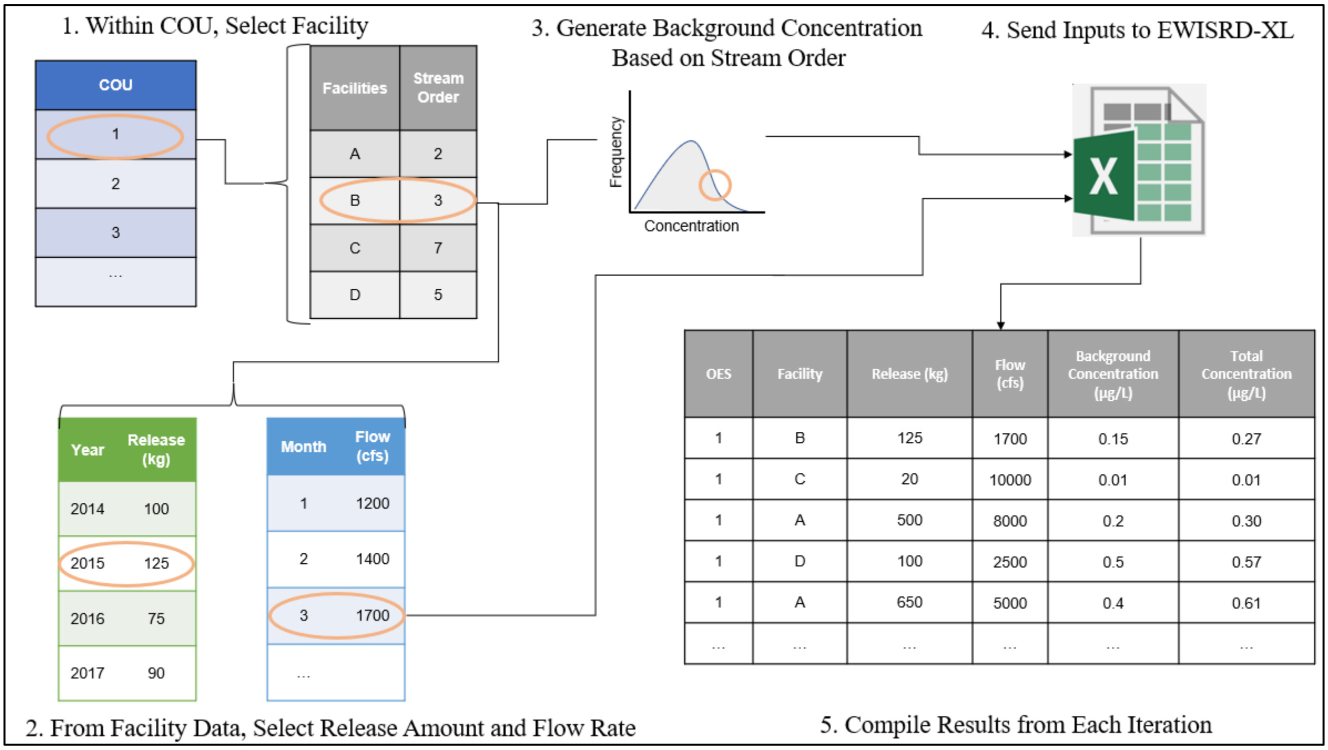
1. **Load and prepare the background concentration data.**
   While the model is capable of estimating DTD loading directly from contributing populations, there is some uncertainty about the distances over which the assumptions inherent in this calculation remain accurate (including assumptions of persistence in the water column, the rates of DTD loading, and that the entire upstream population contributes to the DTD loading). Furthermore, although estimating the population contributing to specific reaches is viable for a case study, that information is not readily available for each facility release. For these reasons, the background component of the probabilistic modeling is estimated using the concentrations detected at PWSs. The background data used to inform this estimation (Figure 2-9) only includes monitoring data for PWSs that were not found to be located downstream from known 1,4-dioxane releasing facilities, in order to represent only concentrations from DTD loading and other unregulated releases.

   To appropriately pair background data with releasing facilities, the background concentrations and facilities were stratified by the Strahler stream order of the associated NHDPlus stream reach. For each stream reach, an empirical cumulative distribution function (ECDF) was created using the Kaplan-Meier method, which has been recommended for estimating the distributions of datasets, particularly with a high percent censored data (Gillespie et al., 2010). The ecdfPlotCensored function within the EnvStats R library is called to develop each ECDF (Millard, 2013), which is then wrapped in a solver function for the inverse of the ECDF. The inverse ECDF solver function can then receive an input of a percentile and return the corresponding background concentrations from the distribution. A random value from the stream-order-specific background distribution can be generated by calling the inverse ECDF solver function with a single input value from a random uniform distribution between 0 and 1.

2. **Load and combine the facility release and flow data.**
   As described in Section B.2.1, stream flow data (mean annual, and mean monthly for each month of the year) are retrieved for each releasing facility, as well as facility flow data. For the probabilistic modeling, all available years of release data, from both TRI and DMR, are loaded into the model, and the monthly flow averages from NHDPlus are joined to them.

3. **Perform a loop of model runs per COU.**
   The Monte Carlo simulations are then conducted with 10,000 model iterations per COU. In each model iteration, a random facility within the COU group and a random year of release is selected. Of the 12 available monthly average flows associated with that facility, one is randomly selected. If the selected flow rate is less than the facility flow rate, the facility flow rate is used instead. For the stream order of the reach associated with the releasing facility, a random background concentration

is generated. The resulting combination of daily loading from a direct release, flow, and upstream concentration are then passed to the EWISRD-XL document as inputs. For the probabilistic modeling, the resulting concentration is calculated at the point of release, so the EWISRD-XL model is configured in an arrangement where the downstream flow is equal to the upstream flow. The resulting stream concentration, as well as the percent contributions of the direct release and the background loading to that concentration, are retrieved from the model outputs and logged. The resulting output table records 10,000 combinations of modeled concentrations from different flow, release, and background concentration combinations. For each iteration, the total stream concentration (facility release + background) and the stream concentration due to only the facility release are recorded. A schematic of the flow of data within the probabilistic model is presented in Figure_Apx G-7.



OES	Facility	Release (kg)	Flow (cfs)	Background Concentration (µg/L)	Total Concentration (µg/L)
1	B	125	1700	0.15	0.27
1	C	20	10000	0.01	0.01
1	A	500	8000	0.2	0.30
1	D	100	2500	0.5	0.57
1	A	650	5000	0.4	0.61
…	…	…	…	…	…

Figure_Apx G-7. Schematic of the Flow of Data within the EWISRD-XL-R Probabilistic Model

4. Summarize and visualize the model output.

The model outputs are then summarized as percentiles and visualized as histograms. A comparison of the modeled facility release and the randomly generated background concentration is conducted for each iteration and summarized. This additional check can indicate whether, within a given COU, the expected concentrations in surface water due to permitted releases from facilities are typically greater than the expected background concentration from DTD and other non-regulated releases. The EWISRD-XL-R script is included as *1,4-Dioxane Supplemental Information File: EWISRD-XL-R probabilistic model code* (U.S. EPA, 2023a).

G.2.3.4 Modeling Ranges of DTD Contributions

The SHEDS-HT model was applied to generate distributions of DTD loading per capita resulting from products listed in Table_Apx G-4. The default scenarios and variables included with version 0.1.9 of SHEDS-HT were used. Product weight fractions generated during the engineering phase of this risk evaluation were used as inputs to the modeling. For each product, 10,000 iterations of the model were run to generate a distribution of results. Only the DTD component of each set of exposure results was pulled from the generated results, and products of the same type were summed together to summarize the per capita DTD loading by product type (Table_Apx G-4).

Table_Apx G-4. Distribution of per Capita DTD Loading, in G/Day, by Product, for Non-commercial Uses Modeled by SHEDS-HT

Product	Q10%	Q25%	Q50%	Q75%	Q90%	Q99%	Mean	SD
Antifreeze	0	0	0	0	0	0	0	0
Dish Soap	0	0	9.97E−03	2.70E−02	5.81E−02	2.08E−01	2.35E−02	5.04E−02
Dishwashing Detergent	0	0	0	5.33E−05	8.65E−04	4.63E−03	3.06E−04	1.06E−03
SPF	0	0	0	0	0	0	0	0
Surface Cleaner	0	0	0	4.57E−04	4.09E−03	2.09E−02	1.43E−03	4.84E−03
Laundry Detergent	0	0	1.50E−04	4.57E−04	1.03E−03	3.53E−03	4.01E−04	8.86E−04
Dye	0	0	0	0	0	0	0	0
Floor Lacquer	0	0	0	0	0	0	0	0
Paint	0	0	0	0	0	0	7.87E−04	2.88E−02

SHEDS-HT models consumer (non-commercial) uses of products, so the mean per capita DTD loading output from the model was applied to represent the average non-commercial per capita DTD loading. To represent increased usage by commercial applications, the 99th percentile per capita DTD loading was applied for commercial uses. The number of commercial users of products was determined using the national average proportion of the population expected to be employed in the following occupations, based on the 2020 U.S. Bureau of Labor Statistics Current Population Survey (Table_Apx G-5) (U.S. BLS, 2022).

Table_Apx G-5. Proportions of Population Expected to Contribute to DTD Loading through Commercial Activities and Product Uses

Product	Occupation	Proportion of Population
Antifreeze	Automotive service technicians and mechanics	0.00225
Dish Soap	Dishwashers	0.00055
Dishwasher Detergent	Dishwashers	0.00055
Spray Polyurethane	Insulation workers	0.00015
Surface Cleaner	Janitors and building cleaners	0.00615
Laundry Detergent	Laundry and dry-cleaning workers	0.00036
Surface Cleaner	Maids and housekeeping cleaners	0.00350

Product	Occupation	Proportion of Population
Textile Dye	Textile machine setters, operators, and tenders	4.82E−05
Floor Lacquer	Carpet, floor, and tile installers and finishers	0.00051
Latex Wall Paint	Painters and paperhangers	0.00157

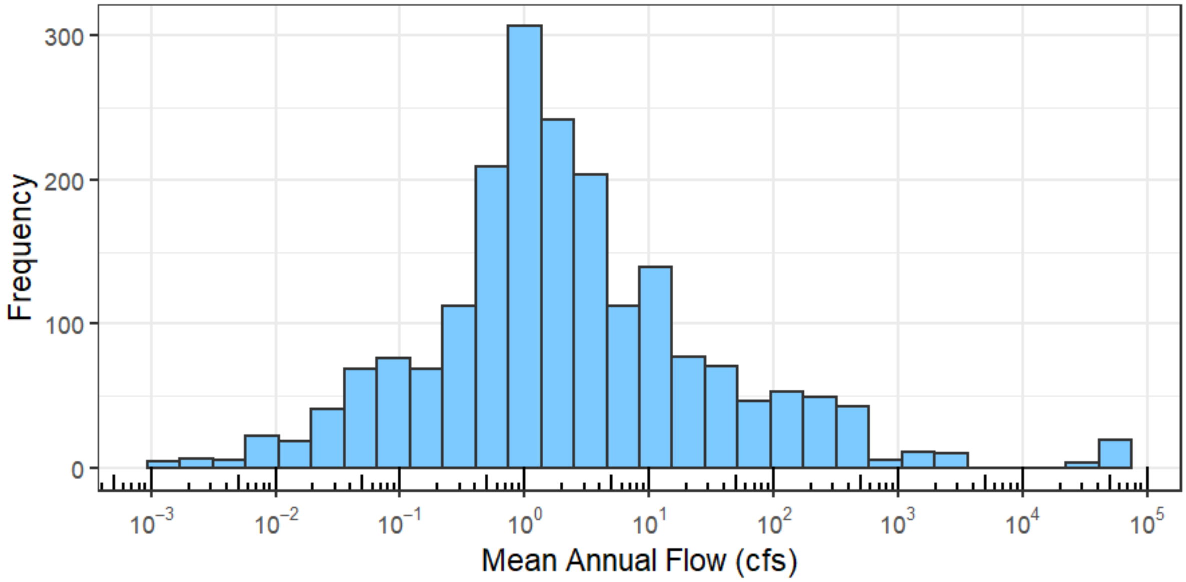
To assess the potential range of concentrations resulting from DTD loading from various populations, the above DTD loading values were applied to a range of population sizes, from 100 to 1,000,000 people. These loadings were applied to a range of mean annual flows, from 300 to 30,000 cfs, as individual runs of the EWISRD-XL-R model. No other input sources were including in the modeling, so that the resulting surface water concentrations were entirely due to the DTD loading. While the largest populations would be expected to discharge wastewater (*i.e.*, from a POTW), to a larger receiving water body, the full range of combinations of flow and contributing populations was analyzed.

G.2.3.5 Modeling Concentrations in Surface Water from Hydraulic Fracturing

The potential concentrations in surface water adjacent to hydraulic fracturing operations were modeled from the distribution of loadings to surface water and stream flow data for reaches located near hydraulic fracturing operations.

A set of 10,000 random values from the Monte Carlo distribution described in Appendix F.7, was generated to represent the range of loading values to surface water. These values were generated by employing a method similar to the generation of random values from background distributions described in Appendix G.2.3.4. The paired percentile and loading values from the Monte Carlo results were used to establish an empirical cumulative distribution function, for which the inverse could then be solved. A uniform distribution of percentile values between 0 and 1 were input into the resulting function to generate the 10,000 loading values used for this analysis.

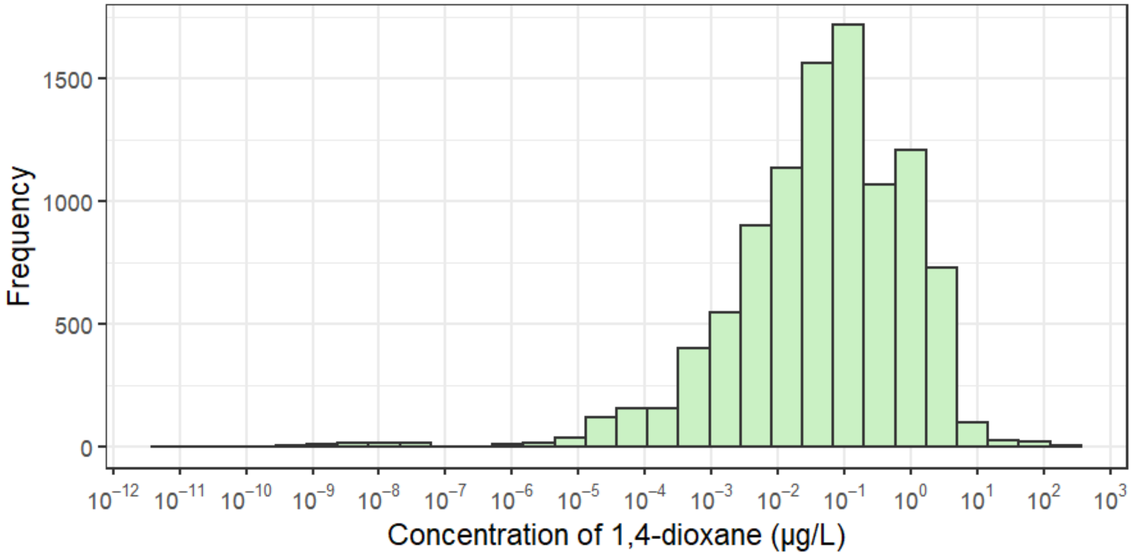
Mapped well locations of hydraulic fracturing operations reporting 1,4-dioxane in the wastewater re retrieved from the Fracfocus database (GWPC and IOGCC, 2022). To identify stream segments near the hydraulic fracturing operations, which can take place across large areas, a 5 km buffer was drawn around each well. Flow data from the 2,053 NHDPlus v2.1 stream segments intersecting these buffers were collected and reviewed. Of the reaches identified, 76 percent were found to have modeled mean annual flows less than 10 cfs (Figure_Apx G-8).



Figure_Apx G-8. Distribution of Mean Annual Modeled Flow Rates for NHDPlus V2.1 Reaches Identified within 5 km of Hydraulic Fracturing Wells Reporting 1,4-Dioxane

While the volumetric rate of discharge from hydraulic fracturing operations to surface water were not readily available, it was assumed that the concentrations in receiving streams with flows less than 10 cfs would be substantially impacted by the volume of wastewater from the operation. For this analysis, flows below 10 cfs were excluded from the pool of flows selected for modeling. Based on the assumption that the rates of discharge from hydraulic fracturing operations are greater, these lower flows would result in unrealistically high estimates of stream concentrations resulting from these releases. From the remaining 486 flow rates, 10,000 values were randomly sampled with replacement.

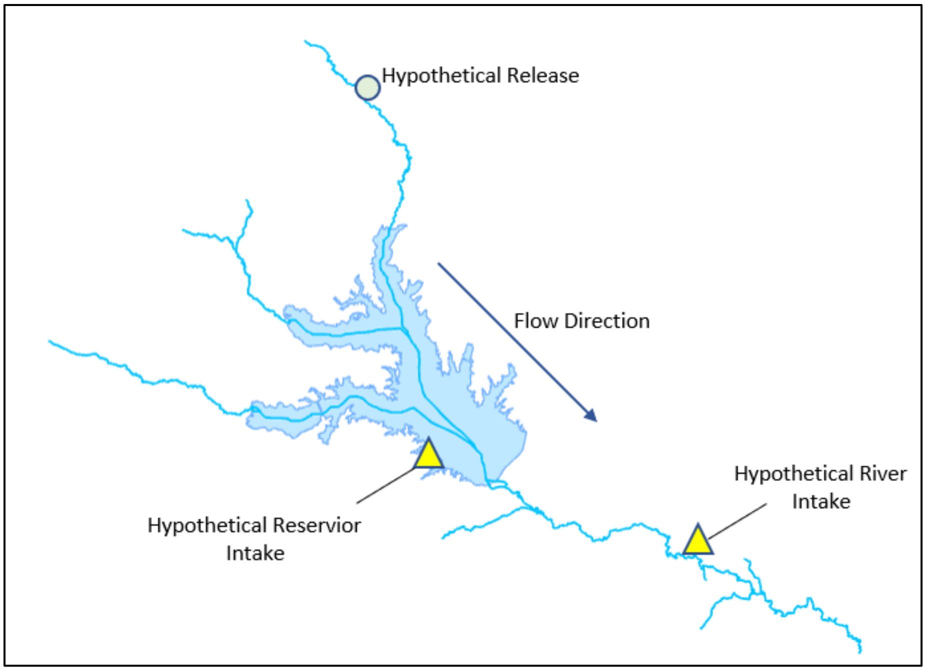
The EWISRD-XL-R model was used to model the concentrations resulting from the 10,000 generated loading values paired with the 10,000 stream flow rates (Figure_Apx G-9). Due to the nature of using a Monte Carlo distribution to generate the release loadings, and the sensitivity to the results of handling the nearby stream flow data, the tails of this distribution (*i.e.*, the highest and lowest percentiles) have a high degree of uncertainty.



Figure_Apx G-9. Distribution of Modeled Ranges of 1,4-Dioxane Concentrations in Streams near Hydraulic Fracturing Wells Reporting 1,4-Dioxane

G.2.4 Assessing Downstream Drinking Water Intakes

The focus of this assessment was to identify known surface water intakes occurring downstream from the facility releases modeled in Section 2. Locations of surface water intakes for PWSs were obtained from the SDWIS Federal Data Warehouse for assessing proximity to facilities releasing 1,4-dioxane to surface water, using the 2nd quarter 2022 version of the data (U.S. EPA, 2022f). The NHDPlus v2.1 national flowline network and water body geospatial information were used to conduct the assessment. As described in Appendix G.2.1, the reach codes to which facilities released 1,4-dioxane were identified. To associate reach codes with surface water intake locations, the nearest flowline or water body to each intake was analyzed using the Near tool in ArcGIS Pro. A crosswalk between reach codes in the flowline network and water bodies in the flowline network was developed from the intersections of the flowline network with the water body coverage (*e.g.*, reservoirs constructed from damming rivers, which may have intakes located closer to their banks than the centerline of the river, Figure_Apx G-10).



Figure_Apx G-10. Generic Schematic of Hypothetical Release Point with Surface Water Intakes for Drinking Water Systems Located Downstream

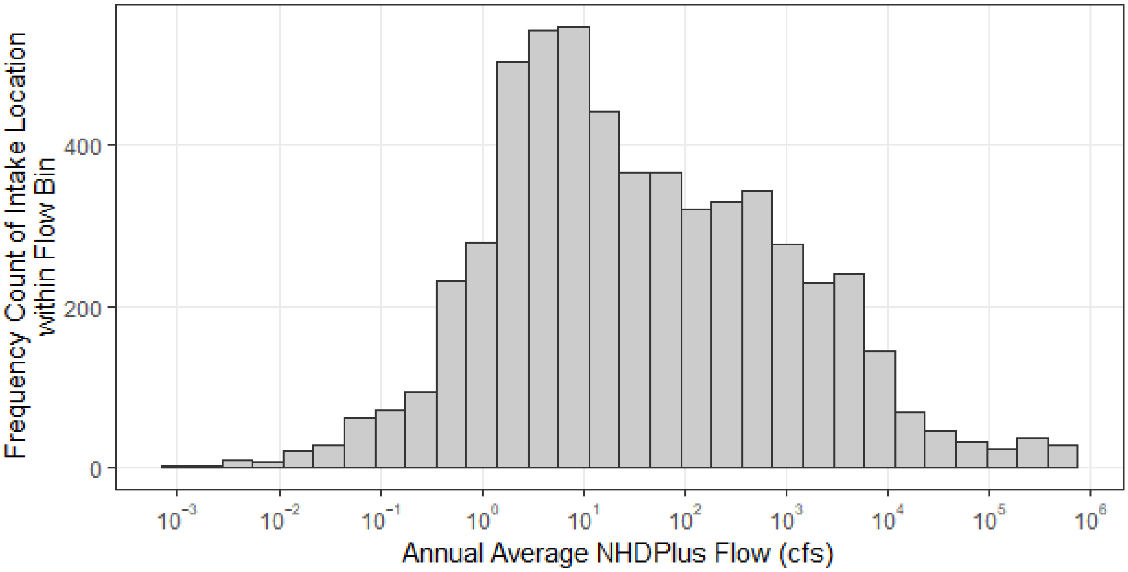
An R script was developed to search downstream from the reach codes with facilities, using the node and reach code sequence information within NHDPlus. The script functions by incrementally stepping downstream to the next reach and evaluating whether a surface water intake is associated with the reach code. When a reach with an intake is identified, the details of the PWS and the distance traveled downstream are recorded, and the script continues until a dead end, or a maximum search distance in achieved for each release. For this assessment, a maximum search length of 500 reaches (approximately 1,000 km) was used.

Overall, about 31 percent of individual facilities found to have an adult lifetime cancer risk for drinking water above 1×10^{-6} were located within 250 km upstream from a known DW). It should be noted that risk estimates are calculated for concentrations in the receiving water at the point of release, and some decrease in concentration due to dilution would be expected at the location of a DWI further downstream. For all OESs other than Functional Fluids and Printing Inks, at least one facility was located within 250 km upstream of a known DWI. Among Industrial Uses, Manufacture, and Remediation, five facilities were located within 10 km upstream of a known DWI. For most facilities identified as being located upstream from any DWI, multiple downstream DWIs were identified.

Table_Apx G-6. Summary of Proximity of Downstream Drinking Water Intakes to Releasing Facilities Resulting in Modeled Risk above 1E−06

			Facilities with Risk above 1E−06 and DWI Downstream				
OES	**Total Facilities Evaluated**	**Facilities With Lifetime Adult Cancer Risk above 1E−06**	**Within 250 km**	**Within 100 km**	**Within 50 km**	**Within 25 km**	**Within 10 km**
Disposal	25	9	4	4	2	1	0
Ethoxylation Byproduct	8	4	2	1	0	0	0
Functional Fluids (Open-System)	6	2	0	0	0	0	0
Import and Repackaging	12	11	3	3	3	2	0
Industrial Uses	32	21	3	3	2	1	1
Manufacture	2	2	2	2	2	2	2
PET Manufacturing	23	18	5	5	4	1	0
Printing Inks	1	1	0	0	0	0	0
Remediation	16	3	3	3	3	2	2
Total	125	71	22	21	16	9	5

To consider the types of waterways potentially used as source water and susceptible to contamination, an additional assessment of reaches associated with intakes was conducted. This simple assessment examined the mean annual flow in NHDPlus V2.1 for each of the reaches matched as being the closest to a drinking water intake. The resulting distribution (Figure_Apx G-11)



Figure_Apx G-11. Summary Distribution of Mean Annual Flow at Stream Reaches Matched with Drinking Water Intakes

As described in Section 5.2.2.1.2, the degree of dilution between the initial receiving water body at the point of release and a downstream drinking water intake was estimated by calculating the ratio of mean annual NHDPlus flows at both locations. The ranges of dilution (as a percent of the concentration at the point of release) ranged from much less than 1 to 100 percent and are presented in Table_Apx G-7 alongside the ranges of diluted downstream harmonic mean concentrations, which ranged from 1.63×10^{-4} to 1.27×10^{4}. These diluted concentrations were used to develop exposure and risk estimates, presented in Table_Apx G-8.

Table_Apx G-7. Ranges of Dilution and Diluted 1,4-Dioxane Concentrations Modeled at Drinking Water Intakes Downstream of Industrial Releases

			Diluted Concentration as a Percent of Concentration at Point of Release (%)			Modeled Harmonic Mean Concentrations at Downstream Intakes (µg/L)		
Distance Range (km)	**Number of Facilities[a] with DWI Downstream**	**Number of PWS with Downstream Intakes**	**Min.**	**Median**	**Max**	**Min.**	**Median**	**Max**
0–10	4	4	<1	1	100	1.63E−02	3.92E−01	1.27E04
10–25	4	7	<1	<1	68	4.42E−02	1.51E−01	8.28E00
25–50	7	8	<1	<1	92	1.81E−03	2.74E−02	3.03E00
50–100	10	15	<1	<1	31	4.42E−03	1.50E−01	2.07E02
100–250	15	57	<1	<1	100	1.63E−04	7.47E−02	1.52E02

[a] Only facilities with an adult lifetime cancer risk for drinking water above 1 in a million were included in this analysis.
DWI = drinking water intake; PWS = public water system

Table_Apx G-8. Ranges of LADD and Adult Lifetime Cancer Risk Estimates for Diluted 1,4-Dioxane Concentrations Modeled at Drinking Water Intakes Downstream of Industrial Releases

	Diluted LADD (mg/kg-day)			Diluted Adult Lifetime Cancer Risk		
Distance Range (km)	**Minimum**	**Median**	**Maximum**	**Minimum**	**Median**	**Maximum**
0–10	7.6E−08	1.8E−06	4.1E−02	9.1E−09	2.1E−07	4.9E−03
10–25	1.4E−07	4.8E−07	2.6E−05	1.7E−08	5.8E−08	3.2E−06
25–50	5.8E−09	9.7E−08	9.7E−06	6.9E−10	1.2E−08	1.2E−06
50–100	1.4E−08	4.8E−07	6.6E−04	1.7E−09	5.7E−08	7.9E−05
100–250	5.2E−10	2.4E−07	4.8E−04	6.2E−11	2.9E−08	5.8E−05

There are important limitations and uncertainties in this analysis. The extent of dilution is highly variable and is driven by site-specific factors that cannot be fully captured in this national-scale analysis. This analysis is based on the conservative assumption that the only decrease in concentration is due to dilution, and the effects of diffusion, advection, or dispersion are not modeled. Additionally, while flows within a river or stream generally increase in the downstream direction, infrastructure like dams and water withdrawal activities can lead to decreases in downstream flows. In lieu of a more robust model to assess each release on a case-by-case basis, this approach allows a rapid assessment of estimated ranges of dilution. Overall confidence in risk estimates is high for drinking water intakes located at or near the point of release, but confidence decreases substantially with increasing distance downstream. This analysis does not provide a comprehensive survey of modeled 1,4-dioxane concentrations at all drinking water intakes. There may be additional drinking water intakes downstream of facilities releasing 1,4-dioxane that are not accounted for in the intake database used in this analysis.

Appendix H GROUNDWATER CONCENTRATIONS AND DISPOSAL PATHWAYS FROM LAND RELEASES

H.1 Groundwater Monitoring Data Retrieval and Processing

The complete set of 1,4-dioxane monitoring results stored in the WQP was retrieved in July 2022, with no filters applied other than the chemical name (NWQMC, 2022). This raw dataset included 12,471 samples. To filter down to only the desired groundwater samples to include in this analysis, only samples with the "ActivityMediaSubdivisionName" attribute of "Groundwater" were kept, and among those, only samples with a "MonitoringLocationTypeName" that was one of the following:

- well;
- subsurface;
- subsurface: groundwater drain; and
- well: multiple wells.

After these steps, 8,046 groundwater samples remained in the dataset. Samples flagged as QC blanks in the "ActivityTypeCode" column were then removed, leaving 7,583 groundwater samples for analysis. Of these remaining samples, only 30 percent (n = 2,284) were results above the respective reported detection limit.

H.2 Review of Land Release Permits

EPA reviewed all Underground Injection Class I Permits to understand if sites were in accordance with regulations. The sites and the corresponding release year, registry number, and disposal weight is available in Table_Apx H-1 for on-site disposal and Table_Apx H-2 for off-site.

Table_Apx H-1. Release Year, TRI Facility ID, Facility Name, State, Registry Number, Disposal Type, and Disposal Weight for On-Site Class I Underground Injection Wells According to TRI

Release Year	TRI Facility ID	Facility Name	State	Registry Number	Disposal Type	Disposal Weight (lb)
2019	77536DSPSL2525B	TM DEER PARK SERVICES LP	Texas	RN100209568	On-Site	23,098
2018	77536DSPSL2525B	TM DEER PARK SERVICES LP	Texas	RN100209568	On-site	23,604
2017	77536DSPSL2525B	TM DEER PARK SERVICES LP	Texas	RN100209568	On-site	23,024
2016	77536DSPSL2525B	TM DEER PARK SERVICES LP	Texas	RN100209568	On-site	12,867
2015	77536DSPSL2525B	TM DEER PARK SERVICES LP	Texas	RN100209568	On-site	94,304
2014	77536DSPSL2525B	TM DEER PARK SERVICES LP	Texas	RN100209568	On-site	731,892

Release Year	TRI Facility ID	Facility Name	State	Registry Number	Disposal Type	Disposal Weight (lb)
2013	77536DSPSL2525B	TM DEER PARK SERVICES LP	Texas	RN100209568	On-site	371,877.95

Table_Apx H-2. Release Year, Source TRI Facility ID, Source State, Receiving Facility RCRA ID, State, Disposal Type, and Disposal Weight for Off-Site Class I Underground Injection Wells According to TRI and RCRAInfo Databases

Release Year	Source TRI Facility ID	Source State	Receiving Facility RCRA ID	Receiving State	Disposal Type	Disposal Weight (lb)
2019	44044RSSNC36790	Ohio	OHD020273819	Ohio[a]	Off-site	0.009
2019	29448GNTCMPOBOX	South Carolina	OHD020273819	Ohio[a]	Off-site	2
2018	29448GNTCMPOBOX	South Carolina	OHD020273819	Ohio[a]	Off-site	23

[a] The state of Ohio provides an overview of its underground injection wells via the Ohio Environmental Protection Agency.

EPA reviewed all RCRA Subtitle C Permits to understand if sites were in accordance with regulations. The sites and the corresponding release year, registry number, and disposal weight is available in Table_Apx H-3 for on-site disposal and Table_Apx H-4 for off-site.

Table_Apx H-3. Release year, TRI Facility ID, Facility Name, State, CERCLIS ID, Disposal Type, and Disposal Weight for RCRA Subtitle C Landfills According to TRI

Release Year	TRI Facility ID	Facility Name	State	FRS ID	Disposal Type	Disposal Weight (lb)
2015	97812CHMCL17629	Chemical Waste Management of the Northwest INC	Oregon	110002059904[a]	On-site	13,368.40
2014	97812CHMCL17629	Chemical Waste Management of the Northwest INC	Oregon	110002059904[a]	On-site	16,108.10
2013	97812CHMCL17629	Chemical Waste Management of the Northwest INC	Oregon	110002059904[a]	On-site	15,400.30

[a] This facility has several violation and compliance issues. The facility was fined $25,000 in 2020 for non-compliance activities. The fine is attributed to inadequate coverage for third party bodily injury and property damage claims. The facility self-reported in 2021 that another compliance issue had been detected.

Table_Apx H-4. Release Year, Source TRI Facility ID, Source State, Receiving Facility RCRA ID, State, Disposal Type, and Disposal Weight for Off-Site Class I Underground Injection Wells According to TRI and RCRAInfo Databases

Release Year	Source TRI Facility ID	Source State	Receiving Facility RCRA ID	Receiving State	Disposal Type	Disposal Weight (lb)
2019	84029SFTYK11600	Utah	UTD991301748[a]	Utah	Off-site	0.08
2018	84029SFTYK11600	Utah	UTD991301748[a]	Utah	Off-site	0.01
2015	84029SFTYK11600	Utah	UTD991301748[a]	Utah	Off-site	0.1488

Release Year	Source TRI Facility ID	Source State	Receiving Facility RCRA ID	Receiving State	Disposal Type	Disposal Weight (lb)
2016	77536SFTYK2027B	Texas	OKD065438376	Oklahoma	Off-site	0.03
2015	77536SFTYK2027B	Texas	OKD065438376	Oklahoma	Off-site	0.16
2019	69145CLNHR5MISO	Nebraska	COD991300484[b]	Colorado	Off-site	0.29
2018	69145CLNHR5MISO	Nebraska	COD991300484[b]	Colorado	Off-site	13.29
2017	69145CLNHR5MISO	Nebraska	COD991300484[b]	Colorado	Off-site	55.49
2019	66736SYSTCCEMEN	Kansas	OKD065438376	Oklahoma	Off-site	750
2019	66736SYSTCCEMEN	Kansas	ALD000622464[c]	Alabama	Off-site	750
2019	44044RSSNC36790	Ohio	MID048090633[d]	Michigan	Off-site	0.011
2015	44044RSSNC36790	Ohio	MID000724831[d]	Michigan	Off-site	0.005
2014	44044RSSNC36790	Ohio	MID000724831[d]	Michigan	Off-site	0.008
2014	43920VNRLL1250S	Ohio	MID000724831[d]	Michigan	Off-site	30.2
2013	43920VNRLL1250S	Ohio	MID048090633[d]	Michigan	Off-site	17
2015	44044RSSNC36790	Ohio	OHD045243706[e]	Ohio	Off-site	0.001
2014	44044RSSNC36790	Ohio	OHD045243706[e]	Ohio	Off-site	0.002
2014	43920VNRLL1250S	Ohio	IND093219012[f]	Indiana	Off-site	72.6
2013	43920VNRLL1250S	Ohio	IND093219012[f]	Indiana	Off-site	44

[a] This facility was found to be non-compliant by the state in 2021 and was fined $20,575. The fine was associated with a formal administrative enforcement action asserting that a remedial action is required.

[b] This facility was found to have significant non-compliance from 2020 to 2021. The facility was fined $12,000 in 2021. The fine was associated with a formal administrative enforcement action asserting that a remedial action is required.

[c] This facility was found to be a significant non-complier by the state in 2020, 2021, and 2022. The facility has been fined a total of $22,650. The fine was associated with a formal administrative enforcement action asserting that a remedial action is required.

[d] These two facilities are likely the same as they have the same address.

[e] This facility has received written informal notices in 2017, 2018, 2019, and 2021. No enforcement actions have occurred.

[f] This facility was found to be a significant non-complier by the state from 2015 to 2021; the facility has been fined a total of $77,385. The fine is associated with a Consent Agreement and Final Order between Region 5 and Heritage Environmental Services, LLC. Heritage violated its permit, the Indiana Administrative Code, and RCRA and its implementing regulations by (1) disposing of hazardous waste in the Roachdale landfill without meeting certain land disposal restriction (LDR) treatment standards; (2) failing to conduct post-treatment verification sampling and analysis of certain waste streams from two stabilization/LDR treatment processes; (3) failing to obtain a detailed chemical and physical analysis of representative samples from such waste streams; (4) failing to follow the acceptable analytical methods in its waste analysis plan (WAP); and (5) failing to determine the proper extraction fluid for TCLP analysis.

H.3 Landfill Analysis Using DRAS

DRAS is an efficient tool developed by EPA Region 6 to provide a multipath risk assessment for the evaluation of Resource Conservation and Recovery Act (RCRA) hazardous waste delisting. For the Supplemental Evaluation to the 1,4-dioxane Risk Evaluation, DRAS was specifically applied to model groundwater concentration estimates from disposing 1,4-dioxane to a hypothetical RCRA Subtitle D landfill at a range of loading rates and leachate concentrations. A comprehensive description of the assumptions and calculations applied in DRAS can be found in the Technical Support Document for the Hazardous Waste Delisting Risk Assessment Software (https://www.epa.gov/hw/technical-support-document-hazardous-waste-delisting-risk-assessment-software-dras). It is worth noting that the underlying assumptions for DRAS are the same as those for EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) described in Appendix Section H.4.

Because DRAS derives calculations based on a survey of drinking water wells located downgradient from waste management units (U.S. EPA, 1988), the model may provide the closest estimate to real world scenarios available. Though there is some uncertainty inherent to applying the model as an assessment tool under the Toxic Substance Control Act (TSCA) for risk evaluations, few other tools are available to effectively address this pathway. This appendix will provide the input variables and calculations used to apply the model determine potential groundwater concentrations. Table_Apx H-5 and Table_Apx H-6 provide the input values used for each parameter in the model. Note that loading volumes were based on the range of TRI release weights and were calculated based on the density of 1,4-dioxane at 20 °C (1.0329 g/cm^3). For each loading volume, the range of leachate concentrations was applied.

Table_Apx H-5. Input Variables for Chemical of Concern

Input Variable for Chemical of Concern	Value
Chem Name	1,4-Dioxane
CASRN	123-91-1
Maximum Contaminant Level	0
Oral Slope Cancer Factor	0.1[a]
Inhalation Slope Cancer Factor (1/mg kg day)	0.018[a]
Oral Reference Dose (mg/kg day)	0.03[a]
Inhalation Reference Dose (mg/kg day)	0.03[a]
Bioconcentration Factor (l/kg)	0.3698
Soil Saturation Level	0
Toxicity Regulatory Rule regulatory level (mg/L)	0[a]
Henry's Law Constant (atm -m^3/mol)	4.25E−06
Diffusion coefficient in Water (cm^2/s)	1.05E−05
Diffusion coefficient in Air (cm^2/s)	0.092[a]
Water Solubility (mg/L)	1,000,000
Landfill Dilution Attenuation Factor	15.4
Surface Impoundment Dilution Attenuation Factor	3.18
Time to Skin Attenuation (hr/event)	0.72[a]
Skin permeability constant (cm/hour)	0.00029[a]
Lag time (hr)	0.3[a]
Bunge constant	4.1E−05[a]
Organic	Yes
Bioaccumulation Factor (L/kg)	0[a]
Chronic Ecological Value (mg/L)	0[a]
Carcinogen	Yes
Molecular Weight (g/mol)	88.1
Vapor Pressure (atm)	0.05
Suspended sediment-surface water partitioning coefficient (mg/L)	0.0549

Input Variable for Chemical of Concern	Value
log Kow (log[mg/l])	–0.27
Chemical Class	VOC[a]
Analytical Method	8260D[a]
Version Description	None[a]
Create Date	None[a]
Creator	None[a]
Cancer Risk Level	1.00E−06[a]
Hazard Quotient	1[a]

[a] Input variables do not directly or indirectly affect groundwater concentrations

Table_Apx H-6. Waste Management Unit (WMU) Properties

Input Variable for WMU Properties	Value(s)
Waste Management Unit Type	Landfill
Loading Volume (m^3)	4.39E−07
	4.39E−06
	4.39E−05
	4.39E−04
	4.39E−03
	4.39E−02
	4.39E−01
	4.39E00
	4.39E01
	4.39E02
Cancer Risk Level	1.00E−06
Hazard Quotient	1.0
Detection Limit	0.5
Waste Management Active Life (Years)	20
TCLP Concentration (mg/L)/ Total Concentration (mg/kg)	0.0001
	0.001
	0.01
	0.1
	1
	10
	100
	1,000
	10,000

Once the model was executed for each loading rate and leachate concentration scenario, the groundwater concentration was calculated using the leachate concentration and the 90th percentile weight-adjusted dilatation attenuation factor using the equation:

$$GW_c = \frac{Leachate\ Concentration}{Weight\ Adjusted\ DAF},$$

Where:

GW_c = Groundwater concentration
Leachate concentration = Input variable for the waste management unit
Weight Adjusted DAF = Weight adjusted dilution attenuation factor.

The results of these analyses are provided in Table 2-13.

H.4 Landfill Analysis Using EPACMTP

EPACMTP is a fate and transport model developed by EPA to simulate the release of constituents from waste managed in land disposal units, and the subsequent impacts of these constituents to the subsurface environment. The model combines two modules to simulate one-dimensional downward flow and transport of constituents in the unsaturated zone beneath a waste disposal unit, as well as ground water flow and three-dimensional constituent transport in the underlying saturated zone. The model is designed to run in a probabilistic or deterministic mode and comes with built-in distributions of national and regional modeling parameters. The output of the model includes estimated concentrations of constituents arriving at a downgradient well under steady-state conditions or as a function of time.

Because EPACMTP derives calculations from based on a survey of drinking water wells located downgradient from a waste management unit (U.S. EPA, 1988), the model may provide the closest estimate to real world scenarios available. Though there is some uncertainty inherent to applying the model as an assessment tool under TSCA for risk evaluations, few other tools are available to effectively address this pathway. This appendix will provide the input variables and calculations used to apply the model determine potential groundwater concentrations. More comprehensive information about the assumptions and calculation embedded in the EPACMTP model can be found on the EPA website.

EPA ran the model under two scenarios. In one scenario, it is assumed that the waste management unit is an unlined landfill. In the other, it is assumed the waste management unit is a clay-lined landfill. In addition to these details, chemical specific input variables are required. For 1,4-dioxane, these included molecular weight (88.1 g/mole), water solubility (10,000 mg/L), K_{OC} (17.0 g/L), rate of abiotic hydrolysis (0.0 $mol^{-1}year^{-1}$), rate of biodegradation (0.0 $mol^{-1}year^{-1}$), and temperature (25 °C). Similarly, initial concentration of the chemical substance was an input and ranged from 1×10^{-4} to 1×10^{4} (Table_Apx H-7). All other variables in the input files were left in their defaults. Each scenario requires a separate input file provided with the executable file package. All files for running the executable model were stored in same folder.

Table_Apx H-7. Potential Groundwater Concentrations (mg/L) Based on Disposal of 1,4-Dioxane to Unlined and Clay-Lined Landfills as Assessed by Applying the EPACMTP Model

Leachate Concentration (mg/L)	Type of Liner			
	No Liner		With Clay Liner	
	Percentile (n = 10,000)	Average Groundwater Concentration (mg/L)	Percentile (n = 10,000)	Average Groundwater Concentration (mg/L)
0.0001	0	0	0	0
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	3.79E−10	75	7.37E−11
	80	6.83E−09	80	4.04E−09
	85	3.3E−08	85	2.55E−08
	90	1.29E−07	90	8.92E−08
	95	7.93E−07	95	7.41E−07
	100	3.42E−05	100	3.34E−05
0.001	0	0	0	0
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	3.35E−09	75	8.57E−10
	80	2.75E−08	80	1.71E−08
	85	1.63E−07	85	8.29E−08
	90	1.4E−06	90	7.64E−07
	95	8.01E−06	95	7.43E−06
	100	0.000342	100	0.000334
0.01	0	0	0	0
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	1.02E−08	75	4.14E−09
	80	1.07E−07	80	5.23E−08
	85	1.62E−06	85	6.82E−07
	90	1.4E−05	90	7.64E−06
	95	8.01E−05	95	7.43E−05
	100	0.003415	100	0.00334
0.1	0	0	0	0
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	3.57E−08	75	1.38E−08
	80	1.05E−06	80	2.77E−07
	85	1.62E−05	85	6.82E−06
	90	0.00014	90	7.64E−05
	95	0.0008	95	0.000743
	100	0.03415	100	0.0334

Leachate Concentration (mg/L)	**Type of Liner**			
	No Liner		**With Clay Liner**	
	Percentile (n = 10,000)	**Average Groundwater Concentration (mg/L)**	**Percentile (n = 10,000)**	**Average Groundwater Concentration (mg/L)**
1	0	0	0	0
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	1.18E−07	75	4E−08
	80	1.02E−05	80	2.76E−06
	85	0.000161	85	6.81E−05
	90	0.001395	90	0.000764
	95	0.00793	95	0.007429
	100	0.3415	100	0.334
10	0	0	0	0
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	1.22E−06	75	1.89E−07
	80	0.000105	80	2.77E−05
	85	0.001622	85	0.000682
	90	0.01395	90	0.00764
	95	0.08004	95	0.07429
	100	3.415	100	3.394
100	0	0	0	0
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	1.21E−05	75	1.89E−06
	80	0.001046	80	0.000277
	85	0.01622	85	0.006816
	90	0.1442	90	0.0764
	95	0.8499	95	0.7429
	100	34.15	100	34.77
1,000	0	0	0	0
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	0.000121	75	1.89E−05
	80	0.01046	80	0.002773
	85	0.1692	85	0.06816
	90	1.537	90	0.764
	95	9.076	95	7.589
	100	341.5	100	347.7
10,000	0	0	0	0

Leachate Concentration (mg/L)	**Type of Liner**			
	No Liner		**With Clay Liner**	
	Percentile (n = 10,000)	**Average Groundwater Concentration (mg/L)**	**Percentile (n = 10,000)**	**Average Groundwater Concentration (mg/L)**
	10	0	10	0
	25	0	25	0
	50	0	50	0
	75	0.00123	75	0.000189
	80	0.1066	80	0.02773
	85	1.819	85	0.6816
	90	15.78	90	7.851
	95	90.89	95	76.19
	100	3,415	100	3,477

Note: The results are a product of Monte Carlo analysis and are organized by leachate concentration (mg/L), percentile, and average concentration of 1,4-dioxane at a well within 1 mile of the disposal facility.

H.5 Surface Impoundment Analysis for the Disposal of Hydraulic Fracturing Produced Water Using DRAS

The Delisting Risk Assessment Software (DRAS) is an efficient tool developed by U.S. Environmental Protection Agency (EPA) region 6 to provide a multipath risk assessment for the evaluation of Resource Conservation and Recovery Act (RCRA) hazardous waste delisting. For the Supplemental Evaluation to the 1,4-dioxane Risk Evaluation, DRAS was specifically applied to model groundwater concentration estimates from disposing 1,4-dioxane in produced waters from a hydraulic fracturing operation to a hypothetical RCRA Surface Impoundment at a range of loading rates and leachate concentrations. A comprehensive description of the assumptions and calculations applied in DRAS can be found in the Technical Support Document for the Hazardous Waste Delisting Risk Assessment Software.

Because the model derives calculations from based on a survey of drinking water wells located downgradient from a waste management unit (U.S. EPA, 1988), the model may provide the closest estimate to real world scenarios available. Although there is some uncertainty inherent to applying the model as an assessment tool under TSCA for risk evaluations, few other tools are available to effectively address this pathway. This appendix will provide the input variables and calculations used to apply the model determine potential groundwater concentrations. Table_Apx H-8 and Table_Apx H-9 provide the input values used for each parameter in the model. Note that loading volume were based on the range of TRI release weights and was calculated based on the density of 1,4-dioxane at 20 °C (1.0329 g/cm^3). For each loading volume, only one potential concentration was applied.

Table_Apx H-8. Input Variables for Chemical of Concern

Input Variable for Chemical of Concern	**Value**
Chem Name	1,4-Dioxane
Chem CASRN	123-91-1
Maximum Contaminant Level	0
Oral Slope Cancer Factor	0.1
Inhalation Slope Cancer Factor (1/mg kg day)	0.018
Oral Reference Dose (mg/kg day)	0.03

Input Variable for Chemical of Concern	Value
Inhalation Reference Dose (mg/kg day)	0.03
Bioconcentration Factor (L/kg)	0.369
Soil Saturation Level	0
Toxicity Regulatory Rule regulatory level (mg/L)	0
Henry's Law Constant (atm -m^3/mol)	4.25E−06
Diffusion coefficient in Water (cm^2/s)	1.05E−05
Diffusion coefficient in Air (cm^2/s)	0.092
Water Solubility (mg/L)	1,000,000
Landfill Dilution Attenuation Factor	15.4
Surface Impoundment Dilution Attenuation Factor	3.18
Time to Skin Attenuation (hour/event)	0.72
Skin permeability constant (cm/hour)	0.00029
Lag time (hours)	0.3
Bunge constant	4.1E−05
Organic	Yes
Bioaccumulation Factor (L/kg)	0
Chronic Ecological Value (mg/L)	0
Carcinogen	Yes
Molecular Weight (g/mol)	88.1
Vapor Pressure (atm)	0.05
Suspended sediment-surface water partitioning coefficient (mg/L)	0.0549
log Kow (log[mg/L])	–0.27
Chemical Class	VOC
Analytical Method	8260D
Version Description	None
Create Date	None
Creator	None
Cancer Risk Level	1.00E−06
Hazard Quotient	1

Table_Apx H-9. Waste Management Unit

Input Variable for WMU Properties	Value(s)
Waste Management Unit Type	Surface Impoundment
Loading Volume (m^3)	1734
	193
	67.1
	15.1
	3.48
	0.0334
	1.09E−08
Cancer Risk Level	1.00E−06
Hazard Quotient	1.0
Detection Limit	0.5
Waste Management Active Life (Years)	50
TCLP Concentration (mg/L)/Total Concentration (mg/kg)	0.06

Once the model was executed for each loading rate and leachate concentration scenario, the groundwater concentration was calculated using the leachate concentration and the 90th percentile weight-adjusted dilatation attenuation factor using the equation:

$$GW_c = \frac{Leachate\ Concentration}{Weight\ Adjusted\ DAF},$$

Where:

GW_c = Groundwater concentration
Leachate concentration = Input variable for the waste management unit
Weight Adjusted DAF = Weight-adjusted dilution attenuation factor.

The results of these analyses are provided in Table 2-14.

Appendix I DRINKING WATER EXPOSURE ESTIMATES

Potential acute and chronic drinking water exposures were estimated based on surface water concentrations estimated in Section 2.3.1 and groundwater concentrations estimated in Section 2.3.2.

Acute and chronic drinking water exposures used to evaluate non-cancer risks are estimated as an Acute Dose Rate (ADR) or Average Daily Dose (ADD), respectively. Lifetime exposures used to evaluate cancer risks are estimated as a Lifetime Average Daily Dose (LADD). The equations used to calculate each of these exposure values are:

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

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

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

Where:

SWC = Surface water concentration (ppb or µg/L)
DWT = Removal during drinking water treatment (%)
IRdw = Drinking water intake rate (L/day)
RD = Release days (days/year for ADD, LADD and LADC; 1 day for ADR)
ED = Exposure duration (years for ADD, LADD and LADC; 1 day for ADR)
BW = Body weight (kg)
AT = Exposure duration (years for ADD, LADD and LADC; 1 day for ADR)
CF1 = Conversion factor (1.0×10^{-3} mg/µg)
CF2 = Conversion factor (365 days/year)

Inputs for body weight, averaging time (AT), and exposure duration were applied the same across the evaluation of drinking water, incidental oral exposure, and incidental dermal exposure, but are described here. For all calculations, mean body weight data were used from Chapter 8, Table 8-1 in the U.S. *Exposure Factors Handbook* (EFH) (U.S. EPA, 2011). To align with the age groups of interest, weight averages were calculated for the infant age group (birth to less than 1 year) and toddlers (1 to 5 years). The ranges given in the EFH were weighted by their fraction of the age group of interest. For example, the EFH provides body weight for 0 to 1 month, 1 to 3 months, 3 to 6 months, and 6 to 12 months. Each of those body weights were weighted by their number of months out of 12 to determine the weighted average for an infant 0 to 1 year old. For all ADR calculations, the AT is 1 day, and the days of release are assumed to be 1 according to the methodology used in E-FAST 2014 (U.S. EPA, 2014). For all ADD calculations, the AT and the ED are both equal to the number of years in the relevant age group up to the 95th percentile of the expected duration at a single residence, 33 years (U.S. EPA, 2011). For example, estimates for a child between 6 and 10 years old would be based on an AT and ED of 5 years. For all LADD and LADC calculations, the AT is based on a lifetime of 78 years, and the ED is the number of years of exposure in the relevant age group, up to 33 years.

Drinking water exposure was estimated for the following age groups: Adult (21+ years), Youth (16 to 20 years), Youth (10 to 15 years), Child (6 to 10 years), Toddler (1 to 5 years), and infant (birth to <1 year). Drinking water intake rates are provided in the 2019 update of Chapter 3 of the *Exposure Factors Handbook* (U.S. EPA, 2019a). Weighted averages were calculated for acute and chronic drinking water intakes for adults 21+ years and toddlers 1 to 5 years. From Table 3-17 in the Handbook, 95th percentile consumer data were used for acute drinking water intake rates. From Table 3-9 in the Handbook, mean per capita data were used for chronic drinking water intake rates.

I.1 Surface Water Sources of Drinking Water

To estimate drinking water exposures that may result from surface water contamination, EPA used water concentrations estimated in Section 2.3.1. Concentrations in estuaries or bays are not considered as they are unlikely to be potable waters. Drinking water exposures are also not considered for large lakes due to high uncertainty in the applicable dilution factors. This is in alignment with the methodology used in E-FAST 2014 (U.S. EPA, 2014)

ADR or acute exposure concentrations used the modeled stream concentrations with the lowest monthly flow rate while the ADD, LADD, and LADC or chronic calculations used the modeled harmonic mean stream concentrations. Drinking water treatment removal (DWT) was set to 0 percent to represent a conservative estimate of possible drinking water exposures.

I.2 Groundwater Sources of Drinking Water

To estimate drinking water exposures that may result from groundwater contamination, EPA used groundwater concentrations estimated in Section 2.3.2.

Chronic and lifetime exposures (ADD and LADD) were calculated based on groundwater concentrations estimated using the DRAS model. Acute exposures to groundwater were not calculated because the available models EPA used for estimating groundwater concentrations are designed to predict long-term trends rather than short peaks in exposure. DWT was set to 0 percent for groundwater under the assumption that home wells are unlikely to remove 1,4-dioxane.

Appendix J AIR EXPOSURE PATHWAY

J.1 Ambient Air Concentrations and Exposures

EPA applied a tiered approach to estimate ambient air concentrations and exposures for members of the general population that are in proximity (between 5 to 10,000 m) to emissions sources emitting the chemicals being evaluated to the ambient air (Figure_Apx J-1). All exposures were assessed for the inhalation route only.

Ambient Air: Screening Methodology

- Methodology is independent of facility and use classifications. Analysis broadly estimates ambient air concentrations and associated exposures/risks based on maximum and mean releases at three pre-defined distances from a releasing facility. Designed to inform whether application of higher-tier analysis methodology is warranted.

Ambient Air: Single Year Methodology (AERMOD or IIOAC)

- Methodology is facility and scenario specific. Analysis evaluates ambient air concentrations and associated exposures/risks resulting from facility-specific releases across multiple distances from the source. Utilizes a single year of release data, but can be expanded to utilize multiple-years of release data from multiple data sets/sources.

Ambient Air: Multi-Year Analysis Methodology (IIOAC)

- Methodology is facility and scenario specific. Analysis evaluates ambient air concentrations and associated exposures/risks resulting from facility-specific releases at three pre-defined distances from a releasing facility. Utilizes multiple years of release data reported to TRI. Developed in response to SACC comments/recommendations on the 2022 Fenceline Report to consider multiple years of release data to estimate exposures and associated risks.

Figure_Apx J-1. Summary of Methodologies Used to Estimate Ambient Air Concentrations and Exposures

J.1.1 Ambient Air: Screening Methodologies and Results Summary – Fenceline

The Ambient Air: Screening Methodology identifies, at a high level, if there are inhalation exposures to select populations from a chemical undergoing risk evaluation which indicates a potential risk. This methodology inherently includes both estimates of exposures as well as estimates of risks to inform the need, or potential need, for further analysis. If findings from the Ambient Air: Screening Methodology indicate any potential risk (acute non-cancer, chronic non-cancer, or cancer) for a given chemical above (or below as applicable) typical Agency benchmarks, EPA generally will conduct a higher-tier analysis of exposures and associated risks for that chemical. If findings from the Ambient Air: Screening Methodology do not indicate any potential risks for a given chemical above (or below as applicable) typical agency benchmarks, EPA would not expect a risk would be identified with higher-tier analyses, but may still conduct a limited higher-tier analysis at select distances to ensure potential risks are not missed (*e.g.*, at distances less than 100 m to ensure risks do not appear very near a facility where people may be exposed).

Model

The Ambient Air: Screening Methodology utilizes EPA's IIOAC model to estimate high-end and central tendency (mean) exposures to select receptors at three pre-defined distances from a facility releasing a chemical to the ambient air (100, 100 to 1,000, and 1,000 m). IIOAC is an Excel-based tool that estimates indoor and outdoor air concentrations using pre-run results from a suite of dispersion scenarios run in a variety of meteorological and land-use settings within EPA's AERMOD. As such, IIOAC is limited by the parameterizations utilized for the pre-run scenarios within AERMOD (meteorologic data, stack heights, distances, receptors, etc.) and any additional or new parameterization would require revisions to the model itself. Readers can learn more about the IIOAC model, equations within the model, detailed input and output parameters, pre-defined scenarios, default values used, and supporting documentation by reviewing the IIOAC users guide (U.S. EPA, 2019b).

Releases

EPA modeled exposures from two release values for 1,4-dioxane. These values were extracted from 2019 TRI data as follows:

1. The maximum individual facility 1,4-dioxane release value among all facilities reporting releases of 1,4-dioxane to TRI.
2. The average (mean) 1,4-dioxane release value across all facilities reporting 1,4-dioxane to TRI.

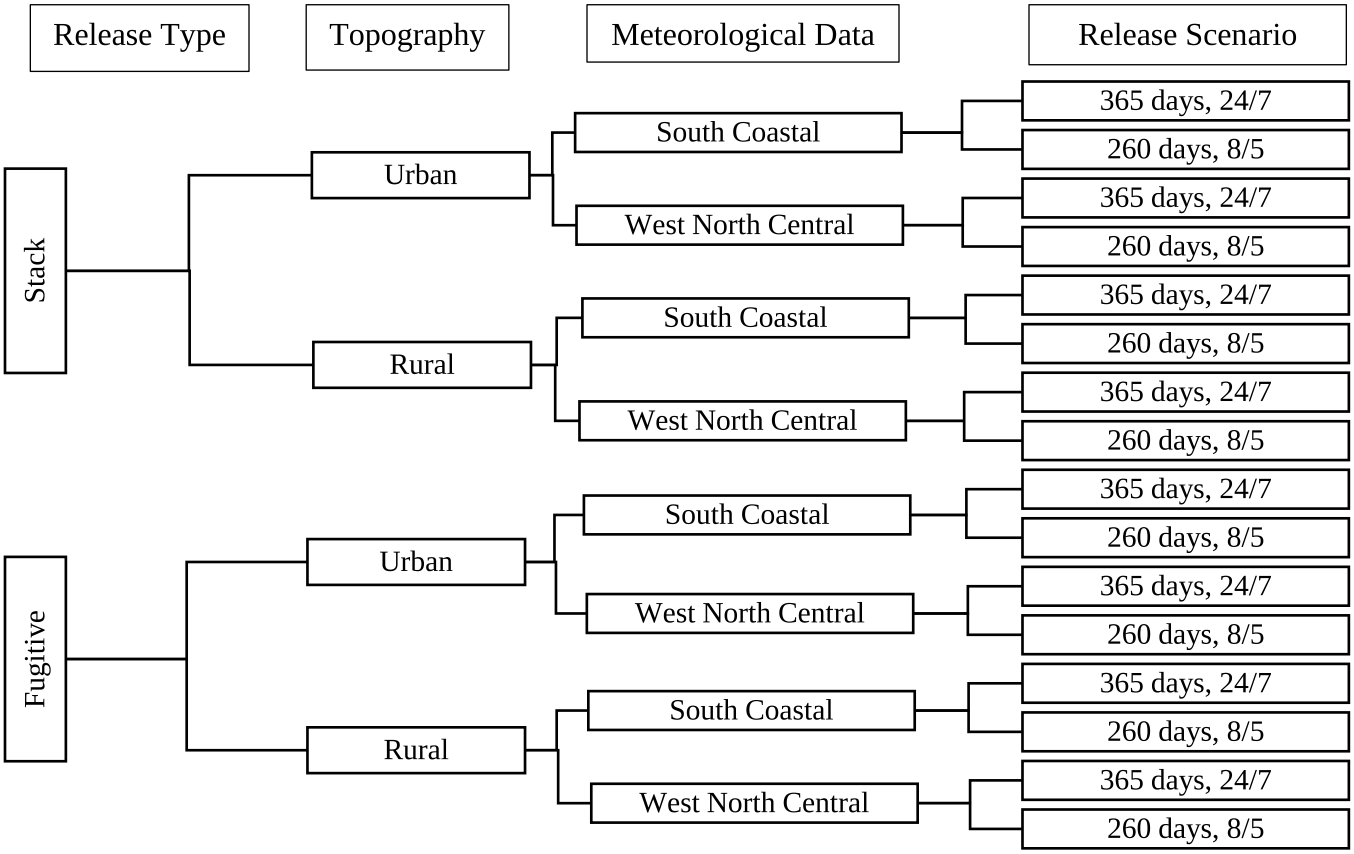
A summary of the releases evaluated for TRI reporting facilities is provided in Table_Apx J-1.

Table_Apx J-1. Release Estimates from 2019 TRI Used for Ambient Air: Screening Methodology for 1,4-Dioxane

Number of Operating Days	**Maximum Facility Release**			**Average Facility Release**		
	Pounds (lb)	**Kilograms (kg)**	**kg/site-day**	**Pounds (lb)**	**Kilograms (kg)**	**kg/site-day**
365	10,442	4,735.601	12.97	792	359.184	0.98
260			18.21			1.38

Exposure Scenarios

EPA developed and evaluated a series of exposure scenarios for the max and mean 1,4-dioxane release values identified above. The scenarios were designed to capture a variety of release types, topography, meteorological conditions, and release scenarios as presented in Figure_Apx J-2. It includes a total of 16 different exposure scenarios, each of which is applied to both the maximum and mean 1,4-dioxane release value resulting in a total of 32 exposure scenarios modeled.



Figure_Apx J-2. Exposure Scenarios Modeled for Max and Mean Release Using IIOAC Model for Ambient Air: Screening Methodology

EPA modeled exposure scenarios for two source types: stack (point source) and fugitive (area source) releases. These source types have different plume and dispersion characteristics accounted for differently within the IIOAC model. The topography represents an urban or rural population density and certain boundary layer effects (like heat islands in an urban setting) that can affect turbulence and resulting concentration estimates at certain times of the day.

IIOAC includes 14 pre-defined climate regions (each with a surface station and upper-air station). Since release data used for the Ambient Air: Screening Methodology was not facility or location specific, EPA selected 2 of the 14 climate regions to represent a central tendency (West North Central) and high-end (South [Coastal]) climate region. This selection was based on a sensitivity analysis of the average concentration and deposition predictions. The two climate regions selected represent meteorological data sets that tended to provide high-end and central tendency concentration estimates relative to the other stations within IIOAC. The meteorological data within the IIOAC model are from years 2011 to 2015 as that is the meteorological data utilized in the suite of pre-run AERMOD exposure scenarios during development of the IIOAC model (see IIOAC users guide (U.S. EPA, 2019b)). While this is older meteorological data, sensitivity analyses related to different years of meteorological data found that although the data does vary, the variation is minimal across years so the impacts to the model outcomes remain relatively unaffected.

The release scenarios consider two potential facility operating conditions. The first represents a facility that operates year-round (365 days per year, 24/7). The second represents a facility that operates generally on a Monday through Friday schedule (260 days per year) for 8 hours per day, 5 days per week. The difference between the two release scenarios is the resulting total daily release, frequency of release, and duration of release. These conditions result in a different exposure pattern that is captured by modeling both release scenarios. As an example, if a facility has a total annual release of 10,000 lb/year, then the daily release from a facility operating 365 days/year, 7 days per week, and 24 hours per day would be 27.4 lb per day for every day of the year over a 24-hour period. If the facility operates 260 days per year, 5 days per week, for 8 hours per day, the daily release would be 38.5 lb per day, but only Monday through Friday and only over an 8-hour period.

Exposure Results and Risks

Modeled exposure concentration results from the Ambient Air: Screening Methodology modeling effort were reviewed and summarized for each scenario modeled. To ensure potential risks were not missed, EPA selected the highest estimated exposure concentrations from the 32 scenarios modeled for 1,4-dioxane for use in risk calculations. These values were used to estimate the MOE and excess cancer risk. The calculated risks were then compared to screening level benchmarks (POD-specific benchmark MOEs for non-cancer risks and 1×10^{-6} for general population cancer risk). Overall, the Ambient Air: Screening Methodology did not identify risk relative to benchmark values for non-cancer risks but did identify risk estimates above the benchmark value for cancer for three of the four release scenarios summarized. Because the results from this methodology indicate potential risks to people near a releasing facility, EPA conducted additional, higher-tier analyses to apply more COU and site-specific data and results to further analyze exposures and associated potential risks resulting from such exposures.

Table_Apx J-2. Exposure and Risk Estimates from the Ambient Air: Screening Methodology for 1,4-Dioxane Releases Reported to TRI

Receptor (Distance in m)	Release Scenario	Maximum, High-End Exposure Concentration (ppm)			Risk Estimates[a] – Inhalation Exposure		
					Non-cancer		Cancer
		AC	ADC	LADC	Acute MOE	Chronic MOE	Chronic IUR
					Liver Effect	Respiratory	Respiratory
Fenceline (100 m)	Max	6.2E−03	6.2E−03	2.6E−03	4,239	137	4.19E−05
	Mean	4.7E−04	4.7E−04	2.0E−04	56,238	1,815	3.16E−06
Community Avg. (100–1,000 m)	Max	7.2E−04	7.2E−04	3.0E−04	36,432	1,175	4.87E−06
	Mean	5.4E−05	5.4E−05	2.3E−05	483,282	15,593	3.67E−07

[a] Details on the methods used to calculate risks are described in Section 5. Shading indicates risk relative to screening level benchmarks.

J.1.2 Ambient Air: IIOAC Methodology and Results for COUs without Site-Specific Data (Hydraulic Fracturing, Industrial, and Institutional Laundry Facilities)

The Ambient Air: IIOAC Methodology for COUs without Site-Specific Data was utilized to evaluate exposures from three new COUs for the ambient air pathway (hydraulic fracturing, industrial laundry, and institutional laundry) previously not included in the published risk evaluation or draft fenceline report. The methodology utilizes IIOAC to estimate high-end and central tendency exposure concentrations at three pre-defined distances from a releasing facility. This methodology is a higher-tier methodology which integrates additional data provided as part of the release assessment. In particular, this additional data included

1. Source attribution (fugitive and stack release types),
2. Days of release,
3. Multiple release percentiles, and

4. Chemical phase/form of release (vapor and particulate phase releases).

Other input parameters like release duration, meteorology, and topography were varied across the scenarios outlined in Figure_Apx J-3. A summary of the various input parameters is provided in Table_Apx J-3. Modeling consisted of evaluating all possible iterations/combinations of the input parameters listed resulting in the following total exposure and release scenarios:

1. <u>Hydraulic Fracturing</u> (fugitive releases only): 8 Exposure Scenarios, each with 28 release scenarios;
2. <u>Industrial Laundry</u> (liquid): 8 exposure scenarios, each with 56 release scenarios for each of two release types (fugitive and stack) and 1 chemical release form (vapor only);
3. <u>Institutional Laundry</u> (liquid): 8 exposure scenarios, each with 56 release scenarios for each of two release types (fugitive and stack) and 1 chemical release form (vapor only);
4. <u>Industrial Laundry</u> (powder): 8 exposure scenarios, each with 56 release scenarios for each of two release types (fugitive and stack) and 3 chemical release form (vapor only, PM10, PM2.5); and
5. <u>Institutional Laundry</u> (powder): 8 exposure scenarios, each with 56 release scenarios for each of two release types (fugitive and stack) and 3 chemical release form (vapor only, PM10, PM2.5).

Table_Apx J-3. Exposure Scenarios and Inputs Utilized for Pre-screening Analysis of Hydraulic Fracturing, Industrial Laundry, and Institutional Laundry COU

COU	Release Percentile	Release Type	Release Duration (h/day)	Release Frequency (Days)	Chemical Phase/Form of Release	Meteorology	Topography
Hydraulic Fracturing	Maximum 99th 95th 50th 5th Minimum Mean	Fugitive	24 8	72 16 1 15	Vapor Only	South (Coastal)-HE West North Central-CT	Rural Urban
Industrial Laundry-Liquid	Maximum 99th 95th 50th 5th Minimum Mean	Fugitive Stack Unknown (Fugitive, Stack, Other)	24 8	365 223 20 260	Vapor Only	South (Coastal)-HE West North Central-CT	Rural Urban
Industrial Laundry-Powder	Maximum 99th 95th 50th 5th Minimum Mean	Fugitive Stack Unknown (Fugitive, Stack, Other)	24 8	365 223 20 260	Vapor Particulate (Coarse) Particulate (Fine)	South (Coastal)-HE West North Central-CT	Rural Urban
Institutional Laundry-Liquid	Maximum 99th 95th 50th 5th Minimum Mean	Fugitive Stack Unknown (Fugitive, Stack, Other)	24 8	365 287 250 260	Vapor Only	South (Coastal)-HE West North Central-CT	Rural Urban

COU	Release Percentile	Release Type	Release Duration (h/day)	Release Frequency (Days)	Chemical Phase/Form of Release	Meteorology	Topography
Institutional Laundry-Powder	Maximum 99th 95th 50th 5th Minimum Mean	Fugitive Stack Unknown (Fugitive, Stack, Other)	24 8	365 287 250 260	Vapor Particulate (Coarse) Particulate (Fine)	South (Coastal)-HE West North Central-CT	Rural Urban

Results

Results for the Ambient Air: IIOAC Methodology for COUs without Site-Specific Data for these three new COUs are summarized in Section 3.2.3.2 for exposure and Section 5.2.2.3.2 for estimated risks. Complete results are presented in *1,4-Dioxane Supplemental Information File: Air Exposure and Risk Estimates for 1,4-Dioxane Emissions from Hydraulic Fracturing Operations* (U.S. EPA, 2023b) and *1,4-Dioxane Supplemental Information File: Air Exposures and Risk Estimates for Industrial Laundry* (U.S. EPA, 2023c). Generally, results from application of this methodology found the following:

1. Hydraulic Fracturing: Lifetime cancer risk estimates for distance within 1000 m of hydraulic fracturing operations range from 1.7×10^{-3} to 7.7×10^{-6} across a range of high-end and central tendency release and exposure scenarios; and
2. Industrial and Institutional Laundry: Lifetime cancer risk estimates for distances within 1,000 m of laundry facilities range from 1.5×10^{-11} to 3.8×10^{-8} across a range of high-end and central tendency release and exposure scenarios.

J.1.3 Ambient Air: Single Year Methodology (AERMOD)

AERMOD was developed to allow EPA to conduct a higher-tier analysis of releases, exposures, and associated risks to people around releasing facilities at multiple distances when EPA has site-specific data like reported releases, facility locations (for local meteorological data), source attribution, and other data, when reasonably available. This methodology can also incorporate additional site-specific information like stack parameters (stack height, stack temperature, plume velocity, etc.), building characteristics, release patterns, different terrains, and other parameters when reasonably available. AERMOD can be performed independent of the Ambient Air: Screening Methodology described above, provides a more thorough analysis, can include wet and dry deposition estimates, and allows EPA to fully characterize identified risks for chemicals undergoing risk evaluation. While the application of this methodology in this supplemental risk evaluation focuses on a single year of data, the methodology can be expanded to include multiple years of data.

Model

AERMOD for this supplemental risk evaluation estimated 1,4-dioxane exposures to fenceline communities at user-defined distances from a facility releasing 1,4-dioxane. AERMOD is a steady-state Gaussian plume dispersion model that incorporates air dispersion based on planetary boundary layer turbulence structure and scaling concepts, including treatment of both surface and elevated sources and both simple and complex terrain. AERMOD can incorporate a variety of emission source characteristics, chemical deposition properties, complex terrain, and site-specific hourly meteorology to estimate air concentrations and deposition amounts at user-specified receptor distances and at a variety of averaging times. Readers can learn more about AERMOD, equations within the model, detailed input and output parameters, and supporting documentation by reviewing the AERMOD users guide (U.S. EPA, 2018d).

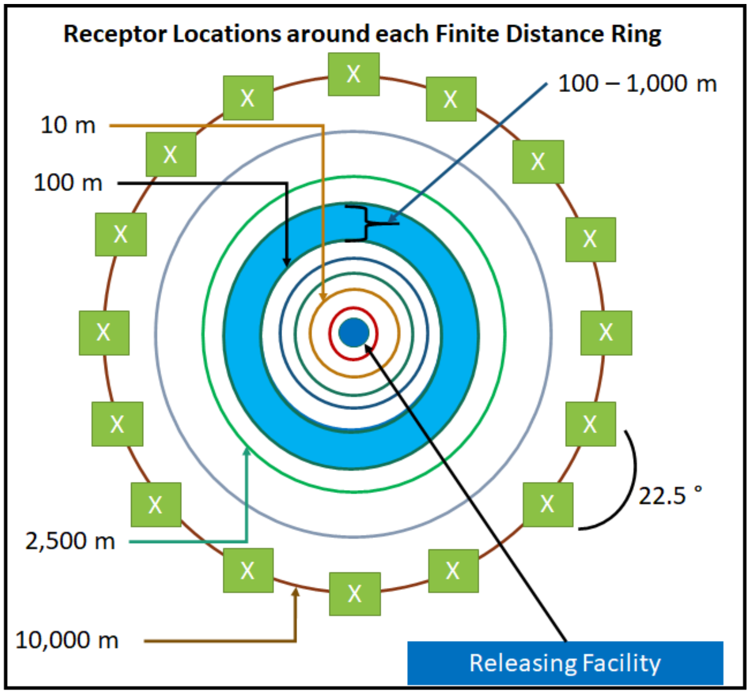
Releases

EPA modeled exposures using the release data developed as described in Section 2.1.1.2 and summarized below. Release data was provided (and modeled) on a facility-by-facility basis:

1. Facility-specific chemical releases (fugitive and stack releases) as reported to the 2019 TRI, where available.
2. Alternative release estimates as described in the decision tree for estimating air releases, where facility specific 2019 TRI data were not available. Alternative release estimates may include facility specific releases reported in previous TRI reporting years (2016 to 2018) or modeled release estimates using existing EPA models or other surrogate data.

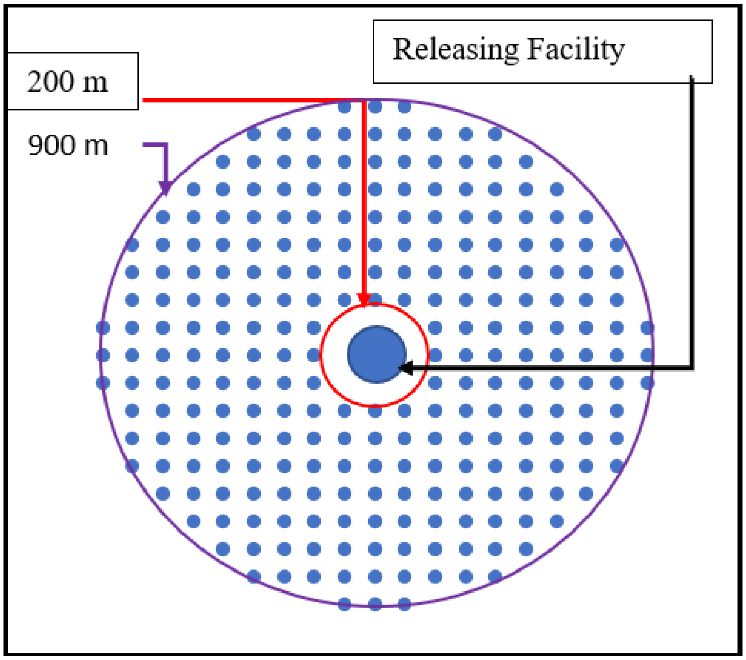
Exposure Scenarios

AERMOD evaluated exposures at eight finite distances (5, 10, 30, 60, 100, 2,500, 5,000, and 10,000 m) and one area distance (100 to 1,000 m) from each releasing facility (or generic facility for alternative release estimates). Receptors for each of the eight finite distances were placed in a polar grid every 22.5 degrees around the respective distance ring. This results in a total of 16 receptors around each finite distance ring for which exposures are modeled. Figure_Apx J-3 provides a visual depiction of the placement of receptors around a finite distance ring. Although the visual depiction only shows receptor locations around a single finite distance ring, the same placement of receptors occurred for all eight finite distance rings



Figure_Apx J-3. Modeled Receptor Locations for Finite Distance Rings

Receptors for the area distance evaluated were placed in a cartesian grid at equal distances between 200 and 900 m around each releasing facility (or generic facility for alternative release estimates). Receptors were placed at 100-meter increments. This results in a total of 456 receptors for which exposures are modeled. Figure_Apx J-4 provides a visual depiction of the placement of receptors (each dot) around the area distance ring.



Figure_Apx J-4. Modeled Receptor Locations for Area Distance

Exposure Concentration Outputs

Hourly-average concentration outputs were provided from AERMOD for each receptor around each distance ring (*i.e.*, each of 16 receptors around a finite distance ring or each receptor within the area distance ring). Daily and Period averages were then calculated from the modeled hourly data. Daily averages for the finite distance rings were calculated as arithmetic averages of all hourly data for each day modeled for each receptor around each ring. Daily averages for the area distance ring were calculated as the arithmetic average of the hourly data for each day modeled across all receptors within the area distance ring. This results in the following number of daily average concentrations at each distance modeled.

1. Daily averages for TRI reporting facilities (using 2016 calendar year meteorological data): One daily average concentration for each of 366 days for each of 16 receptors around each finite distance ring. This results in a total of 5,856 daily average concentration values for each finite distance modeled (366 × 16 = 5,856).
2. Daily averages for EPA estimated releases (using 2011 to 2015 meteorological data): Five daily average concentrations (for each year of meteorological data) for each of 365 (or 366) days for each of 16 receptors around each finite distance ring. This results in a total of 29,216 daily average concentration values for each finite distance modeled.
3. Daily averages for both TRI reporting facilities and EPA estimated releases: One daily average concentration for each of 365 or 366 days across all receptors within the area distance ring. This results in a total of 365 or 366 daily average concentration values for the area distance.

Period averages were calculated from all the daily averages for each receptor for each distance ring over 1 year for TRI reporting facilities and 5 years for facilities where releases were estimated. This results in a total of 16 period average concentration values for each finite distance ring. This is derived from either averaging the daily averages across the single year of meteorological data used (2016) for TRI reporting facilities or across the multi-year meteorological data used (2011 to 2015) for EPA estimated releases.

Daily and period average Outputs were stratified by different source scenarios, such as urban/not urban setting or emission-strengths, where needed. Outputs from AERMOD are provided in units of micrograms per cubic meter ($\mu g/m^3$) requiring conversion to parts per million (ppm) for purposes of calculating risk estimates for 1,4-dioxane. The following formula was used for this conversion:

$$C_{ppm} = (24.45*(C_{AERMOD})/1{,}000)/MW$$

Where:

C_{ppm} = Concentration (ppm)
24.45 = molar volume of a gas at 25 °C and 1 atmosphere pressure
C_{AERMOD} = Concentration from AERMOD ($\mu g/m^3$)
MW = Molecular weight of the chemical of interest (g/mole).

Post-processing scripts were used to extract and summarize the output concentrations for each facility, release, and exposure scenario. The following statistics for daily- and period-average concentrations were extracted or calculated from the results for each of the modeled distances (*i.e.*, each ring or grid of receptors) and scenarios (also see Table_Apx J-4):

- Minimum
- Maximum
- Average
- Standard deviation
- 10th, 25th, 50th, 75th, and 95th percentiles

Table_Apx J-4. Description of Daily or Period Average and Air Concentration Statistics

Statistic	Description
Minimum	The minimum daily or period average concentration estimated at any receptor location on any day at the modeled distance.
Maximum	The maximum daily or period average concentration estimated at any receptor location on any day at the modeled distance.
Average	Arithmetic mean of all daily or period average concentrations estimated at all receptor locations on all days at the modeled distance. This incorporates lower values (from days when the receptor location largely was upwind from the facility) and higher values (from days when the receptor location largely was downwind from the facility).
Percentiles	The daily or period average concentration estimate representing the numerical percentile value across the entire distribution of all concentrations at all receptor locations on any day at the modeled distance. The 50th percentile represents the median of the daily or period average concentration across all concentration values for all receptor locations on any day at the modeled distance.

J.1.4 Ambient Air: Multi-Year Analysis Methodology (IIOAC)

The multi-year analysis incorporates SACC recommendations by evaluating multiple years of chemical release data to estimate exposures and associated risks to fenceline communities. This is achieved by conducting a facility-by-facility evaluation of all 1,4-dioxane releases reported to TRI over six reporting years (2015 through 2020). Data for these 6 years were obtained from the TRI database (TRI basic plus files downloaded on August 5, 2022). Annual release data for 1,4-dioxane were extracted from the entire TRI data set for all facilities reporting air releases of 1,4-dioxane for one or more years between 2015 and 2020. Facilities were categorized into occupational exposure scenarios for modeling purposes and later cross-walked to COUs for risk management purposes.

The TRI data extracted for the multi-year analysis were used as direct inputs to the IIOAC model. An additional arithmetic average of the TRI data for each facility was also calculated when the facility reported releases to TRI for two or more of the years evaluated and used as a direct input to the IIOAC model. EPA then evaluated the more "conservative exposure scenario" of the 16 scenarios evaluated for the Ambient Air: Screening Methodology described above to estimate exposure concentrations. This more conservative exposure scenario consists of a facility that operates year-round (365 days per year, 24 hours per day, 7 days per week), a South Coastal meteorologic region, and a rural topography setting.

The Ambient Air: Multi-Year Analysis Methodology includes a land-use analysis utilizing the same visual methodology described for the 2022 fenceline analysis and the Ambient Air: Single Year Methodology (AERMOD). However, the land use analysis was limited those facilities where the multi-year analysis (1) found risk estimates above the benchmark value extending farther out when compared to the 2022 fenceline analysis, or (2) identified a new facility with risk estimates above the benchmark that was not captured by the 2022 fenceline analysis. Using this methodology, EPA identified if there is an expected exposure for people in fenceline communities to releases from the facility of interest within the distances where the benchmark was exceeded.

J.2 Inhalation Exposure Estimates for Fenceline Communities

Acute and chronic inhalation exposures were estimated based on air concentrations estimated in Section 2.3.3 using the methodologies described above.

Acute and chronic inhalation exposures used to evaluate non-cancer risks are estimated as an Acute Concentration (AC) or Average Daily Concentration (ADC), respectively. Lifetime exposures used to evaluate cancer risks are estimated as a Lifetime Average Daily Concentration (LADC). Methods adequate to quantify the impact of lifestage differences on 1,4-dioxane exposure are not available (see Section 4.3) and air concentration is used as the exposure metric for all lifestages per EPA guidance (U.S. EPA, 2012, 1994b).

The equations used to calculate each of the exposure values are:

$$AC = \frac{DAC \times ET}{AT}$$

$$ADC = \frac{AAC \times ET \times EF \times ED}{AT}$$

$$LADC = \frac{AAC \times ET \times EF \times ED}{AT}$$

Where:

AC = Acute Concentration (ppm)
DAC = Daily Average Air Concentration, model output reflecting average concentrations over a 24-hour period (ppm)
ET = Exposure Time (24 hours/day)
AAC = Annual Average Air Concentration, model output reflecting average concentrations over a year (ppm)
EF = Exposure Frequency (365 days/year)
ED = Exposure Duration (1 year for non-cancer ADC; 33 years for cancer LADC)
AT = Averaging Time

Averaging time for AC = 24 hours
Averaging time for ADC = 24 hours/day × 365 days/year × 1 year
Averaging time for LADC = 24 hours/day × 365 days/year × 78 years

For fenceline communities, all exposure estimates assume continuous exposure (24 hours/day) throughout the duration of exposure. The exposure duration used to calculate the LADC is based on the 95th percentile of the expected duration at a single residence, 33 years (U.S. EPA, 2011) and the averaging time is based on a 78-year lifetime.

Detailed reporting of modeled air concentrations and corresponding AC, ADC, and LADC estimates are provided in *1,4-Dioxane Supplemental Information File: Air Exposures and Risk Estimates for Single Year Analysis* (U.S. EPA, 2023e).

J.3 Land Use Analysis

As described in Section 5.2.2.3, EPA conducted a review of land use patterns around facilities where cancer risk exceeded 1×10^{-6}. The methodology for this analysis is consistent with what was previously described in the *Draft TSCA Screening Level Approach for Assessment Ambient Air and Water Exposures to Fenceline Communities Version 1.0*.[21] This review was limited to those facilities with real Global Information System (GIS) locations that showed risk. The land use analysis does not include generic facilities (since there is no real location around which to conduct the land use analysis) where alternative release estimates were modeled to estimate exposures. The purpose of this review was to determine if EPA can reasonably expect an exposure to fenceline communities to occur within the modeled distances where there was an indication of risk. This detailed review consisted of visual analysis using aerial imagery and interpreting land use/zoning practices around the facility. More specifically, EPA used ESRI ArcGIS (Version 10.8) and Google maps to characterize land use patterns within the radial distances evaluated where there was an indication of risk. For locations where residential or industrial/commercial businesses or other public spaces are present within those radial distances indicating risk, EPA includes those locations within the fenceline communities category and reasonably expects an exposure and therefore an associated potential risk. Where the radial distances showing an indication of risk occur within the boundaries of the facility or is limited to uninhabited areas, EPA does not reasonably expect an exposure to fenceline communities to occur and therefore does not expect an associated risk.

Table_Apx J-5. Summary of Fenceline Community Exposures Expected Near Facilities Where Modeled Air Concentrations Indicated Risk for 1,4-Dioxane

OES	COU	Total Number of Facilities Evaluated	Number of Facilities with Risk Indicated [a]	Number of Facilities with Risk Indicated and Fenceline Community Exposures Expected [a]	Percent of Total Facilities with Risk Indicated and Fenceline Community Exposures Expected [a]
Disposal	Disposal	15	4	1	7%
Ethoxylation Byproduct	Ethoxylation Byproduct	6	3	2	33%
Industrial Uses	Industrial Uses	12	4	1	8%
Manufacturing	Manufacture	1	1	1	100%

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[21] https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/tsca-screening-level-approach-assessing-ambient-air-and.

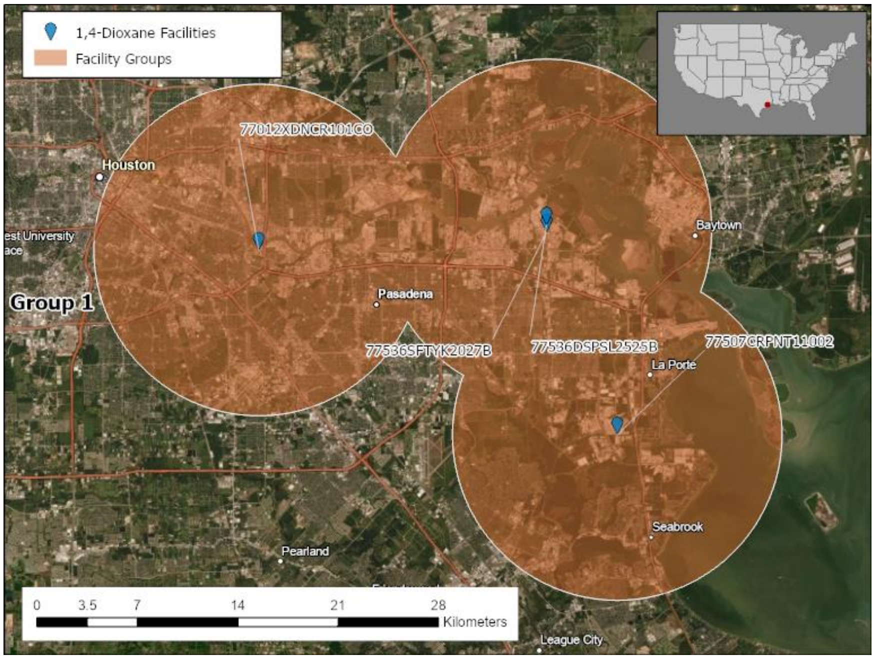
OES	COU	Total Number of Facilities Evaluated	Number of Facilities with Risk Indicated [a]	Number of Facilities with Risk Indicated and Fenceline Community Exposures Expected [a]	Percent of Total Facilities with Risk Indicated and Fenceline Community Exposures Expected [a]
PET Manufacturing	PET Manufacturing	13	10	6	46%
[a] Only includes facilities with TRI ID					

Individual facility summaries are available in *1,4-Dioxane Supplemental Information File: Air Exposures and Risk Estimates for Single Year Analysis* (U.S. EPA, 2023e).

J.4 Aggregate Analysis across Facilities

A conservative screening method for aggregated risk within the air pathway is included to address whether the combined general population exposures to emissions from nearby facilities present any additional risk not represented by the individual facility analysis. By taking a conservative approach, this methodology can effectively screen out aggregate concerns where no additional air risk is identified, and flag groups of facilities that demonstrate the potential for additional aggregate air risk.

The aggregate air approach utilized the existing modeling results for individual facilities, which modeled releases out to 10 km from the point of release. Facilities with releases to air were mapped using location coordinates from the TRI database. A 10 km buffer was drawn around each facility, and groups of facilities were identified by any overlap between these buffers (*i.e.*, any facilities within 20 km of another facility, even if not all of the facilities have overlapping buffers) (Figure_Apx J-5).



Figure_Apx J-5. Example of Group of Air Releasing Facilities with Overlapping 10 km Buffers for Aggregate Air Risk Screening

Next, the modeled air concentrations from each facility in the group were combined to generate hypothetical "worst-case scenario" aggregate air concentrations for the facility group. Due to the modeling methodology for individual facilities producing resulting air concentrations at discrete distances from each facility, the aggregate screening analysis also assesses concentrations and risk at discrete distances. For the sake of the analysis, the facilities are treated as if they are all releasing from the same point. This is a conservative approach, since the facilities with each group all have some distance between them, and the air concentrations tend to decrease with greater distance from the source facility. Within each facility group, the 95th percentile total (stack and fugitive) air concentrations for each facility were summed for each modeled distance interval. Cancer risk levels were similarly added together for each modeled distance interval, due to their proportional relationship to concentration, and non-cancer MOE values were combined using the equation below for each distance interval.

$$\boldsymbol{MOE_{total}} = \frac{\mathbf{1}}{\frac{\mathbf{1}}{\boldsymbol{MOE_1}} + \frac{\mathbf{1}}{\boldsymbol{MOE_2}} + \frac{\mathbf{1}}{\boldsymbol{MOE_3}} + \cdots}$$

Where:

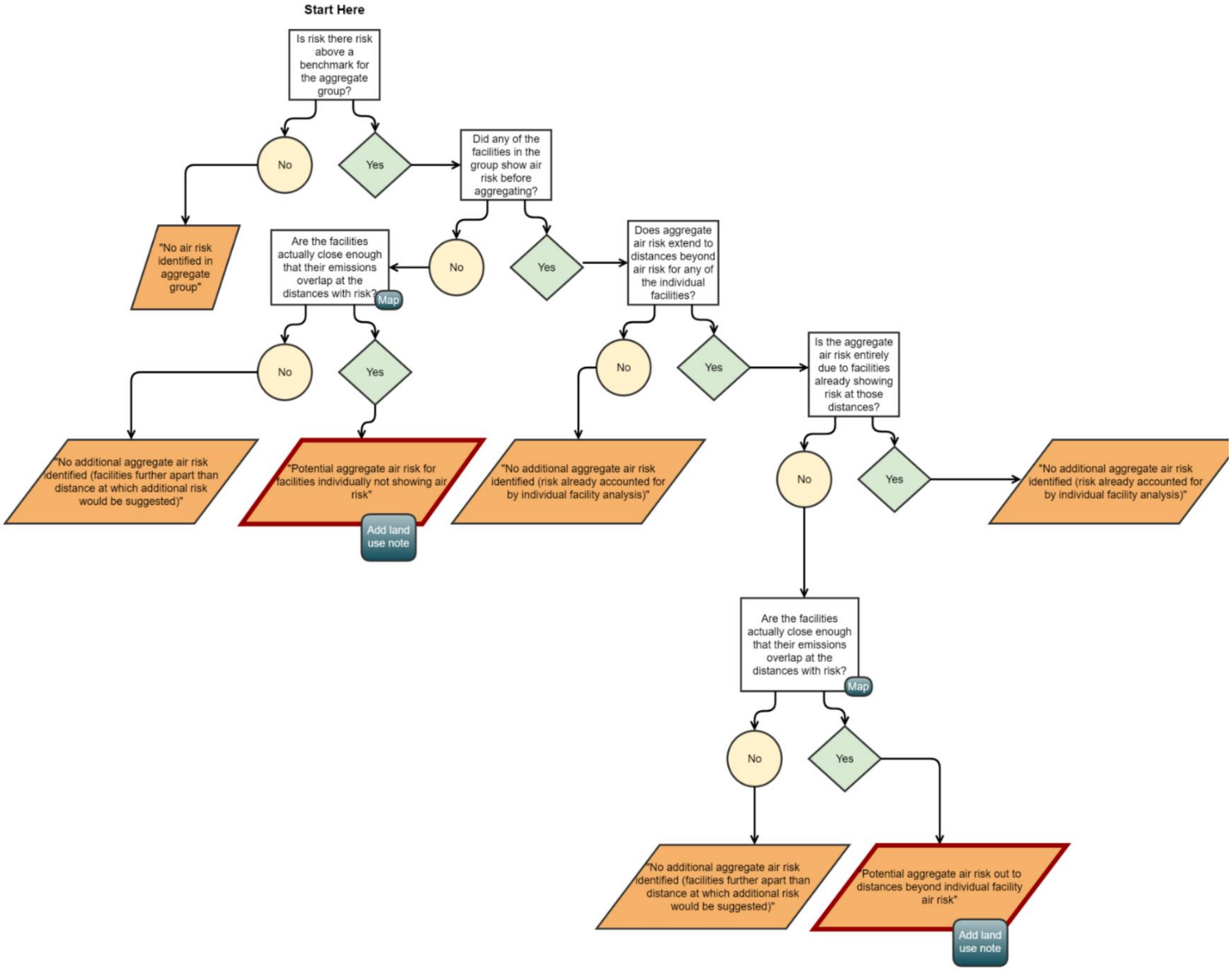
MOE_{total} = The aggregated MOE value for the group
$MOE_{(1,2,3\ldots)}$ = The individual MOE values for each facility in the group

Aggregated risk values were then compared against cancer and non-cancer benchmarks to identify values indicating risk relative to benchmarks. For each facility included in an aggregated group, it was noted whether the individual risk calculation results indicated risk relative to cancer or non-cancer benchmarks before aggregating. Additionally, for each facility group the relative contribution of each facility to the 95th percentile cancer risk was calculated, by dividing the individual facility risk by the aggregated group risk, to determine whether the resulting numbers may be disproportionately due to only one or more facilities. The resulting aggregate risk calculations were reviewed to determine where the numerical results suggested a concern for aggregate air risk that had not been represented by the individual facility risk analysis. Where this additional risk was flagged, the mapped locations of the facilities were then inspected to confirm that the distances between the facilities supported aggregating releases from the facilities at the flagged distance interval. The review of the aggregated results and facility locations was applied to characterize whether aggregate air risk relative to benchmarks is expected for each group.

For example, if the aggregate risk calculations for a group of two facilities indicated cancer risk greater than 1 in 1 million (1×10^{-6}) at the 100 m distance, and the individual facilities only showed that level of risk up to 60 m, the map would be inspected. If the facilities were found to be located 1,000 m apart, the group would be characterized as not showing risk relative to a 1 in 1 million benchmark beyond what was captured by the individual analysis. However, if the facilities were located within 200 m of one another, such that their 100 m distance intervals would intersect, the group would be characterized as showing potential for aggregated air risk beyond what was captured by the individual analysis.

If aggregate air risk relative to benchmarks is identified, then an additional land use check is performed to confirm the potential for a general population exposure at the new distance. In some cases, no additional aggregate air risk is identified, because no distance intervals present risk relative to benchmarks.

12167
12168



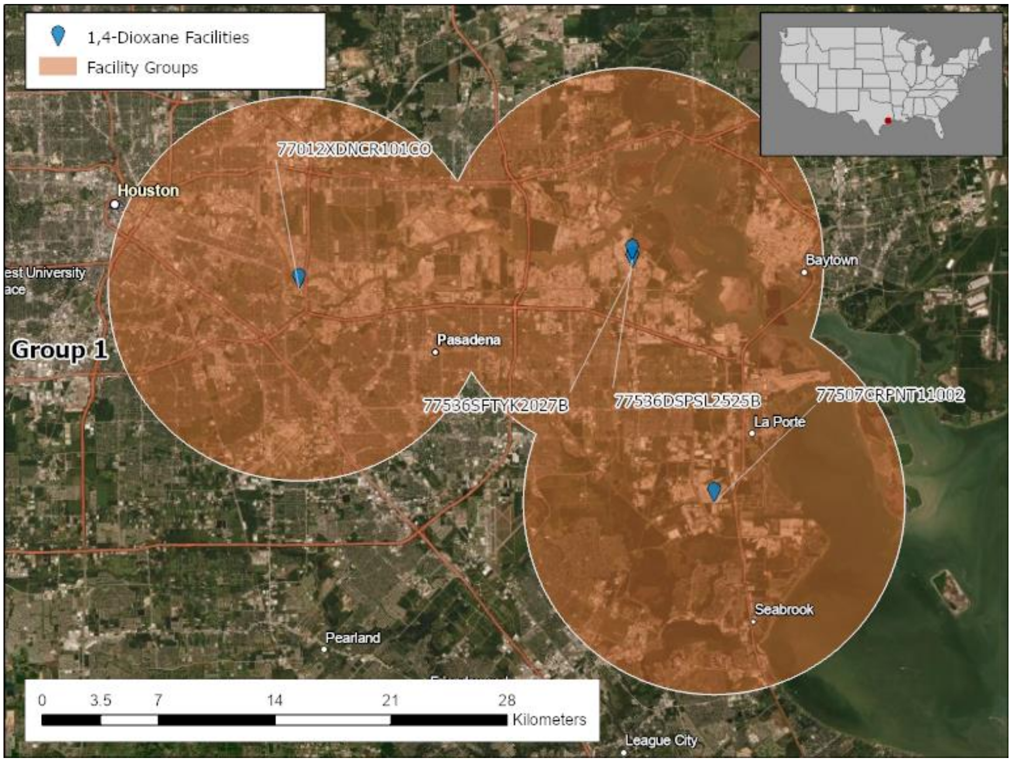
Figure_Apx J-6. Decision Tree for Characterizing Aggregate Air Risk for Multiple Facilities

Table_Apx J-6. Summary of Groups of Facilities Considered in Aggregate Analysis

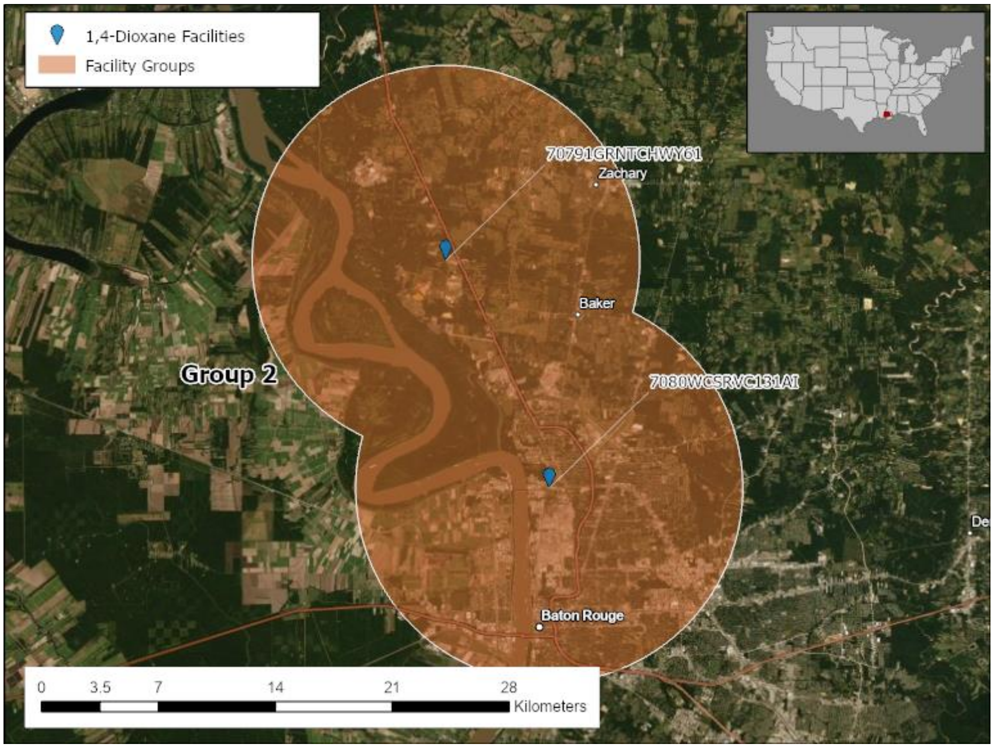
Total Air Facilities with Release Data	Number of Facilities in Groups	Number of Groups	Number of Groups with Additional Aggregate Risk
50	12	5	0

The grouping analysis for 1,4-dioxane resulted in five groups of nearby facilities, ranging from two to four facilities per group. No additional aggregate air risk relative to benchmarks was identified for each of the five groups. Where three groups each contained a single facility showing risk out to some distance, there was no additional distance interval showing risk from the aggregate calculation. While the proximity of the facilities may indicate a reality of greater localized air concentrations than are represented in the individual facility analysis, the aggregated concentrations did not cross any additional risk benchmarks, so any determinations of risk are already accounted for by the individual facility analysis. For the remaining two groups, no aggregated or individual risks were present. Therefore, further inspection and additional land use analysis were not warranted for these facility groups.

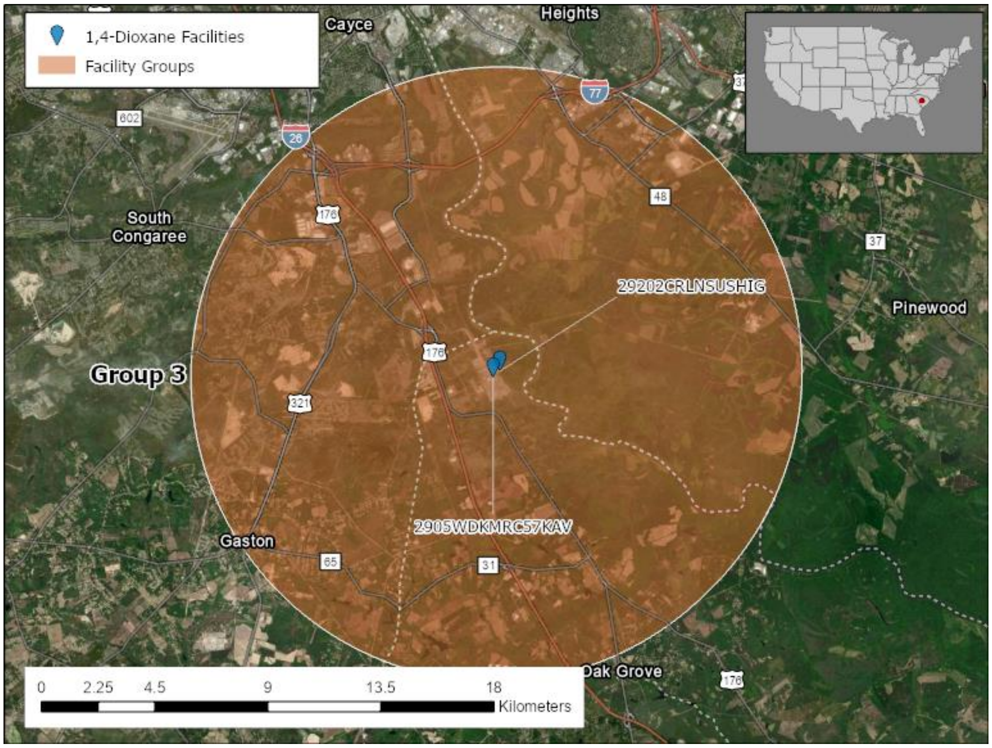
Maps of the five facility groups, with the 10 km buffers used to define them are provided below in Figure_Apx J-7 through Figure_Apx J-11.



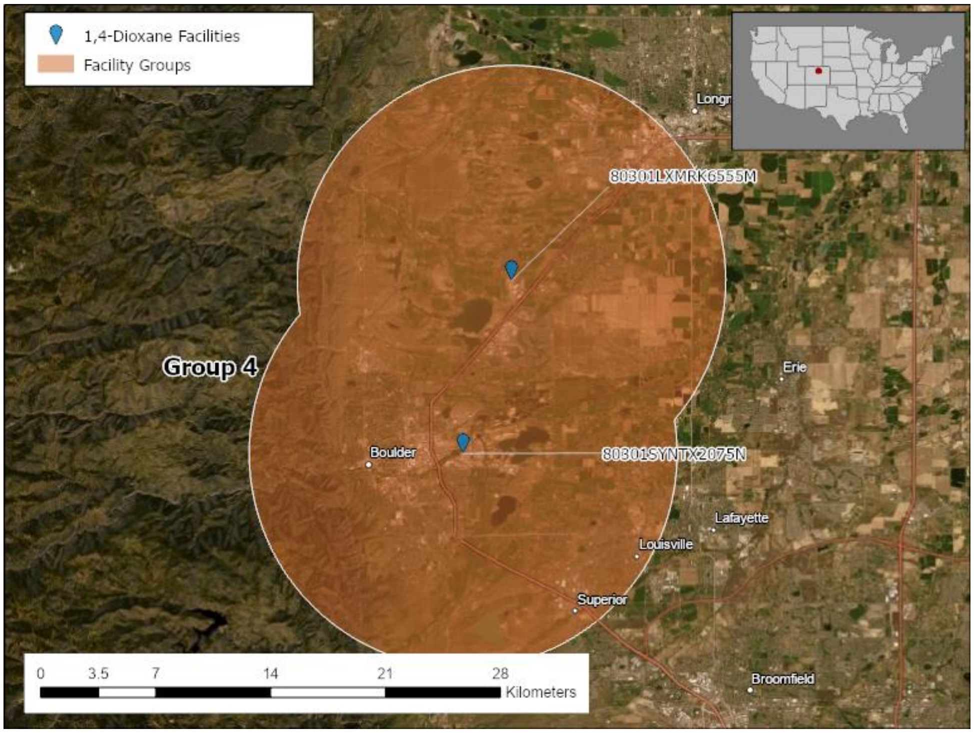
Figure_Apx J-7. Map of Aggregated Air Facilities, Group 1



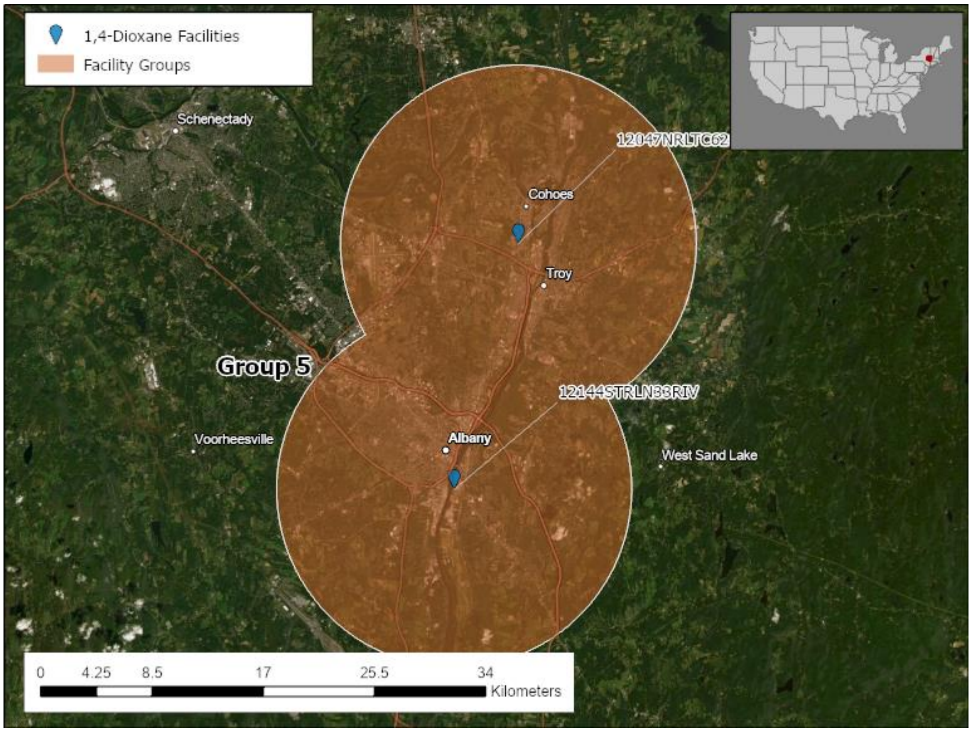
Figure_Apx J-8. Map of Aggregated Air Facilities, Group 2



Figure_Apx J-9. Map of Aggregated Air Facilities, Group 3



Figure_Apx J-10. Map of Aggregated Air Facilities, Group 4



Figure_Apx J-11. Map of Aggregated Air Facilities, Group 5