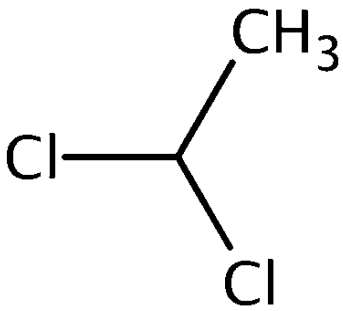
EPA Document# EPA-740-R-25-013
June 2025
Office of Chemical Safety and
Pollution Prevention

United States
Environmental Protection Agency

Risk Evaluation for 1,1-Dichloroethane

CASRN 75-34-3



June 2025

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

KEY ABBREVIATIONS, ACRONYMS, AND GLOSSARY OF SELECT TERMS

Key Abbreviations and Acronyms

7Q10	Lowest 7-day average flow occurring in a 10-year period
30Q5	Lowest 30-day average flow occurring in a 5-year period
ACGIH	American Conference of Governmental Industrial Hygienists
ACS	American Community Survey
ADME	Absorption, distribution, metabolism, and elimination
AF	Assessment factor
AIM	Analog Identification Methodology
AMTIC	Ambient Monitoring Technology Information Center
APF	Assigned protection factor
ATSDR	Agency for Toxic Substances and Disease Registry
BAF	Bioaccumulation factor
BCF	Bioconcentration factor
BMC	Benchmark concentration
BMD	Benchmark dose
BMR	Benchmark response
CAA	Clean Air Act
CAP	Criteria Air Pollutants
CASRN	Chemical Abstracts Service Registry Number
CBI	Confidential Business Information
CDR	Chemical Data Reporting
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
CHRIP	Chemical Risk Information Platform
ChV	Chronic Value
COC	Concentration(s) of concern
CR	Cancer risk
CRD	Chronic retained dose
CSATAM	Community-Scale Air Toxics Ambient Monitoring
CSCL	Chemical Substances Control Law
CWA	Clean Water Act
CWS	Community water systems
CYP	Cytochrome P450
DMR	Discharge Monitoring Report
DOE	Days of exceedance
DOT	Department of Transportation
ECEL	Existing chemical exposure limit
ECHA	European Chemicals Agency
ECHO	Enforcement and Compliance History Online
ECx	Effect concentration at which x percent of test organisms exhibit an effect
EPA	Environmental Protection Agency
EPCRA	Emergency Planning and Community Right-to-Know Act
ERS	Environmental release scenario(s)
ESD	Emission Scenario Document

EU	European Union
GD	Gestation day
GS	Generic Scenario(s)
GSH	Glutathione
HAP	Hazardous Air Pollutant
HC05	Hazardous concentration for 5 percent of species
HEC	Human Equivalent Concentration
HED	Human Equivalent Dose
HERO	Health and Environmental Research Online (Database)
HM	Harmonic Mean
HMTA	Hazardous Materials Transportation Act
HSDB	Hazardous Substances Data Bank
ICIS	Integrated Compliance Information System
IMAP	Inventory Multi-Tiered Assessment and Prioritisation
IRIS	Integrated Risk Information System
ISHA	Industrial Safety and Health Act
IUR	Inhalation Unit Risk
K_{OC}	Organic carbon: water partition coefficient
K_{OW}	Octanol: water partition coefficient
LADC	Lifetime average daily concentration
LADD	Lifetime average daily dose
LCRD	Lifetime chronic retained dose
LCx	Lethal concentration at which x percent of test organisms die
LDx	Lethal dose at which x percent of test organisms die
LOD	Limit of detection
LOAEL	Lowest-observed-adverse-effect-level (LOAEL
LOEC	Lowest-observed-effect-concentration
MACT	Maximum Achievable Control Technology
MCL	Maximum Contaminant Level
MSW	Municipal solid waste
NAAQS	National Ambient Air Quality Standard
NAC	National Advisory Committee
NAICS	North American Industry Classification System
NATA	National Scale Air-Toxics Assessment
NCR	Non-cancer risk
ND	Non-detect
NEI	National Emissions Inventory
NESHAP	National Emission Standards for Hazardous Air Pollutants
NHD	National Hydrography Dataset
NICNAS	National Industrial Chemicals Notification and Assessment Scheme
NIH	National Institutes of Health
NIOSH	National Institute for Occupational Safety and Health
NITE	National Institute of Technology and Evaluation
NOAEL	No-observed-adverse-effect-level
NOEC	No-observed-effect-concentration
NPDES	National Pollutant Discharge Elimination System
NPDWR	National Primary Drinking Water Regulation
NRC	National Response Center
NTP	National Toxicology Program

OCSPP	Office of Chemical Safety and Pollution Prevention
OECD	Organisation for Economic Co-operation and Development
OEHHA	Office of Environmental Health Hazard Assessment
OEL	Occupational exposure limit
OES	Occupational exposure scenario
ONU	Occupational non-user
OPPT	Office of Pollution Prevention and Toxics
ORD	Office of Research and Development
OSHA	Occupational Safety and Health Administration
PBPD	Physiologically based pharmacodynamic
PBPK	Physiologically based pharmacokinetic
PBZ	Personal breathing zone
PECO	Population, exposure, comparator, and outcome
PEL	Permissible exposure limit
POD	Point of departure
POTW	Publicly owned treatment works
PPE	Personal protective equipment
PSC	Point Source Calculator
PV	Production volume
PWS	Public Water Systems
QSAR	Quantitative structure-activity relationship
RCRA	Resource Conservation and Recovery Act
REACH	Registration, Evaluation, Authorisation and Restriction of Chemicals (European Union)
REL	Recommended exposure limit
RfD	Reference Dose
RQ	Reportable Quantity OR Risk Quotient
RTR	Risk and technology review
SADC	Subchronic average daily concentration
SCDD	Subchronic average daily dose
SDS	Safety data sheet
SDWA	Safe Drinking Water Act
SR	Systematic review
SSD	Species sensitivity distribution
STEL	Short-Term Exposure Limit
TGD	European Commission Technical Guidance Document
TLV	Threshold Limit Value
TRI	Toxics Release Inventory
TRV	Toxicity reference value
TSCA	Toxic Substances Control Act
TWA	Time-weighted average
UCMR3	Third Unregulated Contaminant Monitoring Rule
UF	Uncertainty factor
U.S.	United States
USGS	United States Geological Survey
VOC	Volatile organic compound
WHO	World Health Organization
WQP	Water Quality Portal

Glossary of Select Terms

Aggregate exposure (40 CFR 702.33) (accessed June 16, 2025): "means the combined exposures from a chemical substance across multiple routes and across multiple pathways."

Aggregate risk (U.S. EPA, 2003): "The risk resulting from aggregate exposure to a single agent or stressor."

Biomonitoring (U.S. EPA, 2019): "measures the amount of a stressor in biological matrices."

Central Tendency Exposure: EPA's *Guidelines for Human Exposure Assessment* (accessed June 16, 2025) defined central tendency exposures as "an estimate of individuals in the middle of the distribution."

Chemical substance (15 U.S.C. § 2602(2)) (accessed June 16, 2025): "means any organic or inorganic substance of a particular molecular identity, including—(i) any combination of such substances occurring in whole or in part as a result of a chemical reaction or occurring in nature, and (ii) any element or uncombined radical. Such term does not include—(i) any mixture, (ii) any pesticide (as defined in the Federal Insecticide, Fungicide, and Rodenticide Act [7 U.S.C. 136 et seq.]) when manufactured, processed, or distributed in commerce for use as a pesticide, (iii) tobacco or any tobacco product, (iv) any source material, special nuclear material, or byproduct material (as such terms are defined in the Atomic Energy Act of 1954 [42 U.S.C. 2011 et seq.] and regulations issued under such Act), (v) any article the sale of which is subject to the tax imposed by section 4181 of the Internal Revenue Code of 1986 [26 U.S.C. 4181] (determined without regard to any exemptions from such tax provided by section 4182 or 4221 or any other provision of such Code) and any component of such an article (limited to shot shells, cartridges, and components of shot shells and cartridges), and (vi) any food, food additive, drug, cosmetic, or device (as such terms are defined in section 201 of the Federal Food, Drug, and Cosmetic Act [21 U.S.C. 321]) when manufactured, processed, or distributed in commerce for use as a food, food additive, drug, cosmetic, or device."

Conditions of use (COUs) (15 U.S.C. § 2602(4)) (accessed June 16, 2025): "means the circumstances, as determined by the Administrator, under which a chemical substance is intended, known, or reasonably foreseen to be manufactured, processed, distributed in commerce, used, or disposed of."

Consumer exposure (40 CFR § 711.3) (accessed June 16, 2025): Human exposure resulting from consumer use. This exposure includes passive exposure to consumer bystanders.

Consumer use (40 CFR § 711.3) (accessed June 16, 2025): "means the use of a chemical substance or a mixture containing a chemical substance (including as part of an article) when sold to or made available to consumers for their use."

Fenceline exposure: General population exposures occuring in communities near facilities that emit or release chemicals to air, water, or land with which they may come into contact.

General population: The human population potentially exposed to chemicals released into the environment.

High-end exposure: EPA's *Guidelines for Human Exposure Assessment* (accessed June 16, 2025) defined are defined as plausible estimate of individual exposure for those individuals at the upper end of

an exposure distribution, the intent of which is to convey an estimate of exposure in the upper range of the distribution while avoiding estimates that are beyond the true distribution."

Margin of exposure (MOE) (U.S. EPA, 2002a): "a numerical value that characterizes the amount of safety to a toxic chemical–a ratio of a toxicological endpoint (usually a NOAEL [no observed adverse effect level]) to exposure. The MOE is a measure of how closely the exposure comes to the NOAEL."

Mode of action (MOA) (U.S. EPA, 2000b): "a series of key events and processes starting with interaction of an agent with a cell, and proceeding through operational and anatomical changes causing disease formation."

Non-chemical stressors (U.S. EPA, 2022b): "Non-chemical stressors are factors found in the built, natural, and social environments including physical factors such as noise, temperature, and humidity and psychosocial factors (*e.g.*, poor diet, smoking, and illicit drug use)."

Occupational exposure: Exposure to a chemical substance by industrial or commercial workers.

Occupational non-users (ONU): Employed persons who do not directly handle the chemical substance but may be indirectly exposed to it as part of their employment due to their proximity to the substance.

Pathways (40 CFR § 702.33) (accessed June 16, 2025): "means the physical course a chemical substance takes from the source to the organism exposed."

Point of departure (POD) (U.S. EPA, 2002a): "dose that can be considered to be in the range of observed responses, without significant extrapolation. A POD can be a data point or an estimated point that is derived from observed dose-response data. A POD is used to mark the beginning of extrapolation to determine risk associated with lower environmentally relevant human exposures."

Potentially exposed or susceptible subpopulation (PESS) (15 U.S.C. § 2602(12)) (accessed June 16, 2025): "means a group of individuals within the general population identified by the Agency who, due to either greater susceptibility or greater exposure, may be at greater risk than the general population of adverse health effects from exposure to a chemical substance or mixture, such as infants, children, pregnant women, workers, or the elderly."

Risk Quotient (RQ): Risk quotients are unitless values that characterize risk calculated as the environmental concentration divided by the effect level. Environmental concentration is represented by predicted, monitored, and/or literature-based environmental concentrations. The effect level is represented by concentrations of concern (COCs) for aquatic receptors, toxicity reference values (TRVs) for terrestrial receptors, or hazard values when appropriate.

Reasonably available information (40 CFR 702.33) (accessed June 16, 2025): "means information that EPA possesses or can reasonably generate, obtain, and synthesize for use in risk evaluations, considering the deadlines specified in TSCA section 6(b)(4)(G) for completing such evaluation. Information that meets the terms of the preceding sentence is reasonably available information whether or not the information is confidential business information (CBI), that is protected from public disclosure under TSCA section 14."

Routes (40 CFR 702.33) (accessed June 16, 2025): "means the ways a chemical substance enters an organism after contact, *e.g.*, by ingestion, inhalation, or dermal absorption."

Sentinel exposure (40 CFR 702.33) (accessed June 16, 2025): "means the exposure from a chemical substance that represents the plausible upper bound of exposure relative to all other exposures within a broad category of similar or related exposures."

Stressor (U.S. EPA, 2019): "Any chemical, physical or biological entity that induces an adverse response."

ACKNOWLEDGEMENTS

The Assessment Team gratefully acknowledges the participation, input, and review comments from U.S. Environmental Protection Agency (EPA or the Agency) OPPT and OCSPP senior managers and science advisors as well as assistance from EPA contractors Abt Global (Contract No. EP-W-16-009); ICF (Contract No. 68HERC23D0007); ERG (Contract No. 68HERD20A0002; 68HERC21D0003); and SRC (Contract No. 68HERH19D0022; 68HERH19F0213).

Special acknowledgement is given for the contributions of technical experts from EPA ORD-CESER, Randall Ross and David Burden; ORD-CCTE-CCED-CCCB, Tony Williams; OCSPP-OPPT-DGMPD-TAIB-TAIS1, Andrea Hindman; and ORD-CPHEA-CPAD-TEABC, Jonathan Phillip Kaiser—for their joint efforts.

Docket
Supporting information can be found in public docket, Docket IDs EPA-HQ-OPPT-2018-0426 and EPA-HQ-OPPT-2024-0114.

Disclaimer
Reference herein to any specific commercial products, process, or service by trade name, trademark, manufacturer or otherwise does not constitute or imply its endorsement, recommendation, or favoring by the United States Government.

Authors: Janet Burris (Assessment Lead), Seema Schappelle (Branch Supervisor), Clara Hull, Aderonke Adegbule, Katherine Anitole, Albana Bega, Jennifer Brennan, Craig Connolly, Andrea Hindman, Lauren Housley, Jonathan Kaiser, Ryan Klein, William Irwin, David Lynch, Greg Macek, Andrew Middleton, Nerija Orentas, Christina Robichaud, Ali Shohatee, Kelley Stanfield, Nicholas Suek, and Catherine Taylor

Contributors: Sarah Au, Tyler Amrine, Brian Barone, Joshua Booth, Nicholas Castaneda, Jone Corrales, Kellie Fay, Rebecca Feldman, Janine Fetke, Patricia Fontenot, Ross Geredien, Bryan Groza, Annie Jacob, Keith Jacobs, June Kang, Grace Kaupas, Virginia Lee, Yadi Lopez, Matt Lloyd, Benjamin Kunstman , Edward Lo, Bryan Lobar, Kiet Ly, Rony Arauz Melendez, Bethany Masten, Azah Abdalla Mohamed, Brianne Raccor, Simon Regenold, Anthony Rufka, Abhilash Sasidharan, Cory Strope, David Turk, Leora Vegosen, Kevin Vuilleumier, Jason Wight, William Wimbish, and Eva Wong

Technical Support: Marcy Card, Mark Gibson, Hillary Hollinger, S. Xiah Kragie, and Houbao Li

EXECUTIVE SUMMARY

EPA evaluated 1,1-dichloroethane under the Toxic Substances Control Act (TSCA). In this risk evaluation, the Agency found that 1,1-dichloroethane presents an unreasonable risk of injury to human health driven by three conditions of use (COUs) because of risks to workers. EPA did not identify risk of injury to the general population or to the environment associated with any COU that would drive the unreasonable risk determination for 1,1-dichloroethane.

In December 2019, EPA designated 1,1-dichloroethane as a high-priority substance for TSCA evaluation and in August 2020 released the final scope of the risk evaluation. This final risk evaluation assesses human health risk to workers, the general population, and the environment. No consumer or bystander exposures were assessed because no consumer COUs or commercial or consumer products or articles were identified in this final risk evaluation.

1,1-Dichloroethane is manufactured in the United States, is not imported, and is used to produce other chlorinated solvents that have broad industrial applications. Relatively small amounts of 1,1-dichloroethane support commercial use in laboratory chemicals. The reported total domestic production volume in 2020 was between 100 million and 1 billion pounds for two corporations located in the southern United States.[1] 1,1-Dichloroethane is a colorless, oily liquid with a chloroform- or ether-like odor and is volatile and soluble in water. As reported in EPA databases,[2] 1,1-dichloroethane is released to air, surface waters (including sediments), and land and will partition between these environmental media. EPA evaluated facility-specific or modeled releases to air, water, and land for each COU scenario and estimated potential exposures to the general population and to the environment.

The Agency evaluated 1,1-dichloroethane from manufacture to disposal. For exposure assessment, EPA used chemical-specific data where available; however, surrogate data and modeling were used to characterize certain scenarios that lacked monitoring data (*e.g.*, laboratory use of 1,1-dichloroethane). For human health hazard assessment, EPA used 1,2-dichloroethane (an isomer of 1,1-dichloroethane) as an analog. To characterize aquatic environmental hazard, the Agency used 1,1,2-trichlorethane and 1,2-dichloropropane as analogs. In July 2024, EPA released the *Draft Risk Evaluation for 1,1-Dichloroethane* (accessed June 16, 2025) for public comment and external peer review by the Science Advisory Committee on Chemicals (SACC). As part of the SACC deliberations, the Agency held a virtual public meeting to discuss the draft risk evaluation on September 17 to 20, 2024.

Unreasonable Risk to Human Health

EPA evaluated reasonably available information for human health hazards from 1,1-dichloroethane and did not find adequate data for human health hazard assessment and, for this reason, the Agency used hazard data for 1,2-dichloroethane as a read-across analog. Although EPA was not able to quantify the toxicological differences between 1,1-dichloroethane and 1,2-dichloroethane due to the limited data available for 1,1-dichloroethane, the Agency did identify 1,2-dichloroethane as the most appropriate analog for the risk evaluation—recognizing it was a conservative and therefore health protective, read-across approach. This is based on analyses of structural, physical, chemical, metabolic, and qualitative cancer and non-cancer toxicological similarities. Based on hazard read-across data from 1,2-dichloroethane, exposure to 1,1-dichloroethane may increase the risk of (1) non-cancer renal effects from acute, intermediate, and chronic oral/dermal exposure; (2) non-cancer olfactory effects from acute inhalation exposure; (3) non-cancer male reproductive effects from intermediate and chronic inhalation

[1] EPA describes production volumes as a range to protect information claimed as confidential business information.
[2] EPA compiled release data from TRI (Toxics Release Inventory), NEI (National Emissions Inventory), and DMR (Discharge Monitoring Reports) during the 2015 to 2020 timeframe.

exposure; and (4) tumors (combined mammary gland adenomas, fibroadenomas, and adenocarcinomas and subcutaneous fibromas) from inhalation exposure. EPA evaluated risks to workers and the general population using reasonably available monitoring and modeling data for inhalation and dermal exposures, as applicable. EPA also evaluated risk from inhalation and dermal exposure of 1,1-dichloroethane to workers as well as inhalation exposures to occupational non-users (ONUs). Workers with the greatest potential for exposure to 1,1-dichloroethane are those who work directly with the chemical in environments where 1,1-dichloroethane is manufactured, processed, or disposed.

For the general population, EPA evaluated risk from (1) inhalation exposure; (2) dermal exposures to swimmers; and (3) oral exposures via drinking water, fish ingestion, incidental oral ingestion from swimming, and soil. When determining the unreasonable risk of 1,1-dichloroethane to human health, in addition to workers, EPA also accounted for other potentially exposed and susceptible subpopulations (PESS), which included the following: infants exposed to drinking water during formula bottle feeding, subsistence and tribal fishers, individuals with pre-existing conditions such as chronic kidney disease, people with the aldehyde dehydrogenase-2 polymorphism, lifestyle factors such as smoking cigarettes or secondhand smoke, and fenceline communities.

EPA evaluated exposures to the general population associated with (1) breathing the ambient air where 1,1-dichloroethane was released from facilities; and (2) ingesting drinking water, surface water, or soil from 1,1-dichloroethane disposed to land (*i.e.*, land-applied biosolids from public wastewater treatment works treating 1,1-dichloroethane-containing wastewater). The Agency did not identify unreasonable risk to the general population. EPA also evaluated subsistence fishers and did not find unreasonable risk.

EPA's assessment shows unreasonable risks to workers from non-cancer and cancer health effects from exposure to 1,1-dichloroethane driven by the following COUs: Processing as a reactant as an intermediate in all other basic organic chemical manufacturing, Processing as a reactant as an intermediate in all other chemical product and preparation manufacturing, and Processing – recycling.

Unreasonable Risk to the Environment

EPA assessed 1,1-dichloroethane exposures to the environment through the manufacturing, processing, use, or disposal of 1,1-dichloroethane. Exposure to aquatic species was evaluated through surface water and sediment; exposure to terrestrial species was evaluated through soil, surface water, and sediment. EPA's assessment did not identify risk of injury to the environment that would contribute to the unreasonable risk determination for 1,1-dichloroethane.

Conclusions

EPA determined its assessment identifies unreasonable risk to workers from non-cancer and cancer effects due to workplace inhalation exposure. The Agency has confidence in its unreasonable risk determination for 1,1-dichloroethane for workers and in not identifying unreasonable risk for the general population or the environment due to the conservativeness of the analysis and assumptions used in its assessment. Examples of the conservative analysis included the use of 1,2-dichloroethane as a read-across analog and use of modeled data in the absence of measured data (*e.g.*, test order data). The unreasonable risk identified for workers due to inhalation exposure would no longer be unreasonable when using respirators in a manner that achieves minimum assigned protection factor (APF) levels of 10 to 25 (depending on the expected workplace activity, represented in the risk evaluation by Similar Exposure Groups [SEGs]) or implementing other exposure controls (*e.g.*, engineering controls) that may be equally or more effective in reducing worker exposures.

Eight COUs were evaluated for 1,1-dichloroethane. EPA determined that 1,1-dichloroethane presents an unreasonable risk of injury to human health driven by identified risk to workers from three COUs. The following three COUs significantly contribute to the unreasonable risk determination due to identified risk to workers:

- Processing as a reactant as an intermediate in all other basic organic chemical manufacturing;
- Processing as a reactant as an intermediate in all other chemical product and preparation manufacturing; and
- Processing – recycling.

The following five COUs do not significantly contribute to the unreasonable risk determination for 1,1-dichloroethane:

- Manufacturing as an isolated intermediate (domestic manufacture);
- Processing – repackaging;
- Distribution in commerce;
- Commercial use in laboratory chemicals; and
- Disposal.

After considering the risks posed under the COUs, EPA did not identify unreasonable risk of injury to the general population or to the environment associated with any COU.

Key Updates to the Risk Evaluation for 1,1-Dichloroethane

In response to public and SACC comments on the 2024 draft risk evaluation, EPA made the following major revisions to the risk evaluation for 1,1-dichloroethane:

1. Further characterized and incorporated five additional human health hazard studies into the hazard assessment.
2. Incorporated hazard data received from a 1,1-dichloroethane TSCA section 4(a)(2) test order into the environmental hazard assessment, including updates to the Web-based Interspecies Correlation Estimation (Web-ICE) and species sensitivity distribution (SSD) for aquatic toxicity resulting from acute exposures. As a result, the aquatic acute concentration of concern (COC) was revised to a lower value. Associated risk quotients (RQs) were subsequently recalculated but did not result in increased risk.
3. Updated the ecotoxicological similarity analysis within the analog analysis for environmental hazard in response to peer review comments.
4. Refined the characterization of the read-across approach for both environmental hazard and human health hazard in response to SACC recommendations.
5. Removed the laboratory rodent drinking water exposure study (Klaunig et al., 1986) from the mammalian wildlife toxicity reference value (TRV).
6. Revised the endpoint for the point of departure (POD) selection based on SACC recommendations resulting in a higher intermediate and chronic oral/dermal non-cancer POD. These changes resulted in higher revised intermediate and chronic oral/dermal non-cancer risk estimates for general population and workers, respectively.
7. Based on input from the SACC, EPA changed the oral/dermal cancer assessment from quantitative (as proposed in the draft risk evaluation) to qualitative. These updates were based on a re-evaluation of the uncertainties associated with the available data and overall weight of scientific evidence for dose-response analyses.
8. Revised the dermal absorption for the Waste handling COU to use submitted test order data for dilute 1,1-dichloroethane fraction absorbed as recommended by the SACC.
9. Incorporated considerations of personal protective equipment (PPE) use into the risk characterization and risk estimate tables.
10. Added a storm scenario for the facility that reported discharges of 1,1-dichloroethane into surface waters during storm events.
11. Improved characterization of inputs and assumptions along with confidence and uncertainties in exposure and risk estimates throughout the risk evaluation.
12. Revised the unreasonable risk determination for workers based on risk estimate revisions and used the central tendency instead of the high-end for dermal exposure risk determination under all of the COUs.
13. Revised the unreasonable risk determination to no longer identify unreasonable risk for workers, including ONUs, based on revised considerations of risk factors for the Processing – repackaging as well as Disposal COUs. The number of COUs that significantly contribute to the unreasonable risk to human health for 1,1-dichloroethane was lowered from seven to three.
14. Revised the unreasonable risk determination for the environment based on the total number of operating days release scenario. As a result, the Agency did not identify any COU as significantly contributing to unreasonable risk of injury to the environment.

1 INTRODUCTION

EPA has evaluated 1,1-dichloroethane under the Toxic Substances Control Act (TSCA). 1,1-Dichloroethane is a colorless, oily liquid with a chloroform-like odor, which is primarily used in organic chemical manufacturing. Section 1.1 provides production volume, life cycle diagram (LCD), conditions of use (COUs), and conceptual models used for 1,1-dichloroethane; Section 1.2 includes an overview of the systematic review process; and Section 1.3 presents the organization of this risk evaluation. Figure 1-1 describes the major inputs, phases, and outputs/components of the TSCA risk evaluation process (accessed June 16, 2025) from scoping to releasing the final risk evaluation.

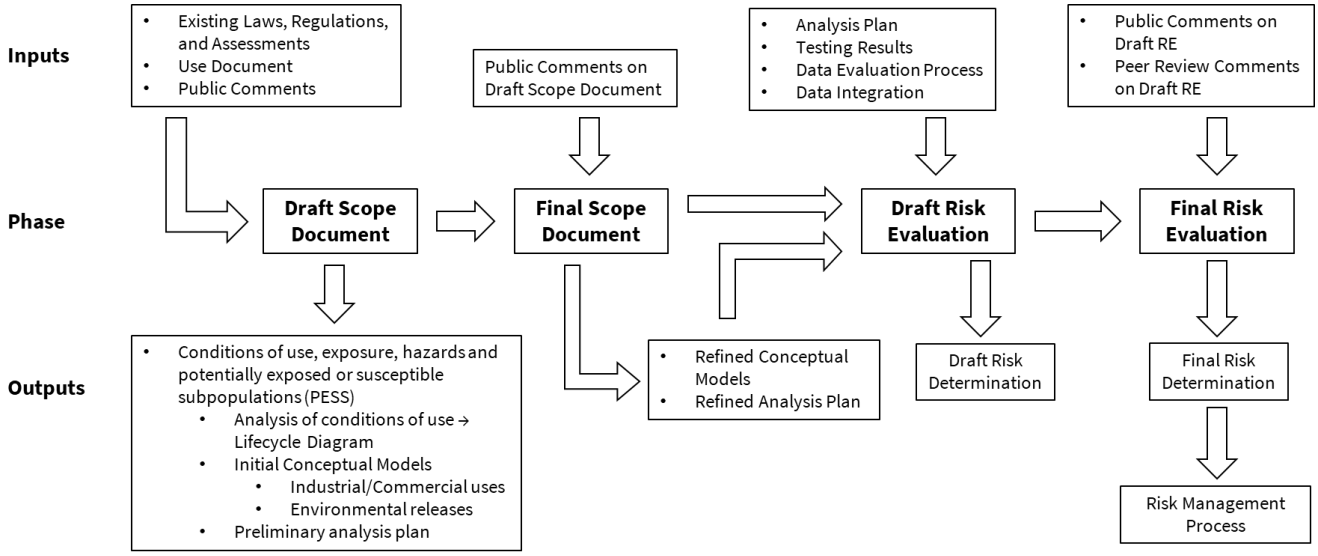


Figure 1-1. TSCA Existing Chemical Risk Evaluation Process

1.1 Scope of the Risk Evaluation

EPA evaluated risk to human and environmental populations for 1,1-dichloroethane. Specifically, for human populations, EPA evaluated risk to (1) workers via inhalation routes; (2) workers via dermal routes; and (3) the general population, including potentially exposed and susceptible subpopulations (PESS; *e.g.*, pregnant women, bottle-fed infants), via oral, dermal, and inhalation routes. For environmental populations, EPA evaluated risk to aquatic species via water and sediment and to terrestrial species via air, water, sediment, and soil pathways leading to dietary and direct ingestion exposure.

1.1.1 Life Cycle and Production Volume

The LCD shown in Figure 1-2 depicts the COUs that are within the scope of the risk evaluation during various life cycle stages, including manufacturing, processing, commercial use, distribution and disposal. The information in the LCD is grouped according to the Chemical Data Reporting (CDR) processing codes and use categories (including functional use codes for industrial uses and product categories for industrial and commercial uses). The CDR Rule under TSCA requires U.S. manufacturers (including importers) to provide EPA with information on the chemicals they manufacture or import into the United States. EPA collects CDR data approximately every 4 years with the latest collections occurring in 2006, 2012, 2016, 2020, and 2024.

The production volume reported in the final scope document was between 100 million and 1 billion pounds (lb), based on total production volume of 1,1-dichloroethane in 2015 from the 2016 CDR

reporting period. The range did not change in the latest 2020 CDR data (the reported total production volume in 2020 was between 100 million and 1 billion lb). Production volume is described here as a range to protect production volumes that were claimed as Confidential Business Information (CBI). For the 2016 CDR cycle, data collected per chemical included the company name, volume of each chemical manufactured/imported, the number of workers at each site, and information on whether the chemical is used in the commercial, industrial, and/or consumer sector(s).

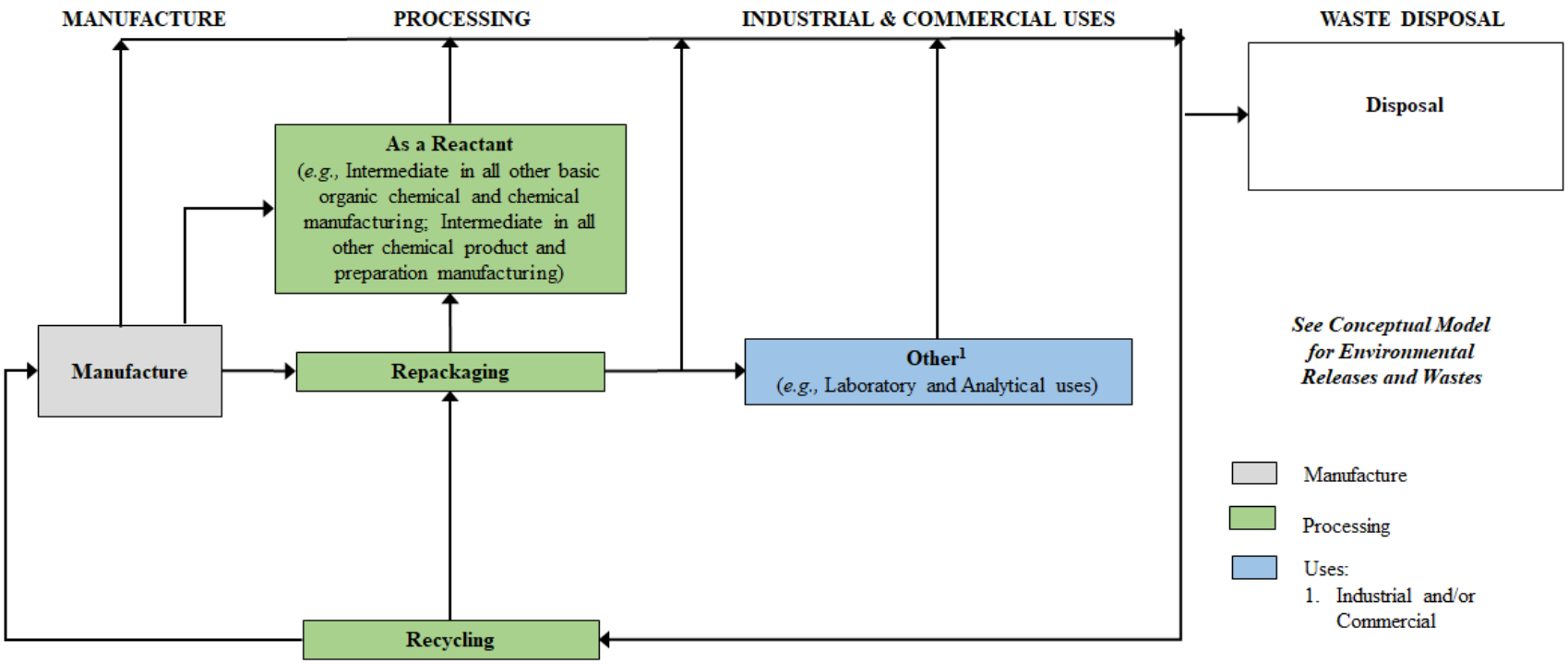


Figure 1-2. 1,1-Dichloroethane Life Cycle Diagram

[a] See (U.S. EPA, 2020b) for additional details on 1,1-dichloroethane uses.

The production volumes shown are for reporting year 2015 from the 2016 CDR reporting period (U.S. EPA, 2016a).

The activities of loading 1,1-dichloroethane product into transport containers and unloading at receiving sites as well as repackaging into smaller containers are considered part of Distribution in Commerce and these are assessed under those occupational exposure scenarios (OESs). Cleanup of accidents/spills that may occur during transport are not within the scope of this risk evaluation.

Descriptions of the categories identified from the 2016 and 2020 CDR are included in the LCD and further described in Appendix P (Figure 1-2)(U.S. EPA, 2016a). The descriptions provide a brief overview of the use category. The *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b) contains more detailed descriptions (*e.g.*, process descriptions, worker activities, process flow diagrams, equipment illustrations) for each manufacture, processing, use, and disposal category.

1.1.2 Conditions of Use Included in the Risk Evaluation

The final scope document (U.S. EPA, 2020b) identified and described the life cycle stages, categories, and subcategories that comprise COUs that EPA planned to consider in the risk evaluation. The COUs included in this final risk evaluation are reflected in the LCD (Figure 1-2) and conceptual models (Section 1.1.2.1). These COUs are evaluated for acute, intermediate, chronic, and lifetime exposures, as applicable, based on reasonably available exposure and hazard data as well as the relevant study populations for each. Table 1-1 below presents all COUs for 1,1-dichloroethane. No consumer uses were identified and therefore, none were evaluated in the 1,1-dichloroethane risk evaluation. In this assessment, EPA added the COU Processing – repackaging to account for the repackaging for distribution of 1,1-dichloroethane for use as a laboratory chemical.

Table 1-1. Categories and Subcategories of Use and Corresponding Exposure Scenario in the Risk Evaluation for 1,1-Dichloroethane

Life Cycle Stage[a]	Category[b]	Subcategory[c]	Reference(s)
Manufacture	Domestic manufacturing	Domestic manufacturing	U.S. EPA (2016a)
Processing	As a reactant	Intermediate in all other basic organic chemical manufacture	(U.S. EPA, 2017a); U.S. EPA (2016a); KemI (2008)
		Intermediate in all other chemical product and preparation manufacturing	U.S. EPA (2016a)
	Repackaging	Repackaging	(Sigma-Aldrich, 2020)
	Recycling	Recycling	U.S. EPA (2016a)
Distribution	Distribution in commerce	Distribution in commerce	Use Document, EPA-HQ-OPPT-2016-0735-0003; U.S. EPA (2016a, 2014b)
Commercial	Other use	Laboratory chemicals	(Sigma-Aldrich, 2020)
Disposal	Disposal	Disposal	KemI (2008)

Life Cycle Stage[a]	Category[b]	Subcategory[c]	Reference(s)

[a] Life Cycle Stage Use Definitions (40 CFR 711.3)
– "Industrial use" means use at a site at which one or more chemicals or mixtures are manufactured (including imported) or processed.
– "Commercial use" means the use of a chemical or a mixture containing a chemical (including as part of an article) in a commercial enterprise providing saleable goods or services.
Although EPA has identified both industrial and commercial uses herein for purposes of distinguishing scenarios in this document, the Agency interprets the authority over "any manner or method of commercial use" under TSCA section 6(a)(5) to reach both.
[b] These categories of COUs appear in the LCD, reflect CDR codes, and broadly represent COUs of 1,1-dichloroethane in industrial and/or commercial settings.
[c] These subcategories reflect more specific COUs of 1,1-dichloroethane.
– The manufacture of 1,1-dichloroethane as an unintentional byproduct during the manufacture of 1,2-dichloroethane (CASRN 107-06-2) (EPA-HQ-OPPT-2018-0426-0027) is not included in this risk evaluation but will be addressed it in the risk evaluation for 1,2-dichloroethane.
– In this risk evaluation, EPA added the COU Processing – repackaging to account for the repackaging for distribution of 1,1-dichloroethane.
– The presence of 1,1-dichloroethane in produced water from hydraulic fracturing is included in the Disposal COU.

1.1.2.1 Conceptual Models

The conceptual model in Figure 1-3 presents the exposure pathways, exposure routes and hazards to human populations from industrial and commercial activities and uses of 1,1-dichloroethane. Figure 1-4 presents general population exposure pathways and hazards for environmental releases and wastes, and Figure 1-5 presents the conceptual model for ecological exposures and hazards from environmental releases and wastes. For general population, only acute, chronic and lifetime exposure scenarios were assessed as exposures resulted from the facility releases that were averaged over annual operating days.

The exposure pathways depicted in Figure 1-4 are based on data EPA compiled regarding the presence of 1,1-dichloroethane in environmental media as well as physical chemical properties that predict the fate and transport and partitioning of 1,1-dichloroethane in the environment. As presented in detail in Section 3.3, monitoring data from EPA databases[3] as well as peer-reviewed literature confirm 1,1-dichloroethane presence in most environmental media. For example, facilities releasing 1,1-dichloroethane into ambient air, surface water and landfills have reported these releases to EPA via the Toxics Release Inventory (TRI) and facility monitoring data of effluent containing 1,1-dichloroethane released to surface receiving waters is reported via Discharge Monitoring Reports (DMRs). Publicly owned water treatment work (POTW) systems report receiving influent containing 1,1-dichloroethane and therefore may have wet biosolids that still contain 1,1-dichloroethane despite potential removal mechanisms such as biodegradation and air stripping.

1,1-Dichloroethane concentrations are reported in a number of air monitoring programs such as EPA's Ambient Monitoring Technology Information Center (AMTIC). Ambient air concentrations of 1,1-dichloroethane are mostly associated with industrial facility releases of 1,1-dichloroethane (see Figure 1-4 and Figure 1-5).

[3] EPA compiled monitoring data from AMTIC, Water Quality Portal (WQP), and DMRs required per National Pollutant Discharge Elimination System [NPDES] permitting requirements) during the 2015 to 2020 timeframe.

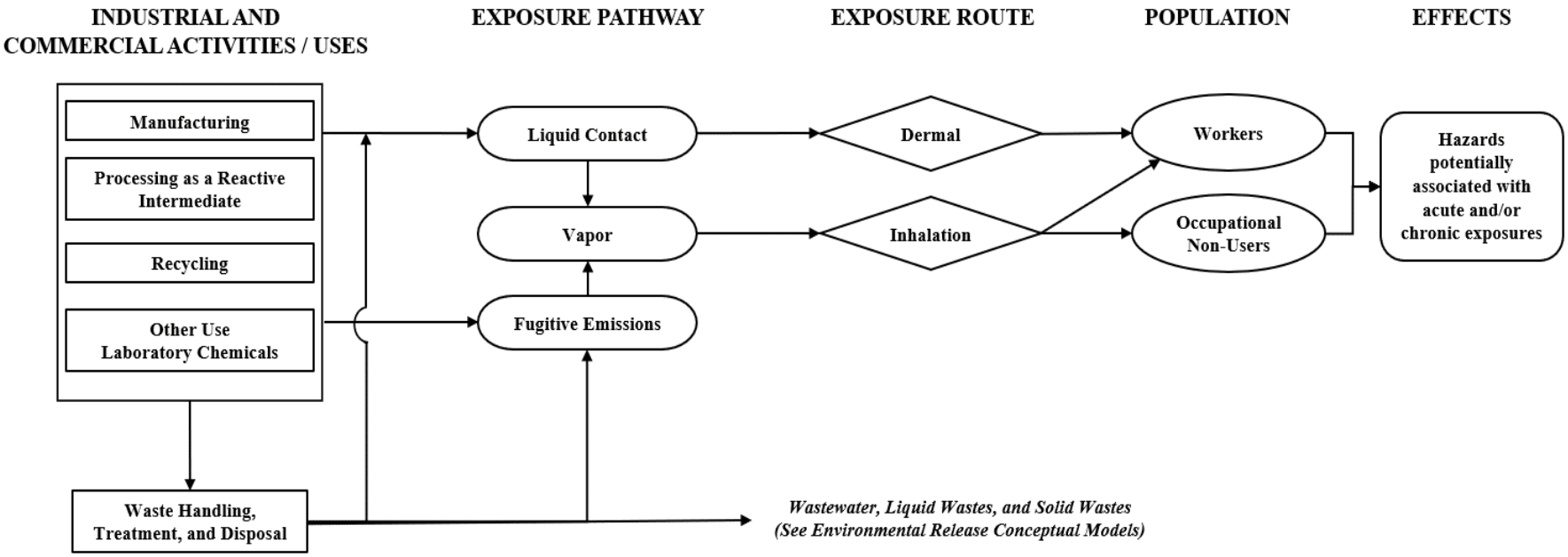


Figure 1-3. 1,1-Dichloroethane Conceptual Model for Industrial and Commercial Activities and Uses: Potential Exposure and Hazards

[a] See Table 1-1 for categories and subcategories of COUs.

[b] Fugitive air emissions are those that are not stack emissions and include fugitive equipment leaks from valves, pump seals, flanges, compressors, sampling connections, and open-ended lines; evaporative losses from surface impoundment and spills; and releases from building ventilation systems.

[c] Exposure may occur through mists that deposit in the upper respiratory tract; however, based on physical and chemical properties, mists of 1,1-dichloroethane will likely be rapidly absorbed in the respiratory tract or evaporate and were evaluated as an inhalation exposure.

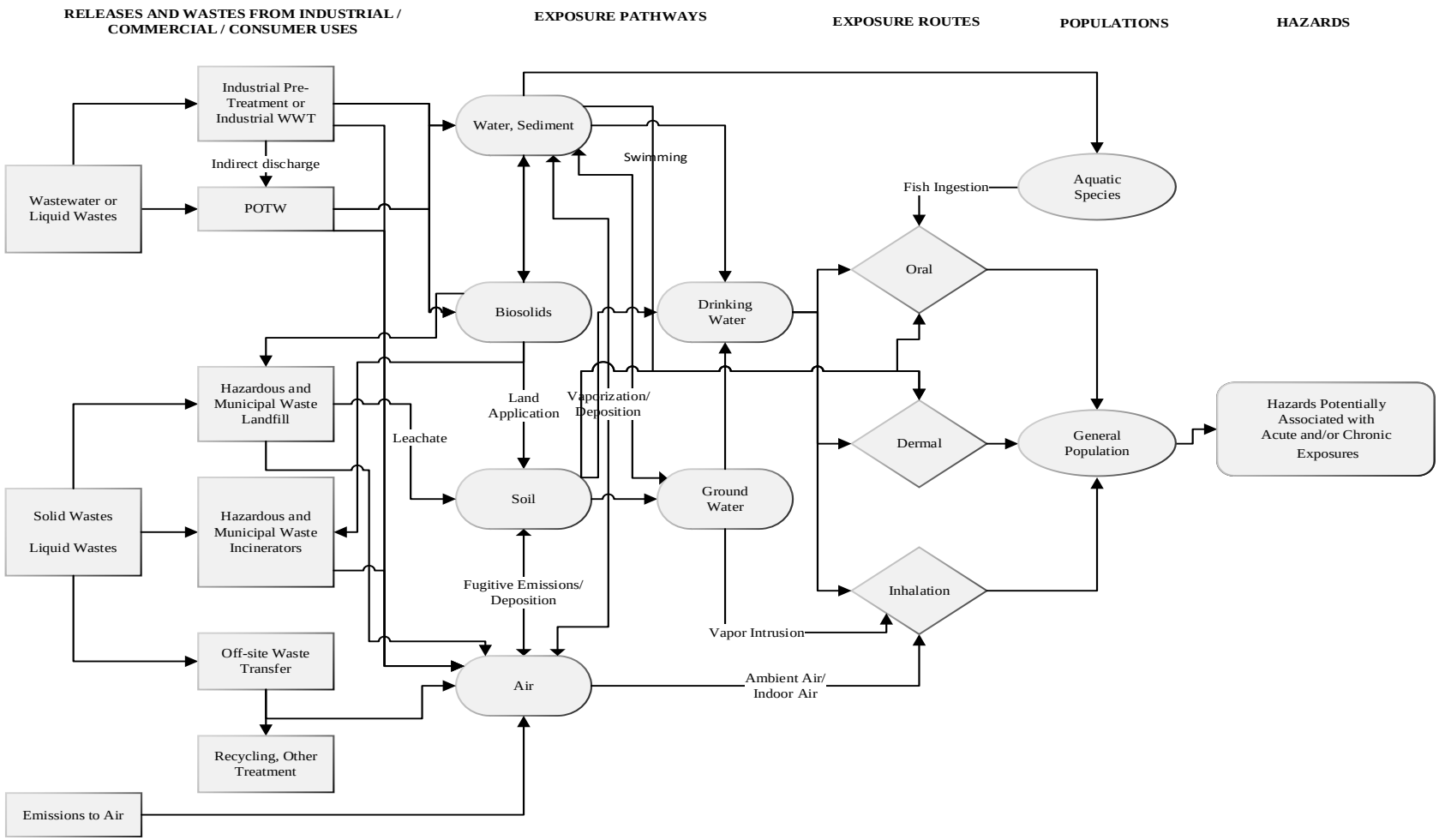


Figure 1-4. 1,1-Dichloroethane Conceptual Model for Environmental Releases and Wastes: General Population Exposures and Hazards

The conceptual model presents the exposure pathways, exposure routes, and hazards to human populations from environmental releases and wastes from industrial and commercial uses of 1,1-dichloroethane.

[a] Industrial wastewater or liquid wastes may be treated on-site and then released to surface water (direct discharge) or pre-treated and released to a publicly owned treatment work (POTW) (indirect discharge).

[b] General population includes people exposed to TSCA releases of 1,1-dichloroethane, including PESS such as infants exposed to drinking water from public drinking water treatment systems during formula bottle feeding; subsistence and tribal fishers; pregnant women, women of reproductive age; individuals with compromised immune systems or neurological disorders; people with the aldehyde dehydrogenase-2 polymorphism; lifestyle factors such as smoking cigarettes or secondhand smoke; and fenceline communities who live near facilities that emit 1,1-dichloroethane.

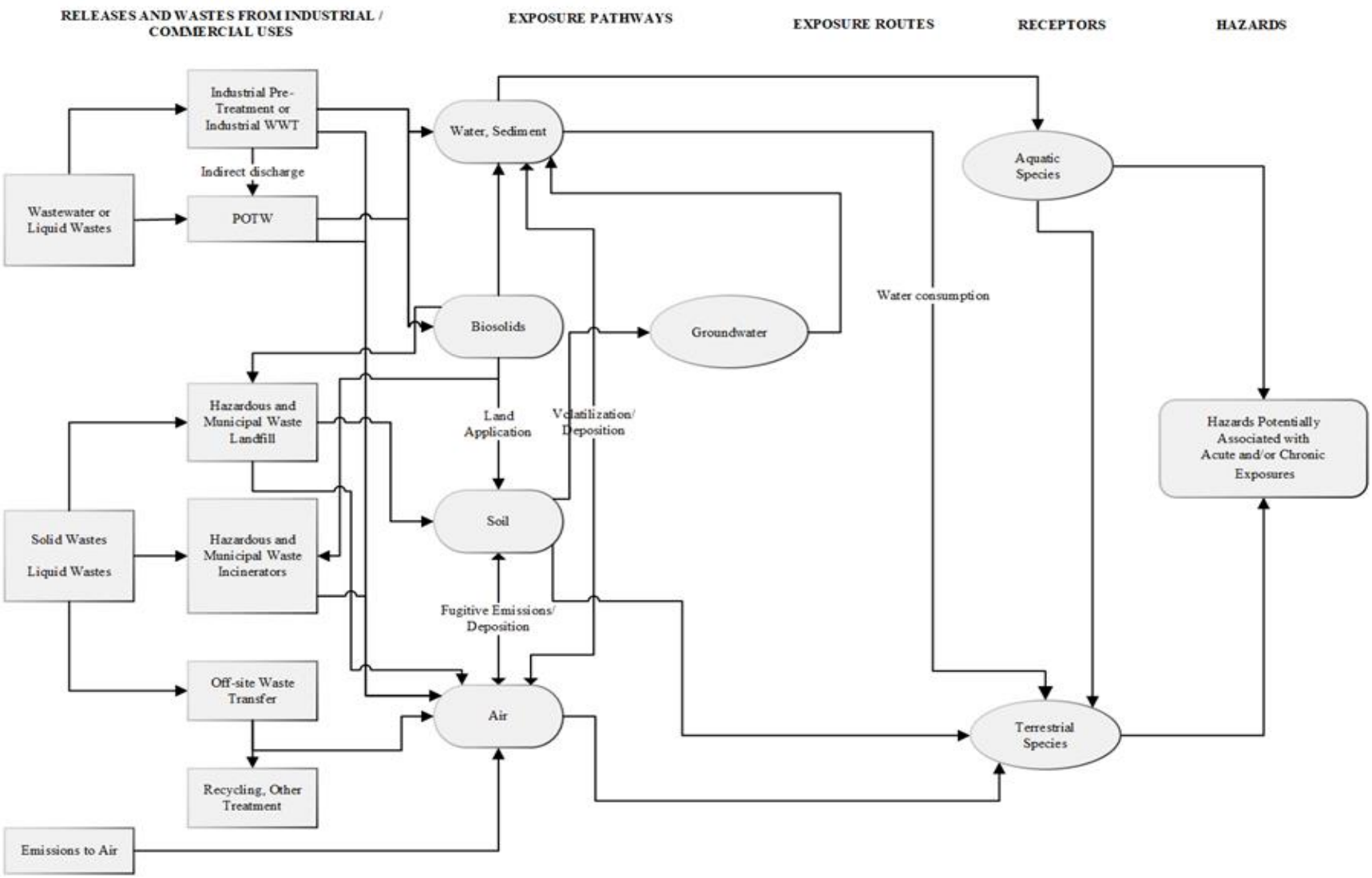


Figure 1-5. 1,1-Dichloroethane Conceptual Model for Environmental Releases and Wastes: Ecological Exposures and Hazards

[a] Industrial wastewater or liquid wastes may be treated on-site and released to surface water (direct discharge) or pre-treated and released to a POTW (indirect discharge).

1.1.3 Populations Assessed

Based on the conceptual models presented in Section 1.1.3.1 below, Figure 1-6 presents the human populations and ecological receptors assessed in this risk evaluation. EPA assessed risk to human populations and environmental receptors for 1,1-dichloroethane. Specifically, for human populations, EPA evaluated risk to (1) workers via inhalation and dermal exposure routes; and (2) the general population via oral, dermal, and inhalation routes depending on the exposure media/pathway and exposure scenario. For environmental receptors, the Agency evaluated risk to aquatic species via water and sediment as well as terrestrial species via air, water, sediment, and soil leading to dietary and direct ingestion exposure. Some analyses of exposure, hazard, and risk for certain populations and environmental receptors are described in greater detail in the appendices rather than the risk evaluation text.

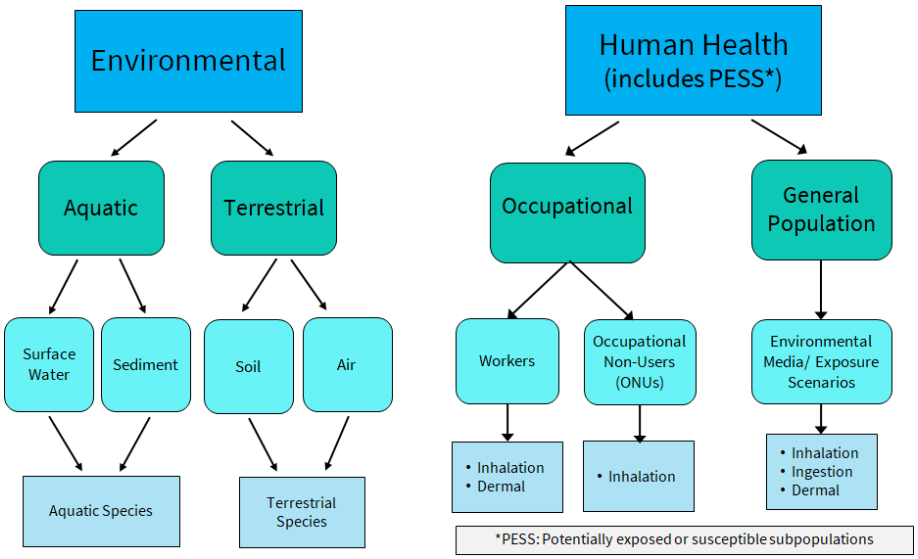


Figure 1-6. Populations Assessed in this Risk Evaluation for 1,1-Dichloroethane

1.1.3.1 Potentially Exposed or Susceptible Subpopulations

TSCA section 6(b)(4)(A) requires that risk evaluations "determine whether a chemical substance presents an unreasonable risk of injury to health or the environment, without consideration of costs or other non-risk factors, including an unreasonable risk to a potentially exposed or susceptible subpopulation identified as relevant to the risk evaluation by the Administrator, under the conditions of use." TSCA section 3(12) states that "the term '*potentially exposed or susceptible subpopulation*' [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."

Evaluation of the qualitative and quantitative evidence for PESS begins as part of the systematic review process. Any available relevant published studies and other data are identified from a broad literature search strategy across several databases, focused only on the chemical name (including synonyms and trade names) with no additional search limits. This broad search process is described in the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances: A Generic TSCA Systematic Review Protocol with Chemical-Specific Methodologies* (also referred to as "2021 Draft Systematic Review Protocol"; see Section 1.2) (U.S. EPA, 2021a). When adequate and complete,

evidence related to PESS informs the derivation of exposure estimates and human health hazard endpoints/values that are protective of those potentially exposed or susceptible subpopulations.

PESS factors can influence the selection of relevant exposure pathways, the sensitivity of derived hazard values, the identification of human subpopulations, and the discussion of uncertainties throughout the assessment. For the 1,1-dichloroethane risk evaluation, EPA integrated and assessed available information on hazards and exposures for the conditions of use of 1,1-dichloroethane, including information relevant to specific risks of injury to PESS. In addition to workers, PESS subpopulations identified as relevant include infants exposed to drinking water during formula bottle feeding, subsistence and Tribal fishers, pregnant women, men and women of reproductive age, people with the aldehyde dehydrogenase-2 polymorphism, lifestyle factors such as smoking cigarettes or secondhand smoke, and communities who live near facilities that emit 1,1-dichloroethane (see Risk Characterization for Potentially Exposed or Susceptible Subpopulations, Section 5.3.2).

1.2 Systematic Review

EPA/OPPT applies systematic review principles in the development of risk evaluations under the amended TSCA. Section 26(h) of TSCA 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.

To meet the TSCA section 26(h) science standards, EPA used the TSCA systematic review process described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and the *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025a) (also called the "1,1-Dichloroethane Systematic Review Protocol"). Systematic review supports the risk evaluation in that data searching, screening, evaluation, extraction, and evidence integration are used to develop the exposure and hazard assessments based on reasonably available information. EPA defines "reasonably available information" to mean information that the Agency possesses or can reasonably obtain and synthesize for use in risk evaluations, considering the deadlines for completing the evaluation (40 CFR 702.33).

The systematic review process is briefly described in Figure 1-7. More detail regarding these steps is provided in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and the 1,1-Dichloroethane Systematic Review Protocol (U.S. EPA, 2025a). The latter provides additional information on the steps in the systematic review process, including literature inventory trees and evidence maps for each discipline (*e.g.*, human health hazard) containing results of the literature search and screening as well as sections summarizing data evaluation, extraction, and evidence integration.

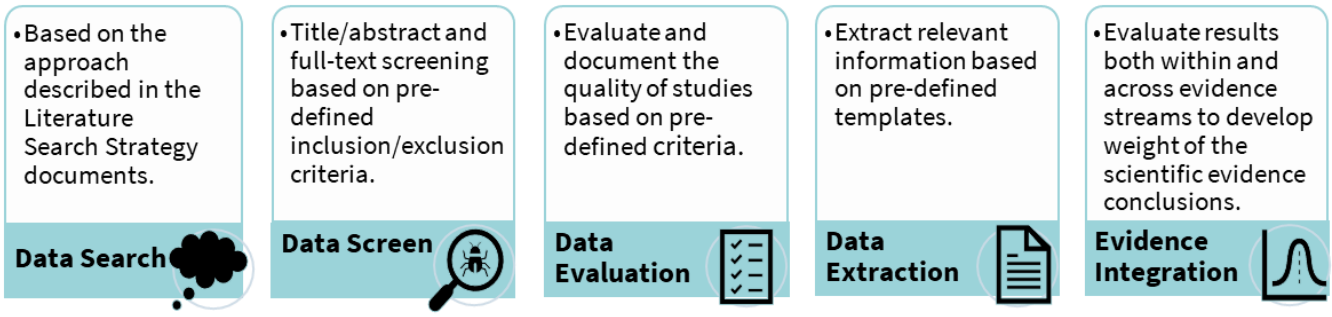


Figure 1-7. Diagram of the Systematic Review Process

EPA reviewed reasonably available information, defined in 40 CFR 702.33, in a fit-for-purpose approach, to develop a risk evaluation that relies on the best available science and is based on the weight of scientific evidence in accordance with TSCA sections 6 and 26. The Agency reviewed reasonably available information and evaluated the quality of the methods and reporting of results of the individual studies using the evaluation strategies described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and the 1,1-Dichloroethane Systematic Review Protocol (U.S. EPA, 2025z).

EPA also identified key assessments conducted by other Agency programs and other U.S. and international organizations. Depending on the source, these assessments may include information on COUs (or the equivalent), hazards, exposures, and PESS. Some of the most pertinent assessments that were consulted for this 1,1-dichloroethane risk evaluation include the following:

- California Environmental Protection Agency, Office of Environmental Health Hazard Assessment (OEHHA) 2003 *Public Health Goals for Chemicals in Drinking Water: 1,1-Dichloroethane in Drinking Water* (accessed June 16, 2025);
- U.S. Department of Human Health Services, Public Health Service, Agency for Toxic Substances and Disease Registry (ATSDR) 2015 *Toxicological Profile for 1,1-Dichloroethane* (accessed June 16, 2025) (also called 2015 ATSDR Tox Profile);
- U.S. EPA 2006 *Provisional Peer Reviewed Toxicity Values for 1,1-Dichloroethane; CASRN 75-34-3* (accessed June 16, 2025); and
- U.S. EPA Integrated Risk Information System (IRIS) Chemical Assessment 1990 *1,1-Dichloroethane; CASRN 75-34-3* (accessed June 16, 2025).

Additionally, as 1,2-dichloroethane was identified as an analog for read-across for human health hazard, documents that were also consulted include the following:

- California Environmental Protection Agency, OEHHA 1999 *Public Health Goals for 1,2-Dichloroethane in Drinking Water* and 2005 update memorandum (accessed June 16, 2025);
- U.S. Department of Human Health Services, Public Health Service, ATSDR 2024 *Toxicological Profile for 1,2-Dichloroethane* (accessed June 16, 2025) (also called 2024 ATSDR Tox Profile);
- U.S. EPA Integrated Risk Information System (IRIS) Chemical Assessment 1987 *1,2-Dichloroethane; CASRN 107-06-2* (accessed June 16, 2025); and
- U.S. EPA 2010 *Provisional Peer Reviewed Toxicity Values for 1,2-Dichloroethane; CASRN 107-06-2* (accessed June 16, 2025).

1.3 Organization of the Risk Evaluation

This final risk evaluation for 1,1-dichloroethane includes 5 additional major sections and a total of 17 appendices:

- Section 2 summarizes basic physical-chemical characteristics as well as the fate and transport of 1,1-dichloroethane.
- Section 3 includes an overview of releases and concentrations of 1,1-dichloroethane in the environment.
- Section 4 provides a discussion and analysis of the environmental risk assessment, including the environmental exposure, hazard, and risk characterization based on the COUs for 1,1-dichloroethane.
- Section 5 presents the human health risk assessment, including the exposure, hazard, and risk characterization based on the COUs. Section 5 also includes a discussion of PESS based on both greater exposure and susceptibility, as well as a description of aggregate and sentinel exposures.

- Section 6 presents EPA's determination of whether 1,1-dicloroethane presents an unreasonable risk to human health or the environment under the assessed COUs.

Appendix A provides a brief summary of the federal, state, and international regulatory history of 1,1-dichloroethane. Appendix B lists all separate supplemental documents associated with this risk evaluation, which can be accessed through hyperlinks included in the references.

All subsequent appendices (C through Q) and supplemental documents listed in Appendix B include more detailed analysis and explanations than are provided in the main body of this final risk evaluation for 1,1-dichloroethane.

2 CHEMISTRY AND FATE AND TRANSPORT OF 1,1-DICHLOROETHANE

Physical and chemical properties determine the behavior and characteristics of a chemical that inform its conditions of use, environmental fate and transport, potential toxicity, exposure pathways, routes, and hazards. Environmental fate includes environmental partitioning, accumulation, degradation, and transformation processes. Transformation or degradation occur through reaction of the chemical in the environment. Environmental transport is the movement of the chemical within and between environmental media. Thus, understanding the environmental fate of 1,1-dichloroethane informs the determination of the specific exposure pathways and potential human and environmental receptors that EPA considered in this risk evaluation.

2.1 Physical and Chemical Properties

EPA gathered and evaluated physical and chemical property data and information according to the process described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). During the evaluation of 1,1-dichloroethane, the Agency considered both measured and estimated physical and chemical property data and information for 1,1-dichloroethane summarized in Table 2-1, as applicable. Information on the fully extracted dataset is available in the supplemental file *Systematic Review of Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties* (U.S. EPA, 2025v).

1,1-Dichloroethane is a colorless oily liquid with a chloroform- or ether-like odor (Government of Canada, 2021; NLM, 2018; NIOSH, 2007). It is soluble in water and is miscible in most organic solvents (NCBI, 2020a; NLM, 2018). With a vapor pressure of 228 mmHg at 25 °C and a boiling point of 57.3 °C, 1,1-dichloroethane is a highly volatile organic compound (VOC) (Elsevier, 2019; Dreher et al., 2014; O'Neil, 2013; RIVM, 2007). The physical and chemical properties of 1,1-dichloroethane are listed in Table 2-1 and a detailed discussion is provided in Appendix C.

Table 2-1. Physical and Chemical Properties of 1,1-Dichloroethane

Property	Selected Value(s)	Reference(s)	Overall Quality Determination
Molecular formula	$C_2H_4Cl_2$	N/A	N/A
Molecular weight	98.95 g/mol	N/A	N/A
Physical form	Colorless oily liquid with a chloroform- or ether-like odor	(Government of Canada, 2021; NLM, 2018; NIOSH, 2007)	High
Melting point	−96.93 °C	(NLM, 2018)	High
Boiling point	57.3 °C	(O'Neil, 2013)	High
Density	1.1757 at 20 °C	(O'Neil, 2013)	High
Vapor pressure	228 mmHg at 25 °C	(Rumble, 2018b)	High
Vapor density	3.44 (air = 1 g/cm^3)	(NCBI, 2020b)	High
Water solubility	5,040 mg/L at 25 °C	(NLM, 2018)	High
Octanol/water partition coefficient (log K_{OW})	1.79 at 25 °C	(Elsevier, 2019)	High
Henry's Law constant	0.00562 atm m^3/mol at 24 °C	(NLM, 2018)	High

Property	Selected Value(s)	Reference(s)	Overall Quality Determination
Flash point	−12 °C	(Dreher et al., 2014)	High
Autoflammability	458 °C	(Rumble, 2018b)	High
Viscosity	0.464 cP at 25 °C	(Rumble, 2018c)	High
Refractive index	1.4164	(Rumble, 2018a)	High
Dielectric constant	10.9 at 20 °C	(NLM, 2018)	High
Heat of evaporation	30.8 kJ/mL at 25 °C	(Dreher et al., 2014)	High

2.2 Environmental Fate and Transport

1,1-Dichloroethane – Environmental Fate and Transport (Section 2.2)
Key Points:

EPA evaluated the reasonably available environmental fate and transport information for 1,1-dichloroethane. The following are key points from EPA's evaluation:

- Environmental Distribution:
  - 1,1-Dichloroethane is a volatile liquid that is soluble in water. Under the COUs, environmental releases are expected to partition primarily to air with lesser amounts to water, sediment, and land.
- Fate and Transport in Air:
  - 1,1-Dichloroethane released to air is expected to primarily remain in air (Henry's Law constant of 0.00562 atm-m^3/mol).
  - In air, 1,1-dichloroethane will react with ·OH radicals with a reported half-life of 39 days and can be subject to transport and wet and dry deposition.
  - Given the relatively large quantities of 1,1-dichloroethane released to air under the COUs and the relatively long half-life, air is expected to be an important medium for exposure.
- Fate and Transport in Soil:
  - 1,1-Dichloroethane released to soil can be subject to volatilization to air, biodegradation, runoff to surface waters, and infiltration to groundwater.
  - Due to its low affinity for soil organic matter (log organic carbon: water partition coefficient 1.48), migration through soil to groundwater will be largely unhindered.
  - Biodegradation in soil will generally occur slowly with half-lives ranging from months to years.
- Fate and Transport in Surface Water and Sediment:
  - In surface water, 1,1-dichloroethane will be subject to volatilization and slow biodegradation as well as advection, dispersion, and dilution.
  - Due to its relatively high-water solubility (5,040 mg/L), continuous releases of 1,1-dichloroethane to deeper, slower moving surface water will result in a portion of the release remaining in water.
  - In sediment, 1,1-dichloroethane will generally biodegrade with half-lives ranging from months to years. Due to its solubility in water 1,1-dichloroethane will partition in sediments to sediment pore-water.
  - Given the relatively low quantity directly released to water under the COUs—coupled with the effects of volatilization, dilution, advection, and dispersion—surface water will generally not be an important medium for exposure. However, exceptions could include sustained direct releases of 1,1-dichloroethane into deep, slower moving, or stagnant surface waters.
- Fate and Transport in Groundwater:
  - Releases of 1,1-dichloroethane to land under the COUs use could migrate over a period of time to groundwater. Modeled groundwater concentrations suggest groundwater will generally not be an important medium for exposure.
  - 1,1-Dichloroethane can be produced as a product in the anaerobic biodegradation of 1,1,1-trichloroethane in groundwater, potentially contributing to 1,1-dichloroethane concentrations.
  - Biodegradation of 1,1-dichloroethane that does reach groundwater generally occurs slowly with half-lives ranging from months to years.
- Persistence and Bioaccumulation:
  - 1,1-Dichloroethane meets criteria for persistence but not criteria to be classified as bioaccumulative based on estimated bioconcentration factor (BCF)/bioaccumulation factor (BAF) values of less than 1,000.

2.2.1 Fate and Transport Approach and Methodology

Reasonably available environmental fate data—including biodegradation rates, removal during wastewater treatment, volatilization from lakes and rivers, and organic carbon: water partition coefficient (K_{OC})—are among selected parameters for use in the current risk evaluation. In assessing the environmental fate and transport of 1,1-dichloroethane, EPA considered the full range of results from sources that were rated via systematic review as medium or high confidence. Data evaluation information and information on the full extracted dataset are available in the supplemental file *Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport* (U.S. EPA, 2025u). Other fate estimates were based on modeling results from EPI Suite™ (U.S. EPA, 2012c), a predictive tool for physical/chemical and environmental fate properties. Information regarding the model inputs is available in Appendix C.2.1.1. EPI Suite™ was reviewed by the EPA Science Advisory Board (SAB, 2007), and the individual models that comprise EPI Suite™ have been peer reviewed through publication in technical journals. Citations for the supporting manuscripts are available in the EPI Suite™ help files.

In addition, methods for estimation of BCF/BAF developed by EPA's Office of Water for the establishment of Ambient Water Criteria for the Protection of Human Health (U.S. EPA, 2003c) are also presented for comparison to EPI Suite™ estimations. Details are presented in Appendix C.2.6.

Table 2-2 provides selected environmental fate data that EPA considered while assessing the fate of 1,1-dichloroethane. The data were updated after publication of the final scope document with additional information identified through the systematic review process and supplemental literature searches.

Table 2-2. Environmental Fate Characteristics of 1,1-Dichloroethane

Property or Endpoint	Value[a]	Reference	Overall Quality Determination
Indirect photodegradation	t ½ = 39 days (based on 12-hour day; 1.5E06·OH/cm^3 from ·OH rate constant of 2.74E−13 cm^3/ molecule·second at 25 °C)	(U.S. EPA, 2012c)	High
Direct photodegradation	Not expected to be susceptible to direct photolysis by sunlight as 1,1-dichloroethane does not contain chromophores that absorb at wavelengths >290 nm	(NCBI, 2020b)	Medium
Hydrolysis half-life	t ½ = 61.3 years at 25 ºC and pH 7	(Jeffers et al., 1989)	High
Aerobic biodegradation water	up to 91% in 7 days after extensive acclimation	(Tabak et al., 1981)	High
Anaerobic biodegradation anaerobic sludge	31% in 25 days	(Van Eekert et al., 1999)	High
Anaerobic biodegradation	t ½ = 1.5–6.9 years	(Huff et al., 2000)	High
	t ½ = 115 days	(Washington and Cameron, 2001)	Medium
Bioconcentration factor (BCF)	7 (estimated)	(U.S. EPA, 2012c)	High
Bioaccumulation factor (BAF)	6.8 (estimated)	(U.S. EPA, 2012c)	High

Property or Endpoint	Value[a]	Reference	Overall Quality Determination
Organic carbon:water partition coefficient (log K_{OC})	1.48	(Poole and Poole, 1999)	High
Removal in wastewater treatment	33–100%	(U.S. EPA, 1982)	High

[a] Measured unless otherwise noted
[b] Information was estimated using EPI Suite™ (U.S. EPA, 2012c)

2.2.2 Summary of Fate and Transport Assessment

1,1-Dichloroethane is a volatile liquid that evaporates at ambient temperature (Rumble, 2018b). Estimated half-lives for volatilization from water range from hours to days depending on environmental conditions. Under the COUs, based on its physical and chemical properties, environmental releases of 1,1-dichloroethane are expected to partition primarily to air (85%) with lesser amounts to water (15%), sediment (<1%), and soil (<1%) using the 2020 TRI releases. Figure 2-1 graphically depicts the relative major and minor partitioning and transport pathways predicted for 1,1-dichloroethane between and within environmental media. Environmental releases of 1,1-dichloroethane reported to the TRI and the National Emissions Inventory (NEI) between 2015 and 2020, indicate most releases are to air. Based on the reported release data, environmental partitioning modeling predicts that approximately 85 percent mass distribution will remain in air, 15 percent in water, and less than 1 percent in soil and sediment. See Appendix C.2.1.2 for further discussion.

In air, 1,1-dichloroethane will react with hydroxyl (·OH) radicals with a half-life of 39 days (U.S. EPA, 2012c) and can be subject to transport and wet and dry deposition. Because the highest releases of 1,1-dichloroethane are to air, and those releases are expected to remain in air, it is expected to be an important transport medium, and inhalation is expected to be an important exposure pathway. The presence of 1,1-dichloroethane in ambient air is confirmed by 2015 to 2020 monitoring data from the AMTIC ambient air monitoring archive, which show national annual average concentrations ranging from 8.0×10^{-2} to 0.13 µg/m^3 (Section 3.3.1). The fate of 1,1-dichloroethane in air is further discussed in Appendix C.2.2 and inhalation exposure in Section 5.1.2.2.1.

In surface water, 1,1-dichloroethane will be subject to volatilization to air (due to its high Henry's Law constant) and biodegradation in anaerobic water. Partitioning from water and adherence onto sediment particles is estimated by the organic carbon:water partition coefficient (log K_{OC} = 1.48 (Poole and Poole, 1999). Due to its relatively high water solubility (5,040 mg/L) (NLM, 2018), continuous releases of 1,1-dichloroethane to water will result in a portion of the release remaining in water and the interstitial sediment pore water spaces. Environmental releases to water and wastewater treatment plants are relatively low and distributed across multiple sites (see Section 3.2). Water Quality Portal (WQP) (NWQMC, 2022) concentrations of 1,1-dichloroethane measured in ambient surface waters from 2015 to 2020 ranged from 0 to 2 µg/L, with a median concentration of 0.25 µg/L and a 95th percentile concentration of 0.5 µg/L. The fate of 1,1-dichloroethane in water is further discussed in Appendix C.2.3.1, environmental aquatic exposure in Section 3.3.3, and human exposure in Section 5.1.2.4.

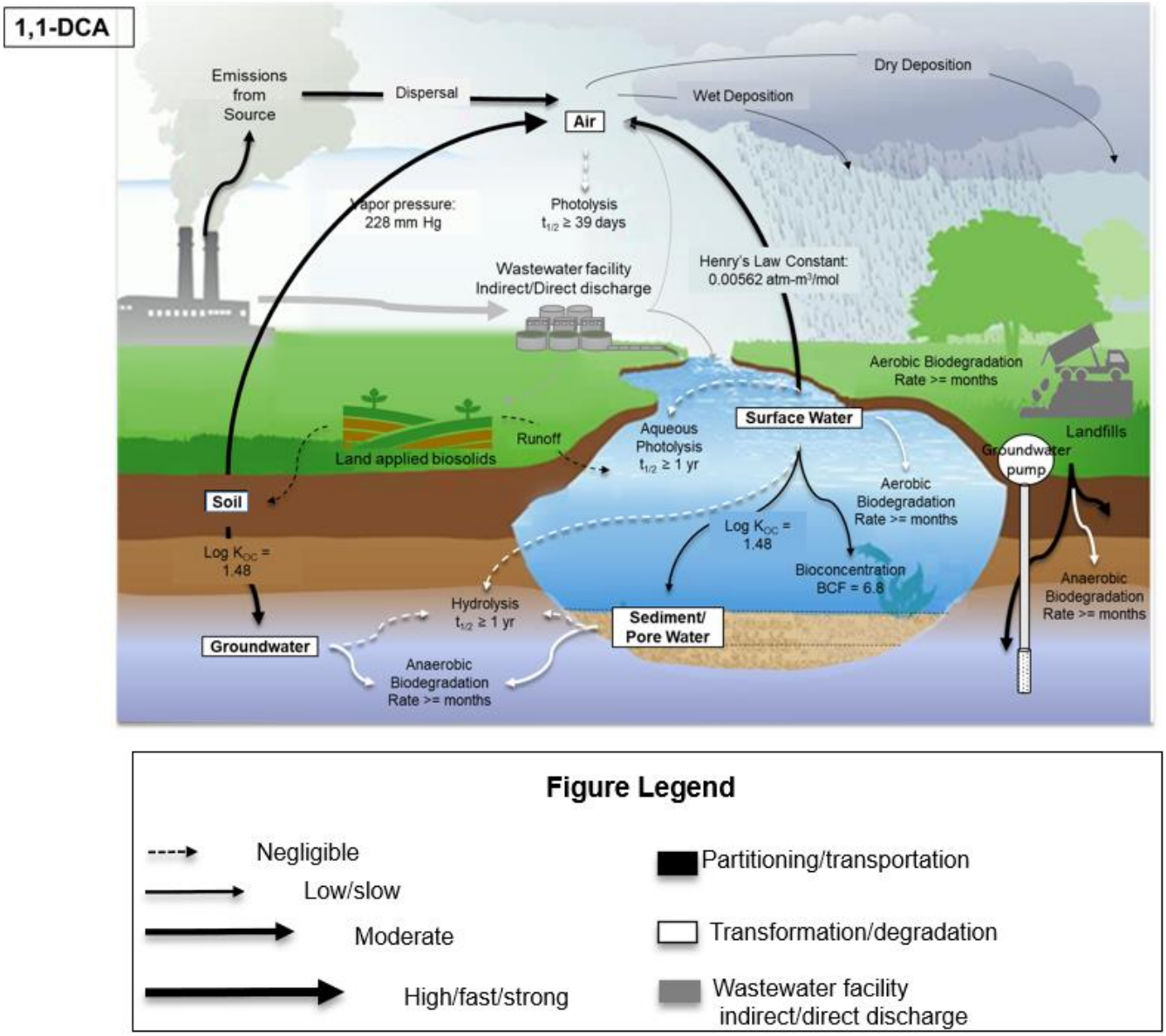


Figure 2-1. Transport, Partitioning, and Degradation of 1,1-Dichloroethane in the Environment[a]

[a] Depicts the distribution (grey arrows), transport, and partitioning (black arrows) as well as the transformation and degradation (white arrows) of 1,1-dichloroethane in the environment. The width of the arrow is a qualitative indication of the likelihood that the indicated partitioning will occur or the rate at which the indicated degradation will occur (*i.e.*, wider arrows indicate more likely partitioning or more rapid degradation of 1,1-dichloroethane).

1,1-Dichloroethane will not partition strongly to sediment particles based on its low measured organic carbon:water partition coefficient (log K_{OC} 1.48) (Poole and Poole, 1999); however, due to its solubility in water 1,1-dichloroethane will partition to sediment pore water. 1,1-Dichloroethane in sediment is expected to biodegrade slowly with half-lives of months to greater than months (Şimşir et al., 2017; Hamonts et al., 2009). No monitoring data were found for exposure of humans and biota to 1,1-dichloroethane via sediment. Relatively low levels of 1,1-dichloroethane in water and low partitioning to sediment suggests low levels of 1,1-dichloroethane would be found in sediment. The fate of 1,1-dichloroethane in sediment is further discussed in Appendix C.2.3.2 and environmental benthic exposure in Section 3.3.3.4.

Releases of 1,1-dichloroethane to land may be subject to volatilization to air, runoff to surface waters, and migration through soil to groundwater due to its low affinity for soil organic matter (log K_{OC} 1.48 (Poole and Poole, 1999)). Biodegradation in soil will generally occur slowly, with half-lives ranging from months to years (U.S. EPA, 2013a). No monitoring data were found for exposure of humans and biota to 1,1-dichloroethane via soil. The releases of 1,1-dichloroethane to land under the conditions of use will be subject to the effects of dilution, advection, and dispersion. Reported releases to TRI for disposal to landfills were no greater than 1 kg/year between 2015 to 2020. TRI data does not specify the details of landfills receiving 1,1-dichloroethane. Where EPA did not have direct release data, EPA modeled generic scenarios such as for laboratory use and assumed that disposal of 1,1-dichloroethane would be less than 22,682 kg/year and disposal would only be to hazardous waste landfills. This assumption is based on regulation of professional laboratories and academic institutions using 1,1-dichloroethane for laboratory use to follow hazardous waste processes and do not dispose of hazardous waste in any other landfills—only in hazardous waste landfills. The fate of 1,1-dichloroethane in soil is further discussed in Appendix C.2.4.1, environmental terrestrial exposure in Appendix J.2, and general population exposure in Section 5.1.2.4.4.

In groundwater, 1,1-dichloroethane will have a low affinity for organic matter based on its measured organic carbon: water partition coefficient of 31 and will not significantly sorb to suspended solids in groundwater. 1,1-Dichloroethane has a reported hydrolysis half-life of approximately 61 years (Jeffers et al., 1989); therefore, losses of 1,1-dichloroethane from groundwater will most likely be due to biodegradation. Biodegradation half-lives are generally on the order of months to years under anaerobic conditions that favor biological reductive dechlorination. Half-lives can also differ markedly within a groundwater plume. Wiedemeier (1999), for example, reports half-lives for cis-1,2-dichloroethylene (cis-1,2-DCE) that are more than an order of magnitude higher in one portion of a plume than in another portion of the same plume. There may be cases where no biodegradation takes place. Wilson (1983) reported no biodegradation in unamended aquifer sediments containing 1,1-dichloroethane after 16 weeks of incubation under aerobic conditions. This indicates that 1,1-dichloroethane entering a pristine oxic aquifer setting may conceivably be recalcitrant to biodegradation. The limited data available in the literature make this difficult to assess. There are no recent studies showing aerobic biodegradation of 1,1-dichloroethane. There are no studies showing aerobic biodegradation of 1,1-dichloroethane in simple mineral culture media. Tabak (1981) reported biodegradation in laboratory experiments, but this was most likely co-metabolic degradation supported by aerobic degradation of the yeast extract or digester solids in their reaction mix.

Wiedemeier (1999) describes three types of biodegradation behavior for chlorinated solvents: Type 1, where anaerobic biodegradation is supported by an anthropogenic electron donor such as landfill leachate or a fuel spill; Type II, where anaerobic biodegradation is supported by natural electron donors such as buried soils or aquifer sediment with high organic matter; and Type III, where the supply of electron donor is inadequate and the chlorinated organic is not biodegraded. This suggests that if a release of 1,1-dichloroethane is not accompanied by landfill leachate or other source of electron donor it may not biodegrade.

Monitoring data confirm the presence of 1,1-dichloroethane in groundwater. 1,1-Dichloroethane concentrations from groundwater monitoring wells retrieved from the Water Quality Portal (NWQMC, 2022) for the years 2015 to 2020 ranged from 0 to 650 µg/L (see Appendix F.1). Groundwater and soil-water leachate concentration data collected through EPA's systematic review of published literature reported ranges from not detected to 1,900 µg/L in 400 samples collected between 1984 and 2005 in the United States (see Appendix G.1.2.1). Monitoring data from EPA's Third Unregulated Contaminant Monitoring Rule (UCMR3) for 1,1-dichloroethane found in finished drinking water from 404 public

water sources across 16 states that draw primarily from groundwater sources indicated a maximum concentration of 1.6 µg/L, indicating that 1,1-dichloroethane in finished drinking water derived from groundwater was measured in relatively low amounts across the nation between 2013 to 2015 (U.S. EPA, 2021b). Modeled groundwater concentrations of 1,1-dichloroethane resulting from migration of its releases to soil suggest groundwater will generally not be an important medium for exposure. However, 1,1-dichloroethane does frequently occur in anaerobic groundwater as a biodegradation product of the compound 1,1,1-trichloroethane. The fate of 1,1-dichloroethane in groundwater is further discussed in Appendix C.2.4.2. 1,1-Dichloroethane groundwater concentrations are further discussed in Appendix F.

Minor amounts of 1,1-dichloroethane in wastewater undergoing biological wastewater treatment may be removed by processes including sorption to wastewater solids. No recent data were found on 1,1-dichloroethane concentrations in biosolids. However, the 1988 National Sewage Sludge Survey sampled 208 representative publicly owned treatment works (POTWs) for a list of substances including 1,1-dichloroethane. 1,1-Dichloroethane had a 0 percent detection frequency. As discussed in Appendix C.2.5.2, less than 1 percent of 1,1-dichloroethane as predicted by modeling is expected to be removed by sorption in biological wastewater treatment based on its K_{OC} value of 31 and EPI Suite™ modeling. Due to assumed low sorption of 1,1-dichloroethane to solids and the low amounts of 1,1-dichloroethane undergoing wastewater treatment (see Section 3.2 for details), land application of biosolids from 1,1-dichloroethane wastewater treatment is not expected to be a significant exposure pathway. However, specific POTW facilities reporting water releases of 1,1-dichloroethane could land apply biosolids containing 1,1-dichloroethane and modeled concentrations of land-applied biosolids are presented in more detail in Appendix G. The fate of 1,1-dichloroethane in biosolids is further discussed in Appendix C.2.5.2, environmental terrestrial exposure to biosolids in Appendix J.2, and general population exposure in Appendix I.

1,1-Dichloroethane does not meet the criteria to be classified as persistent and bioaccumulative (U.S. EPA, 1999). Although 1,1-dichloroethane is expected to have half-lives exceeding 2 months in some environmental compartments, it does not meet bioconcentration/bioaccumulation criteria based on estimated BCF/BAF values of less than 1,000 (U.S. EPA, 2012c). With low bioconcentration/ bioaccumulation potential, fish ingestion and trophic transfer are not expected to be important pathways. The bioconcentration of 1,1-dichloroethane in fish is further discussed in Appendix C.2.6, trophic transfer of 1,1-dichloroethane in Section 4.1.3, and general population exposure through fish ingestion in Section 5.1.2.4.2 (see also Figure 2-1 above).

2.2.3 Weight of Scientific Evidence Conclusions for Fate and Transport

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

The weight of scientific evidence supporting the fate and transport assessment is based on the strengths, limitations, and uncertainties associated with the fate and transport studies evaluated within and outside systematic review. The judgment is summarized using the following confidence descriptors: Robust, Moderate, Slight, or Indeterminate. This approach is consistent with the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a).

The weight of scientific evidence regarding fate and transport as reported in high-moderate quality studies, identified both through systematic review and outside of systematic review, give robust to moderate confidence that 1,1-dichloroethane:

- will not undergo direct photolysis (Appendix C.2.2);
- will not appreciably partition to organic carbon in particulate matter in the air (Appendix C.2.2);

- will exist in the gas phase (Appendix C.2.2);
- will undergo slow indirect photolysis (Appendix C.2.2);
- will not undergo hydrolysis at environmental pH and temperature (Appendix C.2.3);
- will undergo slow or negligible biodegradation in water under aerobic conditions where indigenous microbial communities have not been pre-exposed to 1,1-dichloroethane (Appendix C.2.3.1);
- will undergo slow biodegradation to form chloroethane in soil and sediment under anaerobic conditions (Appendix C.2.3.2);
- will volatilize from surface water and moist soil (Appendixes C.2.3.1 and C.2.4.1);
- will not appreciably partition to organic carbon in sediment and soil thus has the potential to migrate to groundwater (Appendices C.2.3.2 and D.2.4.1);
- is not bioaccumulative in fish (Appendix C.2.6);
- will be removed in wastewater treatment by volatilization with a very low fraction adsorbed onto sludge (Appendix C.2.5.2);
- is minimally removed in conventional drinking water treatment but may be highly removed by certain other treatment technologies (activated carbon adsorption and packed tower aeration) (Appendix H.3);
- is not expected to undergo long-range transport (LRT) relative to LRT benchmark chemicals (Appendixes C.2.2); and
- can be formed under environmental conditions by the anaerobic biodegradation of 1,1,1-trichloroethane (Appendix C.2.4.1).

There is limited evidence on the aerobic biodegradation of 1,1-dichloroethane in water under environmental conditions. The single study identified was a laboratory study that employed extensive efforts to develop microbial populations capable of biodegrading 1,1-dichloroethane. As such, extrapolating rates of biodegradation observed in the laboratory study to environmental biodegradation rates is highly uncertain (Appendix C.2.3.1). A detailed discussion of strengths, limitations, assumptions, and key sources of uncertainty for the fate and transport assessment of 1,1-dichloroethane is available in Appendix C.2.

3 RELEASES AND CONCENTRATIONS OF 1,1-DICHLOROETHANE IN THE ENVIRONMENT

EPA estimated environmental releases of 1,1-dichloroethane that are discussed in Sections 3.1 and 3.2. Specifically, Section 3.1 describes the approach and methodology for estimating releases whereas Section 3.2 presents estimates of environmental releases by geographic location, media of release, and by occupational exposure scenario (OES). Section 3.2 also includes an evaluation of the weight of scientific evidence for the environmental releases. Section 3.3 presents the approach, methodology for estimating environmental concentrations, and the estimates of environmental concentrations that result from environmental releases of 1,1-dichloroethane.

3.1 Approach and Methodology

The assessment of environmental releases for 1,1-dichloroethane focuses on releases from industrial and commercial sources.

3.1.1 Industrial and Commercial

1,1-Dichloroethane is a TRI-reportable substance effective January 1, 1994. It is (1) included on EPA's initial list of hazardous air pollutants (HAPs) under the Clean Air Act (CAA), (2) a designated toxic pollutant under the Clean Water Act (CWA), and (3) currently not subject to National Primary Drinking Water Regulations (NPDWR) under the Safe Drinking Water Act (SDWA).

As mentioned in Section 1.1.1, the total production volume (PV) of 1,1-dichloroethane in 2015 from the 2016 CDR reporting period was between 100 million and 1 billion lb. This range did not change in the 2020 CDR reporting period. Due to a lack of information, EPA was not able to identify the percentage of the PV that goes toward processing as a reactive intermediate or commercial use as a laboratory chemical. The Agency assumes that a high percentage of the PV is used for processing as a reactive intermediate and a small percentage of the PV is used for commercial use as a laboratory chemical.

EPA's approach for estimating releases is illustrated in Figure 3-1 below.

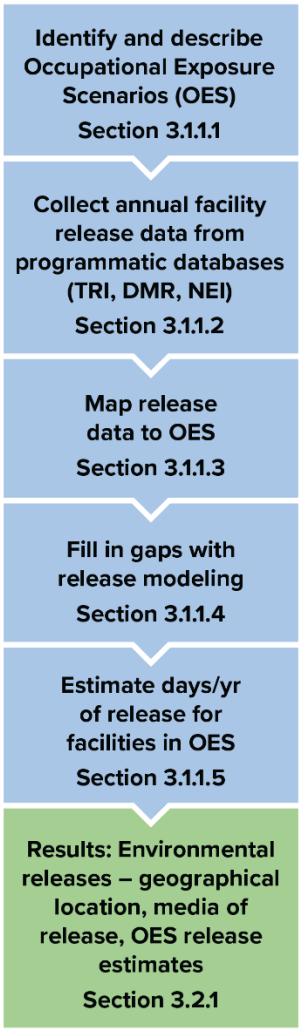


Figure 3-1. Overview of EPA's Approach to Estimate Releases for Each OES

The following Sections (3.1.1.1 through 3.1.1.5) provide information on this approach. A more detailed description of occupational exposures and environmental releases is available in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

3.1.1.1 Identify and Describe OES

COUs are the unique combinations of Lifestyle Stage, Category, and Subcategory that EPA developed and are presented in Table 1-1 of this risk evaluation. The Agency has identified a total of eight COUs in Table 3-1. An OES was identified for each COU with the exception of processing as a reactive intermediate where three COUs were combined into one OES due to expected similarities in release and exposure potential. Table 3-1 also lists the seven OESs that EPA assessed for 1,1-dichloroethane.

Table 3-1. Crosswalk of Conditions of Use to Occupational Exposure Scenarios Assessed

Condition of Use			OES
Life Cycle Stage	**Category[a]**	**Subcategory[b]**	
Manufacturing	Domestic manufacturing	Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate[c]
Processing	As a reactant	Intermediate in all other basic organic chemical manufacturing	Processing as a reactive intermediate
	As a reactant	Intermediate in all other chemical product and preparation manufacturing	
	Recycling	Recycling	
	Processing – repackaging	Processing – repackaging	Processing – repackaging
Distribution in Commerce	Distribution in commerce	Distribution in commerce	Distribution in commerce[d]
Commercial Use	Other use	Laboratory chemicals	Commercial use as a laboratory chemical
Disposal	Disposal	Disposal	General waste handling, treatment, and disposal
			Waste handling, treatment, and disposal (POTW)
			Waste handling, treatment, and disposal (remediation)

[a] These categories of COUs reflect CDR codes and broadly represent COUs for 1,1-dichloroethane in industrial and/or commercial settings.
[b] These subcategories reflect more specific uses of 1,1-dichloroethane.
[c] 1,1-Dichloroethane manufactured as a byproduct during the manufacture of 1,2-dichloroethane will be assessed in the risk evaluation for 1,2-dichoroethane.
[d] EPA considers the activities of loading and unloading of chemical product part of distribution in commerce. These activities were assessed as part of the OESs of Manufacturing of 1,1-dichloroethane as an isolated intermediate; Processing as a reactive intermediate; Processing – repackaging; and Commercial use as a laboratory chemical. EPA's current approach for quantitively assessing releases and exposures for the remaining aspects of distribution in commerce consists of searching Department of Transportation (DOT) and National Response Center (NRC) data and for incident reports pertaining to 1,1-dichloroethane distribution.

After identifying the OES that will be assessed, the next step was to describe the function of 1,1-dichloroethane within each OES (Table 3-2). This would be utilized in mapping release data to an OES as well as in applying release modeling approaches.

Table 3-2. Description of the Function of 1,1-Dichloroethane for Each OES

OES	Role/Function of 1,1-Dichloroethane
Manufacturing of 1,1-dichloroethane as an isolated intermediate	1,1-Dichloroethane may be produced by chlorination of ethane or chloroethane, addition of hydrogen chloride to acetylene or vinyl chloride, or oxychlorination with hydrogen chloride. Additionally, 1,1-dichloroethane is manufactured as a byproduct or impurity during the intentional manufacturing of 1,2-dichloroethane (NCBI, 2020a; Dreher et al., 2014).
Processing as a reactive intermediate	1,1-Dichloroethane is used as an intermediate in the production of other chemicals, primarily 1,1,1-trichloroethane (Dreher et al., 2014; RIVM, 2007; U.S. EPA, 2000a). Additionally, EPA assumes that waste streams containing 1,1-dichloroethane may be recycled on-site and then re-introduced into the facility's process waste stream or recycled as a feedstock to be used in the manufacture of other chemicals.
Processing – repackaging	A portion of the 1,1-dichloroethane manufactured is expected to be repackaged into smaller containers for commercial laboratory use.
Distribution in commerce	1,1-Dichloroethane is expected to be distributed in commerce for processing as a reactive intermediate and commercial laboratory use. EPA expects 1,1-dichloroethane to be transported from manufacturing sites to downstream processing and repackaging sites.
Commercial use as a laboratory chemical	1,1-Dichloroethane is used as a laboratory reference standard domestically for instrument calibration and analytical method validation (Sigma-Aldrich, 2020).
Waste handling, treatment, and disposal	Each OES can generate waste streams of 1,1-dichloroethane that are collected and transported to third-party sites for disposal or treatment, and these cases are assessed under this OES.

3.1.1.2 Collect Facility Release Data from Data Sources

Sections 3.1.1.2.1 through 3.1.1.2.5 describe sources of facility-specific release data for 1,1-dichloroethane and the methods used to collect the data from TRI, DMRs, and the NEI. To help evaluate trends in releases, release data were collected for multiple years from these data sources. The results of the systematic review are also a potential source of release data as described in Section 3.1.1.3.4.

When evaluating releases during distribution in commerce of 1,1-dichloroethane, EPA considered National Response Center (NRC) data and Department of Transportation (DOT) Hazmat Incident Report Search Tool data during the 2015 to 2020 timeframe (NRCE, 2009) (DOT Hazmat Incident Report Data) as described in Section 3.1.1.2.5.

3.1.1.2.1 Toxic Release Inventory (TRI)

The TRI database includes facility-specific information on disposal and other releases of 1,1-dichloroethane to air, water, and land (U.S. EPA, 2022f). The release data are reported in lb/year. EPA downloaded available water, air, and land release data from TRI for six reporting years from 2015 through 2020:

- Air emissions in TRI are reported separately for stack air and fugitive air and occur on-site at the facility. From 2015 to 2020, 23 facilities reported air emissions of 1,1-dichloroethane, and there were 98 total reports.
- Water releases in TRI include both reports of annual direct discharges to surface water and annual indirect discharges to off-site POTWs and wastewater treatment (WWT) facilities. Four facilities reported water releases of 1,1-dichloroethane, with a total of nine reports over the 6 years that were assessed.

- Land releases in TRI provide the type of release media for a particular facility, as well as how the chemical is managed through recycling, energy recovery, or treatment. Two facilities reported land releases of 1,1-dichloroethane to RCRA Subtitle C landfills and other non-site landfills respectively, and there were six non-zero reports over the 6 years assessed.

EPA obtained 2015 to 2020 TRI data for 1,1-dichloroethane from EPA's Basic Plus Data Files that collectively contain all the data submitted by facilities on the TRI reporting Form R and Form A for a specific year. EPA followed a similar approach to estimate air, water, and land releases. The Agency used the reported annual releases directly as reported in TRI. EPA then divided the annual releases over the number of estimated operating days (as discussed in Section 3.1.1.5) to obtain daily average release estimates. EPA presents the release data as high-end and central tendency estimates, as discussed in Section 3.2.1. Release estimates are separated by stack and fugitive air emissions, surface water discharges, and land releases.

A facility is required to report to TRI if it has 10 or more full-time employees; is included in an applicable North American Industry Classification System (NAICS) code; and manufactures, processes, or uses specific chemicals in quantities greater than specified thresholds.[4] Facilities provide on-site release information using readily available data (including monitoring data) collected pursuant to other provisions of law, or where such data are not readily available, "reasonable estimates" of the amounts released.

For each release quantity reported, TRI filers select a "basis of estimate" code to indicate the principal method used to determine the release quantity. TRI provides six basis of estimate codes, which in no particular order, are continuous monitoring, periodic monitoring, mass balance calculations, published emission factors, site-specific emission factors, and engineering calculations/best engineering judgment. For facilities that use a TRI chemical in multiple operations, the filer may use a combination of methods to calculate the overall release quantity. In such cases, TRI instructs the facility to enter the basis of estimate code for the method that corresponds to the largest portion of the reported release quantity.[5] Additional details on the basis for the reported release estimate (*e.g.*, calculations, underlying assumptions) are not reported in TRI.

For further discussion of water, air, and land emission data collection and estimation from TRI, refer to the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

3.1.1.2.2 Discharge Monitoring Reports (DMR)

DMRs include facility-specific information on releases of 1,1-dichloroethane to water. Under the CWA, EPA regulates the discharge of pollutants into receiving waters through the National Pollutant Discharge Elimination System (NPDES). A NPDES permit authorizes facilities to discharge pollutants up to specified limits and requires facilities to monitor their discharges and report the results to EPA and the state regulatory agency in DMRs. EPA makes these reported data publicly available via EPA's Enforcement and Compliance History Online (ECHO) system and EPA's Water Pollutant Loading Tool (Loading Tool). The data collected is annual release data for a given reporting year.

[4] See https://www.epa.gov/toxics-release-inventory-tri-program/tri-threshold-screening-tool (accessed June 16, 2025).
[5] See TRI Program Guidance on EPA's GuideME website under Reporting Forms and Instructions, Section 5. Quantity of the Toxic Chemical Entering Each Environmental Medium On-Site (Form R).

EPA downloaded DMR data from reporting years 2015 through 2020 (U.S. EPA, 2022c) using ECHO system and the Loading Tool. Over the 6 reporting years, 79 facilities reported water releases in DMR for 1,1-dichloroethane with a total of 219 reports.

Where available, EPA used DMR data to estimate annual wastewater discharges, average daily wastewater discharges, and high-end daily wastewater discharges. For DMR, annual discharges are automatically calculated by the Loading Tool based on the sum of the discharges associated with each monitoring period in DMR. Monitoring periods in DMR are set by each facility's NPDES permit and can vary between facilities. Typical monitoring periods in DMR include monthly, bimonthly, quarterly, biannual, and annual reporting.

In instances where a facility reports a period's monitoring results as below the limit of detection (LOD) (also referred to as a non-detect or ND) for a pollutant, the Loading Tool applies a hybrid method to estimate the wastewater discharge for the period. The hybrid method sets the values to half of the LOD if there was at least one detected value in the facility's DMRs in a calendar year. If all values were less than the LOD in a calendar year, the annual load is set to zero. EPA included emissions below the LOD in the release estimates. To estimate daily discharges, EPA divided the annual discharges over the number of estimated operating days (as discussed in Section 3.1.1.5). In some cases, the same facility reported water releases to both TRI and DMR for a given reporting year. EPA presented data from both sources for the water release assessment.

For further discussion on the collection of DMR data, refer to *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

3.1.1.2.3 National Emissions Inventory (NEI)

NEI was established to track emissions of Criteria Air Pollutants (CAPs)[6] and CAP precursors and assist with National Ambient Air Quality Standard (NAAQS) compliance under the CAA. 1,1-Dichloroethane is on EPA's initial list of HAPs under the CAA.[7] Air emissions data for the NEI are collected at the state, local, and tribal (SLT or S/L/T) level.[8] SLT air agencies then submit these data to EPA through the Emissions Inventory System (EIS). In addition to CAP data, many SLT air agencies voluntarily submit data for pollutants on EPA's list of HAPs. EPA uses the data collected from SLT air agencies, in conjunction with supplemental HAP data, to build the NEI. EPA releases an updated NEI every 3 years.

For this risk evaluation, NEI emissions data for 1,1-dichloroethane were collected for both point sources and area (or nonpoint) sources. Point sources are stationary sources of air emissions from facilities with operating permits under Title V of the CAA, also called "major sources." Point source facilities include large energy and industrial sites and are reported at the emission unit[9] and release point-level.[10] As documented in the Technical Support Document for the 2017 NEI.

For point sources (in general, large facilities), emissions are inventoried at a process-level within a facility. The point data are collected from S/L/T air agencies and EPA emissions programs including the TRI, the Acid Rain Program, and Maximum Achievable Control Technology

[6] The CAA requires EPA to set National Ambient Air Quality Standards (NAAQS) for five CAPs: ground-level ozone (O_3), particulate matter (PM), carbon monoxide (CO), lead (Pb), sulfur dioxide (SO_2), and nitrogen dioxide (NO_2).
[7] See EPA's initial list of HAPs (accessed June 16, 2025) and subsequent modifications.
[8] See EPA Air Emissions Reporting Requirements (AERR) (accessed June 16, 2025).
[9] Defined as any activity at a stationary source that emits or has the potential to emit a regulated air pollutant.
[10] Defined as the point from which air emissions from one or more processes are released into the atmosphere (*e.g.*, a stack).

(MACT) standards development. For nonpoint sources (typically smaller, yet pervasive sources) and mobile sources[11] (both onroad and nonroad), emissions are given as county totals.[12]

Area or nonpoint sources are stationary sources that do not qualify as major sources. The nonpoint data are reported at the county-level and include emissions from smaller facilities as well as agricultural emissions, construction dust, and open burning. Industrial and commercial/institutional fuel combustion, gasoline distribution, oil and gas production and extraction, publicly owned treatment works, and solvent emissions may be reported in the point or nonpoint source categories depending upon source size.[13]

EPA downloaded NEI data from reporting years 2014 and 2017, which were the most recent datasets available at the time of this evaluation. In 2017, there were 2,111 facilities that reported point source air emissions of 1,1-dichloroethane to NEI and 5,136 point source reports, and 13,527 area source reports. In 2014, there were 2,111 facilities that reported point source air emissions to NEI, 4,192 total reports, and 13,269 area source reports.

Where available, EPA used NEI data to estimate annual and average daily fugitive and stack air emissions. Facility-level annual emissions are available for major sources in NEI. EPA then divided the annual stack and fugitive emissions over the number of estimated operating days (as discussed in Section 3.1.1.5) to develop daily release estimates. In some cases, the same facility reported air releases to both TRI and NEI for a given reporting year. EPA presented data from both sources for the air release assessment.

See the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b) for additional information on obtaining NEI data.

3.1.1.2.4 Systematic Review

EPA conducted a systematic review of the literature to supplement release data of 1,1-dichloroethane from DMR, TRI, and NEI. The systematic review process is briefly described in Section 1.2. More detail regarding these steps is provided in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b). Upon review of the literature, EPA did not identify release data pertaining to 1,1-dichloroethane.

3.1.1.2.5 National Response Center and DOT Hazmat

Section 103 of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) requires the person in charge of a vessel or an onshore or offshore facility to immediately notify the National Response Center (NRC) when a CERCLA hazardous substance is released at or above the reportable quantity (RQ) in any 24-hour period, unless the release is federally permitted (40 CFR 302). The NRC is an emergency call center maintained and operated by the U.S. Coast Guard that fields initial reports for pollution and railroad incidents. Information reported to the NRC is available on the NRC website. The DOT Hazmat Incident Report Data uses submissions from Hazardous Materials

[11] Note that the NEI provides data for marine vessel and railroad sources at the sub-county, "polygon" shape-level. "For wildfires and prescribed burning, the data are compiled as day-specific, coordinate-specific (similar to point) events in the 'event' portion of the inventory, and these emission estimates are further stratified by smoldering and flaming components (Section 1.2 of EPA's Technical Support Document for the 2017 NEI)."
[12] See Section 1.2 of EPA's Technical Support Document for the 2017 NEI.
[13] See EPA's 2017 National Emissions Inventory: January 2021 Updated Release, Technical Support Document (accessed June 16, 2025)

Incident Reports (DOT Form F 5800.1 [01/2004]) that are required to be reported within 30 days of the discovery of an incident (49 CFR 171).

EPA reviewed NRC data and DOT data for the 2015 to 2020 calendar years for incident reports pertaining to distribution of 1,1-dichloroethane (NRCE, 2009) (DOT Hazmat Incident Report Data). The Agency did not identify reported releases for 1,1-dichloroethane during distribution of the chemical.

3.1.1.3 Map Facility Release Data to OES

EPA developed the OES to group processes or applications with similar sources of release that occur at industrial and commercial workplaces within the scope of the risk evaluation. There are data available in each of these data sources that can be utilized to map the facility to an OES. The full details of the methodology for mapping facilities from EPA reporting programs is described in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b). In brief, mapping consists of using facility reported industry sectors (typically reported as either North American Industry Classification System [NAICS] or Standard Industrial Classification [SIC] codes), and chemical activity, processing, and use information to assign the most likely OES to each facility. A brief overview of the mapping process is shown in Figure 3-2. Mapping results, as well as the associated release data, are provided in *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

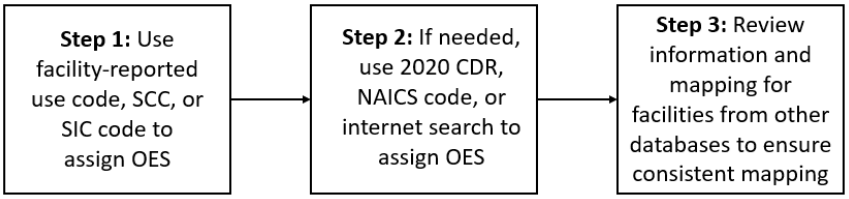


Figure 3-2. Overview of EPA's Approach to Map Facility Release Data to OES

3.1.1.3.1 Mapping TRI Release Data to an OES

TRI provides facility-specific information such as name, address, and other facility identification information. However, TRI does not include descriptive information on the activity of the chemical at the facility. There is information in the TRI that can be utilized to map the facility to a particular OES. For example, the Olin Blue Cube Facility in Freeport, Texas, reported releases of 1,1-dichloroethane to TRI. The facility reported a TRI use code that indicates 1,1-dichloroethane is processed as a reactant at the facility. Using the provided use code, EPA mapped the facility to the Processing as a reactive intermediate OES.

In some cases, there are multiple TRI uses reported by a given facility. To determine the OES for these facilities, EPA used the 2020 CDR, NAICS codes, and internet searches to determine the type of products and operations at the facility. *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b) for further discussion on mapping TRI data to an OES.

3.1.1.3.2 Mapping DMR Release Data

DMR provides facility-specific information such as name, address, and other facility identification information. However, DMR does not include descriptive information on the activity of the chemical at the facility, and unlike the TRI mapping, DMR facilities do not include any use/sub-use codes. There is information in the DMR that can be utilized to map the facility to a particular OES. For example, Amcol

Health and Beauty Solutions, Inc. reported water discharges of 1,1-dichloroethane to DMR. For a particular facility in DMR, the report will include a SIC code. The SIC code provided for this facility is 8731 – Commercial Physical and Biological Research. EPA mapped the facility to the Commercial use as a laboratory chemical OES based on the reported SIC code. In some cases, EPA assigned the OES by reviewing 2020 CDR for 1,1-dichloroethane (U.S. EPA, 2020a) or conducting an internet search of the types of products and operations at the facility.

3.1.1.3.3 Mapping NEI Release Data

NEI provides facility-specific information, such as name, address, site description, and other facility identification information. Additionally, there is information in NEI that can be used to assign a facility to a particular OES. For example, the Northwest Tennessee Disposal Corporation reported air emissions of 1,1-dichloroethane to NEI. According to NEI reporting, the facility is included in the waste disposal sector. The Source Classification Codes (SCC) also indicate waste disposal operations at the facility. Based on the sector and SCC, EPA mapped the facility to Waste handling, treatment, and disposal. In some cases, EPA assigned an OES using NAICS codes or conducting an internet search of the types of products and operations at the facility.

3.1.1.3.4 Mapping Systematic Review Data

EPA did not identify release data pertaining to 1,1-dichloroethane from systematic review data.

3.1.1.4 Fill in Gaps with Modeling to Estimate Releases for OES with No Data

Generally, EPA performs modeling to estimate releases when

- releases are expected for an OES but TRI, DMR, and/or NEI data or release data from systematic review are not available; or
- the Agency determines that the facility release data collected do not capture the entirety of environmental releases for an OES.

Standard models that have been previously developed by EPA are used to estimate releases. The models include loss fraction models as well as models for estimating chemical vapor generation rates. If EPA determines that an existing model does not capture the entirely of releases for a given scenario, a new model may be developed.

EPA modeled releases for two OESs: Processing – repackaging as well as the Commercial use as a laboratory chemical. The Agency modeled releases for both scenarios due to uncertainty in whether facility release data captured the entirety of environmental releases. For the Repackaging OES, EPA identified three relevant facilities in DMR. The release estimates reported by those facilities were below the LOD and there were no releases reported to air and land media. For modeling of releases, a Generic Scenario on Repackaging was utilized which includes estimation methods for key parameters to estimate magnitude of releases from a facility such as the facility throughput in kg/site-day. EPA lacked specific information on 1,1-dichloroethane for the repackaging OES so there is uncertainty in the values of facility throughput used in the modeling.

For the Laboratory chemicals OES, EPA identified four relevant facilities in DMR and NEI. One of the facilities reported a release estimate that was below the LOD in DMR. Additionally, there were no releases reported to land media for this OES. Due to uncertainty in whether the data from these four facilities were sufficient to capture the entirety of releases for this OES, the Agency also modeled releases. There was uncertainty in the number of sites that could use 1,1-dichloroethane as a laboratory chemical and therefore uncertainty in the facility throughput parameter.

Additionally, EPA identified the following Generic Scenarios (GSs) that are applicable to the OES: The July 2022 Chemical Repackaging – Generic Scenario for Estimating Occupational Exposures and Environmental Releases (U.S. EPA, 2023b) and Use of Laboratory Chemicals – Generic Scenario for Estimating Occupational Exposures and Environmental Releases (U.S. EPA, 2023b). Both GSs list standard models that are applicable to the release scenarios. For both scenarios, EPA used the following approach to obtain high-end and central tendency release estimates:

1. Identify release sources and media of release for the OES.
2. Identify model input parameters from relevant literature sources, GSs, or Emission Scenario Document (ESDs). Model input parameters include the estimated number of sites, container size, mass fractions, and 1,1-dichloroethane's physical properties. If a range of input values is available for an input parameter, determine the associated distribution of input values.
3. Identify model equations based on standard models from relevant GSs or ESDs.
4. Conduct a Monte Carlo simulation to calculate the total 1,1-dichloroethane release (by environmental media) across all release sources during each iteration of the simulation.
5. Select the 50th percentile and 95th percentile values to estimate the central tendency and high-end releases, respectively.

EPA performed a Monte Carlo simulation to variability in the model input parameters. The simulation used the Latin hypercube sampling method in @Risk Industrial Edition, Version 7.0.0, which generates a sample of possible values. The Agency performed the model at 100,000 iterations to capture a broad range of possible input values. The model generates statistics, and any desired percentile may be selected. EPA selected the 50th percentile and 95th percentile to estimate releases.

Detailed descriptions of the model approaches used for each OES, model equations, input parameter values and associated distributions are provided both in Section 3.3 and the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b). Additionally, input parameters and modeling results are provided in *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Laboratory Chemical Occupational Exposure and Environmental Release Modeling Results* (U.S. EPA, 2025j); *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Repackaging Environmental Release Modeling Results* (U.S. EPA, 2025k); and *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Repackaging Occupational Exposure Modeling Results* (U.S. EPA, 2025l).

3.1.1.5 Estimate the Number of Release Days per Year for Facilities in the OES

EPA's general approach is to estimate both an annual (kg/site-year) and a daily (kg/site-day) release rate for a facility. Data on the number of release days for a facility are not available from data sources such as DMR and TRI. As a surrogate, EPA uses generic estimates of the number of operating days (days/year) for facilities in each OES as presented in Table 3-3.

Table 3-3 lists generic estimates of the number of operating days/year for a facility in the OES for the 1,1-dichloroethane release assessment. A daily release rate for a facility with TRI data; for example, can be estimated by using the annual facility release from TRI and dividing it by the number of operating days/yr. The annual release and average daily release of 1,1-dichloroethane can be utilized in evaluating potential environmental concentrations, as discussed in Section 3.3. See *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b) for further discussion on the methodologies used to estimate the number of operating days. Additionally, see Section 3.3 for assumptions of release days applied to exposure modeling.

Table 3-3. Generic Estimates of Number of Operating Days per Year for Each OES

OES	Operating Days (days/year)	Basis
Manufacturing of 1,1-dichloroethane as an isolated intermediate	350	For the manufacture of the large-PV solvents, EPA assumes 350 days/year for release frequency. This assumes the plant runs 7 days/week and 50 weeks/year (with 2 weeks down for turnaround) and assumes that the plant is always producing the chemical.
Processing as a reactive intermediate	350	1,1-Dichloroethane is largely used to manufacture other commodity chemicals, such as 1,1,1-trichloroethane, which will likely occur year-round. Therefore, EPA assumes 350 days/year for release frequency.
Processing – repackaging	260	The July 2022 Chemical Repackaging GS (U.S. EPA, 2023b) estimates a default of 260 operating days/year per the U.S. Bureau of Labor Statistics Occupational Employment Statistics (BLS OES) data (US BLS, 2020).
Commercial use as a laboratory chemical	260	The Draft GS on Use of Laboratory Chemicals (U.S. EPA, 2023b) estimates a default of 260 operating days/year per the BLS OES data (US BLS, 2020).
General waste handling, treatment, and disposal	250	It is unlikely that non-POTW waste handling, treatment, and disposal facilities use 1,1-dichloroethane every day; therefore, EPA assumes 250 days/year (5 days/week, 50 weeks/year).
Waste handling, treatment, and disposal (POTW)	365	POTWs are expected to operate continuously over 365 days/year; therefore, 365 days/year should be used.
Waste handling, treatment, and disposal (remediation)	365	Remediate sites are expected to operate continuously over 365 days/year; therefore, 365 days/year should be used.

3.2 Environmental Releases

Estimates of releases for 1,1-dichloroethane in this section are from industrial and commercial sources.

3.2.1 Industrial and Commercial Releases

This section provides results of EPA's 1,1-dichloroethane environmental release analysis. Although data on the exact percentage are not available, publicly available information states that the use of 1,1-dichloroethane is as a feedstock. Therefore, EPA assumes that most of the production volume for 1,1-dichloroethane is reactive intermediate use where 1,1-dichloroethane would be reacted to make another chemical and therefore the 1,1-dichloroethane would be consumed and not available at that point for environmental release.

EPA developed environmental release information by estimating and summarizing the following:

- number of facilities with 1,1-dichloroethane environmental releases,
- facility releases according to geographic location,
- releases according to media of release, and
- releases per OES facility.

3.2.1.1 Number of Facilities with 1,1-Dichloroethane Emissions

EPA compiled the number of facilities reporting 1,1-dichloroethane releases from TRI, NEI, and DMR. Each programmatic database provides facility-specific release information. DMR data provides annual

effluent measured or monitored concentrations of 1,1-dichloroethane into receiving water bodies as well as other NPDES permit information. TRI provides both facility-specific annual water release as well as air emissions and land disposal quantities and NEI provides facility's unit-specific annual ambient air release estimates. For the Processing – repackaging OES and Commercial use as a laboratory chemical OES, the number of sites were estimated as part of the release modeling. EPA assumed a range. The number of facilities is presented by OES and shown in Table 3-4.

Table 3-4. Number of Sites with 1,1-Dichloroethane Environmental Releases

OES	Number of Sites from Programmatic Databases				Number of Sites Estimated During Release Modeling
	DMR[a]	TRI	NEI	Unique Sites[b]	
Manufacturing of 1,1-dichloroethane as an isolated intermediate	1	9	10	10	–
Processing as a reactive intermediate	58	6	32	90	–
Processing – repackaging	3	–	–	3	2
Commercial use as a laboratory chemical	2	–	2	4	43–138
General waste handling, treatment, and disposal	22	8	650	672	–
Waste handling, treatment, and disposal (POTW)	125	–	–	125	–
Waste handling, treatment, and disposal (remediation)	42	–	–	42	–
Natural gas fired reciprocating engines	–	–	1,380	1,380	–
Facilities not mapped to an OES	68	–	35	103	–

[a] Includes sites in DMR that reported releases of 1,1-dichloroethane below the LOD.
[b] Due to the nature of DMR/TRI/NEI reporting, some facilities appear in multiple programmatic databases. The Laboratory chemical OES was assessed from both the limited facility-specific data as well as modeling based on the assumption that more sites may use 1,1-dichloroethane as a laboratory chemical than represented by the facility data. The modeling approach used a Monte Carlo simulation produced a range of site estimates, with the 50th percentile at 43 sites and the 95th percentile at 138 sites.

EPA expects that the major contributor to the large number of landfills sites in NEI reporting 1,1-dichloroethane in the air emissions must be sources other than 1,1-dichloroethane COUs of Manufacture, Processing, and Commercial use. The 2015 ATSDR Tox Profile (ATSDR, 2015) states that emissions of 1,1-dichloroethane in landfills come from the anaerobic decomposition of the organic material in the landfill; decomposition of 1,1,1-trichloroethane forms 1,1-dichloroethane as a major product. 1,1-Dichloroethane has a presence in landfills, either by direct disposal of 1,1-dichloroethane or decomposition of 1,1,1-trichloroethane. However, it is unclear how much 1,1,1-trichloroethane is disposed to landfills and how much 1,1-dichloroethane is generated.

Sites were mapped to "Natural gas fired reciprocating engines" in NEI due to sites that reported 1,1-dichloroethane emissions during natural gas combustion. However, upon further review, these 1,1-dichloroethane emissions were likely due to the use of an AP-42 natural gas-fired reciprocating engine emissions factor, which was not based on quantitative measurements of 1,1-dichloroethane, but non-detects. Therefore, EPA does not believe there are actual 1,1-dichloroethane emissions from these NEI sites. It should be noted that the number of records in NEI may differ from the number of sites, as multiple records may exist for a single site.

Facilities were not mapped to an OES in cases where information on the 1,1-dichloroethane use at the site was not available. These sites do not fit in any of the 1,1-dichloroethane OES since they are mainly tire manufacturing, pulp and paper, and alloy production.

3.2.1.2 Environmental Releases by Geographic Location

This section provides mapping of the location of facilities reporting air emissions of 1,1-dichloroethane from TRI and NEI respectively. Ambient air releases as reported by TRI from reporting years 2015 to 2020 are presented below in Figure 3-3.

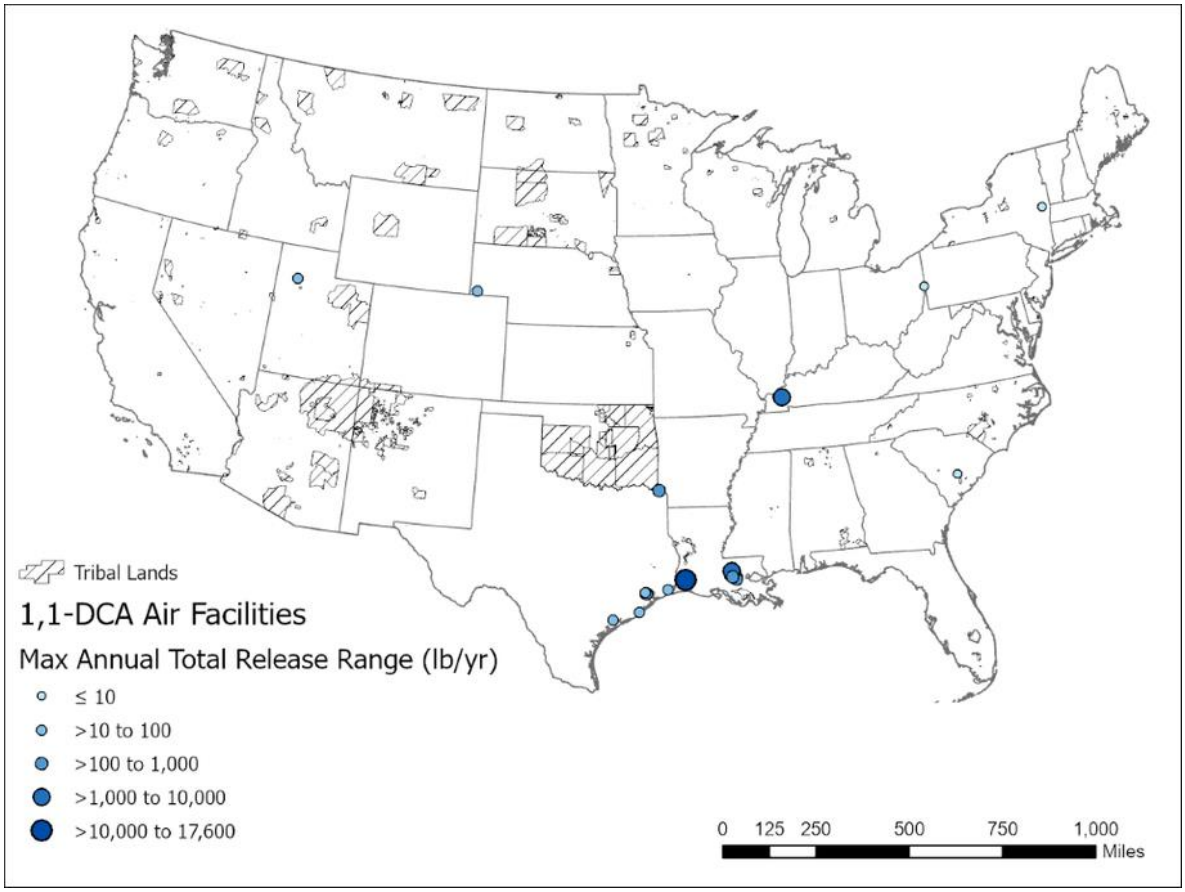


Figure 3-3. 1,1-Dichlorothane Annual Releases to Air as Reported by TRI, 2015–2020
Note: Some symbols for individual years may overlap and obscure annual releases at each site.
Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown as there are no known releases for these territories reported to TRI.

Ambient air releases as reported by NEI from reporting years 2014 and 2017 are presented below in Figure 3-4.

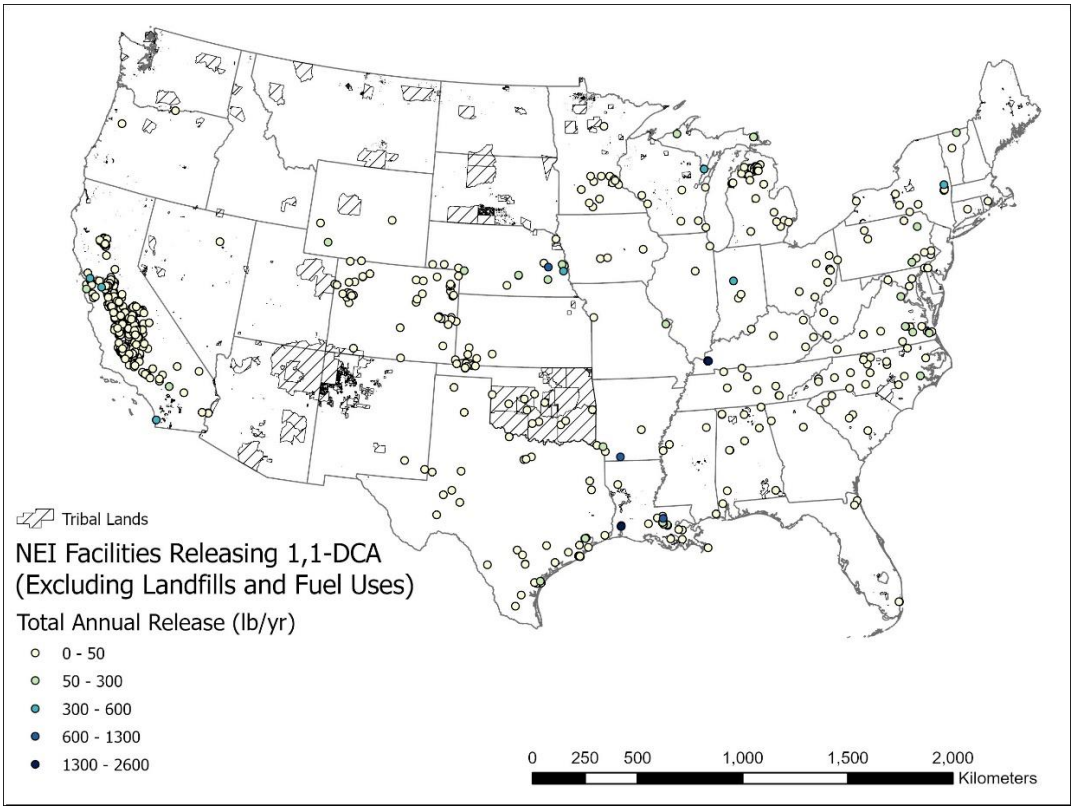


Figure 3-4. 1,1-Dichloroethane Annual Releases to Air as Reported by NEI, 2014 and 2017

3.2.1.3 Environmental Releases by Media of Release

EPA compiled the annual environmental releases by air, water, and disposal media as presented in Table 3-5. The data used to compile the release estimates from TRI and DMR are from reporting years 2015 to 2020, and the data from NEI are from reporting years 2014 and 2017. The release estimates are presented by media of release. NEI releases from natural gas fired reciprocating engines and landfills are not included in Table 3-5. However, TRI reported disposal of 1,1-dichloroethane to landfills are included in subsequent land/soil/groundwater estimates.

EPA estimated the releases by media by summing annual releases that were reported directly by facilities from the programmatic databases and then averaging across the corresponding number of years of release. For example, for fugitive air releases, the Agency averaged the total yearly releases from 2015 to 2020 TRI and 2014 and 2017 NEI to develop an average annual release estimate. The yearly fugitive releases from 2015 to 2020 TRI are as follows: 2,565 kg/year, 2,238 kg/year, 2,260 kg/year, 2,662 kg/year, 1,990 kg/year, and 4,000 kg/year. The fugitive releases from 2014 and 2017 NEI are 38,576 kg/year, and 37,879 kg/year, respectively. The average annual fugitive release estimate from 2015 to 2020 TRI and 2014 and 2017 NEI data is 11,521 kg/year.

Table 3-5. Average Annual Environmental Release Estimates by Media of Release

Media of Release[a]	Subcategory[b]	Average Annual Release Estimate (kg/yr)	Sources
Air	Fugitive air (data)	11,521	TRI/NEI
	Stack air (data)	3,505	TRI/NEI
	Fugitive or stack air (modeled release estimates)	<777	Environmental release modeling
Water	Surface water	1,052	TRI/DMR
Disposal	Land (data)	1.0	TRI
	1,1-Dichloroethane sent to a hazardous waste landfill or to incineration for combustion of the waste stream	<22,682[c]	Environmental release modeling

[a] Categories broadly represent the media of release for 1,1-dichloroethane in industrial and/or commercial settings.
[b] Subcategories reflect more specific releases of 1,1-dichloroethane.
[c] Ninety-seven percent of the hazardous waste landfill or incineration releases are from the Commercial use as a laboratory chemical OES. Because 1,1-Dichloroethane is included on the list of hazardous wastes pursuant to RCRA section 3001 (40 CFR 261.33) as a listed waste on the list, EPA assumed all disposal for the scenario would be to hazardous waste landfill or incineration.

3.2.1.4 Environmental Releases by OES

EPA compiled the annual and daily release estimates by OES as presented in Table 3-6. The release estimates are also separated by release media (*e.g.,* surface water, fugitive air, stack air, etc.). Annual release estimates were reported directly by facilities in TRI, DMR, and NEI. The facility release data were then mapped to an OES as discussed in Section 3.1.1.3. Annual fugitive air and stack air release data were provided by TRI and NEI, surface water discharge release data were provided by TRI and DMR, and land release data were provided by TRI.

For example, one site was mapped to the Manufacturing of 1,1-dichloroethane as an isolated intermediate OES that reported land releases to TRI. The site-reported land releases for reporting years 2015 to 2017 and 2019 to 2020, with the following release values: 2.3, 1.5, 1.4, 0.4, and 0.2 kg/year. EPA then selected the 50th and 95th percentile land release estimates for this site that are presented in Table 3-6 (1.4 and 2.1 kg/site-year, respectively). EPA then divided the annual release estimate by the estimated number of release days as discussed in Section 3.1.1.5, which is 350 days/year for the Manufacturing of 1,1-dichloroethane as an isolated intermediate OES. The 50th and 95th percentile daily land releases for the Manufacturing of 1,1-dichloroethane as an isolated intermediate OES are 3.9×10^{-3} kg/day and 6.0×10^{-3} kg/day, respectively.

Table 3-6. Summary of EPA's Annual and Daily Release Estimates for Each OES

OES	Estimated Annual Release (kg/site-year)		Type of Discharge,[b] Air Emission,[c] or Transfer for Disposal[c]	Estimated Daily Release (kg/site-day) [e]		Number of Facilities [f]	Source(s)
	Central Tendency	High-End [a]		Central Tendency	High-End		
Manufacturing of 1,1-dichloroethane as an isolated intermediate	1.6	1,299	Surface water	4.7E−03	3.7	3	TRI/DMR
	8.4	2,184	Fugitive air	2.4E−02	6.2	8	TRI
	34	74	Fugitive air	9.5E−02	0.20	4	NEI
	45	499	Stack air	0.13	1.4	9	TRI
	33		Stack air	9.1E−02		1	NEI
	1.4	2.1	Land	3.9E−03	6.0E−03	1	TRI
Processing as a reactive intermediate	3.8E−03	7.5E−02	Surface water	1.1E−05	2.1E−04	60	TRI/DMR
	2.3	155	Fugitive air	1.0E−02	0.44	5	TRI
	4.1	327	Fugitive air	1.2E−02	0.93	16	NEI
	14	610	Stack air	4.0E−02	1.7	4	TRI
	3.8	526	Stack air	1.1E−02	1.5	23	NEI
	0.45		Land	1.3E−02		1	TRI
Processing – repackaging	1.7E−02	0.40	Surface Water	5.0E−05	1.1E−03	3	DMR
	11	19	Fugitive or stack air	0.24	0.46	2 generic sites	Environmental release modeling
	275	320	Hazardous landfill or incineration	6.0	9.4		
Commercial use as a laboratory chemical	1.1E−03	9.4E−03	Surface water	4.3E−06	3.7E−05	2	DMR
	3.4	6.2	Fugitive air	9.5E−03	1.7E−02	2	NEI
	2.0E−03	2.0E−03	Stack air	7.9E−06	7.9E−06	2	NEI
	17	32	Fugitive or stack air	7.2E−02	0.14	43–138 generic sites	Environmental release modeling
	504	882	Hazardous landfill or incineration	2.2	3.7		
General waste handling, treatment, and disposal	9.3E−04	6.0E−03	Surface water	3.7E−06	2.4E−05	22	TRI/DMR
	0.63	7.3	Fugitive air	2.5E−03	2.9E−02	7	TRI
	34	200	Fugitive air	0.14	0.81	575	NEI
	1.8E−02	0.82	Stack air	7.3E−05	3.3E−03	8	TRI
	2.5	134	Stack air	1.0E−02	0.54	153	NEI

OES	Estimated Annual Release (kg/site-year)		Type of Discharge,[b] Air Emission,[c] or Transfer for Disposal[c]	Estimated Daily Release (kg/site-day) [e]		Number of Facilities [f]	Source(s)
	Central Tendency	High-End [a]		Central Tendency	High-End		
Waste handling, treatment, and disposal (POTW)	5.1E−03	8.9E−02	Surface water	1.4E−05	2.4E−04	126	DMR
Waste handling, treatment, and disposal (remediation)	2.9E−04	8.5E−03	Surface water	8.0E−07	2.3E−05	42	DMR
Distribution in commerce	N/A[f]						

[a] "High-end" are defined as 95th percentile releases
[b] Direct discharge to surface water; indirect discharge to non-POTW; indirect discharge to POTW
[c] Emissions via fugitive air; stack air; or treatment via incineration
[d] Transfer to surface impoundment, land application, or landfills
[e] Where available, EPA used peer-reviewed literature (*e.g.*, GSs or ESDs to provide a basis to estimate the number of release days of 1,1-dichloroethane within a COU).
[f] EPA reviewed NRC data and DOT data for the 2015–2020 calendar years for incident reports pertaining to distribution of 1,1-dichloroethane (NRCE, 2009) (DOT Hazmat Incident Report Data). EPA did not identify reported releases for 1,1-dichloroethane during distribution of the chemical.

3.2.2 Weight of Scientific Evidence Conclusions for the Estimates of Environmental Releases from Industrial and Commercial Sources

EPA develops a conclusion on the weight of scientific evidence supporting the environmental release estimates based on the strengths, limitations, and uncertainties associated with the environmental release estimates. The conclusion is summarized using confidence descriptors: Robust, Moderate, Slight, or Indeterminate. EPA 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. Table 3-7 summarizes EPA's overall weight of scientific evidence conclusions for its release estimates for each of the assessed OES.

Table 3-7. Summary of Weight of Scientific Evidence Ratings for Environmental Release Estimates by OES

OES	Weight of Scientific Evidence Conclusion	Weight of Scientific Evidence Conclusion
Manufacturing of 1,1-dichloroethane as an isolated intermediate	Moderate to Robust	Water releases are assessed using reported releases from 2015–2020 TRI and DMR. The primary strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. The primary limitation is that the water release assessment is based on three reporting sites, and EPA did not have additional sources to estimate water releases from this OES. Based on other reporting databases (CDR, NEI, etc.), there are seven additional manufacturing sites that are not accounted for in this assessment. Air releases are assessed using reported releases from 2015–2020 TRI, and 2014 and 2017 NEI. A strength of NEI data is that NEI captures additional sources that are not included in TRI due to reporting thresholds. Factors that decrease the overall confidence for this OES include the uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI and NEI may not capture all relevant sites. Additionally, EPA made assumptions on the number of operating days to estimate daily releases. Land releases are assessed using reported releases from 2015–2020 TRI. The primary limitation is that the land releases assessment is based on one reporting site, and EPA did not have additional sources to estimate land releases from this OES. Based on other reporting databases (CDR, DMR, NEI, etc.), nine additional manufacturing sites are not accounted for in this assessment. Based on this information, EPA has concluded that the weight of 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.
Processing as a reactive intermediate	Moderate to Robust	Water releases are assessed using reported releases from 2015–2020 TRI and DMR, which both have a medium overall data quality determination from the systematic review process. The primary strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. The water release assessment is based on 60 reporting sites. Based on other reporting databases (CDR, NEI, etc.), 30 additional sites are not accounted for in this assessment. Air releases are assessed using reported releases from 2015–2020 TRI, and 2014 and 2017 NEI. A strength of NEI data is that NEI captures additional sources that are not included in TRI due to reporting thresholds. Factors that decrease the overall confidence for this OES include the uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI and NEI may not capture all relevant sites. Land releases are assessed using reported releases from 2015–2020 TRI. The primary limitation is that the land release assessment is based on one reporting site, and EPA did not have additional sources to estimate land releases from this OES. Based on other reporting databases (CDR, DMR, NEI, etc.), 89 additional sites are not accounted for in this assessment.

OES	Weight of Scientific Evidence Conclusion	Weight of Scientific Evidence Conclusion
		Based on this information, EPA has concluded that the weight of 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.
Processing – repackaging	Moderate	All facility release data were below the limit of detection, therefore, EPA assessed releases to the using the assumptions and values from the July 2022 Chemical Repackaging GS (U.S. EPA, 2023b), which the systematic review process rated medium for data quality. EPA used EPA/OPPT models combined with Monte Carlo modeling to estimate releases to the environment, with media of release assessed using assumptions from the ESD and EPA/OPPT models. EPA believes a strength of the Monte Carlo modeling approach is that variation in model input values and a range of potential releases values is more likely than a discrete value to capture actual releases at sites. EPA lacks 1,1-dichloroethane facility production volume data and number of importing/repackaging sites; therefore, throughput estimates are based on CDR reporting thresholds with an overall release using a hypothetical scenario of two facilities. For modeling of releases, a key parameter in determining the magnitude of releases from a facility is the facility throughput kg/site-day. EPA lacked specific information on 1,1-dichloroethane for the repackaging OES so there is uncertainty in the values of facility throughput used in the modeling. Based on this information, EPA has concluded that the weight of 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.
Commercial use as a laboratory chemical	Moderate	EPA identified four facilities reporting water and air releases of 1,1-dichloroethane, However, EPA determined this data is not sufficient to capture the entirety of environmental releases for this scenario. Therefore, releases to the environment are assessed using the Draft GS on the Use of Laboratory Chemicals, which has a high data quality rating from the systematic review process (U.S. EPA, 2023b). EPA used EPA/OPPT models combined with Monte Carlo modeling to estimate releases to the environment, with media of release assessed using assumptions from the ESD and EPA/OPPT models. EPA assumed that the media of release for disposal of laboratory waste is to hazardous waste landfill or incineration. EPA believes a strength of the Monte Carlo modeling approach is that variation in model input values and a range of potential releases values is more likely than a discrete value to capture actual releases at sites. EPA believes the primary limitation to be the uncertainty in the representativeness of values toward the true distribution of potential releases. In addition, EPA lacks 1,1-dichloroethane laboratory chemical throughput data and number of laboratories; therefore, number of laboratories and throughput estimates are based on stock solution throughputs from the *Draft GS on the Use of Laboratory Chemicals* and on CDR reporting thresholds. Based on this information, EPA has concluded that the weight of 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 Scientific Evidence Conclusion	Weight of Scientific Evidence Conclusion
General waste handling, treatment, and disposal	Moderate to Robust	Water releases for non-POTW sites are assessed using reported releases from 2015–2020 TRI and DMR. The primary strength of TRI data is that TRI compiles the best readily available release data for all reporting facilities. For non-POTW sites, the primary limitation is that the water release assessment is based on 22 reporting sites, and EPA did not have additional sources to estimate water releases from this OES. Based on other reporting databases such as NEI, there are additional sites that are not accounted for in this assessment. Air releases for non-POTW sites are assessed using reported releases from 2015–2020 TRI, and 2014 and 2017 NEI. A strength of NEI data is that NEI captures additional sources that are not included in TRI due to reporting thresholds. Factors that decrease the confidence for this OES include the uncertainty in the accuracy of reported releases, and the limitations in representativeness to all sites because TRI and NEI may not capture all relevant sites. The air release assessment is based on 650 reporting sites. Based on other reporting databases (CDR and DMR), there are 22 additional non-POTW sites that are not accounted for in this assessment. Additionally, EPA made assumptions on the number of operating days to estimate daily releases. EPA found that major sources of air emissions of 1,1-dichloroethane in landfills come from sources other than 1,1-dichloroethane COUs of Manufacture, processing, and commercial use; specifically, the decomposition of 1,1,1-trichloroethane. However, it is unclear how much 1,1,1-trichloroethane is disposed to landfills and how much 1,1-dichloroethane is generated. Based on this information, EPA has concluded that the weight of 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.
Waste handling, treatment, and disposal (POTW)	Moderate to Robust	Water releases for POTW sites are assessed using reported releases from 2015–2020 DMR. 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 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. The release assessment is based on 126 reporting sites. Based on other reporting databases (CDR, TRI, etc.), all sites are accounted for in this assessment. Based on this information, EPA has concluded that the weight of 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.
Waste handling, treatment, and disposal (remediation)	Moderate to Robust	Water releases for remediation sites are assessed using reported releases from 2015–2020 DMR. 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 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. The release assessment is based on 42 reporting sites. Based on other reporting databases (CDR, TRI, etc.), all sites are accounted for in this assessment.

OES	Weight of Scientific Evidence Conclusion	Weight of Scientific Evidence Conclusion
		Based on this information, EPA has concluded that the weight of 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.

3.3 Concentrations of 1,1-Dichloroethane in the Environment

1,1-Dichloroethane – Concentrations in the Environment (Section 3.3): Key Points

EPA evaluated the reasonably available information on concentrations of 1,1-dichloroethane in the environment, including air, water, and land (soil, biosolids, and groundwater). The key points on environmental concentrations are summarized in the bullets below:

- For the air pathway, data obtained from the EPA's ambient air monitoring databases provided 1,1-dichloroethane concentrations near facilities and locations represent general population exposure.
  - EPA modeled ambient air concentrations and air deposition to soil from facilities releasing 1,1-dichloroethane resulting from TSCA COU activities to air as reported to TRI and NEI databases from 2015 to 2020.
  - AERMOD modeled concentrations of NEI-reported facility releases ranged from 0 to 32 $\mu g/m^3$ with the maximum modeled concentration being similar to the maximum monitored concentration of 26 $\mu g/m^3$ from AMTIC (approximately 97% of the samples were non-detects), which is approximately an order of magnitude lower that the AERMOD maximum modeled concentration of TRI-reported facility releases of 232 $\mu g/m^3$.
  - EPA has high confidence in the modeled results representing 1,1-dichloroethane ambient air concentrations because (1) AERMOD is EPA's primary regulatory model for ambient air modeling and is peer-reviewed; (2) EPA used industry reported TRI and NEI releases as inputs for modeling; and (3) the ranges of the ambient air modeled concentrations from AERMOD are within the ranges of monitored concentrations from AMTIC data.
  - EPA has medium confidence in the modeled 1,1-dichloroethane air deposition results due to the medium confidence in the input parameter values for AERMOD deposition modeling.
- For the water pathway, measured data from a variety of locations (surfaces waters and groundwaters) within and outside of the United States provided 1,1-dichloroethane concentrations to understand general occurrence. However, these locations are not typically in receiving water bodies associated with the TSCA COU facility releases investigated or were not measured at relevant timeframes. Thus, it remains difficult to use monitoring data to assess general population exposure and compare with EPA modeled results.
  - EPA modeled aqueous concentrations in surface waters and groundwater from TSCA COU facilities reporting in DMR releases of 1,1-dichloroethane directly to a receiving surface water body or from reporting in TRI the disposal to landfill in the case of groundwater.
  - Facility releases to surface waters as reported in DMR and disposal to landfills as reported in TRI result in concentrations of 1,1-dichloroethane that present an exposure to the general population; however, these aqueous concentrations are expected to be low even for the conservative scenarios that were modeled.
- For the land pathway, EPA evaluated potential 1,1-dichloroethane concentrations in biosolids based on DMR POTW reported releases as well as potential for partitioning to solids. Biosolids application to soil was also considered a potential pathway for 1,1-dichloroethane presence in soil. Modeled concentrations of 1,1-dichloroethane resulting from POTW biosolids were estimated to be low.

The environmental exposure characterization focuses on releases of 1,1-dichloroethane from facilities that report use, manufacture, or processing of 1,1-dichloroethane under industrial and/or commercial COUs subject to TSCA regulations as described in Section 3.2.1. To characterize environmental exposure, EPA assessed point estimate exposures derived from both measured and modeled concentrations of 1,1-dichloroethane in ambient air, surface water, and groundwater resulting from landfills in the United States. Measured concentrations of 1,1-dichloroethane in groundwater are presented from monitoring data and predicted concentrations in soil are noted as a possible source of environmental exposures.

A literature search was also conducted to identify peer-reviewed sources of 1,1-dichloroethane measured and reported modeled data. Searches in sources not found in standard, peer-reviewed literature databases were also conducted, such as white papers, conference proceedings, technical reports, reference books, dissertations, information on various stakeholder websites and various databases. The tornado plots and associated tables in Appendix C and in the *Final Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025z) are a summary of the measured and reported modeled data for the various environmental media. The plots provide the range of media concentrations in monitoring various studies. The plots show U.S. and non-U.S. data, fraction (*e.g.*, vapor, gas, particle) and the studies are ordered from top-to-bottom from newer-to-older data. The plots are colored to indicate general population, remote, near facility, and unknown population information. An example of a tornedo plot and additional details on the location type such as near facility, general population, are provided in Appendix C.3.

3.3.1 Ambient Air Pathway

EPA searched peer-reviewed literature and reported release and monitoring databases to obtain concentrations of 1,1-dichloroethane in ambient air. Section 3.3.1.1 shows the results of reported measured concentrations for ambient air found in the peer-reviewed and gray literature from the systematic review and from the EPA AMTIC archive (accessed June 16, 2025). Section 3.3.1.2 reports EPA modeled ambient air concentrations and air deposition 1,1-dichloroethane from facility releases.

3.3.1.1 Measured Concentrations in Ambient Air

Ambient air concentrations of 1,1-dichloroethane were measured in one study in the United States (Figure 3-5). Logue et al. 2010) reported concentrations of 1,1-dichloroethane in ambient air from non-detect to 4.0×10^{-2} µg/m^3 at four locations across Pittsburgh, Pennsylvania (2 residential areas near chemical and industrial facilities, 1 downtown residential area with high traffic, and 1 residential area with distant industrial facilities), from 2006 to 2008.

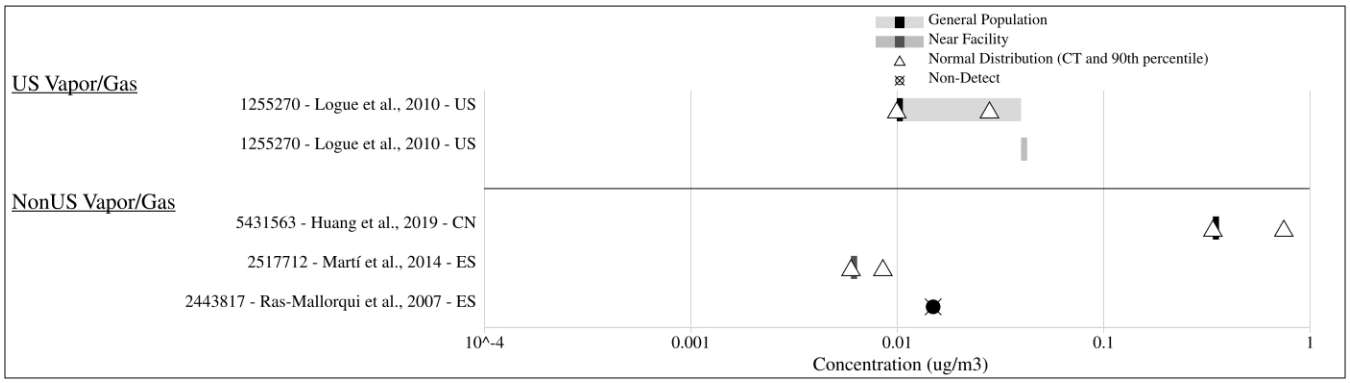


Figure 3-5. Concentrations of 1,1-Dichloroethane (µg/m^3) in the Vapor/Gas Fraction of Ambient Air from U.S.-Based and International Studies, 2005–2017

Additional ambient air concentrations of 1,1-dichloroethane were obtained from EPA's AMTIC archive. The AMTIC archive houses data from 2,800 ambient air monitoring sites across the United States from 1990 to 2020, with 90 percent of the data from the years 2000 to 2020, resulting from the air toxics program. The air toxics program includes the National Air Toxics Trends Sites (NATTS) Network, Community-Scale Air Toxics Ambient Monitoring (CSATAM), and Urban Air Toxics Monitoring Program (UATMP) that monitor for hazardous air pollutants (HAPs), including 1,1-dichloroethane. These data are reported from federal, state, local, and tribal monitoring networks. AMTIC HAPs monitoring data are summarized in Table 3-8 for the years 2015 to 2020. These years were selected to be consistent with the TRI and NEI data used in the modeled ambient air concentrations (Section 3.3.1.2). As shown in Table 3-8, measured concentrations from the AMTIC archive ranged from non-detect to 26 µg/m^3. Because most of the TRI reporting facilities are either in Texas (7 of 23) or in Louisiana (9 of 23), EPA focused on AMTIC data in these states. Approximately 25 percent of the monitoring data were reported by the State of Texas where nearly 99 percent of the samples were considered non-detects. The State of Louisiana reported approximately 8 percent of the monitoring data and about 95 percent of the data reported were considered non-detects.

For more information on 1,1-dichloroethane in ambient air monitoring data, see the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Ambient Monitoring Technology Information Center (AMTIC), 1,1-Dichloroethane Monitoring Data 2015 to 2020* (U.S. EPA, 2025d).

Table 3-8. Summary of Selected Statistics of 1,1-Dichloroethane Ambient Air Concentrations (µg/m^3) from EPA Ambient Monitoring Technology Information Center

Chemical	Statistic[a]	Year					
		2015	2016	2017	2018	2019	2020
1,1-Dichloroethane	Number of samples	12,332	11,954	11,849	11,495	10,234	9,581
	Percent ND	96.6	93.8	97.4	98.3	98.7	98.0
	Minimum[b]	ND	ND	ND	ND	ND	ND
	Mean	8.0E−02	8.5E−02	8.6E−02	0.11	0.12	0.13
	Max	7.6	2.0	26	1.2	8.9	6.1

ND = non-detect

[a] Approximately 97% of samples were NDs. For samples with a reported method detection limit (MDL), EPA considered any sample with a concentration below the MDL to be an ND. Additionally, for samples with no reported MDL, the Agency considered any sample with a concentration ≤0 to be an ND. For calculation of summary statistics, EPA did not include data points where no concentration was reported. EPA also did not include data points in the summary statistics where no MDL was reported and the concentration was ≤0. For data points where the concentration was less than the reported MDL, a concentration of half the MDL was used for calculating the mean.

[b] According to AMTIC's Technical Guide (accessed June 16, 2025), NDs are to be reported in AQS as zeros. Therefore, EPA is unable to distinguish between ND and 0 measured values. MDLs range from 4.0×10^{-3} to 2.3 µg/m^3

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

EPA developed and applied tiered methodologies and analyses to estimate ambient air concentrations and air deposition of 1,1-dichloroethane from facility releases. These methodologies and analyses focus on inhalation exposures to a sub-set of the general population residing nearby facilities general population residing within 10,000 m of a releasing facility. EPA considered multiple years of data and multiple data sets (TRI and NEI) for this analysis. The methodology and analyses were first presented in the *Draft TSCA Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities* (accessed June 16, 2025) referred to herein as the "2022 Draft Fenceline Report."[14] The specific methodologies used in this assessment to evaluate general population exposures to 1,1-dichloroethane in air are briefly described in Figure 3-6 and below. Additional details on the methodologies and the full set of inputs are provided in Appendix C.3 and in the risk evaluation reporting 1,1-dichloroethane releases to TRI and NEI. For purposes of these analyses, EPA focused on a subset of the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: AERMOD Input Specifications* (U.S. EPA, 2025c).

[14] See 2022 Fenceline Report.

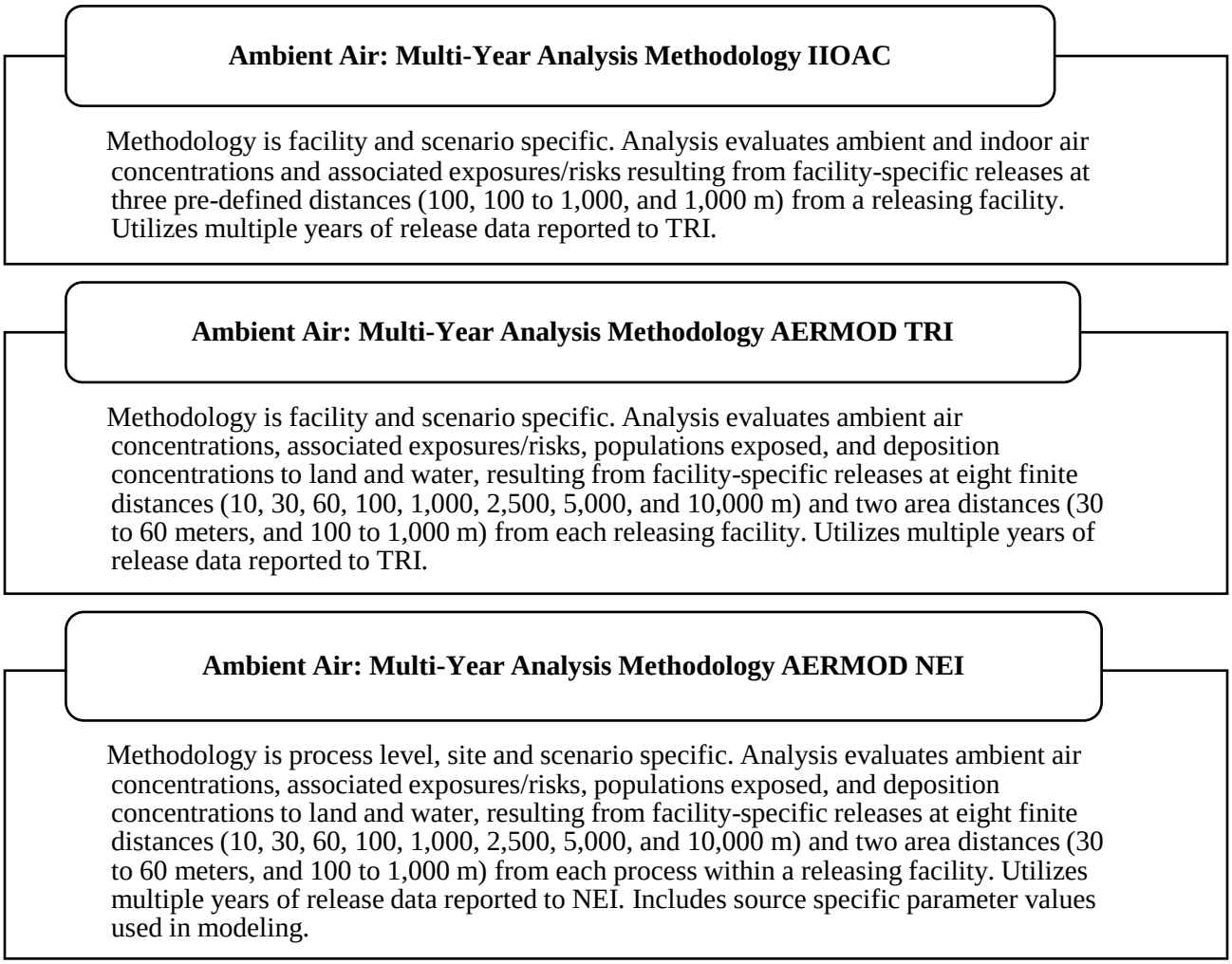


Figure 3-6. Brief Description of Methodologies and Analyses Used to Estimate Air Concentrations and Exposures

1,1-Dichloroethane ambient air concentrations were modeled using facility releases reported in TRI and NEI or alternative release estimates where facility specific data were not available. EPA performed a full analysis using the American Meteorological Society/Environmental Protection Agency Regulatory Model (AERMOD)[15] and EOA's Integrated Indoor/Outdoor Air Calculator (IIOAC).[16] EPA used the air release estimates obtained using the methodology described in Section 3.1 as direct inputs for the models to estimate exposure concentrations at various distances from a releasing facility. EPA expanded upon the methods described in the 2022 Draft Fenceline Report by evaluating air deposition and potential aggregate concentrations from multiple TRI and NEI reporting facilities.

Specifically, to estimate ambient air concentrations of 1,1-dichloroethane from facility releases EPA used the Ambient Air: Multi-Year Analysis Methodology IIOAC. This analysis relies upon TRI data and basic model inputs (IIOAC) and evaluates ambient and indoor air concentrations and associated exposures/risks at three pre-defined distances from a releasing facility to inform whether additional, more specific, higher-tier analysis may be warranted. For 1,1-dichloroethane, the results of the Ambient Air: Multi-Year Methodology IIOAC identified risk estimates above 1 in a million (1×10^{6}) for cancer at

[15] See https://www.epa.gov/scram/air-quality-dispersion-modeling-preferred-and-recommended-models#aermod (accessed June 16, 2025) for more information.
[16] See IIOAC website (accessed June 16, 2025) for more information.

all distances modeled and for multiple releases (high-end and central tendency). Due to results of the Ambient Air: Multi-Year Methodology IIOAC EPA conducted a higher-tier analysis (Ambient Air: Multi-Year Analysis Methodology AERMOD TRI) of all facilities reporting releases of 1,1-dichloroethane to TRI and NEI.

The Ambient Air: Multi-Year Analysis Methodology AERMOD TRI relies upon TRI data as the previous tier analysis but uses a higher tier model (AERMOD) and evaluates ambient air concentrations and associated exposures/risks at eight finite distances and two area distances from each releasing facility. This tier also evaluates total (wet and dry) deposition concentrations to land and water at each distance/area distance modeled. For 1,1-dichloroethane, the results of the Ambient Air: Multi-Year Analysis Methodology AERMOD TRI identified risk estimates above 1 in a million for cancer for multiple releases (high-end and central tendency). In order to confirm the conclusions, EPA utilized the next tier and the NEI dataset as described below.

The final tier EPA used in this assessment is the Ambient Air: Multi-Year Analysis Methodology AERMOD NEI. Compared to the previous two tiers of analyses that are facility- and scenario-specific, this analysis uses process unit releases and is site and scenario specific. It includes source-specific parameter values used in modeling like stack parameters (stack height, stack temperature, plume velocity, etc.), and releases of facilities that may not report to TRI.

3.3.1.2.1 Ambient Air: Multi-Year Methodology IIOAC

The Ambient Air: Multi-Year Methodology IIOAC utilizes EPA's IIOAC Model to estimate high-end (95th percentile) and central tendency (mean) 1,1-dichloroethane exposure concentrations in ambient air and indoor air at three distances from an emitting facility: 100, 100 to 1,000, and 1,000 m. EPA considered 6 years of TRI release data (2015–2020) for this analysis. The TRI data were used as direct inputs to the IIOAC. EPA modeled releases reported to TRI considering source attribution (fugitive and stack releases) for each facility and each year of reported releases. Facilities were categorized into OESs and later cross-walked to COUs. Indoor air concentrations were calculated by multiplying the outdoor air concentration by the indoor-outdoor ratio of 0.65 and 1 for the mean and high-end exposure concentrations, respectively.

The Ambient Air: Multi-Year Methodology IIOAC includes both estimates of exposures as well as estimates of risks to inform the need, or potential need, for further analysis. For 1,1-dichloroethane, the results of the Ambient Air: Multi-Year Methodology IIOAC identified risk estimates above typical Agency benchmarks for cancer at all distances modeled and for multiple releases (high-end and central tendency). Due to results of the Ambient Air: Multi-Year Methodology IIOAC and the inability to model gaseous deposition, EPA conducted a higher-tier analysis (AERMOD) of all facilities reporting releases of 1,1-dichloroethane to TRI and NEI.

The full set of inputs and results of IIOAC are provided in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis* (U.S. EPA, 2025p).

3.3.1.2.2 Ambient Air: Multi-Year Methodology AERMOD TRI

The Ambient Air: Multi-Year Methodology AERMOD TRI utilizes AERMOD to estimate 1,1-dichloroethane concentrations in ambient air and air deposition concentrations to land and water, at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and two area distances (30–60 m and 100–1,000 m) from an emitting facility (Appendix D.1.2.3). EPA modeled two different types of release estimates for 1,1-dichloroethane: (1) facility-specific chemical releases with source attribution

when TRI data were available, and (2) alternative release estimates representing a generic facility when TRI data were not available for an OES. When TRI data were available, EPA considered 6 years of release data (2015–2020), and modeled releases reported to TRI considering source attribution (fugitive and stack releases) for each facility and each year of reported releases as well as an arithmetic average release for each facility across all reported releases across all years. Not all facilities reported releases for all 6 years. Facilities were categorized into OESs and later cross-walked to COUs. 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 1,1-dichloroethane concentrations at each distance modeled.

A summary of the air concentration ranges estimated using the Ambient Air: Multi-Year Methodology AERMOD TRI is provided in Table 3-9. The summary includes three OESs and select statistics (maximum, 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. For the maximum 95th percentile, range of modeled concentrations varied by as much as four orders of magnitude between minimum and maximum concentrations across all modeled distances for the Manufacturing of 1,1-dichloroethane as an isolated intermediate OES, three orders of magnitude for the Processing as a reactive intermediate OES, and 12 orders of magnitude for the General waste handling, treatment, and disposal OES. This occurs because within each OES there are multiple facilities with varying releases. These varying releases, in turn, affect the range of estimated exposure concentrations at a given distance. AERMOD-modeled concentrations for the 95th percentile ranged from 0 to 232 $\mu g/m^3$ across all modeled distances, with the maximum modeled concentration being approximately one order of magnitude higher than the maximum monitored concentration of 26 $\mu g/m^3$ from AMTIC, where approximately 97 percent of the samples were non-detects (Table 3-8).

A summary of the air deposition rate ranges estimated using the Ambient Air: Multi-Year Methodology AERMOD TRI is provided in Table 3-10 and Table 3-11. The summary includes three OESs and select statistics (maximum, mean, median, and minimum) calculated from the TRI modeled deposition rates distributions within each OES at each distance modeled. The associated range of estimated deposition rates is based on the maximum 95th percentile daily (Table 3-10) and annual (Table 3-11) deposition rates for each distance. Table 3-12 provides a summary of the air concentrations estimated using the Ambient Air: Multi-Year Methodology AERMOD TRI for the Commercial use as a laboratory chemical and Processing – repackaging OESs where there was no site-specific location data available for modeling. The associated range of estimated concentrations is based on the maximum 95th percentile annual average exposure concentrations. The ambient air modeled concentrations values are presented for high-end modeled releases, high-end meteorology (Lake Charles, Louisiana), and both rural and urban settings. The high-end meteorological station used represents meteorological datasets that tended to provide high-end concentration estimates relative to the other stations within IIOAC (see Appendix D.1.2.4). The modeled results indicate a maximum ambient air concentration of 0.9 $\mu g/m^3$ at 10 m from the facility for the Processing – repackaging OES, 22,680 kg/year production volume, and 95th percentile release estimate scenario for both rural and urban land category scenarios. For the Commercial use as a laboratory chemical OES, results indicate a maximum ambient air concentration of 1.5 $\mu g/m^3$ at 10 m from the facility, 22,680 kg/year production volume, and 95th percentile release estimate scenario for both rural and urban land category scenarios.

The full inputs and results are presented in the *Risk Evaluation for 1,1-Dichloroethane — Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025o) and in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File:*

Supplemental Information on AERMOD Generic Releases Exposure and Risk Analysis (U.S. EPA, 2025m).

Table 3-9. Summary of Select Statistics for the 95th Percentile Annual Average Concentrations for 1,1-Dichloroethane Releases Reported to TRI[a]

OES	# Facilities Evaluated in OES [b]	Statistics	95th Percentile Annual Average Concentration (µg/m^3) Estimated Within 10–10,000 m of Releasing Facilities									
			10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	2,500 m	5,000 m	10,000 m
Manufacturing of 1,1-dichloroethane as an isolated intermediate	9	Max	2.3E02	9.0E01	6.9E01	3.7E01	1.8E01	2.5	4.1E−01	9.3E−02	3.0E−02	1.0E−02
		Mean	2.0E01	8.7	6.1	3.6	1.7	2.4E−01	4.3E−02	1.0E−02	3.5E−03	1.2E−03
		Median	6.1E−01	2.9E−01	1.8E−01	1.3E−01	6.2E−02	1.2E−02	3.3E−03	1.3E−03	5.7E−04	2.1E−04
		Min	4.0E−02	1.7E−02	1.1E−02	6.5E−03	3.0E−03	3.6E−04	6.4E−05	1.4E−05	4.6E−06	1.5E−06
Processing as a reactive intermediate	6	Max	1.5E01	6.4	4.3	2.5	1.2	1.6E−01	2.7E−02	1.3E−02	6.8E−03	2.9E−03
		Mean	3.2	1.4	9.7E−01	5.8E−01	3.0E−01	4.9E−02	1.3E−02	5.1E−03	2.3E−03	9.2E−04
		Median	2.2E−02	1.0E−02	3.8E−02	5.4E−02	1.1E−01	5.5E−02	1.7E−02	4.5E−03	1.5E−03	4.9E−04
		Min	0	0	0	0	0	0	0	0	0	0
General waste handling, treatment, and disposal	8	Max	1.9E01	9.3	6.1	3.9	1.9	1.4E−01	4.8E−02	1.1E−02	3.4E−03	1.1E−03
		Mean	8.4E−01	4.0E−01	2.6E−01	1.7E−01	8.2E−02	6.3E−03	2.0E−03	4.4E−04	1.5E−04	4.8E−05
		Median	4.1E−02	1.6E−02	1.1E−02	5.7E−03	2.4E−03	3.0E−04	4.9E−05	1.3E−05	4.5E−06	1.5E−06
		Min	7.6E−11	6.5E−08	3.6E−07	5.4E−07	9.4E−07	3.1E−07	1.1E−07	4.4E−08	2.4E−08	1.1E−08

[a] The full inputs and results are presented in the *Risk Evaluation for 1,1-Dichloroethane — Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025o).

[b] For each OES, EPA modeled all TRI-reported releases considering source attribution (fugitive and stack releases) for each facility from 2015–2020. Not all facilities reported releases for all 6 years.

Table 3-10. Summary of Select Statistics for the 95th Percentile Daily Average Air Deposition Rates for 1,1-Dichloroethane Releases Reported to TRI[a]

OES	# Facilities Evaluated in OES [b]	Statistics	95th Percentile Daily Average Air Deposition Rate (g/m^2-day) Estimated Within 10–10,000 m of Releasing Facilities									
			10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	2,500 m	5,000 m	10,000 m
Manufacturing of 1,1-dichloroethane as an isolated intermediate	9	Max	4.0E−02	3.9E−02	2.2E−02	1.3E−02	5.4E−03	1.8E−04	5.8E−05	1.0E−05	2.9E−06	8.9E−07
		Mean	3.3E−03	3.1E−03	1.7E−03	1.1E−03	4.1E−04	1.5E−05	4.6E−06	7.9E−07	2.4E−07	7.7E−08
		Median	2.8E−05	2.9E−05	1.7E−05	1.3E−05	1.3E−05	1.7E−06	6.1E−07	7.7E−08	2.1E−08	8.0E−09
		Min	1.5E−08	1.3E−08	6.9E−09	4.3E−09	1.7E−09	5.3E−11	1.8E−11	3.4E−12	1.1E−12	3.6E−13
Processing as a reactive intermediate	6	Max	8.9E−04	7.9E−04	4.6E−04	2.8E−04	1.2E−04	2.3E−05	9.3E−06	1.6E−06	4.2E−07	1.2E−07
		Mean	2.0E−04	2.0E−04	1.2E−04	8.0E−05	5.4E−05	5.9E−06	2.1E−06	3.8E−07	1.1E−07	3.5E−08
		Median	9.4E−06	1.3E−05	1.4E−05	3.0E−05	7.5E−05	2.7E−06	8.7E−07	1.4E−07	4.1E−08	1.4E−08
		Min	0	0	0	0	0	0	0	0	0	0
General waste handling, treatment, and disposal	8	Max	2.1E−05	2.7E−05	1.6E−05	1.1E−05	4.2E−06	1.3E−07	4.8E−08	7.8E−09	2.4E−09	8.8E−10
		Mean	2.9E−06	3.1E−06	1.9E−06	1.2E−06	4.8E−07	1.7E−08	6.2E−09	1.1E−09	3.3E−10	1.1E−10
		Median	8.0E−08	4.7E−08	2.3E−08	1.8E−08	2.2E−08	5.2E−10	1.6E−10	3.2E−11	1.0E−11	3.6E−12
		Min	2.9E−14	4.7E−12	5.6E−11	1.3E−10	2.2E−10	1.6E−11	4.0E−12	6.5E−13	2.3E−13	8.3E−14

[a] The full inputs and results are presented in the *Risk Evaluation for 1,1-Dichloroethane — Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025o).

[b] For each OES, EPA modeled all TRI-reported releases considering source attribution (fugitive and stack releases) for each facility from 2015–2020. Not all facilities reported releases for all 6 years.

Table 3-11. Summary of Select Statistics for the 95th Percentile Annual Average Air Deposition Rates for 1,1-Dichloroethane Releases Reported to TRI[a]

OES	# Facilities Evaluated in OES [b]	Statistic	95th Percentile Annual Average Air Deposition Rates (g/m^2-year) Estimated Within 10–10,000 m of Releasing Facilities									
			10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	2,500 m	5,000 m	10,000 m
Manufacturing of 1,1-dichloroethane as an isolated intermediate	9	Max	2.2E01	2.2E01	1.5E01	7.9	3.1	2.2E−01	3.8E−02	7.4E−03	2.3E−03	7.4E−04
		Mean	8.5E−01	8.6E−01	6.0E−01	3.1E−01	1.2E−01	9.4E−03	1.7E−03	3.3E−04	1.0E−04	3.3E−05
		Median	7.0E−03	6.9E−03	4.9E−03	3.0E−03	2.5E−03	5.3E−04	1.5E−04	3.8E−05	1.3E−05	4.3E−06
		Min	1.5E−06	1.3E−06	9.0E−07	4.5E−07	1.8E−07	2.0E−08	3.2E−09	7.4E−10	2.7E−10	1.1E−10
Processing as a reactive intermediate	6	Max	4.0E−01	4.5E−01	3.3E−01	2.0E−01	2.2E−01	4.3E−02	1.7E−02	3.5E−03	1.1E−03	3.3E−04
		Mean	4.4E−02	5.5E−02	4.2E−02	2.9E−02	2.6E−02	4.3E−03	1.4E−03	3.0E−04	9.0E−05	2.8E−05
		Median	2.3E−03	3.3E−03	9.4E−03	1.4E−02	1.8E−02	1.4E−03	3.0E−04	5.7E−05	1.9E−05	5.9E−06
		Min	0	0	0	0	0	0	0	0	0	0
General waste handling, treatment, and disposal	8	Max	5.1E−03	7.8E−03	5.6E−03	3.2E−03	1.3E−03	1.1E−04	1.7E−05	3.3E−06	9.9E−07	3.2E−07
		Mean	6.1E−04	7.9E−04	5.5E−04	3.2E−04	1.4E−04	1.0E−05	2.0E−06	4.0E−07	1.2E−07	4.2E−08
		Median	1.5E−05	1.5E−05	1.0E−05	6.7E−06	4.9E−06	4.6E−07	9.3E−08	2.4E−08	8.0E−09	2.6E−09
		Min	5.9E−12	3.2E−09	3.4E−08	7.2E−08	1.2E−07	1.5E−08	3.6E−09	6.7E−10	2.4E−10	1.0E−10

[a] The full inputs and results are presented in the *Risk Evaluation for 1,1-Dichloroethane — Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025o).

[b] For each OES, EPA modeled all TRI-reported releases considering source attribution (fugitive and stack releases) for each facility from 2015–2020. Not all facilities reported releases for all 6 years.

Table 3-12. Summary of Maximum 95th Percentile Annual Average Concentrations for 1,1-Dichloroethane for Commercial Use as a Laboratory Chemical, and Processing – Repackaging for Laboratory Chemicals OESs[a]

OES	Meteorology [b]	Source	Land	95th Percentile Annual Average Concentration (µg/m^3) Estimated Within 10–10,000 m of Releasing Facilities									
				10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	2,500 m	5,000 m	10,000 m
Processing – repackaging	High	Stack and Fugitive	Urban	9.3E−01	2.6E−01	2.1E−01	1.5E−01	1.4E−01	3.8E−02	1.3E−02	3.8E−03	1.3E−03	4.7E−04
	High	Stack and Fugitive	Rural	9.3E−01	2.6E−01	2.0E−01	1.2E−01	1.0E−01	3.4E−02	1.5E−02	4.5E−03	1.9E−03	9.8E−04
Commercial use as a laboratory chemical	High	Stack and Fugitive	Urban	1.5	4.4E−01	3.9E−01	3.1E−01	3.5E−01	1.0E−01	3.4E−02	1.0E−02	3.7E−03	1.3E−03
	High	Stack and Fugitive	Rural	1.5	4.3E−01	3.5E−01	2.5E−01	2.4E−01	9.0E−02	4.0E−02	1.3E−02	5.1E−03	2.5E−03

[a] The full inputs and results are presented in *the Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD Generic Releases Exposure and Risk Analysis* (U.S. EPA, 2025m). Releases for these two OESs were modeled as described in Section 3.1.1.4.
[b] High refers to meteorological conditions from Lake Charles, LA. Because the scenarios do not have location data, they were modeled using a meteorological station that represents meteorological datasets that tended to provide high-end concentration estimates relative to the other stations within IIOAC.

3.3.1.2.3 Ambient Air: Multi-Year Methodology AERMOD NEI

The Ambient Air: Multi-Year Methodology AERMOD NEI utilizes AERMOD to estimate 1,1-dichloroethane concentrations in ambient air and air deposition rates to land and water, at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and two area distance from an emitting facility. EPA considered the most recent 2 years of NEI release data (2014 and 2017) for this analysis. The NEI data were used as direct inputs to the AERMOD. NEI releases were categorized into OESs and later cross-walked to COUs. 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 1,1-dichloroethane concentrations at each distance modeled. A summary of the concentration ranges estimated using the Ambient Air: Multi-Year Methodology AERMOD NEI is provided in Table 3-13. The summary includes four OESs and select statistics (maximum, mean, median, and minimum) calculated from the NEI 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. EPA grouped all the NEI releases currently not mapped to an OES in the "Facilities not mapped to an OES" OES (Section 3.2).

Ambient Air: Multi-Year Methodology AERMOD NEI modeled concentrations ranged from 0 to 32 $\mu g/m^3$ (Table 3-13) with the maximum modeled concentration being similar to the maximum monitored concentration of 26 $\mu g/m^3$ (approximately 97% of the samples were non-detects) from AMTIC (Table 3-8), which is approximately an order of magnitude lower that the AERMOD TRI maximum modeled concentration of 232 $\mu g/m^3$ (Section 3.3.1.2.2). Like the AERMOD TRI, there are many instances where within an OES the range of maximum modeled concentrations extends across as many as five orders of magnitude across all modeled distances. This occurs because within each OES there are multiple facilities with varying releases. These varying releases, in turn, affect the range of estimated exposure concentrations at a given distance.

The full inputs and results are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD NEI Exposure and Risk Analysis* (U.S. EPA, 2025n).

Table 3-13. Summary of Select Statistics for the 95th Percentile Estimated Annual Average Concentrations for 1,1-Dichloroethane Releases Reported to NEI[a]

OES	# Releases Evaluated in OES [b]	Statistic	Annual Average Concentration (µg/m^3) Estimated Within 10–10,000 m of Releasing Facilities									
			10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	2,500 m	5,000 m	10,000 m
Commercial use as a laboratory chemical	2	Max	3.7E−02	1.2E−02	7.2E−03	4.2E−03	1.9E−03	1.9E−04	3.8E−05	8.2E−06	2.6E−06	8.4E−07
		Mean	1.2E−02	3.8E−03	2.4E−03	1.4E−03	6.2E−04	6.4E−05	1.3E−05	2.7E−06	8.7E−07	2.8E−07
		Median	1.7E−06	8.1E−07	5.6E−07	3.4E−07	1.7E−07	1.8E−08	4.1E−09	8.9E−10	2.9E−10	9.2E−11
		Min	4.2E−07	2.0E−07	1.4E−07	8.4E−08	4.1E−08	4.4E−09	1.0E−09	2.2E−10	7.1E−11	2.3E−11
Manufacturing of 1,1-dichloroethane as an isolated intermediate	9	Max	2.1	6.1	6.1	6.0	5.7	1.0	1.2E−01	2.6E−02	8.3E−03	2.6E−03
		Mean	7.0E−01	3.6E−01	3.0E−01	2.2E−01	1.6E−01	3.3E−02	4.7E−03	1.0E−03	3.3E−04	1.1E−04
		Median	3.8E−03	3.1E−03	4.2E−03	4.0E−03	2.7E−03	7.1E−04	1.7E−04	4.5E−05	1.7E−05	5.5E−06
		Min	0	0	0	0	0	0	0	0	0	0
Processing as a reactive intermediate	50	Max	3.2E01	1.2E01	8.2	4.9	2.2	2.7E−01	4.8E−02	1.7E−02	6.7E−03	2.4E−03
		Mean	9.9E−01	4.7E−01	3.1E−01	1.9E−01	8.9E−02	1.1E−02	3.0E−03	8.1E−04	3.1E−04	1.2E−04
		Median	1.3E−06	2.5E−05	1.7E−04	2.0E−04	4.4E−04	2.3E−04	7.2E−05	2.5E−05	1.1E−05	5.5E−06
		Min	0	0	0	0	0	0	0	0	0	0
General waste handling, treatment, and disposal	102	Max	1.3E01	8.2	6.5	4.1	2.1	2.1E−01	5.2E−02	1.1E−02	3.4E−03	1.0E−03
		Mean	8.3E−01	3.5E−01	2.5E−01	1.5E−01	7.6E−02	9.8E−03	2.0E−03	4.5E−04	1.5E−04	4.8E−05
		Median	3.1E−04	6.3E−04	6.9E−04	5.0E−04	3.3E−04	5.4E−05	1.8E−05	6.5E−06	2.5E−06	9.8E−07
		Min	0	0	0	0	0	0	0	0	0	0
Facilities not mapped to an OES	57	Max	9.2	3.7	2.8	1.5	7.3E−01	1.2E−01	1.8E−02	3.9E−03	1.3E−03	4.0E−04
		Mean	1.3E−01	5.7E−02	4.1E−02	2.3E−02	1.1E−02	1.7E−03	2.9E−04	6.6E−05	2.2E−05	7.6E−06
		Median	2.8E−09	2.9E−06	1.7E−05	2.4E−05	3.2E−05	1.4E−05	7.3E−06	2.8E−06	1.2E−06	4.4E−07
		Min	0	0	0	0	0	0	0	0	0	0

[a] The full inputs and results are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD NEI Exposure and Risk Analysis* (U.S. EPA, 2025n).

[b] For each OES, EPA modeled all NEI-reported releases considering source attribution (fugitive and stack releases) for each facility for 2014 and 2017 reported data. Not all facilities reported releases for both years.

3.3.1.2.4 Population Analysis

The Ambient Air: Multi-Year Methodology AERMOD TRI and NEI includes a detailed population analysis described in Appendix D.4. This includes an evaluation of the general population in terms of characterization of those members of the general population that are considered PESS (see Section 5.3.2), that are living within 1,000 m of TRI releasing facilities—locations with highest 1,1-dichloroethane ambient air concentrations (see Table 3-12). The analysis also includes an examination of the environments and community infrastructure surrounding the TRI release sites, such as residential neighborhoods, parks, schools, childcare centers, places of worship, and hospitals.

3.3.2 Indoor Air Pathway

Concentrations of 1,1-dichloroethane in the indoor environment may be limited to a few sources, the most likely from outdoor air intrusion to indoor air through heating, ventilation, air conditioning systems, and open windows. There are no consumer products or articles currently identified containing and off-gassing 1,1-dichloroethane and thus not anticipated to contribute to indoor 1,1-dichloroethane concentrations. Also, given the very low estimated groundwater concentrations (see Appendix G.1.2.3), vapor intrusion is not expected to be a source of 1,1-dichloroethane exposures.

3.3.2.1 Measured Concentrations in Indoor Air

Indoor air concentrations of 1,1-dichloroethane were measured in one study in the United States and one study in Canada (Figure 3-7). Lindstrom 1995) reported non-detect concentrations of 1,1-dichloroethane in indoor air in 34 homes (conventional single-family homes and townhomes) in the Rocky Mountains, United States between 1992 (pre-occupancy) and 1993 (during occupancy). Due to the lack of any additional U.S. studies, EPA also included an international study measuring 1,1-dichloroethane in indoor air in Canada.

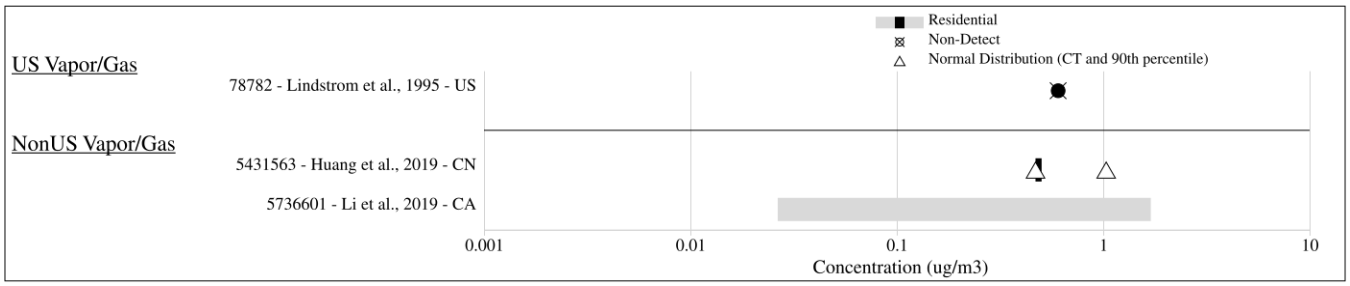


Figure 3-7. Concentrations of 1,1-Dichloroethane (µg/m^3) in the Vapor/Gas Fraction in Indoor Air, from U.S.-Based and International Studies, 1992–2017

3.3.2.2 Modeled Concentrations in Indoor Air

IIOAC calculates a mean and high-end indoor air concentration based on the outdoor/ambient air concentration and the mean and high-end indoor-outdoor ratios. In IIOAC, the indoor-outdoor ratio of 0.65 is used to calculate indoor air concentrations corresponding to the mean outdoor air concentration for each potentially exposed population. The indoor-outdoor ratio of 1 is used to calculate the indoor air concentration corresponding to the 95th percentile of outdoor air concentration of each potentially exposed population.

IIOAC-modeled high-end indoor air concentrations ranged from 9.9×10^{-8} to 18 µg/m^3 (Table 3-14). The range of concentrations can vary by as much as six orders of magnitude between minimum and maximum concentrations. This occurs because within each OES there are multiple facilities with

varying releases. These varying releases, in turn, affect the range of estimated exposure concentrations at a given distance.

The full inputs and results of IIOAC are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis* (U.S. EPA, 2025p).

Table 3-14. Summary of Select Statistics for the 95th Percentile Estimated Annual Average Indoor Air Concentrations for 1,1- Dichloroethane Releases Reported to TRI[a]

OES	# Facilities Evaluated in OES[b]	Statistic	Annual Average Indoor Air Concentration (µg/m^3) Estimated Within 100–1,000 m of Releasing Facilities		
			100 m	100–1,000 m	1,000 m
Manufacturing of 1,1-dichloroethane as an isolated intermediate	9	Max[c]	18	2.0	8.3E−01
		Mean[d]	1.5	1.8E−01	7.2E−02
		Median	4.1E−02	7.1E−03	3.3E−03
		Min	3.2E−03	3.7E−04	1.5E−04
Processing as a reactive intermediate	6	Max[c]	9.5E−01	1.1E−01	4.5E−02
		Mean[d]	2.1E−01	2.9E−02	1.3E−02
		Median	7.9E−02	2.5E−02	1.3E−02
		Min	0	0	0
Waste handling, treatment, and disposal	8	Max[c]	6.4E−01	7.5E−02	3.0E−02
		Mean[d]	2.7E−02	3.1E−03	1.3E−03
		Median	3.2E−03	3.8E−04	1.5E−04
		Min	5.9E−07	1.9E−07	9.9E−08

[a] The full inputs and results of IIOAC are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis* (U.S. EPA, 2025p).

[b] For each OES, EPA modeled all TRI-reported releases considering source attribution (fugitive and stack releases) for each facility from 2015–2020. Not all facilities reported releases for all 6 years.

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

[d] In IIOAC, the indoor-outdoor ratio of 0.65 is used to calculate indoor air concentrations corresponding to the mean outdoor air concentration for each potentially exposed population.

3.3.3 Surface Water Pathway

Surface water contamination from 1,1-dichloroethane occurs primarily from the direct discharge of wastewater from industrial operations and wastewater treatment plants. To understand the possible exposure scenarios from these ongoing practices, EPA assessed exposures to the general population from ambient surface waters and drinking water. The Agency also evaluated exposures to ecological species dwelling in the water column and benthic zone of ambient surface waters. These exposures are due to the release of 1,1-dichloroethane from direct facility discharges to receiving surface water bodies.

The evaluation of these exposures considered the review of available monitoring data collected from ambient surface waters and finished drinking water, as well as model results generated by EPA. Although EPA identified a robust set of surface and drinking water monitoring data (Section 3.3.3.1), indicating the presence of 1,1-dichloroethane in both sources of exposure, the timing and location that samples were collected as a part of these datasets typically do not coincide with locations and timeframes most relevant to modeled estimates of 1,1-dichloroethane concentrations using available

release information. Therefore, EPA relied primarily on a series of modeling approaches to estimate concentrations of 1,1-dichloroethane in surface waters near known release locations (Section 3.3.3.2.1) and at known downstream drinking water intake locations that serve public water systems (PWSs). To the degree possible, the relationship between monitoring and modeled data is further evaluated in Section 3.3.4.

3.3.3.1 Measured Concentrations in Surface Water

Measured aqueous concentration data for 1,1-dichloroethane in ambient surface water (*i.e.*, collected from rivers, streams, lakes, and ponds—rather than within industrial operations or drinking water systems) from across the country—were collected from public databases and peer-reviewed publications. Measured concentrations of 1,1-dichloroethane in finished (*i.e.*, treated) drinking water as a part of routine monitoring conducted by PWSs were likewise collected from public databases and peer-reviewed publications. The methods for retrieving this ambient surface water and PWS monitoring data are described in detail in Appendix E.

Measured concentrations of 1,1-dichloroethane from surface waters were retrieved from the Water Quality Portal (WQP) (NWQMC, 2022) to characterize the distribution of 1,1-dichloroethane levels found in ambient surface water from across the nation, and to provide context for the modeled surface water concentrations of 1,1-dichloroethane presented in Section 3.3.3.2.2. Measured data were retrieved irrespective of the reason for sample collection in order to assess trends in the observed concentrations more broadly. WQP data were downloaded in May 2023 for samples collected between 2015 to 2020, resulting in 6,274 data points (Figure 3-8 and Figure 3-9). Full details of the retrieval and data processing steps of ambient surface water monitoring data from the WQP are presented in Appendix E.

WQP concentrations of 1,1-dichloroethane measured in ambient surface waters ranged from the detection limit to 2 µg/L, with a median concentration of 0.25 µg/L and a 95th percentile concentration of 0.5 µg/L. Figure 3-8 shows the national spatial distribution of these results, with a strong bias of samples collected from New Mexico, Louisiana, North Carolina, and New Jersey. In the absence of a national standardized study of 1,1-dichloroethane in ambient surface water (that would be analogous to EPA's UCMR3 for drinking water), and without greater national coverage and metadata, it is difficult to characterize the national occurrence of 1,1-dichloroethane in surface waters. However, over-representation of certain states or regions may reflect targeted sampling campaigns of specific locations expected to have potentially high concentrations of 1,1-dichloroethane. Conclusions about areas without monitoring data cannot be drawn without further exploration through modeling. However, for those areas containing sufficient data coverage, it is apparent that 1,1-dichloroethane is found in relatively low quantities in ambient surface waters.

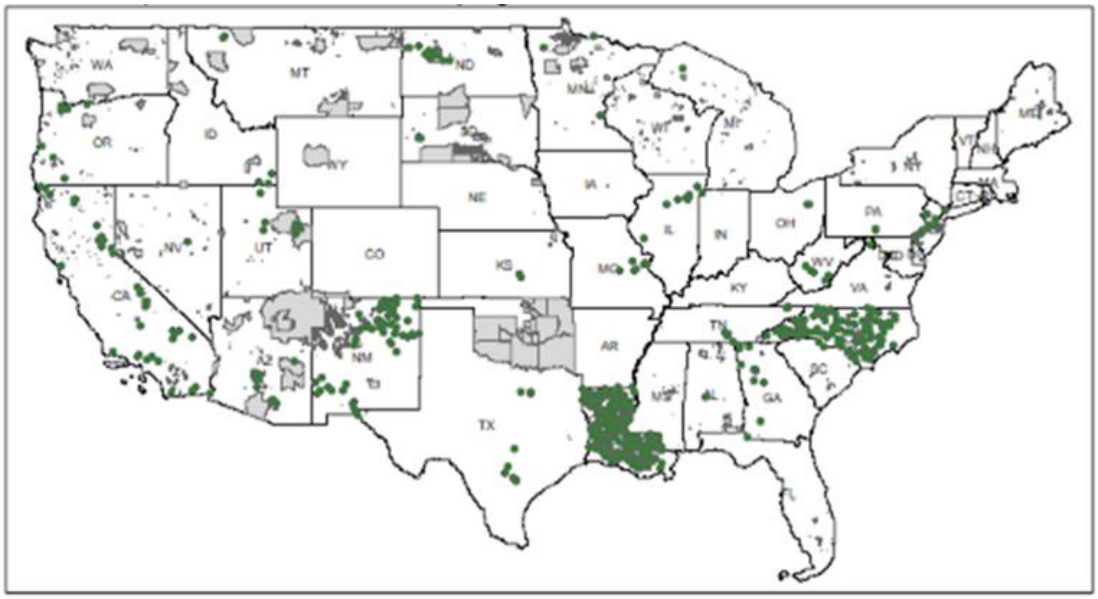


Figure 3-8. Locations of 1,1-Dichloroethane Measured in Ambient Surface Waters Obtained from the WQP, 2015–2020

American Indian, Alaska Native and Native Hawaiian (AIANNH) tribal boundaries are shaded gray. Note: Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown because they do not contain surface water monitoring data within the WQP.

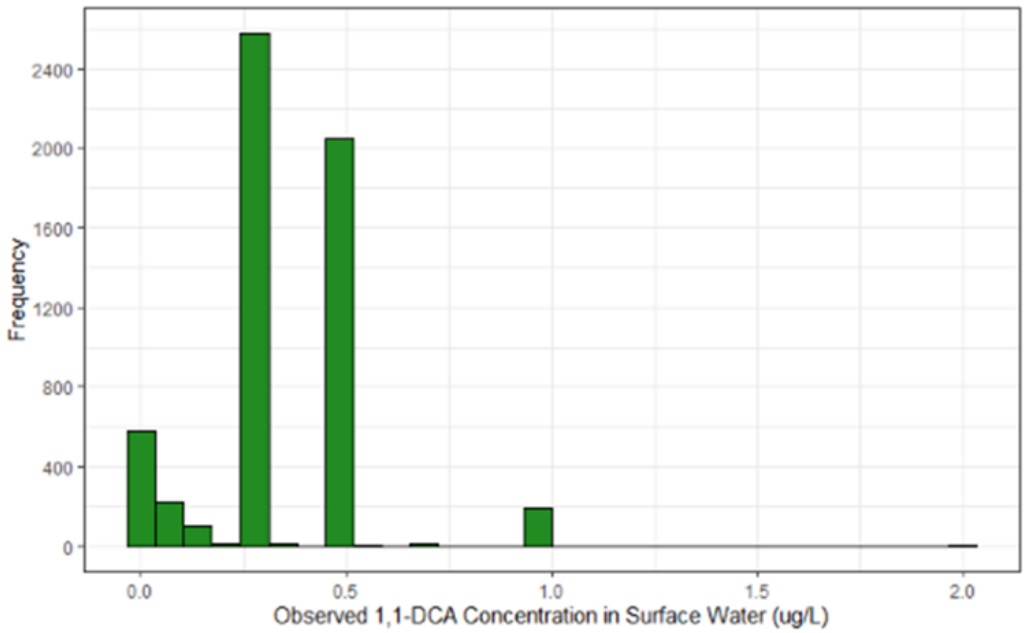


Figure 3-9. National Distribution of 1,1-Dichloroethane Concentrations Measured in Ambient Surface Waters from Surface Waters Obtained from the WQP, 2015–2020

A limited amount of 1,1-dichloroethane concentration data were identified through EPA's systematic review of published literature. A summary of the individual studies is shown in Figure 3-10. Results from peer-reviewed studies showed that concentrations of 1,1-dichloroethane ranged from not detected

to 48.7 µg/L from 155 surface water samples, from near facility release sites or not associated with release sites of 1,1-dichloroethane, collected between 1984 and 2005 in three countries: Australia, United Kingdom, and the United States. Reported detection frequency ranged from 0 to 0.5 µg/L. While these results collected from EPA's systematic review process are few, they do indicate that relatively high concentrations of 1,1-dichloroethane have been observed in ambient surface waters in years past.

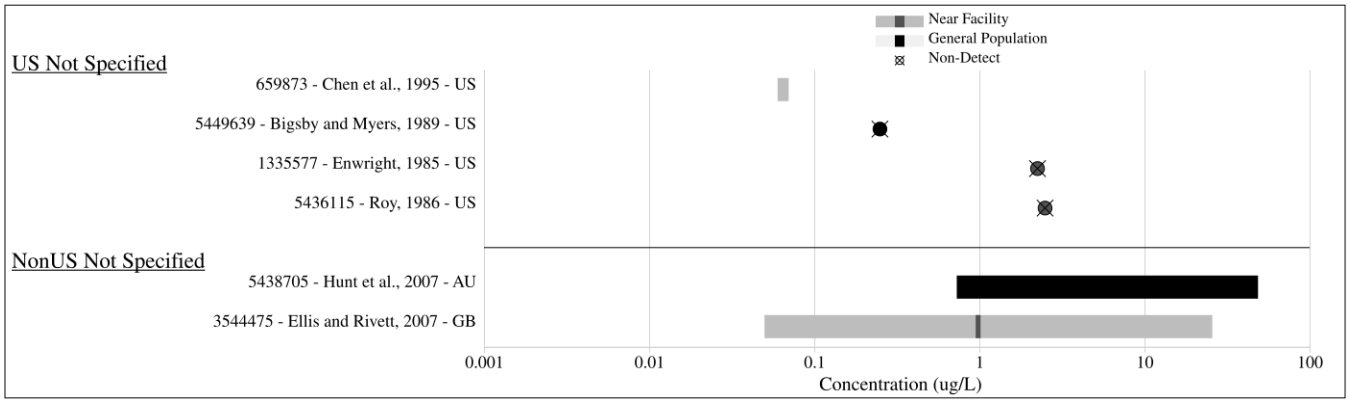


Figure 3-10. Concentrations of 1,1-Dichloroethane (µg/L) in Surface Water from U.S.-Based and International Studies, 1984–2005

3.3.3.2 Modeled Concentrations in Surface Water

To assess general population and aquatic ecological species exposures to 1,1-dichloroethane via industrial releases to surface waters, aqueous concentrations of 1,1-dichloroethane were modeled in the receiving water bodies of individual facility releases. These estimates reflect the highest potential aqueous concentrations resulting from reported 1,1-dichloroethane facility discharges.

3.3.3.2.1 Surface Water Modeling Methodology

A full description of the modeling approach to estimate concentrations of 1,1-dichloroethane in surface waters from direct facility-specific releases can be found in Appendix E.

As described in Section 3.2.1, annual releases of 1,1-dichloroethane to surface waters from regulated facility discharges were retrieved from the TRI and DMR public data records. To the extent possible, modeled hydrologic flow data (*i.e.,* stream flow) associated with the facility's receiving water body was retrieved from the NHDPlus V2.1 dataset (U.S. EPA, 2016c). The receiving water body was identified from NPDES permit information of the releasing facility for the 2015 to 2020 reporting period. Detailed methods for the retrieval and processing steps with the flow data are presented in Appendix E.2.1. Surface water (water column) concentrations of 1,1-dichloroethane were calculated for general population and human health exposures as well as exposure to aquatic ecological species.

Individual Facility Modeling

Individual facility modeling was conducted to estimate concentrations in receiving water bodies resulting from the highest facility-specific annual release reported between 2015 through 2020. An exception was made for the release data of the manufacturing COU facility where the next highest release data that occurred in 2016 was used in lieu of the highest release data corresponding with a hurricane event (see Section 3.3.3.2.3) in 2020 (U.S. EPA, 2025f). In some cases, a calculated facility effluent hydrologic flow was prioritized over a modeled NHD receiving water body stream flow value (see Appendix E for more details). This modeling approach employed the equations used to model releases from facilities in the E-FAST 2014 model (U.S. EPA, 2014a), which is described in Appendix E. Each facility and annual release amount were applied to a 1-day maximum release scenario, which

assumes that the annual release amount occurs in a single operation day as well as a scenario in which releases are equal to the facility's OES operating days (see Table 3-3). The former scenario provides more upper bound estimates of resulting surface water concentrations and are intended to evaluate the highest possible facility release pattern based on the best available information. The latter scenario which assumes releases occur during all facility operating days provides a refined analysis and provides more realistic surface water concentrations for estimating drinking water and fish ingestion exposure estimates.

Two flow metrics based on NHD hydrologic stream flow or the facility effluent hydrologic flow value were used to estimate concentrations associate with general population exposure and human health outcomes: a 30Q5 (the lowest 30-day average flow within a 5-year period) and the harmonic mean flow. The resulting modeled water column concentrations for each facility release site were used to calculate exposures related to human dermal contact, oral ingestion, and fish consumption.

The 7Q10 flow metric (the lowest measured 7-day average flow within a 10-year period) was used to estimate concentrations and exposures to aquatic ecological species. These 7Q10 flow values were also based on NHD stream flow or a calculated facility effluent flow. Aqueous concentrations of 1,1-dichloroethane for acute and chronic aquatic ecological exposures were calculated as described in Appendix E. To estimate concentrations for acute or water column ecological exposure, the highest annual facility load was divided by one and then paired with the respective receiving water body flow value, which assumes the annual release occurred in a single operation day. To estimate concentrations for chronic ecological exposure, the highest annual facility load was divided by 21, which thereby assumes the annual release occurred in equal daily amounts over the course of 21 consecutive facility operation days.

The acute (highest 1-day daily) and chronic (highest 21-day daily) concentrations were then compared with identified concentrations of concern (COCs) for acute water column ecological exposure (8,931 μg/L) and chronic water column ecological exposure (93 μg/L). Details that describe how the COCs were chosen can be found in Section 4.2.5.1. Facility releases that result in modeled acute and chronic aqueous concentrations of 1,1-dichloroethane that exceed these water column COCs formed a new list of facility releases to re-model estimates of water column concentration using the Point Source Calculator (PSC). A description of the PSC and modeling steps taken herein can be found in Section 3.3.3.2.3. The PSC allows for a refined estimation of chemical concentrations in the water column of receiving water bodies that takes into consideration several key physicochemical and fate properties of the chemical following its release into surface water (*e.g.*, biological and physical degradation). The PSC is a preferred model for estimating concentrations of 1,1-dichloroethane for ecological species exposures, but the model in its present version is impractical to apply for multiple sites without making certain assumptions surrounding the model's input parameters. Details on the assumptions made can be found in Section 3.3.3.2.3. After applying PSC, refined estimates of 1,1-dichloroethane concentration in the water column were again compared with their respective acute and chronic water column COCs. Those facility releases with modeled aqueous concentrations that exceed their respective COC formed a final list of facility releases. This list was carried through to estimate acute and chronic water column 1,1-dichloroethane concentrations for the ecological exposure assessment using the PSC. In addition, the modeled number of days that the concentration exceeds the respective acute or chronic COC was calculated by PSC and considered in the ecological exposure evaluation.

3.3.3.2.2 Surface Water Modeling Results

The locations where surface water concentrations of 1,1-dichloroethane were modeled are shown in Figure 3-11. The number of facilities and the corresponding annual release amounts used to generate

concentration estimates are shown in Figure 3-12. Concentrations of 1,1-dichloroethane for each individual direct facility release to their respective receiving surface water body or within a calculated facility effluent flow were estimated using facility releases as reported to EPA via the NPDES permit reporting requirements. These results reflect estimates of the potential 1,1-dichloroethane concentration at the site of facility release into surface water, where the entire annual release derived from the Pollutant Loading Tool is assumed to occur over the period of facility operating days. It is important to note that these results do not consider aggregate contribution of 1,1-dichloroethane from other sources, including instances where multiple facility releases combine within the same stream/river network.

The lowest modeled 30Q5-based 1,1-dichloroethane concentrations were near detection limit. The 25th, 50th, 75th, and 95th percentiles of the modeled concentrations were 3.6, 49.6, 194, and 913 µg/L, respectively. A similar distribution of data was found for modeled harmonic mean based 1,1-dichloroethane concentrations. The highly variable estimates are due to variability in the annual facility release amounts and the receiving water body or calculated facility effluent hydrologic flow values.

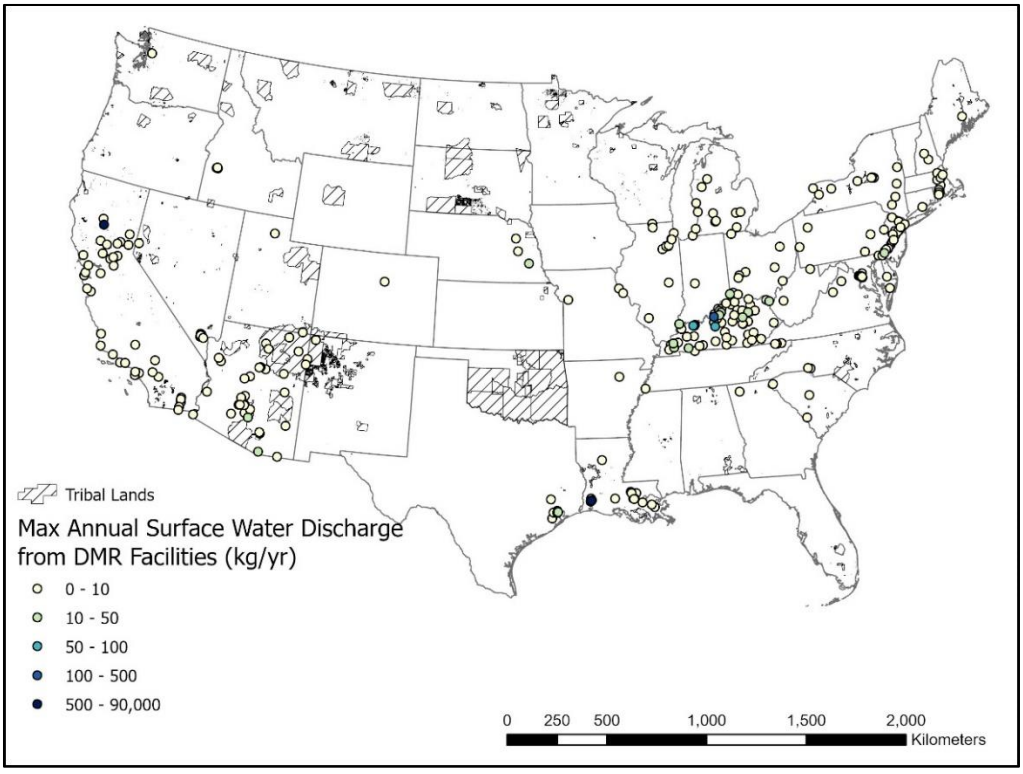


Figure 3-11. Locations of Modeled Estimates of 1,1-Dichloroethane Concentration from Facility Releases to Ambient Surface Waters, 2015–2020

AIANNH tribal boundaries are shaded in gray.

Note: Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown because they do not contain surface water monitoring data within the WQP.

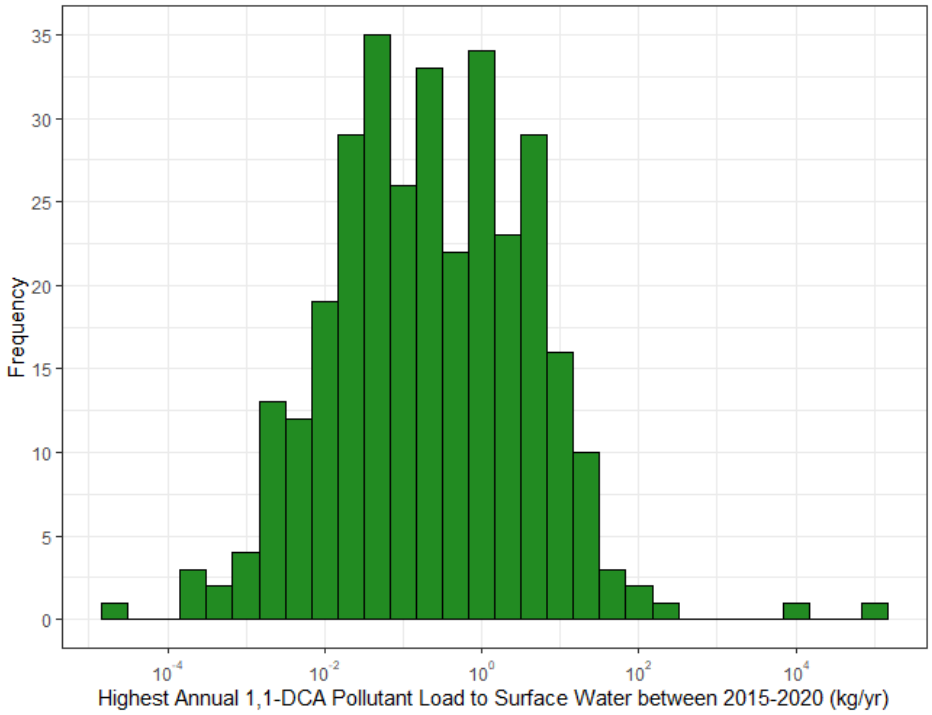


Figure 3-12. Distribution of Highest Facility Annual Releases of 1,1-Dichloroethane to their Receiving Water Body Between 2015–2020

3.3.3.2.3 Lake Charles, Louisiana Surface Water Estimates During Storm Events

EPA generally does not include exposures associated with extreme weather events within the scope of the risk evaluation. However, when specific chemical information is available to the Agency and can provide additional characterization of facility operations and associated exposures, EPA considers this as part of a fact-specific, chemical-specific analysis. The Eagle US 2 LLC – Lake Charles Complex facility submitted 6 years of release data with the largest releases associated with storm events (see Table 3-15). Based on the chemical- and facility- specific data received, EPA considered the exposures associated with these storm events. The Agency is presenting the data that are reflective of the range of releases and corresponding conditions, particularly the frequency of storm events in Louisiana. EPA also considered the 2020 releases resulting from extreme storm events separately and considered the 2016 releases as representative of normal operating conditions.

Table 3-15. Six Years of Eagle US 2 LLC Facility Release Data in Louisiana[a]

Year		Event 1	Event 2	Event 3	Event 4		TOTAL (lb/yr)
2016	Date	12/3–4/2016	8/13–14/2016	4/30/2016	5/1/2016		
	Flow (GPM)	1,042	1,115	42	2,916		
	1,1-Dichloroethane release (lb)	0.55	0	0.02	4.69		**5.26**
			Power Failure	**200 Year Rain**			
		Event 1	**Event 2**	**Event 3**	**Event 4**		
2017	Date	3/29/2017	4/28/2017	5/3/2017	6/21/2017		
	Flow (GPM)	7	76	2764	208		
	1,1-Dichloroethane release (lb)	0.1	0.2	2	0		**2.3**
		Event 1	**Event 2**	**Event 3**	**Event 4**		
2018	Date	10/9/2018	10/15/2018	10/16/2018	10/31/2018		
	Flow (GPM)	1.6	59	144	2		
	1,1-Dichloroethane release (lb)	0.1	1	2	0		**3.1**
		Event 1	**Event 2**	**Event 3**	**Event 4**		
2019	Date	4/4/2019	5/10/2019				
	Flow (GPM)	333	729				
	1,1-dichloroethane release (lb)	0.4	0				**0.4**
			Post-Hurricane Laura		**Hurricane Delta**		
		Event 1	**Event 2**	**Event 3**	**Event 4**	**Event 5**	
2020	Date	4/29/2020	9/21–25/2020	9/28/2020	10/9/2020	10/20/2020	
	Flow (GPM)	7	1651	44	2,640	17	
	1,1-Dichloroethane release (lb)	0	987	16.4	35	0	**1,038.4**
		Winter Storm Uri					
		Event 1	**Event 2**	**Event 3**	**Event 4**		
2021	Date	2/16–17/2021	5/19/2021				
	Flow (GPM)	2,900	430	0	0		
	1,1-Dichloroethane release (lb)	171	4	0	0		**175**

[a] All data provided by Westlake, Eagle2 LLC, April 23, 2023.

EPA estimated the 1,1-dichloroethane surface water concentration resulting from these releases during storm events. The NPDES permit data listed the receiving water body as Bayou Verdine, but during a significant storm it is assumed that the Bayou and the Calcasieu River will flood at their confluence such that the Calcasieu River becomes the major flow at the point of discharge. Thus, EPA used Calcasieu River flow from NHDPlus (12,069 million L/day) to estimate 1,1-dichloroethane concentrations in the receiving water body resulting from the 987 lb released from September 21 to 25, 2020. The corresponding 1,1-dichloroethane surface water concentration was estimated to be 45 µg/L—well below

aquatic concentrations of concern and below concerns for human exposures. This exposure scenario was, therefore, not further quantitatively assessed.

3.3.3.2.4 Model Estimates from Point Source Calculator (PSC)

Industrial Releases to Surface Waters

Of the 319 unique sites releasing 1,1-dichloroethane to surface water, 11 sites' concentrations initially modeled using methodology described in 3.3.3.2.1, exceeded the chronic water column COC (93 µg/L). EPA used PSC for the next tier analysis that utilized physical-chemical and fate properties to estimate partitioning between media and estimate water column, benthic, and sediment concentrations as well as a 21-day release scenario as relevant for the aquatic species exposure analysis (see Section 4.1.2). After estimating the 11 facility water column concentrations again using the PSC, Table 3-16 presents 7 site concentration estimates were identified for environmental assessment (see Section 4.1.2). The receiving water body 7Q10 low hydrologic flow values were applied to these facility releases as a conservative estimate of aquatic species' exposures to 1,1-dichloroethane. One facility, CA0083721 site, was excluded from further analysis because of a data reporting error.

Table 3-16. Results from the PSC, Showing Facility Release Information, 7Q10 Flow Values, and Modeled Chronic Surface Water (Water Column) Concentrations for Ecological Species Exposure

Facility NPDES ID	21-Day Highest Release (kg/day)	7Q10 Flow (MLD)	Surface Water Concentration (µg/L)
LA0000761	5.788	4.051	1,430
KY0022039	3.881	27.334	143
NE0043371	2.368	10.996	218
TX0119792	1.056	4.656	236
CA0064599	0.243	0.416[a]	580
OH0143880	0.025	0.073	312
NV0021067	0.019	0.129	139

[a] For CA0064599, permit reported plant flow was used to estimate surface water concentrations instead of estimated receiving water body 7Q10.

Air Deposition to Surface Waters

Concentrations of 1,1-dichloroethane in surface waters resulting from air deposition were estimated for a small, slow moving, stream scenario using the PSC. The intention was to estimate aquatic water column concentrations resulting from air deposition that represent a conservative scenario, appropriate for a tier-1 screening style evaluation. The highest 95th percentile daily average air deposition rate and associated AERMOD modeled distance for each OES was first identified using the results from Table 3-10. These air deposition rates were then applied to the following scenario in PSC: constant 365 consecutive days-on of release (and deposition) that overlaps entirely with a stream having a 200 m^2 surface area and 200 m^3 volume (40 m length × 5 m width × 1 m depth), and a constant streamflow of 10 m^3/day. The same 1,1-dichloroethane physicochemical properties, biogeochemical parameters, and weather file described in the wastewater discharge analysis was used for the PSC runs. PSC results for the 1- and 21-day average surface water column concentrations were compared with their respective acute (1-day) and chronic (21-day) water column COCs for exposure to aquatic ecological species. The distances between the facility air release sites (*i.e.*, the TRI coordinates) and the nearest neighboring NHD hydrological flowlines were estimated using GIS software to inform whether the highest 95th percentile daily average air deposition rate and associated modeled distance for each OES were reasonably representative to choose. If the PSC-estimated concentrations exceeded their respective acute or chronic COC, but the

distance between the facility release site and nearest neighboring NHD flowline was deemed too far away relative to the AERMOD modeled distance or areal range, a new daily average air deposition rate was chosen based on the distance between the release site and nearest NHD flowline. PSC was then run again using the new deposition rate. Results of the air deposition rates and surface water column concentrations of 1,1-dichloroethane are shown in Table 3-18.

The PSC-simulated 1-day average concentrations of 1,1-dichloroethane in the water column resulting from air deposition of 1,1-dichloroethane from TRI-reported fugitive emissions to the small, slow-moving stream scenario did not exceed the acute water column COC of 8,931; however, an initial 21-day average concentration did exceed the chronic water column COC of 93 µg/L for the Manufacturing of 1,1-dichloroethane as an isolated intermediate OES designation. Under this conservative stream scenario, the air deposition of 1,1-dichloroethane to surface waters from facilities with a Manufacturing of 1,1-dichloroethane as an isolated intermediate OES may result in exposure levels that pose a concern to water-column dwelling ecological species. It is important to note however, that the air deposition rate for this specific manufacturing facility applies to a distance of 10 m from the facility release site. EPA found that the nearest NHD flowline to this facility release site was approximately 340 m away, indicating the scenario modeled is unrealistic and should be further evaluated. The Agency repeated the PSC run using the highest p95 daily average air deposition rate at 100 m (0.003 g/m^2/day), which resulted in a 21-day average water column concentration of 64 µg/L that no longer exceeded its respective chronic COC. Thus, it is more likely that the air deposition of 1,1-dichloroethane to surface waters results in exposure levels that do not pose a concern for ecological species dwelling in the water column.

Table 3-17. Results from the Point Source Calculator, Showing the Highest 95th Percentile Daily Average Air Deposition Rate for the OES of Manufacturing of 1,1-Dichloroethane as an Isolated Intermediate and Modeled Surface Water (Water Column) Concentrations for a 21-Day Chronic Scenario for Ecological Species Exposure 10 m from Releasing Facility of TRI-Reported Fugitive Emissions

OES	Highest 95th Percentile Daily Average Air Deposition (g/m^2/day) [a]	Water Column Concentration (µg/L)
		21-Day Average
Manufacturing of 1,1-dichloroethane as an isolated intermediate	0.0402	791
Processing as a reagent	0.0402	791
Waste handling, disposal, treatment, and recycling	0.000114	2.24

[a] Air deposition rates are estimated using the Ambient Air: Multi-Year Methodology AERMOD TRI as shown in Section 3.3.1.2.2, Table 3-10. The values represent the maximum 95th percentile daily deposition rate at 10 m from the TRI-releasing facility for reported fugitive emissions within the OES.

3.3.3.3 Measured Concentrations in Benthic Pore Water and Sediment

No relevant data on measured concentrations of 1,1-dichloroethane in ambient aquatic benthic pore waters or sediments were found in the WQP for the 2015 to 2020 timeframe. Likewise, no relevant ambient monitoring data on these sample types were collected through EPA's systematic review process.

3.3.3.4 Modeled Concentrations in Benthic Pore Water and Sediment

To assess exposures of 1,1-dichloroethane via industrial releases to ecological species dwelling in the aquatic benthic environment, benthic pore water and bulk sediment concentrations at the facility release sites were modeled using the PSC.

3.3.3.4.1 Benthic Pore Water and Sediment Modeling Methodology

A full description of the modeling approach to estimate concentrations of 1,1-dichloroethane in benthic pore waters and bulk sediment from facility-specific releases can be found in Appendix E.

3.3.3.4.2 Benthic Pore Water and Sediment Modeling Results

Industrial Releases to Benthic Pore Waters and Sediment

Of the 319 unique sites releasing 1,1-dichloroethane to surface water, 3 sites had initially modeled (water column) concentrations that exceeded the acute benthic pore water aquatic COC (8,931 µg/L), but no sites had modeled concentrations that exceeded the chronic benthic pore water aquatic COC (6,800 µg/L. After estimating their benthic porewater concentrations again using the PSC, no PSC-estimated concentrations exceeded the acute benthic porewater COC. For the sites that had initially modeled (water column) concentrations that exceeded the chronic benthic pore water COC, the PSC-modeled estimates of their chronic benthic sediment concentrations did not exceed the benthic chronic sediment COC (2,900 µg/L).

Air Deposition to Benthic Pore Waters and Sediment

EPA did not find that any PSC-simulated estimates of benthic pore water or sediment concentrations exceeded their respective aquatic acute and chronic benthic pore water COCs (8,931 and 6,800 µg/L, respectively) or chronic benthic sediment COC (2,900 µg/kg) (Table 3-18).

Table 3-18. Results from the Point Source Calculator, Showing the Highest 95th Percentile Daily Average Air Deposition Rate per OES, and Modeled Benthic Pore Water and Sediment Concentrations for a 1-Day Acute and 21-Day Chronic Scenario for Ecological Species Exposure

OES	Highest 95th Percentile Daily Average Air Deposition ($g/m^2/day$) [a]	Benthic Pore Water Concentration (µg/L)	Benthic Sediment Concentration (µg/kg)
		21-Day Average	35-Day Average
Manufacturing of 1,1-dichloroethane as an isolated intermediate	0.000736	12.8	19.9
Processing as a reagent	0.0402	700	1,090
Waste handling, disposal, treatment, and recycling	0.000114	1.99	3.08

[a] Air deposition rates are estimated using the Ambient Air: Multi-Year Methodology AERMOD TRI as shown in Section 3.3.1.2.2, Table 3-10. The values represent the maximum 95th percentile daily deposition rate at 10 m from the TRI-releasing facility for reported fugitive emissions within the OES.

3.3.3.5 Measured Concentrations in Drinking Water

Public Water Systems are regulated under the SDWA to enforce common standards for drinking water across the country. Although individual primacy agencies, such as state governments, may require monitoring or impose limits for contaminants beyond those regulated under SDWA, currently there are no national requirements to routinely monitor or limit 1,1-dichloroethane in finished water from PWSs. To assess concentrations in surface water known to be distributed as drinking water, monitoring data collected by PWSs were evaluated. Concentrations of 1,1-dichloroethane found in finished (*i.e.,* treated) drinking water were collected from EPA's published UCMR3 dataset, which includes samples collected

between 2013 to 2015 (U.S. EPA, 2017b). To the extent that it could be determined from the database records, only those PWSs that draw from surface water as their primary source were included for this assessment. Similarly, only treated water that was sent to the distribution system were included. Descriptions of these data retrieval and processing methods are presented in Appendix E.

The UCMR3 dataset from EPA's Final Regulatory Determination 4 Support Document was used to gather concentrations of 1,1-dichloroethane found in finished drinking water from PWSs that draw primarily from surface water sources (U.S. EPA, 2017b). This portion of the UCMR3 dataset includes 21,336 samples from surface water sources from a total of 36,848 samples from all finished water samples collected from 4,916 systems across the United States. 1,1-Dichloroethane was measured above the maximum reporting level (MRL) of 0.03 µg/L in only 2.27 percent of the samples. The maximum concentration of 1,1-dichloroethane measured in finished drinking water from surface water source water was 1.5 µg/L. These results indicate that 1,1-dichloroethane in finished drinking water from PWSs was measured in low levels across the nation between 2013 and 2015.

Two studies that reported concentrations of 1,1-dichloroethane in drinking water for general population locations were found through EPA's systematic review process (see Figure 3-13). Overall, concentrations ranged from not detected (0.035 µg/L) to 367 µg/L from 170 samples collected between 2002 and 2012 in the United States.

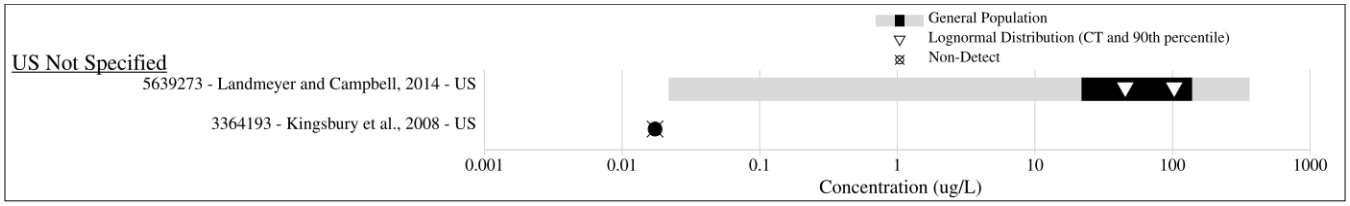


Figure 3-13. Concentrations of 1,1-Dichloroethane (µg/L) in Drinking Water from a U.S.-Based Study, 2002–2012

3.3.3.6 Modeled Concentrations in Drinking Water

To assess general population exposures to 1,1-dichloroethane via industrial releases to surface waters, aqueous concentrations of 1,1-dichloroethane in potential drinking water sources were modeled at PWS intake locations downstream of known 1,1-dichloroethane release sites. Estimates of 1,1-dichloroethane concentrations in drinking water account for upstream-to-downstream dilution and were adjusted for applicable treatment processes that remove of 1,1-dichloroethane in source water.

3.3.3.6.1 Drinking Water Modeling Methodology

To provide more robust estimates of 1,1-dichloroethane concentrations in drinking water, known facility releases were mapped to drinking water sources using PWS data stored in EPA's Safe Drinking Water Information System Federal Data Warehouse (U.S. EPA, 2022e). This dataset is updated quarterly, and the 2nd quarter 2022 version was used for this analysis. Following the mapping, the colocation of and proximity of facility release sites to PWS drinking water intake locations were evaluated. These drinking water data are considered sensitive by EPA's Office of Water and are protected from public release. Geospatial analysis using the NHDPlus V2.1 flowline network was used to determine PWS intake locations within 250 km downstream of facility 1,1-dichloroethane release sites. Provided a PWS may have multiple intake locations, concentrations of 1,1-dichloroethane were estimated at the most upstream intake for a given PWS, thus reflecting a more conservative estimate. Results of surface water concentrations of 1,1-dichloroethane modeled from the highest annual facility releases between 2015 and 2020 for a 1-operating day per year scenario were adjusted by a dilution factor that was calculated

from the change in hydrologic flow between the facility release site and receiving water body associated with the identified PWS intake location. The resulting drinking water source concentration was then adjusted for the removal of 1,1-dichloroethane during the respective PWS treatment processes, if applicable. It is important to note that multiple facility releases can be upstream of the same PWS intake. Estimates of 1,1-dichloroethane concentration in finished drinking water were evaluated independently for each facility-intake linkage. Details of the methodology used for this analysis are provided in Appendix E.

3.3.3.6.2 Drinking Water Modeling Results

Drinking water concentrations of 1,1-dichloroethane were modeled from the highest annual facility releases to surface waters between 2015 to 2020 utilizing a first tier, 1-operating day per year scenario as well as a less conservative facility operating day release scenario. The 1-day release scenario assumes that facility reported annual releases are all discharged in one day and was used as a screening level assessment. The 1-day release assumption was used to represent a worst-case scenario, resulting in the highest possible modeled surface water and therefore, highest drinking water concentrations. The distribution of these results is shown in Figure 3-14.

EPA refined the analysis by dividing the annual release load by the maximum facility operating days. The 16 facility releases and highest corresponding 1,1-dichloroethane drinking water concentration estimates are presented in Table 3-19. Table 3-19 shows for each facility release site, the modeled drinking water concentration at the most upstream intake location of each PWS within 250 km of the release site. Calculated 30Q5 hydrologic flow values were used to estimate the drinking water concentrations shown in Table 3-19, accounting for dilution with changes in the flow values between the facility release site and PWS intake location. Those differences in flow, as well as the distance between the facility release site and PWS intake location modeled, are included. In addition, the population served for each PWS is shown in Table 3-19. This table excludes facility CA0083721 because of an error in the 1,1-dichloroethane wastewater discharge data.

Modeled drinking water concentrations within the top five percent of modeled values ranged between 0.12 µg/L to 1.1 µg/L. As a conservative analysis, low 30Q5 hydrologic flow values were applied. That is, EPA assumed that in the event the downstream flow value was lower than the upstream flow value, the upstream flow value was used in the calculation step and so no adjustment to the amount of dilution was applied. Despite this conservative assumption, the resulting estimates presented in Table 3-19 are similar to the EPA Office of Water measured occurrence data for 1,1-dichloroethane in drinking water (range: 0.03–1.5 µg/L) (U.S. EPA, 2021b).

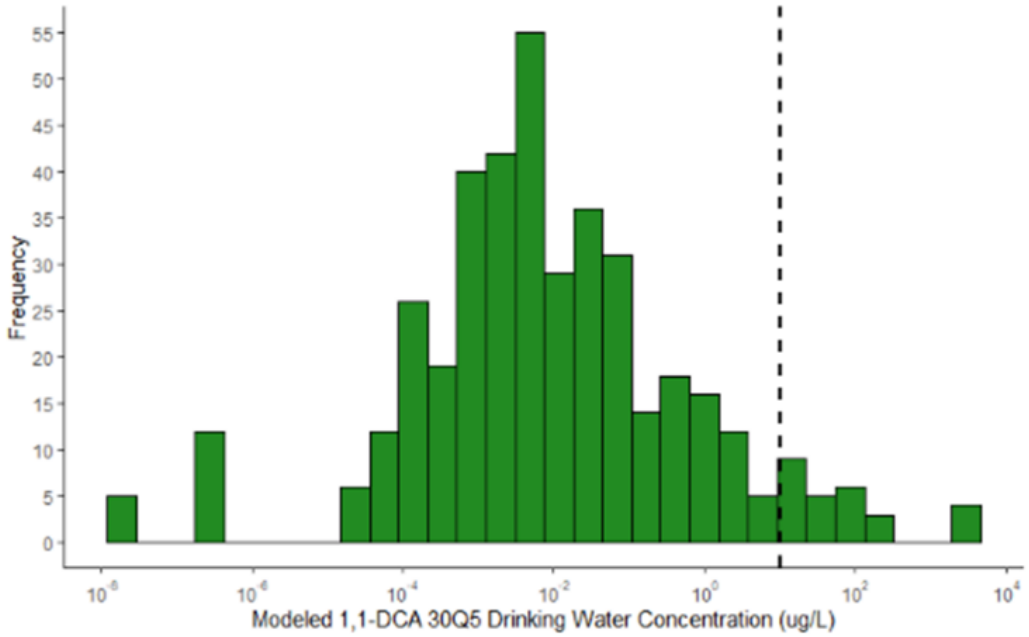


Figure 3-14. Distribution of Drinking Water Concentrations of 1,1-Dichloroethane Modeled from the Highest Annual Facility Releases Between 2015–2022 for a 1 Operating Day per Year Scenario
Estimates of 30Q5 hydrologic flow were used to generate these concentration estimates. The dashed black line indicates concentrations at 10 µg/L.

Table 3-19. Modeled 30Q5 Concentrations of 1,1-Dichloroethane in Drinking Water at PWSs Within 250 km Downstream of a Facility Release Site, Changes in Hydrologic Flow Between the Release Site and PWS Intake Location, as Well as the Population Served by the PWS

Facility NPDES *ID*[a]	PWSID	Facility 30Q5 Flow[b] (MLD)	Intake 30Q5 Flow[b] (MLD)	30Q5 Drinking Water Concentration[c] (µg/L)	Population Served
KY0022039	KY0470175	45	214	1.1	76,326
MI0004057	MI0006101	1.1	0.0	5.2 E−01	9,133
MI0004057	IN5245012	1.1	0.0	5.2 E−01	29,500
CA0048143	CA4210010	20	0.1	5.0 E−01	95,628
CA0048127	CA4210010	12	0.1	5.0 E−01	95,628
CA0022764	CA2110001	43	0.3	2.5 E−01	1,445
CA0048194	CA4410010	30	0.1	2.5 E−01	87,957
CA0048194	CA2710004	30	0.0	2.5 E−01	N/A
CA0048194	CA4000684	30	0.1	2.5 E−01	N/A
AZ0020559	AZ0407093	122	0.2	1.8 E−01	234,766
AZ0020559	AZ0407096	122	0.2	1.8 E−01	135,975
KY0066532	KY1110054	52	297	1.5 E−01	6,165
CA0084271	CA0710003	2.9	0.4	1.4 E−01	198,000
MI0044130	MI0006101	7.5	0.0	1.2 E−01	9,133
MI0044130	IN5245012	7.5	0.0	1.2 E−01	29,500
MI0044130	IN5245020	7.5	0.0	1.2 E−01	78,384

Facility NPDES *ID*[a]	PWSID	Facility 30Q5 Flow[b] (MLD)	Intake 30Q5 Flow[b] (MLD)	30Q5 Drinking Water Concentration[c] (µg/L)	Population Served

[a] Facility data, including NPDES ID, are from DMRs, as reported in the EPA Pollutant Loading Tool.
[b] Modeled hydrologic flow data (*i.e.*, stream flow) are associated with the facility's receiving water body at the point of release (facility 30Q5 flow). The point of drinking water intake (intake 30Q5) was retrieved from the NHDPlus V2.1 dataset (U.S. EPA, 2016c). The receiving water body was identified from NPDES permit information of the releasing facility for the 2015–2020 reporting period.
[c] Modeled 1,1-dichloroethane drinking water concentration is at the point of drinking water facility (public water system) intake. Estimate considers dilution from the point of discharge and does not consider drinking water treatment removal.

3.3.4 Land Pathway (Soils, Groundwater, and Biosolids)

A full description of the modeling approach to estimate concentrations of 1,1-dichloroethane in soils, groundwater and biosolids deposition from facility-specific releases can be found in Appendix G.

3.3.5 Weight of Scientific Evidence Conclusions for Environmental Concentrations

3.3.5.1 Strengths, Limitations, and Sources of Uncertainty in Assessment Results for Monitored and Modeled Concentrations

According to the *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025z), the selection of data and information are informed by the hierarchy of preferences, which considers the use of both measured (monitoring) and estimated (modeled) data. Monitoring data from both published literature and sampling databases provides strong evidence for the presence of 1,1-dichloroethane in ambient air, surface water, and groundwater (see Sections 3.3.1.1, 3.3.3.1, and 3.3.4). EPA modeling of TSCA releases as reported in TRI, NEI and DMR also predicts presence in ambient air and surface water. Facility-reported levels in effluent as required by NPDES permits as well as the corresponding receiving water body (referenced in permit data) and USGS flow data result in very accurate 1,1-dichloroethane concentration estimates at the point of discharge. Fate and physical-chemical properties provide additional context; that is, high water solubility of 1,1-dichloroethane and low potential for hydrolysis are factors that strengthen the evidence of 1,1-dichloroethane presence in water and the volatility of 1,1-dichloroethane and low potential for photolysis provides evidence of its presence in air.

Ambient and Indoor Air Monitored and Modeled Concentrations

EPA modeled air concentrations from TRI and NEI facility releases. The TRI and NEI data are reported by facilities and state/county government entities and provide EPA with data on the level of 1,1-dichloroethane being emitted into ambient air. EPA monitoring of HAPs via the AirToxic monitoring program provides high-quality data for the monitoring location. Therefore, EPA has high confidence in the air concentrations estimates because AERMOD is appropriate for modeling air pollutants such as 1,1-dichloroethane and has been thoroughly peer reviewed. EPA also has high confidence in the estimates because the Agency used reported facility specific TRI and NEI release data as input data for AERMOD modeling. The Agency has high confidence in the deposition concentrations estimated to land and water from TRI and NEI release data using AERMOD. For the full analysis, EPA used releases reported to the TRI and NEI as direct inputs to AERMOD. Furthermore, EPA conducted a multi-year analysis using 6 years of TRI and 2 years of NEI data that strengthens the confidence that all relevant releases of 1,1-dichloroethane were assessed.

EPA has medium confidence in the indoor air concentrations estimated from TRI release data using IIOAC. Indoor air concentrations within IIOAC are calculated by multiplying the modeled ambient air

concentrations by an indoor-outdoor ratio. Per IIOAC Model guidance, EPA used the default indoor-outdoor ratios of 0.65 and 1 for the mean and high-end ratios, respectively. The indoor-outdoor ratio is influenced by many factors including the characteristics of the building such as building footprint and architecture, interior sources or sinks, physical form of the chemical substance (particulate or gas), HVAC system air flow rates, and activity patterns such as how often are windows and doors opened, how the HVAC system is operated. However, in many screening models, the indoor-outdoor ratio is set to a value of one, which represents the upper bound of this ratio if there are no indoor sources, as it is the case for 1,1-dichloroethane.

AERMOD uses 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 (*e.g.,* 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. The TRI reported data used for AERMOD 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/s (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 that 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 m area source for fugitive releases may impact the exposure estimates very near a releasing facility (*i.e.,* 10 m from a fugitive release). This assumption places the 10-meter exposure point just off the release point that may result in either an over or underestimation of exposure depending on other factors like meteorological data, release heights, and plume characteristics. In addition, EPA also used meteorology data for Lake Charles, Louisiana, for the Commercial use as a laboratory chemical and Processing – repackaging OESs where facility data were not available to represent meteorological datasets that tended to provide high-end concentration estimates.

Contrary to the TRI reported data, the NEI reported data used for AERMOD include source-specific stack parameters. Therefore, specific parameter values were used in modeling, when available. When parameters were not available, and/or values were reported outside of normal bounds, reported values were replaced using procedures outlined in Appendix C.3.

AERMOD modeled concentrations of releases from TRI reporting facilities ranged from 0 to 232 $\mu g/m^3$ (Table 3-9) with the maximum modeled concentration being one order of magnitude higher than the maximum monitored concentration of 26 $\mu g/m^3$ (≈97% of the samples were NDs) from AMTIC (Table 3-8). EPA has high confidence in the modeled results representing 1,1-dichloroethane ambient air concentrations because the ranges of the ambient air modeled concentrations from AERMOD are within the ranges of monitored concentrations from AMTIC data.

As an example, Figure 3-15 shows the location of a 1,1-dichloroethane releasing facility as reported in TRI and six AMTIC ambient air monitoring sites located within 10 km of the facility. AERMOD TRI modeled concentrations of 1,1-dichloroethane and the corresponding years of monitoring data are listed in Table 3-20. As shown in Table 3-20, modeled concentrations are within an order of magnitude with the monitored 1,1-dichloroethane concentrations.

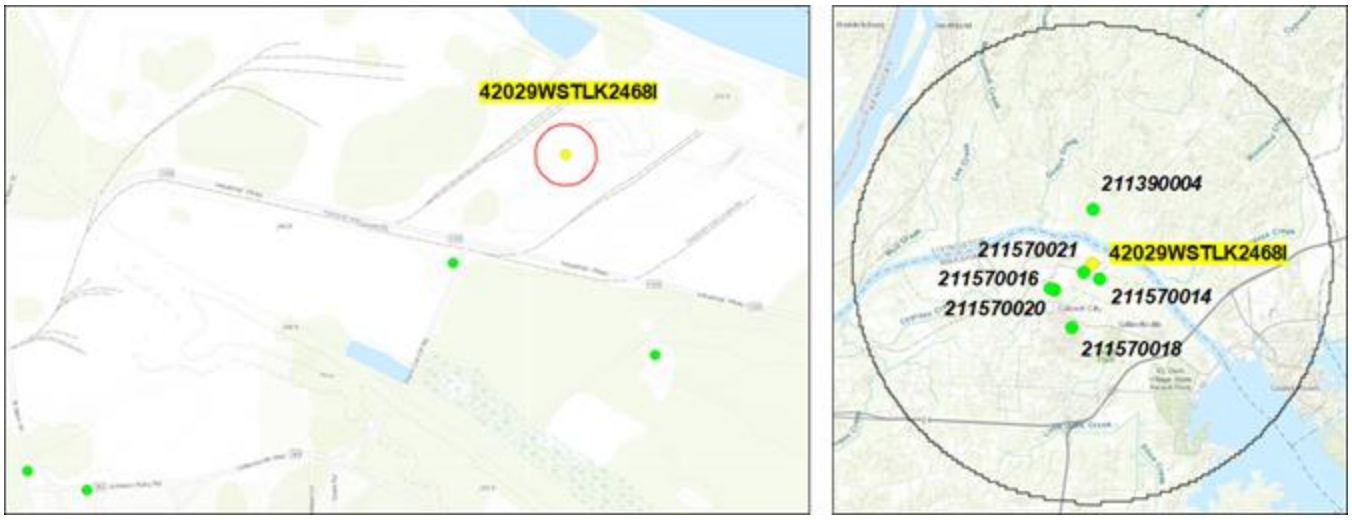


Figure 3-15. Location of TRI Facility (TRI ID 42029WSTLK2468I, Yellow Dot) and AMTIC Monitoring Sites Within 10 km of the TRI Facility (Green Dots)

Table 3-20. Comparison of 1,1-Dichloroethane AERMOD Modeled Concentrations for a TRI Facility with 1,1-Dichloroethane Ambient Air Monitoring Data from Six AMTIC Monitoring Sites Within 10 km of the Facility from 2015 to 2020[a]

Facility TRI ID	Year	Lowest P95 Modeled Daily Concentration (ppb)	Max 1 Day Monitoring Concentration (ppb)	Distance from TRI Reporting Facility to Monitoring Site (m)	Difference Between Modeled and Monitored Concentrations
42029WSTLK2468I	2015	0.212	0.097	2,268	0.115
42029WSTLK2468I	2015	0.212	0.063	719	0.149
42029WSTLK2468I	2015	0.212	0.013	2,049	0.199
42029WSTLK2468I	2016	0.221	0.109	2,268	0.112
42029WSTLK2468I	2016	0.221	0.274	719	−0.053
42029WSTLK2468I	2016	0.221	0.228	2,049	−0.007
42029WSTLK2468I	2017	0.228	0.091	2,268	0.137
42029WSTLK2468I	2017	0.228	0.183	719	0.045
42029WSTLK2468I	2018	0.291	0.268	2,268	0.023
42029WSTLK2468I	2018	0.291	0.206	719	0.085
42029WSTLK2468I	2019	0.132	0.028	2,268	0.104
42029WSTLK2468I	2019	0.132	0.123	719	0.009
42029WSTLK2468I	2020	0.157	0.013	2,813	0.144
42029WSTLK2468I	2020	0.157	0.054	1,919	0.103
42029WSTLK2468I	2020	0.157	0.361	513	−0.204

[a] A facility in Calvert City, Kentucky, reported 1,1-dichloroethane releases to TRI that were modeled to estimate ambient air concentrations of 1,1-dichloroethane using AERMOD. Modeled 95th percentile average daily concentrations were calculated and compared to the maximum 1-day 1,1-dichloroethane concentrations at the closest monitoring station to the TRI facility. Modeled concentrations are within an order of magnitude with the monitored 1,1-dichloroethane concentrations.

AERMOD was used to model daily ($g/m^2/day$) and annual ($g/m^2/year$) deposition rates from air to land and water from each TRI and NEI releasing facility. Based on physical and chemical properties of 1,1-dichloroethane (Section 2.1), EPA considered only gaseous deposition. The Agency used chemical-specific parameters as input values for AERMOD deposition modeling. Three of the chemical-specific parameters (diffusivity in air, diffusivity in water, and cuticular resistance) used for deposition modeling were obtained outside of the systematic review process used for obtaining other physical and chemical properties; therefore, EPA has moderate confidence in these values. Thus, the Agency has moderate confidence in the deposition rates estimated from TRI and NEI release data using AERMOD.

Surface and Drinking Water Monitored and Modeled Concentrations

Unlike the example given above correlating ambient air modeling/monitoring, the available measured surface water concentration data are poorly co-located with 1,1-dichloroethane facility release sites. EPA relied primarily on modeling to estimate aqueous concentrations resulting from releases to surface waters as reported in the EPA Pollutant Loading Tool. The tool compiles and makes public discharges as reported in DMRs required in NPDES permits and provides data on the amount of 1,1-dichloroethane in discharged effluent and the receiving water body. EPA assesses the overall confidence of estimated releases for various OESs. For those OESs releasing to surface water, confidence is rated as robust for releases as reported in DMR.

The modeling used, as well as the associated default and user-selected inputs, can affect the overall strength in evaluating exposures to the general population. The facility-specific releases methodology described in Section 3.2.1, and the results in 3.3.3.2.2 rely on a modeling framework that does not consider downstream fate. Drinking water estimates do account for downstream transport and treatment removal processes, while concentration estimates to evaluate exposure to ecological species account for key source/sink fate processes at the facility release site. To reduce uncertainties, EPA incorporated an updated hydrologic flow network and flow data into this assessment that allowed a more site-specific consideration of release location and associated receiving water body flows. However, these releases are evaluated on a per facility basis that do not account for additional sources of 1,1-dichloroethane that may be present in the evaluated waterways. Finally, drinking water exposures are not likely to occur from the receiving water body at the point of facility-specific releases. Specifically, the direct receiving water bodies may or may not be used as drinking water sources. To address this limitation, EPA evaluated the proximity of known 1,1-dichloroethane releases to known drinking water sources as well as known drinking water intakes as described in Section 3.3.3.6.

The measured data encompassed both ambient surface water monitoring as well as drinking water system monitoring data. For ambient surface water, data are 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 near modeled releases were rare, often making direct comparisons of modeled results unavailable. In most cases, monitoring data represented water bodies without identified releases of 1,1-dichloroethane nearby. To an extent, monitoring data in finished drinking water data provided a comparison for the low-range of modeled concentrations at individual PWS, though it is important to recognize that even this comparison is weak given the poor temporal alignment between modeled and measured concentrations of 1,1-dichloroethane in drinking water.

At the higher end, the modeled surface water concentrations of 1,1-dichloroethane from facility releases are several orders of magnitude greater than those observed in the 1,1-dichloroethane monitoring data (Figure 3-8). All measured concentrations in surface waters acquired from the WQP fall below 2 μg/L, with 95 percent of the concentrations below 0.5 μg/L. In comparison, the median of 1,1-dichloroethane concentrations in surface waters (based on 30Q5 hydrologic values) was approximately 50 μg/L.

Validation of facility-specific 1,1-dichloroethane surface water concentration estimates is not available as EPA did not identify monitoring data associated spatially and temporally to facility-specific releases.

There are a few reasons that can help explain why higher aqueous concentrations of 1,1-dichloroethane were modeled in comparison to those that have been observed from measured samples. The locations where measurements were taken could have been collected further downstream or on-stream segments not impaired by facility releases of 1,1-dichloroethane. In addition, many of the facilities release into very small streams or industrial canals, which can elevate modeled concentration at the point of release when release amounts are high. As this water travels downstream, it is expected to eventually join with larger water bodies, where some decrease in concentration due to dilution would be expected to occur.

Measured concentrations of 1,1-dichloroethane in finished drinking water from the UCMR3 and state database were compared to 30Q5-based model estimates for individual PWSs where co-located data were available. It is important to note, however, both the timing and location of release and sample collection must align to make a true comparison of the modeled versus measured results. Thus, the comparison described herein provides a broader sense of agreement. For the low range of modeled drinking water estimates (<1–5 μg/L), there was a strong agreement with measured data from UCMR3 data, provided these results were all less than 1 μg/L.

To further refine the possible distribution and concentrations of 1,1-dichloroethane between water column, benthic pore water and sediment, EPA used the PSC to estimate 1,1-dichloroethane concentrations in the corresponding media resulting from TSCA releases. PSC is a thoroughly reviewed model and is an appropriate tool for soluble chemicals such as 1,1-dichloroethane. Because EPA used chemical-specific physical-chemical properties and facility-specific releases as input data, the Agency is confident in the tool's ability to estimate 1,1-dichloroethane concentrations in the corresponding media. In addition, estimates of water column concentrations and surface water concentrations are closely aligned, demonstrating that PSC is an appropriate tool for 1,1-dichloroethane concentration estimates in aqueous environments. Benthic pore water and sediment concentrations of 1,1-dichloroethane were estimated using physical and chemical properties such as log K_{OC}, a measure of chemical adsorption to organic materials such as sediment or soils. EPA has robust confidence in estimates of 1,1-dichloroethane concentrations in benthic pore water and sediments.

Land Pathway (Soils, Groundwater, and Biosolids)

Current reported releases to landfills are not anticipated to result in any measurable 1,1-dichloroethane groundwater concentrations. Uncertainties and limitations are inherent in the modeling of groundwater concentrations from disposing chemical substances into poorly managed RCRA Subtitle D landfills as well as those that are not regulated as closely. These uncertainties include, but are not limited to, (1) determining the total and leachable concentrations of waste constituents, (2) estimating the release of pollutants from the waste management units to the environment, and (3) estimating and 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. The DRAS Model is 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.

Based on NEI data, 1,1-dichloroethane is reported to be emitted from several landfills, which also report methane as an indicator of anaerobic activity and degradation. Those landfills reporting measured anaerobic activity presumably emit 1,1-dichloroethane as an anaerobic degradant of 1,1,1-trichloroethane-containing materials disposed in landfills. EPA therefore has moderate confidence in estimates of 1,1-dichloroethane in groundwater from TSCA releases.

EPA did estimate additional possible media for 1,1-dichloroethane exposures, specifically via air deposition from air releases and releases from POTWs via land application of biosolids. These media concentrations are further used for ecological species exposure estimates (Section 4.1.3) and for limited general population exposures (Appendix G). Given the lack of soil and biosolids monitoring data, and the reliance on estimates based on reported releases and assumptions of POTW biosolids use in land application, EPA is not highly confident in the quantitative estimates of 1,1-dichloroethane in biosolids/soils.

Appendix Q presents a summary of the weight of scientific evidence conclusions for each of the media concentrations considered in environmental and human exposures to 1,1-dichloroethane. Evidence for 1,1-dichloroethane presence in each media is most dependent on the releases reported in TRI and NEI for ambient air, TRI and DMR for surface water, and TRI for releases to land. The confidence in these releases is reported in Table 3-7 and presented in Appendix Q.

4 ENVIRONMENTAL RISK ASSESSMENT

Environmental Exposures (Section 4.1):
Key Points

EPA evaluated the reasonably available information for environmental exposures of 1,1-dichloroethane to aquatic and terrestrial species. The key points of the environmental exposure assessment are summarized below:

- EPA expects the main environmental exposure pathways for 1,1-dichloroethane to be surface water and air. The ambient air exposure pathway was assessed for its contribution via deposition to soil.
- 1,1-Dichloroethane exposure to aquatic species through surface water and sediment were modeled to estimate concentrations near industrial and commercial uses.
  - Modeled data based on number of operating days per year estimate surface water concentrations range from 0.7 to 85 µg/L, benthic pore water concentrations range from 0.55 to 78 µg/L, and sediment concentrations range from 0.85 to 124 µg/kg from facility releases to surface waters.
  - EPA also estimated fish tissue and crayfish tissue concentrations by COU using the modeled water releases from industrial uses.
- 1,1-Dichloroethane exposure to terrestrial species through soil, surface water, and sediment was also assessed using modeled data. These data are available in Appendix J.

EPA assessed environmental risks of 1,1-dichloroethane exposure to aquatic and terrestrial species. Section 4.1 describes the environmental exposures through surface water, sediment, soil, air, and diet via trophic transfer. Environmental hazards for aquatic and terrestrial species are described in Section 4.2, while environmental risk is described in Section 4.3.

4.1 Environmental Exposures

4.1.1 Approach and Methodology

The major environmental compartments for 1,1-dichloroethane exposures to ecological receptors are surface water and air (see Section 2.2.2). EPA assessed 1,1-dichloroethane exposures via surface water, sediment, soil, and air, which were used to determine risks to aquatic and terrestrial species (see Section 4.3). Ambient air is assessed for its contribution via deposition to soil.

EPA used two models, PSC and AERMOD, to assess the environmental concentrations resulting from the industrial and commercial release estimates (Section 3.2). Additional information on these models is available in Section 3.3. EPA modeled 1,1-dichloroethane surface water, benthic pore water, and sediment concentrations using PSC as described in Section 3.3. EPA modeled 1,1-dichloroethane concentrations in soil via air deposition near facility (10 m from the source) as described in Appendix G.1.1. The distance of 10 m from source was selected as the most conservative scenario because the highest concentrations occurred at this distance. Modeled surface water, sediment, and benthic pore water concentrations were used to assess 1,1-dichloroethane exposures to aquatic species.

EPA used calculated soil concentrations to assess risk to terrestrial species via trophic transfer (see Section 4.1.3). Specifically, EPA based trophic transfer of 1,1-dichloroethane and potential risk to terrestrial animals on modeled air deposition to soil from AERMOD as well as estimated biosolids land application. Potential risk to aquatic dependent wildlife used surface water and benthic pore water

concentrations modeled via PSC for each COU in combination with 1,1-dichloroethane fish and crayfish concentrations, respectively, using the estimated BCFs shown in Table 2-2. Exposure factors for terrestrial organisms used within the trophic transfer analyses are presented in Section 4.1.3. Application of exposure factors and hazard values for organisms at different trophic levels is detailed within Section 4.3 and used equations described in the *U.S. EPA Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a).

4.1.2 Exposures to Aquatic Species

4.1.2.1 Measured Concentrations in Aquatic Species

There are very limited data available on 1,1-dichloroethane concentrations in fish or other aquatic biota. Only one study was identified where 1,1-dichloroethane was detected, in oysters in Lake Pontchartrain (33 ng/g) (Ferrario et al., 1985). Other similar chlorinated solvents, including 1,1,1-trichloroethane, 1,2-dichloroethane, and trichloroethylene, reported concentrations in bivalves between 0.6 and 310 ng/g. (Gotoh et al., 1992; Ferrario et al., 1985). No reasonably available data on 1,1-dichloroethane concentrations in fish tissue were identified; however, data in fish muscle and liver tissue for other chlorinated solvents ranged from 0.51 to 4.89 ng/g for 1,1,1-trichloroethane and 0.36 to 29.3 ng/g trichloroethylene (Roose and Brinkman, 1998). Therefore, 1,1-dichloroethane concentrations in fish and crayfish were calculated as described below to estimate exposure.

4.1.2.2 Calculated Concentrations in Aquatic Species

EPA used PSC to estimate maximum daily average 1,1-dichloroethane surface water, benthic pore water, and sediment concentrations as described in Section 3.3.3.2 and Section 3.3.3.4. The days of exceedance modeled in PSC are not necessarily consecutive and could occur throughout a year at different times. Days of exceedance is calculated as the probability of exceedance multiplied by the total modeled days of release as described in Appendix J.1.

EPA calculated 1,1-dichloroethane concentrations in fish and crayfish for each industrial and commercial release scenario (see Table_Apx J-6 and Table_Apx J-7). The highest calculated concentrations of 1,1-dichloroethane in fish and crayfish were 590 ng/g and 550 ng/g, respectively, for the Manufacturing of 1,1-dichloroethane as an isolated intermediate OES, with the lowest calculated concentrations as 4.5 and 3.8 ng/g for fish and crayfish, respectively for the OES commercial use as a laboratory chemical. These calculated concentrations are similar to the 1,1-dichloroethane concentration reported in oysters (Ferrario et al., 1985) and the highest reported concentrations of other chlorinated solvents in fish tissues (Roose and Brinkman, 1998). Concentrations of 1,1-dichloroethane in fish were calculated by multiplying the maximum PSC modeled surface water concentrations based on the number of operating days per year for each industrial and commercial release scenario (Table 3-3) by the EPI Suite™ generated BCF of 7 (Table 2-2). Similarly, concentrations of 1,1-dichloroethane in crayfish were calculated by multiplying the maximum PSC-modeled benthic pore water concentrations based on the number of operating days per year for each industrial and commercial release scenario (Table 3-3) by the estimated BCF. These whole fish and crayfish 1,1-dichloroethane concentrations were utilized within the screening level assessment for trophic transfer described in Section 4.1.4.

4.1.3 Exposures to Terrestrial Species

No reasonably available measured data on 1,1-dichloroethane concentrations in terrestrial biota were identified. Modeled concentrations were used to assess 1,1-dichloroethane exposures to terrestrial mammals and birds through diet and indirect ingestion. These concentrations are available in Appendix J.2.

4.1.4 Trophic Transfer Exposure

Trophic Transfer is the process by which chemical contaminants can be taken up by organisms through dietary and media exposures and be transferred from one trophic level to another. Representative species were chosen at each trophic level for both terrestrial and aquatic pathways. Details on these species and the resulting concentrations are available in Appendix J.3.

4.1.5 Weight of Scientific Evidence Conclusions for Environmental Exposures

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

EPA used a combination of chemical-specific parameters and generic default parameters when estimating surface water, sediment, soil, and fish-tissue concentrations.

Concentrations of 1,1-dichloroethane in environmental media are expected to vary by exposure scenario. Release from industrial facilities, either by water or air, contribute to concentrations of 1,1-dichloroethane in the environment. Proximity to facilities and other sources is likely to lead to elevated concentrations via air deposition compared to locations that are more remote. The ability to locate releases by location reduces uncertainty in assumptions when selecting model input parameters that are typically informed by location (*e.g.*, meteorological data, land cover parameters for air modeling, flow data for water modeling).

Measured surface water monitoring data for 1,1-dichloroethane is available but does not generally align well either geographically or temporally with modeled releases. In most cases, comparison between measured and modeled surface water concentrations was not possible. Environmental exposures of aquatic invertebrates, vertebrates, and plants to 1,1-dichloroethane were assessed using modeled surface water, benthic pore water, and sediment concentrations resulting from 1,1-dichloroethane releases to surface water (Section 3.3.3.2) using site-specific information such as flow data for the receiving water body at a release location. The confidence in the estimated surface water, benthic pore water, and sediment concentrations resulting from surface water releases is characterized as Robust. For additional details see Section 3.3.5.1.

Neither 1,1-dichloroethane soil monitoring data reflecting releases to air and deposition to soil or reflecting releases to soil via land application of biosolids were found for comparison to modeled concentration estimates. Environmental exposures of soil invertebrates, terrestrial plants, and mammals to 1,1-dichloroethane were assessed using modeled air deposition of 1,1-dichloroethane releases to soil (Appendix G.1.1) and estimation of resulting bulk soil and soil porewater concentrations using conservative assumptions regarding persistence and mobility. Exposure of these receptors via land application of biosolids was assessed using modeled biosolids concentrations, both screening level calculations, modeling, and similar conservative assumptions (see Appendix G.1 for details). Although the screening level models and methods used to estimate soil concentrations from air deposition and land application of biosolids are scientifically sound and largely peer reviewed, some key inputs such as the concentration of 1,1-dichloroethane in land applied biosolids and biosolids land application practices are highly variable or unknown. Thus, the confidence in the estimated soil concentrations resulting from land application of biosolids is characterized as Moderate.

4.2 Environmental Hazards

1,1-Dichloroethane – Environmental Hazards (Section 4.2):
Key Points

EPA evaluated the reasonably available information for environmental hazard endpoints associated with 1,1-dichloroethane exposure. The key points of the environmental hazard assessment are summarized below:

- Aquatic species hazard:
  - Few empirical data were reasonably available on aquatic species for 1,1-dichloroethane; therefore, EPA used analog data and predictions to supplement the data for hazard characterization.
  - To estimate aquatic and benthic hazards (mortality) from acute exposures, EPA supplemented empirical data on 1,1-dichloroethane with an identified analog, 1,2-dichloropropane. Data from 1,2-dichloroproane were used to generate hazard predictions from an EPA predictive tool, Web-based Interspecies Correlation Estimation (Web-ICE). These data were used with the empirical aquatic and benthic invertebrate and fish data to create a species sensitivity distribution (SSD) and calculate a concentration of concern (COC) for acute exposures of aquatic species (1,769 ppb) using the lower 95th percentile of an HC05, a hazardous concentration threshold for 5 percent of species.
  - EPA also calculated a COC for chronic exposures (reproduction in *Daphnia magna*) to aquatic species (93 ppb) using empirical 1,1-dichloroethane data.
  - EPA calculated two COCs for chronic exposures in benthic pore water and sediment to benthic-dwelling species (reproduction of *Ophryotrocha labronica* and growth and development of *Chironomus riparius*, 6,800 ppb in benthic pore water and 2,900 µg/kg in sediment, respectively) using empirical, sediment-dwelling invertebrate data on a close analog, 1,1,2-trichloroethane.
  - EPA also calculated an algal COC for exposures (growth of *Skeletonema costatum*) to aquatic plants (1,000 ppb) using empirical 1,2-dichloropropane data on algae.
- Terrestrial species hazard:
  - Terrestrial hazard data for 1,1-dichloroethane were available for plants and mammals.
  - Based on empirical toxicity data for Canadian poplar, the chronic hazard threshold for terrestrial plants is 802 mg/kg soil.
  - Empirical toxicity data for mice and rats were used to estimate a chronic toxicity reference value (TRV) for terrestrial mammals of 1,189 mg/kg-bw/day.

4.2.1 Approach and Methodology

During scoping, EPA reviewed potential environmental hazards associated with 1,1-dichloroethane and identified the eight sources of environmental hazard data shown in Figure 2-9 of the final scope document (U.S. EPA, 2020b).

EPA completed the review of environmental hazard data/information sources during risk evaluation using the data quality review evaluation metrics and the rating criteria described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025z). Studies were assigned an overall quality of high, medium, low, or uninformative.

EPA assigned overall quality determinations of high or medium to six acceptable aquatic toxicity studies, including data generated from a 1,1-dichloroethane TSCA section 4(a)(2) test order, and three acceptable terrestrial toxicity studies. There were few aquatic toxicity data for 1,1-dichloroethane, so the Agency also used environmental hazard information for the analog 1,2-dichloropropane in a read-across to 1,1-dichloroethane. 1,2-Dichloropropane was selected as an analog for 1,1-dichloroethane aquatic hazard read-across due to similar structure, physical, chemical, and environmental fate and transport, and toxicity. Because no chronic benthic hazard data were identified for 1,1-dichloroethane or analog 1,2-dichloropropane, chronic benthic hazard data from a second analog 1,1,2-trichloroethane were used to read-across to 1,1-dichloroethane. Although 1,1,2-trichloroethane was not considered as robust an analog as 1,2-dichloropropane for read-across of certain aquatic hazard (*e.g.*, algal hazard), 1,1,2-trichloroethane was considered a sufficient analog for a targeted read-across of chronic benthic hazard to 1,1-dichloroethane. See Section 4.2.1.1 for the analog selection rationale.

EPA identified eight sources of environmental hazard analog data, including six sources shown in Figure 2-9 of *Final Scope of the Risk Evaluation for 1,2-Dichloropropane; CASRN 78-87-5* (U.S. EPA, 2020f) to assess hazard to aquatic species, and two sources shown either in Figure 2-9 of *Final Scope of the Risk Evaluation for 1,1,2-Trichloroethane; CASRN 79-00-5* (U.S. EPA, 2020d) or generated from a 1,1,2-trichloroethane section 4(a)(2) test order (Smithers, 2023) to assess hazards to benthic species under chronic exposure duration. Studies on the analogs were also reviewed and assigned an overall quality of high, medium, low, or uninformative, and only those assigned medium- or high-quality were used in the read-across. In lieu of terrestrial wildlife studies, controlled laboratory studies that used mice and rats as human health model organisms were used to calculate a toxicity reference value (TRV), which is expressed as doses in units of mg/kg-bw/day. Although the TRV for 1,1-dichloroethane is derived from laboratory mice and rat studies, because body weight is normalized, the TRV can be used with ecologically relevant wildlife species to evaluate chronic dietary exposure to 1,1-dichloroethane. Chronic hazard thresholds for representative wildlife species are evaluated in the trophic transfer assessments using the TRV (Section 4.2.5.2).

4.2.1.1 Analog Selection for Environmental Hazard

Few 1,1-dichloroethane environmental hazard data were identified for aquatic and benthic invertebrates, fish, and algae, and no 1,1-dichloroethane environmental hazard data were identified for earthworms. Analog selection was performed to identify an appropriate analog to read-across to 1,1-dichloroethane. A tiered approach was used to select analogs for read-across to 1,1-dichloroethane environmental hazard (Figure 4-1). 1,2-Dichloropropane was selected as an analog for read-across of aquatic hazard data to supplement the 1,1-dichloroethane aquatic hazard based on structural similarity, physical and chemical similarity, ecotoxicological similarity in aquatic taxa, and availability of 1,2-dichloropropane aquatic hazard data from data sources that received ratings of either high or medium (Figure 4-1). Strengths and weakness of the analog selection are described in Section 4.2.1.1.4. Ability to read-across sediment invertebrate and soil invertebrate hazard to 1,1-dichloroethane was inferred from similarities in structure, physical chemical and fate properties relevant to soil, and ecotoxicological behavior in benthic and aquatic invertebrates between 1,1-dichloroethane and its analogs. No chronic benthic hazard data were reasonably available for 1,1-dichloroethane or its primary analog, 1,2-dichloropropane; therefore, 1,1,2-trichloroethane was selected as an analog for read-across of chronic benthic environmental hazard to 1,1-dichloroethane based on structural similarity, physical and chemical similarity, ecotoxicological similarity, and availability of 1,1,2-trichloroethane chronic benthic hazard data from data sources receiving a high or medium rating (Figure 4-1). Although this comparison also indicated that 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane soil invertebrate hazard data would be suitable for read-across to 1,1-dichloroethane (Table 4-6), no reasonably available analog data were

identified that could be used quantitatively to derive a soil invertebrate hazard threshold. The similarities between 1,1-dichloroethane and its analogs are described in detail below.

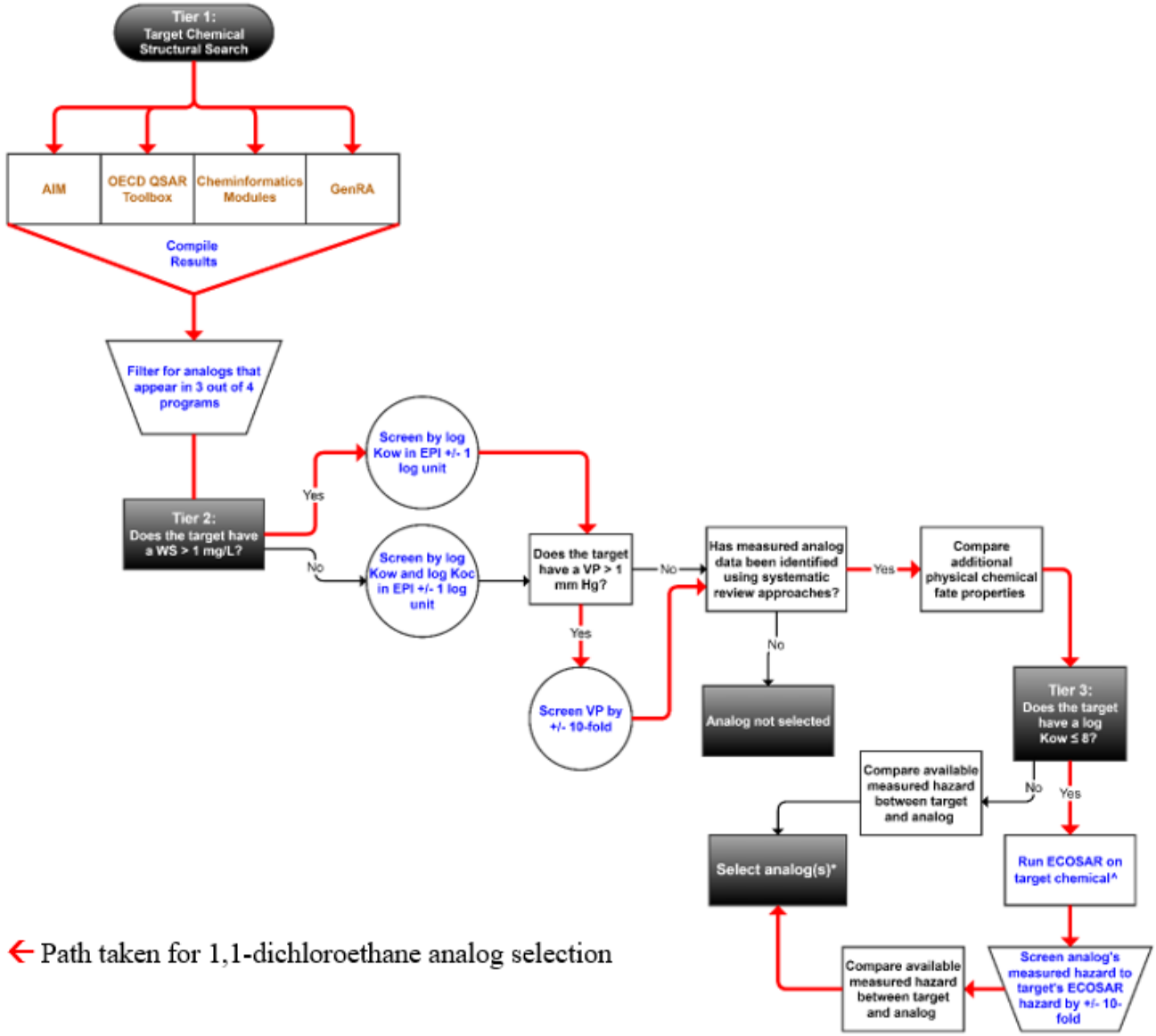


Figure 4-1. Framework for 1,1-Dichloroethane Environmental Hazard Analog Selection
ECOSAR acute and chronic toxicity predictions for vertebrates and invertebrates generated for chemicals with log $K_{OW} \leq 5$ and chronic toxicity predictions generated if log $K_{OW} \leq 8$, and algal toxicity predictions generated if log $K_{OW} \leq 6.4$ should the chemical meet the definition of an ECOSAR class.
*Weight of scientific evidence and professional judgement involved in finalizing selection.

4.2.1.1.1 Structural Similarity

Structural similarity between 1,1-dichloroethane and candidate analogs was assessed using two TSCA NAMs (the Analog Identification Methodology [AIM] program and the Organisation of Economic Cooperative Development Quantitative Structure Activity Relationship [OECD QSAR] Toolbox) and two EPA Office of Research and Development (ORD) products (Generalized Read-Across [GenRA; accessed June 16, 2025] and the Search Module within the Cheminformatics Modules [accessed June 16, 2025]), as shown in Table 4-1 and Figure 4-1. These four programs provide complementary methods of assessing structural similarity. There are several different methods for determining structural

similarity. A fragment-based approach (*e.g.*, as implemented by AIM) searches for compounds with similar structural moieties or functional groups. EPA's TSCA New Chemicals Program utilizes the Confidential Business Information (CBI) version of AIM to identify analogs with data (including analogs with CBI; however, no analogs with CBI were included in the environmental or human health hazard analog selection for 1,1-dichloroethane). Analogs with CBI are not found in the public-facing version of AIM in order to protect business confidentiality, and CBI-AIM has undergone updates not found in the public-facing version of AIM. Therefore, CBI-AIM can provide a more robust list of analogs—including analogs without CBI and has undergone updates not found in the public-facing version of AIM with the latest applied to the non-CBI-AIM platform in 2012 and CBI-AIM update in 2016 in relation to data sources from other EPA programs.

A structural identifier approach (*e.g.*, the Tanimoto coefficient) calculates a similarity coefficient based on molecular fingerprinting (Belford, 2023). However, although a CBI version of AIM was used, no CBI analogs were found in the analog selection for 1,1-dichloroethane that differed when compared to those found in the non-CBI version of the AIM tool. Additionally, AIM was used as an initial screening tool for analog identification that was complemented by the additional tools outlined in the analog selection framework in Section 4.2.1.1 . As AIM is a tiered set of search approaches for identifying analogs based on measured data, a 1st and 2nd pass was applied to the search criteria. A 1st pass is an initial more stringent search criteria in which chemicals are selected when all fragments and atoms in the query chemical are contained in the analog identified (1:1 match) and is the default search if no additional pass is applied. During the 2nd pass, many of the larger molecular fragments identified in the 1st pass that specify the orientation of the atoms are not part of this search and are more inclusive of additional analog candidates.

Molecular fingerprinting approaches look at similarity in atomic pathway radius between the analog and target chemical substance (*e.g.*, Morgan fingerprint in GenRA that calculates a Jaccard similarity index). Some fingerprints may be better suited for certain characteristics and chemical classes. For example, substructure fingerprints like PubChem fingerprints perform best for small molecules such as drugs, while atom-pair fingerprints, which assigns values for each atom within a molecule and thus computes atom pairs based on these values, are preferable for large molecules. Some tools implement multiple methods for determining similarity. Regarding programs that generate indices, it has been noted that because the similarity value is dependent on the method applied, that these values should form a line of evidence rather than be utilized definitively (Pestana et al., 2021; Mellor et al., 2019).

AIM analogs were obtained using the CBI version of AIM and described as 1st or 2nd pass (only analogs not considered CBI are included in Table 4-2). Tanimoto-based PubChem fingerprints were obtained in the OECD QSAR Toolbox (v4.4.1, 2020) using the Structure Similarity option and are presented as a range. Chemical Morgan Fingerprint scores were obtained in GenRA (v3.1) (limit of 100 analogs, no ToxRef filter). Tanimoto scores were obtained in the Cheminformatics Search Module using Similar analysis comparing Tanimoto scores. AIM 1st and 2nd pass analogs were compiled with the top 100 analogs with indices greater than 0.5 generated from the OECD QSAR Toolbox and the Cheminformatics Search Module and indices greater than 0.1 generated from GenRA. These filtering criteria are displayed in Table 4-1. Analogs that appeared in three out of four programs were identified as potential analog candidates (Figure 4-1). Using these parameters, 17 analogs were identified as potentially suitable analog candidates for 1,1-dichloroethane based on structural similarity (Table 4-2). The results for structural comparison of 1,1-dichloroethane to 1,2-dichloropropane (CASRN 78-87-5), 1,1,2-trichloroethane (CASRN 79-00-5), and 1,2-dichloroethane (CASRN 107-06-2) are further described below due to those analog candidates having completed data evaluation and extraction.

Table 4-1. Structure Program Filtering Criteria

Program	Index	Filtering Parameters
Analog Identification Methodology (AIM)	Fragment-based	1st or 2nd pass
OECD QSAR Toolbox	Tanimoto-based PubChem fingerprints	Top 100 analogs ≥ 0.5
Cheminformatics Search Module	Similarity-type: Tanimoto	Top 100 analogs with index ≥ 0.5
GenRA	Morgan Fingerprints	Top 100 analogs with index ≥ 0.1 (ToxRef data filter off)

1,2-Dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane were indicated as structurally similar to 1,1-dichloroethane in AIM (analogs were 2nd pass), OECD QSAR Toolbox (PubChem features = 0.7–0.8). 1,1,2-trichloroethane and 1,2-dichloroethane were indicated as structurally similar to 1,1-dichloroethane in the Cheminformatics Search Module (Tanimoto coefficient 0.63–0.78), and 1,2-dichloropropane was indicated as structurally similar to 1,1-dichloroethane in GenRA (Morgan Fingerprint = 0.45) (Table 4-2). The structural similarity of 1,1-dichloroethane to its analogs indicated in these tools supported the selection of 1,2-dichloropropane and 1,1,2-trichloroethane in the read-across to 1,1-dichloroethane aquatic and benthic environmental hazard. 1,2-Dichloropropane and 1,1,2-trichloroethane were ultimately selected for read-across of aquatic and benthic hazard to 1,1-dichloroethane based on the additional lines of evidence (physical-chemical and environmental fate and transport similarity and toxicological similarity).

Table 4-2. Structural Similarity Between 1,1-Dichloroethane and Analog Candidates that Met Filtering Criteria in at Least 3 of 4 Structure Programs

Chlorinated Solvent	CASRN	AIM	OECD QSAR Toolbox	Cheminformatics	GenRA	Count
1,1-Dichloroethane (target)	75-34-3	Exact match	1.00	1.00	1.00	4
2-Chloropropane	75-29-6	1st pass	0.7–0.8	0.56	0.83	4
2,3-Dichlorobutane	7581-97-7	1st pass	0.6–0.7	–	0.63	3
1,1,2-Trichloropropane	598-77-6	1st pass	–	0.50	0.56	3
Chloroethane	75-00-3	2nd pass	0.8–0.9	0.71	–	3
1,1-Dichloropropane	78-99-9	2nd pass	0.8–0.9	0.70	–	3
1,1,2,2-Tetrachloroethane	79-34-5	2nd pass	0.7–0.8	0.78	–	3
1,1,2-Trichloroethane[a]	79-00-5	2nd pass	0.7–0.8	0.78	–	3
1,1,1-Trichloroethane	71-55-6	2nd pass	0.7–0.8	0.78	–	3
2,2-Dichloropropane	594-20-7	2nd pass	0.7–0.8	0.70	–	3
1,1,1,2-Tetrachloroethane	630-20-6	2nd pass	0.7–0.8	0.64	–	3
Pentachloroethane	76-01-7	2nd pass	0.7–0.8	0.64	–	3
1,2-Dichloroethane[a]	107-06-2	2nd pass	0.7–0.8	0.63	–	3
1-Chloropropane	540-54-5	2nd pass	0.7–0.8	0.56	–	3
1,3-Dichloropropane	142-28-9	2nd pass	0.7–0.8	0.50	–	3
1,2-Dichloropropane[a]	78-87-5	2nd pass	0.7–0.8	–	0.45	3
2-Chlorobutane	78-86-4	2nd pass	0.6–0.7	–	0.45	3
Chloroform	67-66-3	2nd pass	–	0.50	0.43	3

Chlorinated Solvent	CASRN	AIM	OECD QSAR Toolbox	Cheminformatics	GenRA	Count
OECD QSAR = Organisation of Economic Cooperative Development Quantitative Structure Activity Relationship [a] Analogs that have completed data evaluation and extraction are bolded. Dashes ("–") indicate structural similarity scores were not available for those analogs using the filtering parameters described in Table 4-1.						

4.2.1.1.2 Physical and Chemical and Environmental Fate and Transport Similarity

1,1-Dichloroethane analog candidates from the structural similarity analysis were preliminarily screened based on similarity in log octanol-water partition coefficient (log K_{OW}) and vapor pressure obtained using EPI Suite™ with measured values, if available, used in place of predicted values. For this screening step, 1,1-dichloroethane, 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane values were obtained from Table 2-1, the *Final Scope of the Risk Evaluation for 1,2-Dichloropropane; CASRN 78-87-5;* the *Final Scope of the Risk Evaluation for 1,1,2-Trichloroethane;* and the *Final Scope of the Risk Evaluation for 1,2-Dichloroethane;* (U.S. EPA, 2020c, e, f). Analog candidates with log K_{OW} and vapor pressure within one log unit relative to 1,1-dichloroethane were considered potentially suitable analog candidates for 1,1-dichloroethane (Figure 4-1). This preliminary screening analysis narrowed the analog candidate list from 17 candidate analogs to 11 candidate analogs (Table 4-3). Three of the 11 candidate analogs were 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane. Because these three solvents' environmental hazard data had completed evaluation and extraction, a more expansive analysis of physical, chemical, environmental fate and transport similarities between 1,1-dichloroethane and candidate analogs 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane was conducted (Figure 4-1, Table 4-4).

Table 4-3. Analog Candidates with Similar Log K_{OW} and Vapor Pressure Values to that of 1,1-Dichloroethane

Chemical	CASRN	Log K_{OW}	Vapor Pressure (mmHg)
1,1-Dichloroethane (target)	75-34-3	1.79	228
2-Chloropropane	75-29-6	1.90	515
2,3-Dichlorobutane	7581-97-7	2.67[a]	24.4
Chloroethane	75-00-3	1.43	1,010
1,1-Dichloropropane	78-99-9	2.25[a]	50.8
1,1,2-Trichloroethane[b]	79-00-5	1.89	23.0
1,1,1-Trichloroethane	71-55-6	2.49	124
1,2-Dichloroethane[b]	107-06-2	1.48	78.9
1-Chloropropane	540-54-5	2.04	345
1,2-Dichloropropane[b]	78-87-5	1.98	53.3
2-Chlorobutane	78-86-4	2.33	157
Trichloromethane	67-66-3	1.97	197
[a] Values predicted using EPI Suite™ [b] Analogs which have completed data evaluation and extraction are bolded.			

Physical, chemical, and environmental fate and transport similarities between 1,1-dichloroethane and its analog candidates 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane were assessed based on properties relevant to the aquatic, benthic, and soil compartments (Table 4-4). These properties

were selected based on their general importance in determining similar exposure potential in the aquatic, benthic, and soil compartments. Physical, chemical, and environmental fate and transport values for 1,1-dichloroethane, 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane are specified in Appendix C, the *Final Scope of the Risk Evaluation for 1,2-Dichloropropane; CASRN 78-87-5* (U.S. EPA, 2020f) and the *Final Scope of the Risk Evaluation for 1,1,2-Trichloroethane; CASRN 79-00-5* (U.S. EPA, 2020c), respectively. Similar values are observed for 1,1-dichloroethane, 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane water solubilities (2,800–8,600 mg/L), log K_{OW} (1.48–1.99), and log K_{OC} (1.28–2.32) indicating all four solvents as highly water soluble with low affinity for sediment and soil (Table 4-4). 1,1-Dichloroethane and its analogs 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane have relatively low bioconcentration factors (BCF, 0.5–7) and bioaccumulation factors (3.8–7.1), indicating low bioaccumulation potential in aquatic and terrestrial environments. Although hydrolysis half-lives are relatively long for all four solvents, other properties of 1,1-dichloroethane and its analogs 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane indicate that the chemicals will likely volatilize well before hydrolyzing in aqueous environments.

All four chlorinated solvents are highly volatile (Henry's Law constants 8.24×10^{4} to 5.62×10^{-3} atm-m^3/mol and vapor pressures 23 to 228 mmHg), indicating volatilization from both water and soil will occur. The vapor pressures indicate some difference in volatility between the four chlorinated solvents; vapor pressures of 40, 23, and 78 mmHg for 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane, respectively, compared to 228 mmHg for 1,1-dichloroethane. However, potential impacts of volatility differences on read-across to 1,1-dichloroethane for environmental hazard can be addressed by factoring in experimental design considerations in the analog's aquatic hazard dataset such as chemical measurement of the substance in the test medium, regular renewal with chemical solution, capping of test vessels, and/or use of flow-through/dilutor systems. See Section 4.2.1.1.4 for further discussion of the impact of volatility differences between 1,1-dichloroethane and its analogs on the analog selection. All four solvents exist as colorless liquids at room temperature and have similar low molecular weights (Table 4-4). The similarity of the physical, chemical, fate, and environmental transport behavior of these four chlorinated solvents in aquatic, benthic, and soil environments support the ability to read-across to 1,1-dichloroethane from 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane environmental hazard data. 1,2-Dichloropropane and 1,1,2-trichloroethane were ultimately selected for read-across of aquatic and benthic hazard to 1,1-dichloroethane based on an additional line of evidence (ecotoxicological similarity).

Table 4-4. Comparison of 1,1-Dichloroethane and Analog Candidates 1,2-Dichloropropane, 1,1,2-Trichloroethane, and 1,2-Dichloroethane for Several Physical and Chemical and Environmental Fate Properties Relevant to Water, Sediment, and Soil

Property	1,1-Dichloroethane (Target)	1,2-Dichloropropane	1,1,2-Trichloroethane	1,2-Dichloroethane
Water solubility	5,040 mg/L	2,800 mg/L	4,590 mg/L	8,600 mg/L
Log K_{OW}	1.79	1.99	1.89	1.48
Log K_{OC}	1.48	1.67	1.9–2.05, 2.2–2.32	1.28–1.62
BCF	7	0.5–6.9	0.7–6.7	2
BAF	6.8	7.1	6.9	3.8
Hydrolysis t½	61.3 years	15.8 years	85 days	65 years, 72 years
Henry's Law constant (atm-m^3/mol)	5.62E−03	2.82E−03	8.24E−04	1.18E−03
Vapor pressure (mmHg)	228	40	23	79
Molecular weight	98.95 g/mol	112.99 g/mol	133.41 g/mol	98.96 g/mol
Physical state of the chemical	Colorless liquid	Colorless liquid	Colorless liquid	Colorless liquid

4.2.1.1.3 Ecotoxicological Similarity

Ecotoxicological similarity between 1,1-dichloroethane and its analog candidates was assessed based on a comparison of the analogs' measured hazard data proposed for aquatic hazard read-across to corresponding ECOSAR toxicity predictions of the target 1,1-dichloroethane (Figure 4-1). Similarity in empirical hazard evidence for benthic and aquatic invertebrates exposed in water to 1,1-dichloroethane and 1,1-dichloroethane's analogs was also assessed to determine suitability of using analog sediment invertebrate and soil invertebrate hazard data to read-across to 1,1-dichloroethane. Although a soil invertebrate read-across was attempted, no reasonably available soil invertebrate hazard data for 1,2-dichloropropane, 1,1,2-trichloroethane, or 1,2-dichloroethane were identified for use as a hazard threshold to read-across to 1,1-dichloroethane. All measured data used in the ecotoxicological similarity comparisons were from studies with overall quality determinations of high and medium except when noted otherwise in table footnotes. 1,1-Dichloroethane toxicity predictions for acute and chronic exposure to fish, aquatic invertebrates (mysid), and green algae were generated using ECOSAR v2.2 (neutral organics category) using inputs CASRN and measured log K_{OW} value for 1,1-dichloroethane (U.S. EPA, 2022d).

Comparison of the analog empirical hazard data to corresponding ECOSAR toxicity predictions for 1,1-dichloroethane shows agreement of hazard values well within 10-fold (Figure 4-1, Table 4-5). Average ratio of empirical 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane aquatic hazard data to predicted 1,1-dichloroethane hazard values are 0.75 ± 0.15, 1.22 ± 0.38, and 3.07 ± 1.07 (standard error), respectively (Table 4-5), which indicates very similar ecotoxicological behavior between 1,2-dichloropropane and 1,1-dichloroethane when aquatic vertebrates, aquatic invertebrates, and algae are exposed under acute and chronic conditions. Therefore, due to 1,2-dichloropropane's similarity to 1,1-dichloroethane using multiple lines of evidence (structure, physical chemical, and ecotoxicological) and 1,2-dichloropropane's availability of aquatic hazard data, 1,2-dichloropropane is an appropriate analog for an aquatic hazard read-across to 1,1-dichloroethane. 1,1,2-Trichloroethane was also considered a suitable analog for aquatic hazard read-across to 1,1-dichloroethane; however, the aquatic hazard profile for 1,1,2-trichloroethane was less complete than that of 1,2-dichloropropane and empirical aquatic hazard data for 1,1,2-trichloroethane was slightly less protective when compared to the 1,1-dichloroethane predicted aquatic hazard (Table 4-5). 1,2-Dichloroethane was ultimately not selected for read-across of aquatic hazard to 1,1-dichloroethane due to empirical aquatic toxicity data that were less conservative than the toxicity predictions for 1,1-dichloroethane (Table 4-5).

Ecotoxicological similarity for sediment invertebrate chronic hazard and soil invertebrate hazard read-across is inferred by the aquatic and benthic invertebrate toxicity comparisons made between 1,1-dichloroethane and its analogs (Table 4-6), similar to the environmental hazard read-across approach used for phthalates (U.S. EPA, 2024 U.S. EPA, 2025, 11799662). The comparison of 1,1-dichloroethane's measured hazard in *Chironomus riparius* (48-hour EC50) and *Daphnia magna* (48-hour EC50 and LC50, 21 day ChV) to that of its analogs indicated the same trend as the first ecotoxicological similarity analysis in Table 4-5 in that 1,2-dichloropropane was the closest in toxicity to 1,1-dichloroethane followed by 1,1,2-trichloroethane and lastly 1,2-dichloroethane (Table 4-6). However, because 1,2-dichloropropane did not have reasonably available chronic benthic hazard data, 1,1,2-trichloroethane chronic benthic invertebrate data were used in a read-across to 1,1-dichloroethane (Table 4-6). Although all three analogs were generally suitable for a soil invertebrate hazard read-across to 1,1-dichloroethane based on the comparisons of the empirical *Daphnia magna* and *Chironomus riparius* data (Table 4-6), no reasonably available soil invertebrate hazard data were identified for 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane that could be used to derive a quantitative hazard threshold for 1,1-dichloroethane.

Table 4-5. Ecotoxicological Similarity Between 1,1-Dichloroethane (Predicted Hazard) and its Analogs 1,2-Dichloropropane (Measured Hazard), 1,1,2-Trichloroethane (Measured Hazard), and 1,2-Dichloroethane (Measured Hazard) in Aquatic Taxa

Taxa	Duration	Endpoint	1,1-Dichloroethane (Target)		1,2-Dichloropropane		1,1,2-Trichloroethane		1,2-Dichloroethane	
			Measured Hazard (mg/L)	Predicted Hazard (mg/L)	Measured Hazard (mg/L)	Ratio to 1,1-Dichloroethane Toxicity	Measured Hazard (mg/L)	Ratio to 1,1-Dichloroethane Toxicity	Measured Hazard (mg/L)	Ratio to 1,1-Dichloroethane Toxicity
Fish	96-hour	LC50	**Read-across**	125.5	**133.34** [a]	1.06	71.9 [c]	0.57	153 [f]	1.22
Mysid	96-hour	LC50	**Read-across**	135.2	**24.79** [a]	0.18	–	–	64.8 [g]	0.48
Green algae	96-hour	EC50	**Read-across**	48.1	**35.4** [a]	0.74	57 [d]	1.19	124 [d]	2.58
Fish		ChV	**Read-across**	12	**11.6** [a b]	0.97	22.7 [e]	1.89	48.4 [h]	4.03
Green algae		ChV	**Read-across**	12.1	**10** [a]	0.83	–	–	85.3 [i]	7.05
Average fold hazard analog:1,1-dichloroethane						0.75 ± 0.15		1.22 ± 0.38		3.07 ± 1.07

[a] Hazard values used in the read-across to 1,1-dichloroethane are bolded and data sources are listed in Table 4-7.
[b] Value for 1,2-dichloropropane represents a geometric mean of fish (*Pimephales promelas*) NOEC/LOEC pairs for Mortality (11/25 mg/L) and Development/Growth (6/11 mg/L) endpoints from (Benoit et al., 1982). Exposure and study duration was 32–33 days.
[c] Value for 1,1,2-trichloroethane represents a geometric mean of 96-hour fish (*Pimephales promelas*, *Jordanella floridae*) LC50 data (81.6 mg/L, 89.1 [66.6–110.0] mg/L, 45.1 [42.0–48.5] mg/L, 81.6 [60.9–109] mg/L) from (Smith et al., 1991 Walbridge, 1983, 4259619; Geiger et al., 1985).
[d] Data derived from 72-hour exposures to 1,1,2-trichloroethane or 1,2-dichloroethane from (Brack and Rottler, 1994) and (CITI, 1996b), respectively, and were the closest representative measured hazard data available for comparison to the 96-hour green algae EC50 hazard prediction for 1,1-dichloroethane. CIs were 54.0–60.6 mg/L and 106–144 mg/L, respectively.
[e] Value for 1,1,2-trichloroethane represents a geometric mean of fish (*Pimephales promelas*, *Jordanella floridae*) NOEC/LOEC pairs for Mortality (29.0/74.8 mg/L; 14.8/48.3 mg/L) and Development/Growth (6.0/14.8 mg/L) endpoints from (Smith et al., 1991; Ahmad et al., 1984). Exposure and study durations were 28–32 days.
[f] Value for 1,2-dichloroethane represents a geometric mean of 96-hour fish (*Pimephales promelas*, *Oncorhynchus mykiss*) LC50 data (116 [110–123] mg/L, 136 [129–144] mg/L, 225 mg/L) from (Mayer and Ellersieck, 1986; Geiger et al., 1985; Walbridge et al., 1983).
[g] Value for 1,2-dichloroethane was the closest representative hazard data for comparison to the 96-hour predicted LC50 for mysid and represents a geometric mean of 24-hour brine shrimp (*Artemia salina*) EC50 data (36.4 [30.6–43.0] mg/L, 79.4 [69.7–96.0] mg/L, 93.64 [77.0–113.6] mg/L) from (Foster and Tullis, 1985; Foster and Tullis, 1984).
[h] Value for 1,2-dichloroethane represents a geometric mean of fish (*Pimephales promelas*, *Oryzias latipes*) NOEC/LOEC pairs for Mortality (41.3/78.9 mg/L) and Development/Growth (29/59 mg/L) endpoints from (CITI, 1996c; Benoit et al., 1982). Exposure and study durations were 21-33 days.
[i] Value for 1,2-dichloroethane represents a geometric mean of green algae (*Raphidocelis subcapitata*) NOEC/LOEC pairs for Growth/Development (65.6/111 mg/L) from (1996b). Exposure was 72 hours.

Table 4-6. Comparison of Measured 1,1-Dichloroethane and Analog Hazard Values in Aquatic and Benthic Invertebrates

Species	Outcome	Endpoint(s)	1,1-Dichloroethane (Target)	1,2-Dichloropropane		1,1,2-Trichloroethane		1,2-Dichloroethane	
			Measured Hazard (mg/L)	Measured Hazard (mg/L)	Ratio to 1,1-Dichloroethane Toxicity	Measured Hazard (mg/L)	Ratio to 1,1-Dichloroethane Toxicity	Measured Hazard (mg/L)	Ratio to 1,1-Dichloroethane Toxicity
Daphnia magna	Mortality and Immobilization	EC50, LC50	34 [a]	39.2 [b]	1.14	80.8 [c]	2.35	194.7 [d]	5.68
Chironomus riparius	Mortality and Immobilization	EC50	150 [e]	49 [f]	0.33	–	–	–	–
Daphnia magna	Reproduction	ChV	0.93 [g]	4.16 [h]	4.49	3.17 [i]	3.42	1.62 [j]	1.74
Chironomus riparius	Growth/Development	ChV	**Read-across**	–	–	**29** [k l], **93** [k]	–	–	–
Ophryotrocha labronica	Reproduction	EC10	**Read-across**	–	–	**68** [m]	–	309 [m]	–
Average fold hazard analog:1,1-dichloroethane					1.98 ± 1.27		2.89 ± 0.53		3.71 ± 1.97

[a] Data are from (2009a) and have CI [30.0–39.1] mg/L. Exposure was 48 hours.
[b] Value for 1,2-dichloropropane represents a geometric mean of *Daphnia magna* EC50 and LC50 data (29.5 [26.5–32.8] mg/L; 52 [42–68] mg/L) from (1995a; 1980). Data from (1995a) are unrated for data quality. Exposure was 48 hours.
[c] Value for 1,1,2-trichloroethane represents a geometric mean of *Daphnia magna* EC50 and LC50 data (81 [58–97] mg/L; 18 [11–32] mg/L; 190 [160–210] mg/L; 170 [150–200] mg/L; 81 [58–110] mg/L; 78 [57–110] mg/L) from (3M Environmental Lab, 1984; 1983; 1980). Exposure was 48 hours.
[d] Value for 1,2-dichloroethane represents a geometric mean of *Daphnia magna* EC50 and LC50 data (220 [160–280] mg/L; 320 [270–410] mg/L; 270 [250–290] mg/L; 160 [140–190] mg/L; 180 [150–230] mg/L; 99.4 [88.3–115] mg/L) from (1996a; 1983; 1980). Exposure was 48 hours.
[e] Data are from (2024b) and have CI 130–180 mg/L. Exposure was 48 hours.
[f] Data are from (2024a) and have CI 43–56 mg/L. Exposure was 48 hours.
[g] Value for 1,1-dichloroethane represents a geometric mean of *Daphnia magna* NOEC/LOEC pair for Reproduction endpoints (1.64/0.525 mg/L) from (Mitsubishi Chemical Medience Corporation, 2009d). Exposure and study durations were 21 days.
[h] Value for 1,2-dichloropropane represents a geometric mean of *Daphnia magna* NOEC/LOEC pairs for Reproduction endpoints (8.3/15.8 mg/L; 0.96/2.40 mg/L) from (NITE, 1995b; Dow Chemical, 1988). Data from (NITE, 1995b) are unrated for data quality. Exposure and study durations were 21 days.
[i] Value for 1,1,2-trichloroethane represents a geometric mean of *Daphnia magna* NOEC/LOEC pair for Reproduction endpoints (2.4/4.2 mg/L) from (3M Environmental Lab, 1984). Exposure and study durations were 21 days.
[j] Value for 1,2-dichloroethane represents a geometric mean of *Daphnia magna* NOEC/LOEC pair for Reproduction endpoints (1.02/2.56 mg/L) from (CITI, 1996d). Exposure and study durations were 21 days.
[k] Value for 1,1,2-trichloroethane represents a geometric mean of *Chironomus riparius* NOEC/LOEC pair for Growth Development endpoints (66/130 mg/L; 19/44 mg/kg) from (Smithers, 2023). Exposure and study duration was carried out over two generations.
[l] Hazard value in mg/kg.
[m] Data are from (Rosenberg et al., 1975). Exposure and study duration were 15 days.

4.2.1.1.4 Read-Across Weight of Scientific Evidence and Conclusion

1,1-Dichloroethane presented with minimal aquatic and benthic hazard data and no soil invertebrate data. Analog selection was carried out to address these data gaps. Several chlorinated solvents of interest (1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane) were indicated as structurally similar to 1,1-dichloroethane. A screening by log K_{OW} values and further comparison of additional physical, chemical, and environmental fate and transport properties indicated that all three analog candidates were similar to 1,1-dichloroethane with some differences in volatility. A comparison of available 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane empirical hazard data to corresponding 1,1-dichloroethane toxicity predictions for aquatic taxa targeted for read-across showed high concordance between analogs 1,2-dichloropropane and 1,1,2-trichloroethane and the target 1,1-dichloroethane. However, due to 1,2-dichloropropane being slightly more protective and having more aquatic hazard data available to be used in the read-across than 1,1,2-trichloroethane, 1,2-dichloropropane was selected for read-across of aquatic hazard to 1,1-dichloroethane. Due to 1,2-dichloropropane lacking benthic hazard data, 1,1,2-trichloroethane was selected as an analog to supplement the 1,1-dichloroethane benthic hazard dataset.

1,2-Dichloroethane was generally considered less toxic to aquatic taxa than 1,1-dichloroethane or its analogs 1,2-dichloropropane and 1,1,2-trichloroethane. To determine if any of the three candidate analogs would be suitable for sediment invertebrate and soil invertebrate read-across, a second toxicity comparison was made in benthic invertebrate *Chironomus riparius* and aquatic invertebrate *Daphnia magna* exposed in water to either 1,1-dichloroethane or the three solvent analogs for 48 hours (*C. riparius* or *D. magna*) or 21 days (*D. magna*). The comparisons indicated that 1,1,2-trichloroethane was the most suitable analog which had chronic benthic hazard data to read-across to 1,1-dichloroethane (Table 4-6). Although all three analogs were within 10-fold agreement of 1,1-dichloroethane's toxicity values and were considered suitable for a soil invertebrate hazard read-across, no reasonably available soil invertebrate hazard were identified for 1,2-dichloropropane, 1,1,2-trichloroethane, or 1,2-dichloroethane which could be used to derive a hazard threshold for soil invertebrates.

Ecotoxicological similarity for a soil invertebrate hazard read-across is inferred by the aquatic and benthic invertebrate toxicity comparisons made between 1,1-dichloroethane and its analogs (Table 4-6), although this inference has slightly greater uncertainty than when it was made in a previous read-across (U.S. EPA, 2024). The greater uncertainty is due to a lack to 1,1-dichloroethane sediment exposure data with which to compare to analog sediment exposure data as a more relevant ecotoxicological comparison for a soil invertebrate hazard read-across. Because ECOSAR hazard predictions do not encompass benthic invertebrates (Table 4-5) and 1,1,2-trichloroethane lacked acute hazard data for *C. riparius* with which to compare to that of 1,1-dichloroethane (Table 4-6), there is also some uncertainty in the chronic benthic hazard read-across from 1,1,2-trichloroethane to 1,1-dichloroethane. However, the structural agreement and similar chemical behavior in sediment inferred from the physical, chemical, environmental fate and transport properties as well as the similar aquatic toxicity of 1,1,2-trichloroethane and 1,1-dichloroethane support the ability to read-across for benthic hazard. Another uncertainty in the analog selection for 1,1-dichloroethane is that the relatively small chemical structures of 1,1-dichloroethane and its analogs could result in lower structural similarity scores. However, looking for concordance across multiple structure programs increases the confidence that structurally similar analogs were identified for 1,1-dichloroethane in Table 4-2.

Regarding uncertainty in the physical, chemical, environmental fate and transport line of evidence used in the analog selection, lower vapor pressure of analog candidates 1,2-dichloropropane and 1,1,2-trichloroethane relative to 1,1-dichloroethane (although still within 10-fold) could result in volatility

differences between target and analog. However, by considering the experimental design in the analog's empirical hazard data used in the read-across (chemical measurement, chemical renewal, capping test vessels, use of flow-through, *etc*.), confidence is increased such that the volatility differences do not impact the strength of the read-across. The neutral organics class in ECOSAR v2.2 has a robust dataset for predicting environmental hazard which increases the confidence in the predicted toxicological similarity observed between 1,1-dichloroethane and its analogs, giving high confidence to the aquatic hazard read-across followed by moderate confidence in the benthic and soil invertebrate hazard read-across. Looking across the multiple lines of evidence (structural, physical/chemical, ecotoxicological), 1,2-dichloropropane and 1,1,2-trichloroethane are appropriate analogs with high- and medium-quality aquatic and benthic hazard data to be used in a read-across to 1,1-dichloroethane.

4.2.2 Aquatic Species Hazard

Toxicity to Aquatic Organisms

EPA assigned overall quality determinations of high to six acceptable aquatic toxicity studies for 1,1-dichloroethane, high or medium to six acceptable aquatic studies for analog 1,2-dichloropropane, and high or medium to two acceptable aquatic study for analog 1,1,2-trichloroethane. Analog selection for environmental hazard is discussed in Section 4.2.1.1. EPA identified 13 aquatic toxicity studies, displayed in Table 4-7, as the most relevant for quantitative assessment. The remaining study was represented by a short-term exposure (1 hour) of a single low-dose of 1,1-dichloroethane, resulting in a no-effect for ventilation frequency, ventilation amplitude, or swimming behavior in rainbow trout (*Oncorhynchus mykiss*) (Kaiser K et al., 1995), and was therefore considered less relevant for establishing a hazard threshold. The Web-ICE application was used to predict LC50 toxicity values for 75 additional aquatic organisms (25 fish, 2 amphibians, and 48 aquatic invertebrate species) from the 1,1-dichloroethane *Daphnia magna* 48-hour effective concentration 50 (EC50) and 1,2-dichloropropane fathead minnow and opossum shrimp 96-hour LC50 data (Raimondo and Barron, 2010). *Chironomus riparius* was not available as a surrogate species in Web-ICE; however, the TSCA section 4(a)(2) test order for *C. riparius* immobilization and mortality data from 48-hour exposure to 1,1-dichloroethane were included in the SSD alongside the other empirical hazard data and predicted hazard. The test species (n = 4) and predicted species (n = 75) toxicity data were then used to calculate the distribution of species sensitivity.

Aquatic Vertebrates

EPA assigned overall quality determinations of high to a single study with 1,1-dichloroethane fish hazard data and high or medium to three studies with analog 1,2-dichloropropane fish hazard data as relevant for quantitative assessment. The 1,1-dichloroethane study and two of the 1,2-dichloropropane studies contained fish hazard resulting from acute exposures whereas the remaining 1,2-dichloropropane study contained fish hazard data for acute and chronic exposures to 1,2-dichloropropane (Table 4-7).

For acute toxicity studies in fish, Japanese medaka (*Oryzias latipes*) no greater than 6 months old exposed to measured concentrations of 1,1-dichloroethane for 96 hours under semi-static conditions (renewal every 24 hours) had abnormal swimming behavior with a derived EC50 value of 70.7 mg/L (Mitsubishi Chemical Medience Corporation, 2009b). Authors noted abnormal swimming behavior if any of the following were observed: inactivity, hyperactivity, surface swimming, loss of balance, directionless swimming, or convulsions (Mitsubishi Chemical Medience Corporation, 2009b). Details on EC50 derivation are described in Appendix K.2.1.3. Twenty-eight to 34-day-old fathead minnow (*Pimephales promelas*) exposed to measured concentrations of analog 1,2-dichloropropane for 96 hours in flow-through conditions exhibited loss of equilibrium, swimming near the surface, loss of schooling behavior, hypoactivity, and mortality with a reported LC50 for mortality of 127 mg/L (Geiger et al., 1985). Similarly, 30- to 35-day old fathead minnow exposed to measured concentrations of 1,2-

dichloropropane for 96 hours under flow-through conditions had a reported mortality LC50 of 140 mg/L (Walbridge et al., 1983) (Table 4-7).

For chronic toxicity in fish, no data were reasonably available for 1,1-dichloroethane; therefore, the data are represented by exposure to 1,2-dichloropropane. In the fish early life stage test, fathead minnow exposed to measured concentrations of 1,2-dichloropropane under flow-through conditions for 32 to 33 days resulted in a no-observed-effect-concentration (NOEC) and lowest-observed-effect-concentration (LOEC) for survival of 11 and 25 mg/L, respectively, and a NOEC and LOEC for decreased weight of 6 and 11 mg/L, respectively (Benoit et al., 1982). EPA calculated the 32- to 33-day survival NOEC and LOEC geometric mean of 16.58 mg/L as the chronic value (ChV) for survival and the growth NOEC and LOEC geometric mean of 8.12 mg/L (Table 4-7).

Amphibians

No amphibian studies were reasonably available to assess potential hazards from 1,1-dichloroethane exposure. However, modeled data from Web-ICE predicted a bullfrog (*Lithobates catesbeianus*) 96-hour LC50 of 133,397 µg/L and African clawed frog (*Xenopus laevis*) 96-hour LC50 of 46,684 µg/L from the empirical data of 1,1-dichloroethane and analog 1,2-dichloropropane (Table_Apx K-1). Therefore, amphibian acute toxicity is accounted for within the Web-ICE and SSD results (Figure_Apx K-4).

Aquatic Invertebrates

EPA assigned overall quality determinations of high to three studies with 1,1-dichloroethane aquatic invertebrate hazard data, including hazard data received from a 1,1-dichloroethane TSCA section 4(a)(2) test order, and high or medium to three studies with 1,2-dichloropropane or 1,1,2-trichloroethane aquatic invertebrate hazard data as relevant for quantitative assessment. Three of these studies contained hazard data for acute and/or chronic exposures of water column-dwelling invertebrates to 1,1-dichloroethane or 1,2-dichloropropane and the other three studies contained hazard data for acute and/or chronic exposures of benthic invertebrates to 1,1-dichloroethane or 1,1,2-trichloroethane.

For acute toxicity studies for water column-dwelling invertebrates, *Daphnia magna* exposed to measured concentrations of 1,1-dichloroethane for 48-hours in semi-static conditions (renewal every 24 hours) in covered beakers had an immobilization EC50 value of 34.3 mg/L (Mitsubishi Chemical Medience Corporation, 2009a). In a saltwater-dwelling invertebrate study, opossum shrimp (*Americamysis bahia* or *Mysidopsis bahia*) less than 24 hours old had a LC50 of 24.79 mg/L when exposed to measured concentrations of analog 1,2-dichloropropane for 96-hours under flow-through conditions (Dow Chemical, 1988). In the same study, the 96-hour LC50 for 3-to 4-day old *A. bahia* was greater than 26.65 mg 1,2-dichloropropane/L (also based on measured concentrations), suggesting neonates are more sensitive to 1,2-dichloropropane than more developed shrimp. The mortality NOEC for neonate opossum shrimp was 4.92 mg 1,2-dichloropropane/L; therefore, EPA assigned the mortality LOEC as the next highest concentration tested in the study, which was 6.89 mg 1,2-dichloropropane/L (Table 4-7).

For chronic toxicity studies for water-column dwelling invertebrates, *D. magna* exposed to measured concentrations of 1,1-dichloroethane for 21 days in semi-static conditions (renewal daily) in covered beakers had a chronic 21-day NOEC of 0.525 mg/L and LOEC of 1.64 mg/L for reproductive inhibition (based on number of young produced), resulting in a reproductive ChV of 0.93 mg/L (Mitsubishi Chemical Medience Corporation, 2009d). A median EC50 of 6.67 mg/L was also reported for reproductive inhibition (Mitsubishi Chemical Medience Corporation, 2009d).

Benthic Invertebrates

Potential hazards from acute 1,1-dichloroethane exposures to sediment-dwelling organisms were assessed using freshwater midge *Chironomus riparius* hazard data received from a 1,1-dichloroethane TSCA section 4(a)(2) test order. In a study which received an overall quality determination of high, 3-day old *C. riparius* larvae exposed in water to measured concentrations of 21, 43, 95, 180, and 380 mg/L 1,1-dichloroethane (nominal 31, 63, 130, 250, and 500 mg/L) demonstrated immobilization and/or mortality with 48-hour EC50 of 150 mg/L 1,1-dichloroethane (95th confidence intervals 130–180 mg/L) (Smithers, 2024b). *C. riparius* larvae were exposed to 1,1-dichloroethane test solutions in sealed containers and transferred to freshly prepared 1,1-dichloroethane test solution in sealed containers after 24 hours for an additional 24-hour exposure period (resulting in a total exposure period of 48 hours). Behavior, mortality, and immobilization of the larvae were monitored during the 48-hour exposure duration. In addition to the *C. riparius* hazard data in (2024b), modeled data from Web-ICE predicted 96-hour LC50 values for 36 benthic invertebrates from the empirical data of 1,1-dichloroethane and analog 1,2-dichloropropane (Table_Apx K-1, Figure_Apx K-4). Therefore, acute toxicity to sediment-dwelling invertebrates is accounted for within the Web-ICE and SSD results.

No reasonably available data on chronic hazard of sediment-dwelling invertebrates were available for 1,1-dichloroethane or its primary analog 1,2-dichloropropane. Therefore, chronic hazard data from two high or medium-rated studies for sediment-dwelling invertebrates on a secondary analog, 1,1,2-trichloroethane were considered. EPA deemed 1,1,2-trichloroethane suitable for targeted read-across of chronic benthic hazard to 1,1-dichloroethane as described in Section 4.2.1.1. The marine polycheate worm species *Ophryotrocha labronica* exposed to increasing nominal concentrations of 1,1,2-trichloroethane in water for 15 days under semi-static renewal conditions had reduced hatching with a modeled EC10 of 68 mg/L (Rosenberg et al., 1975). Derivation of the EC10 is described in Appendix K.2.1.3. Larvae of the freshwater midge *C. riparius* exposed over two generations to measured concentrations of 1,1,2-trichloroethane in sediment had significantly decreased emergence in second-generation (F1) larvae exposed to the highest tested concentration of 1,1,2-trichloroethane (measured 44 mg 1,1,2-trichloroethane/kg sediment dry weight, nominal 1,000 mg/kg), resulting in a chronic 28-day NOEC of 19 mg/kg and LOEC of 44 mg/kg, which EPA then calculated a ChV of 29 mg/kg for growth and development (Table 4-7). The decrease in F1 larval emergence at the LOEC was approximately half of control value (42 ± 24% emergence in the 44 mg 1,1,2-trichloroethane/kg treatment group compared to 77 ± 8% emergence in the control group; values presented as average ± standard deviation) (Smithers, 2023). The NOEC and LOEC for the same endpoint within this study were also expressed in measured pore water concentrations at 66 and 130 mg/L, which the EPA then calculated a growth and development ChV of 93 mg/L in benthic pore water (Table 4-7).

None of the other measured endpoints for F1 midges or parent midges (F0) in the definitive study resulted in a definitive LOEC; however, it should be noted that percent emergence was significantly decreased in F0 larvae (44 ± 16% compared to 81 ± 8% emergence in the controls) exposed to the second highest tested 1,1,2-trichloroethane concentration (measured 10 mg/kg) but not the highest tested 1,1,2-trichloroethane concentration (30 mg/kg); therefore, a LOEC was not established for percent emergence in the F0 larval midges. In the preliminary 2-generation sediment screening portion of this same study, decreased emergence was also noted in F1 larval midges exposed to the highest tested concentration of 1,1,2-trichloroethane (14 ± 6% emergence of F1 larval midges exposed to nominal 1,000 mg 1,1,2-trichloroethane/kg sediment dry weight compared to 90 ± 11% emergence in the control larval midges (Smithers, 2023). Although the preliminary 2-generation sediment screening study did not report measured concentrations of 1,1,2-trichloroethane in the sediment and nominal concentrations are not expected to be representative of actual concentrations, the results support decreased emergence in F1 larvae in the definitive study.

Aquatic Plants

EPA assigned overall quality determinations of high to one study with 1,1-dichloroethane aquatic plant hazard data and high or medium to three studies with analog 1,2-dichloropropane aquatic plant hazard data as relevant for quantitative assessment.

For studies that reported growth inhibition in the form of EC50 values, green algae species (*Chlamydomonas reinhardtii*) exposed to measured concentrations of 1,2-dichloropropane for 96-hours under flow-through conditions had an EC50 of 83 mg/L for growth rate (Schäfer et al., 1994). This study also reported *C. reinhardtii* EC50 values for 7 to 10-days of exposure ranging from 50 to 62 mg/L and NOECs ranging from 29 to 31.5 mg/L, demonstrating increasing toxicity with increasing exposure durations. EPA used the 96-hour EC50 value from (Schäfer et al., 1994) and the 96-hour EC50 hazard value of 15.1 mg/L for marine diatom (*Skeletonema costatum*) growth rate exposed to measured concentrations of 1,2-dichloropropane in closed vessels (Dow Chemical, 2010) to calculate a geometric mean of 35.4 mg/L, representing multiple algal species.

For studies reporting growth inhibition NOECs and LOECs, the 1,2-dichloropropane data presented in Dow Chemical (2010) are a reanalysis of *S. costatum* 120-hour NOEC and LOEC biomass data originally presented in Dow Chemical (1988). In Dow Chemical (2010), the authors report data for additional hazard values (EC10 and EC50 in addition to NOEC and LOEC), growth endpoints (growth rate and abundance in addition to biomass), and durations (72 and 96 hours in addition to 120 hours). The authors also used the geometric means of the daily measured chemical concentrations to establish the hazard values in the reanalysis presented in Dow Chemical (2010).

From the 72-, 96-, and 120-hour EC10 values of 8.47 mg/L, 8.49 mg/L, and 6.19 mg/L 1,2-dichloropropane, respectively, EPA calculated the geometric mean of 72- to 120-hour biomass (area under the growth curve) EC10 as 7.64 mg/L 1,2-dichloropropane in *S. costatum*. From the 72-, 96-, and 120-hour NOECs of 8.50 mg/L, 7.12 mg/L, and 6.87 mg/L 1,2-dichloropropane, respectively, and 72-, 96-, and 120-hour LOECs of 16.5 mg/L, 13.2 mg/L, and 10.9 mg/L 1,2-dichloropropane, respectively, EPA also calculated geometric means for 72- to 120-hour biomass NOEC and LOEC from Dow Chemical (2010) as 7.46 mg/L 1,2-dichloropropane and 13.3 mg/L 1,2-dichloropropane, respectively, in *S. costatum*. EPA calculated the geometric mean of this NOEC and LOEC, generating a ChV of 10.0 mg/L 1,2-dichloropropane for growth in *S. costatum*. In comparison, the 96-hour NOEC for green algae species *C. reinhardtii* was 38.0 mg/L (Schäfer et al., 1994). Green algae species (*Raphidocelis subcapitata*, previously *Pseudokirchneriella subcapitata*) exposed to measured concentrations of 1,1-dichloroethane for 72 hours in closed vessels reported no observed effects for growth at the highest tested concentration, 94.3 mg/L 1,1-dichloroethane (Mitsubishi Chemical Medience Corporation, 2009c). Similarly, green algae species (*Raphidocelis subcapitata*, previously *Selenastrum capricornutum*) exposed to measured concentrations of 1,2-dichloropropane for 120-hours in closed vessels (Dow Chemical, 1988) reported no observed effects for growth at the highest tested concentration (23.33–675.93 mg/L 1,2-dichloropropane), for which EPA calculated the geometric mean as 162 mg/L 1,2-dichloropropane (Table 4-7).

Table 4-7. Aquatic Organisms Environmental Hazard Studies for 1,1-Dichloroethane, Supplemented with 1,2-Dichloropropane and/or 1,1,2-Trichloroethane Data as Analogs

Study Type	Test Organism	Species	Endpoint	Hazard Values[a] (mg/L)	Geometric Mean[b] (mg/L)	Effect Endpoint(s)	Citation (Study Quality)
Acute	Fish	Japanese medaka (*Oryzias latipes*)	96-hour freshwater EC50	70.7		Behavior (abnormal swimming)	(Mitsubishi Chemical Medience Corporation, 2009b) (High)
		Fathead minnow (*Pimephales promelas*)	96-hour freshwater LC50	127 [119–135] [c]; 140 [131–150] [c]	**133.34**	Mortality	(Walbridge et al., 1983) (Medium); (Geiger et al., 1985) (High)
	Aquatic invertebrates	*Daphnia magna*	48-hour freshwater EC50	**34.3** [30.0–39.1]		Immobilization	(Mitsubishi Chemical Medience Corporation, 2009a) (High)
		Mysid shrimp (*Americamysis bahia*)	96-hour saltwater LC50	**24.79** [>4.92–26.62] [c] >26.65 [c]		Mortality	(Dow Chemical, 1988) (High)
		Mysid shrimp (*Americamysis bahia*)	96-hour saltwater NOEC/LOEC	4.92/6.89 [c]			(Dow Chemical, 1988) (High)
	Benthic invertebrates	*Chironomus riparius*	48-hour EC50	**150** [130–180]		Mortality and/or immobilization	(Smithers, 2024b) (High)
Chronic	Fish	Fathead minnow (*Pimephales promelas*)	32- to 33-day freshwater NOEC/LOEC	11/25[c]	16.58 (ChV)	Mortality (survival)	(Benoit et al., 1982) (High)
		Fathead minnow (*Pimephales promelas*)	32- to 33-day freshwater NOEC/LOEC	6/11[c]	8.12 (ChV)	Growth/ development (weight)	(Benoit et al., 1982) (High)
	Aquatic invertebrates	*Daphnia magna*	21-day freshwater EC50	6.67 [5.43–8.41]		Reproduction (young produced)	(Mitsubishi Chemical Medience Corporation, 2009d) (High)
		Daphnia magna	21-day freshwater NOEC/LOEC	0.525/1.64	**0.93** (ChV)	Reproduction (young produced)	(Mitsubishi Chemical Medience Corporation, 2009d) (High)
	Benthic invertebrates	*Ophryotrocha labronica*	15-day saltwater EC10	**68**[d]		Reproduction (hatchability)	(Rosenberg et al., 1975) (High)
		Chironomus riparius	2-generation freshwater NOEC/LOEC	66/130[d] 19/44[d e]	93 (ChV) **29** (ChV) [e]	Growth/development (decreased emergence)	(Smithers, 2023) (High)

Study Type	Test Organism	Species	Endpoint	Hazard Values[a] (mg/L)	Geometric Mean[b] (mg/L)	Effect Endpoint(s)	Citation (Study Quality)
Algae	*Skeletonema costatum*, *Chlamydomonas reinhardtii*		EC50	15.4 [12.9–17.9] –83[c]	35.4	Growth/development	(Schäfer et al., 1994) (Medium), (Dow Chemical, 2010) (Medium)
	Skeletonema costatum		NOEC	6.19–8.49[c]	7.64		(Dow Chemical, 2010) (Medium)
	Chlamydomonas reinhardtii		NOEC	38.0[c]			(Schäfer et al., 1994) (Medium)
	Skeletonema costatum		NOEC/LOEC	6.87–8.50/ 10.9–16.5[c]	**10.0** (ChV)		(Dow Chemical, 2010) (Medium), (Dow Chemical, 1988) (High)
	Raphidocelis subcapitata		NOEC	≥94.3			(Mitsubishi Chemical Medience Corporation, 2009c) (High)
	Raphidocelis subcapitata		NOEC	≥29.33–675.93[c]	162		(Dow Chemical, 1988) (High)

[a] Hazard values presented as ranges represent the range of all the definitive values in the citations and are presented with the number of significant figures reported by the authors.
[b] Geometric mean of definitive values only.
[c] Hazard values represented by analog 1,2-dichloropropane data.
[d] Hazard values represented by analog 1,1,2-trichloroethane data.
[e] Hazard values in mg/kg sediment.
Bolded values were used to derive COCs as described in Section 4.2.4. All values are listed individually with study quality in (U.S. EPA, 2025w) and (U.S. EPA, 2025t).

4.2.3 Terrestrial Species Hazard

EPA assigned overall quality determinations of high or medium to three acceptable terrestrial toxicity studies. These studies contained relevant 1,1-dichloroethane terrestrial toxicity data for one Norway rat (*Rattus norvegicus*) strain (Sprague-Dawley), one mouse (*Mus musculus*) strain (B6C3F1), and the Canadian poplar (*Populus x canadensis*). EPA identified these three terrestrial toxicity studies, displayed in Table 4-8, as the most relevant for quantitative assessment.

Terrestrial Vertebrates

Two relevant chronic toxicity studies for terrestrial vertebrates that reported no-observed-adverse-effect-level (NOAEL) and/or lowest-observed-adverse-effect level (LOAEL) information for 1,1-dichloroethane were assigned an overall quality level of high or medium with behavior (central nervous system [CNS] depression), growth, and/or mortality endpoints for rodents (species n = 2). No acceptable hazard studies were identified for avian species exposed to 1,1-dichloroethane. For terrestrial mammals and birds, relative contribution to total exposure associated with inhalation is generally minor in comparison to exposures by diet and indirect ingestion. EPA has quantitatively evaluated the relative contribution of inhalation exposures for terrestrial mammals and birds in previous peer-reviewed *Guidance of Ecological Soil Screening Levels (Eco-SSL)* (U.S. EPA, 2003a, b); therefore, EPA selected toxicity studies with oral exposure to 1,1-dichloroethane and not inhalation exposure to represent ecological hazard to terrestrial vertebrates.

Mammals

Observed effects occurred at relatively high doses (*e.g.*, LOAELs ≥1,000 mg/kg-bw/day) in rats and mice.

Behavior: EPA identified behavior data for terrestrial mammalian vertebrates from one studies (Muralidhara et al., 2001). In Muralidhara et al. (2001), authors observed moderate central nervous system depression (*e.g.*, progressive motor impairment and sedation) in Sprague-Dawley rats gavaged for 13 weeks with 2 g/kg-bw/day 1,1-dichloroethane. EPA subsequently adjusted that value as shown in (U.S. EPA, 2025s) for dosing number of days per week and maximum body weight (200 g) to calculate a NOAEL and LOAEL of 714 mg/kg-bw/day and 1,429 mg/kg-bw/day, respectively (Table 4-8).

Reproduction: No ecologically relevant adverse reproductive effects from 1,1-dichloroethane treatment were identified in rats and mice.

Growth: EPA identified growth data for terrestrial mammalian vertebrates from two studies (Muralidhara et al., 2001; NCI, 1978). Adverse growth effects were observed in rats but not mice. In a 10-day study where Sprague Dawley rats were gavaged daily with 1,1-dichloroethane, significantly decreased body weight was observed at the lowest dose administered, which was reported as a LOAEL of 1 g/kg-bw/day (Muralidhara et al., 2001) which the EPA then converted to a LOAEL of 1,000 mg/kg-bw/day (Table 4-8). In the same study, Sprague-Dawley rats were gavaged 5 times weekly for 13 weeks with 1,1-dichloroethane, and a NOAEL and LOAEL were established in the 13-week study for decreased body weight compared to the control group at 1.0 g/kg-bw/day and 2.0 g/kg-bw/day, respectively, which the EPA adjusted as shown in (U.S. EPA, 2025s) for dosing number of days per week to calculate a NOAEL and LOAEL of 714 mg/kg-bw/day and 1,429 mg/kg-bw, respectively.

A 78-week study tested for effects on several endpoints, including growth, in B6C3F1 mice gavaged 1,1-dichloroethane in corn oil five times weekly (NCI, 1978). No effect was observed for growth (mean body weight) in the 1,1-dichloroethane-treated B6C3F1 mice when compared to the control; therefore, a

time-weighted average NOAEL for growth was established as 2,885 mg/kg-bw/day for males and 3,331 mg/kg-bw/day for females as reported by NTP (NCI, 1978), which the EPA then adjusted for dosing number of days per week to 2,061 mg/kg-bw/day and 2,379 mg/kg-bw/day, respectively (Table 4-8). Within the same report (NCI, 1978), no effect on body weight was observed in male and female B6C3F1 mice gavaged five times weekly for 6 weeks with 1,1-dichloroethane in corn oil up to doses of 10,000 mg/kg/day. Therefore, a NOAEL of 10,000 mg/kg-bw/day was established by the authors, which EPA then adjusted as shown in (U.S. EPA, 2025s) for dosing number of days per week to 7,143 mg/kg-bw/day (Table 4-8).

Survival: EPA identified mortality data for terrestrial mammalian vertebrates from two studies (Muralidhara et al., 2001; NCI, 1978). Both studies demonstrated adverse effects on survival in rat and mice. In Muralidhara et al. (2001), a NOAEL and LOAEL for survival was established in male Sprague-Dawley rats gavaged five times weekly for 13 weeks with 1,1-dichloroethane. The highest tested dose group (4.0 g/kg) experienced significant mortality and were terminated at 11 weeks into the study with a NOAEL and LOAEL of 2 g/kg-bw/day and 4 g/kg-bw/day, respectively, which EPA then adjusted as shown in (U.S. EPA, 2025s) for dosing number of days per week and converted into a NOAEL of 1,429 mg/kg-bw/day and a LOAEL of 2,857 mg/kg-bw/day (Table 4-8). A 78-week NOAEL and LOAEL for survival were established in B6C3F1 female mice gavaged 1,1-dichloroethane in corn oil 5 times weekly (NCI, 1978), with the NOAEL and LOAEL reported as time-weighted averages of 1,665 mg/kg-bw/day and 3,331 mg/kg-bw/day, respectively, which EPA then adjusted for dosing number of days per week to a NOAEL of 1,189 mg/kg-bw/day and a LOAEL of 2,379 mg/kg/bw/day, respectively. Survival for female mice in the control, vehicle control, low dose and high dose groups within this study was 80%, 80%, 80%, and 50%, respectively.

Avian

No avian studies were available to assess potential hazards from 1,1-dichloroethane exposure.

Soil Invertebrates

No soil invertebrate studies were reasonably available to assess potential hazards from 1,1-dichloroethane exposure. Although comparison of 1,1-dichloroethane benthic and aquatic invertebrate hazard data to that of its analogs indicated that 1,2-dichloropropane, 1,1,2-trichloroethane, and 1,2-dichloroethane soil invertebrate hazard data would be suitable for read-across to 1,1-dichloroethane (Table 4-6), no reasonably available analog data were identified that could be used quantitatively to derive a hazard threshold. Available soil invertebrate hazard data for analog 1,2-dichloropropane was determined Uninformative (Neuhauser et al., 1986). Available soil invertebrate hazard data for analogs 1,1,2-trichloroethane and 1,2-dichloroethane were assigned overall quality determination of high (Neuhauser et al., 1985). A 48-hour contact exposure of earthworms to 1,1,2-trichloroethane or 1,2-dichloroethane applied to filter paper reported mortality LC50 values of 42 and 60 μg/cm^2, respectively (Neuhauser et al., 1985). However, because the filter paper contact test is not considered a relevant exposure pathway for soil invertebrates due to the absorbed amount of chemical to earthworm via dermal contact being uncertain, EPA did not establish a hazard threshold from the 1,1,2-trichloroethane earthworm hazard data. A 14-day LC50 toxicity prediction of 181 mg/L 1,1-dichloroethane for earthworm can be generated from the neutral organics category using U.S. EPA's Ecological Structure Activity Relationships (ECOSAR) Prediction Model (v2.2) (U.S. EPA, 2022d). The neutral organics category in ECOSAR includes data from several species of earthworm, including data from *Eisenia fetida* (U.S. EPA, 2022d).

Terrestrial Plants

For terrestrial plant species, one medium-quality study was identified by EPA as relevant for quantitative assessment (Table 4-8). (Dietz and Schnoor, 2001) reported zero-growth and 50 percent transpiration reduction concentrations in Canadian poplar seedlings for a 2-week exposure to 1,1-dichloroethane in growth medium (EC0 and EC50 values of 1,059 mg/L and 802 mg/L, respectively).

Table 4-8. Terrestrial Organisms Environmental Hazard Studies Used for 1,1-Dichloroethane

Duration	Test Organism (Species)	Endpoint	Hazard Values (mg/kg-bw/day) [a]	Effect	Citation (Data Evaluation Rating)
Terrestrial vertebrates					
13 weeks (subchronic)	Sprague-Dawley Rat (*Rattus norvegicus*)	NOAEL/ LOAEL	714/1,429	Behavior (CNS depression)	(Muralidhara et al., 2001) (Medium)
10 days (short-term)	Sprague-Dawley Rat (*Rattus norvegicus*)	LOAEL	1,000	Growth (body weight)	(Muralidhara et al., 2001) (High)
13 weeks (subchronic)	Sprague-Dawley Rat (*Rattus norvegicus*)	NOAEL/ LOAEL	714/1,429	Growth (body weight)	(Muralidhara et al., 2001) (High)
78 weeks (chronic)	B6C3F1 Mouse (*Mus musculus*)	NOAEL	2,061	Growth (body weight, male)	(NCI, 1978) (High)
78 weeks (chronic)	B6C3F1 Mouse (*Mus musculus*)	NOAEL	2,379	Growth (body weight, female)	(NCI, 1978) (High)
6 weeks (subchronic)	B6C3F1 Mouse (*Mus musculus*)	NOAEL	7,143	Growth (body weight)	(NCI, 1978) (High)
13 weeks (subchronic)	Sprague-Dawley Rat (*Rattus norvegicus*)	NOAEL/ LOAEL	1,429/2,857	Survival	(Muralidhara et al., 2001) (High)
78 weeks (chronic)	B6C3F1 Mouse (*Mus musculus*)	NOAEL/ LOAEL	1,189/2,379	Survival	(NCI, 1978) (High)
Terrestrial plants					
14 days (short-term)	Canadian poplar (*Populus x canadensis*)	EC50	**802 mg/L**	Transpiration	(Dietz and Schnoor, 2001) (Medium)

Bolded values used to derive hazard thresholds for terrestrial species as described in Section 4.2.4. All values are listed individually with study quality in (U.S. EPA, 2025x) and (U.S. EPA, 2025t).

4.2.4 Weight of Scientific Evidence Conclusions for Environmental Hazards

Based on the weight of the scientific evidence and uncertainties, a confidence statement was developed that qualitatively ranks (*i.e.*, robust, moderate, slight, or indeterminate) the confidence in the hazard threshold. The evidence considerations and criteria detailed within the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025z) guide the application of strength-of-evidence judgments for environmental hazard effect within a given evidence stream as described in Appendix K. That appendix also summarizes how these considerations were determined for each environmental hazard threshold.

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

Due to the robust confidence in quality of the database, consistency, strength and precision, and biological response, an overall hazard confidence rating of robust was assigned to the acute aquatic assessment (Appendix Q). As a result of moderate confidence in most considerations and having

empirical acute hazard data for only a single benthic species, an overall hazard confidence rating of moderate was assigned to the acute benthic assessment. Due to the robustness in strength and precision, observed dose-response, and relevance, a robust confidence was assigned to the chronic aquatic assessment. Because of the moderate confidence in quality of the database and consistency and read-across from 1,1,2-trichloroethane data, a moderate confidence was assigned to the chronic benthic assessment. Due to the moderate confidence in the number of studies, consistency, and relevance, an overall hazard confidence rating of moderate was assigned to the algal assessment (Appendix Q). Owing to the moderate confidence in number of studies, consistency, and strength and magnitude of effect, an overall hazard confidence of moderate was assigned to the terrestrial mammalian assessment. Due to the slight confidence in number of studies, consistency, and relevance, an overall hazard confidence of slight was assigned to the terrestrial plant assessment (Appendix Q). Indeterminate ratings were assigned to the confidence for the avian and soil invertebrate assessments due to lack of reasonably available data. A more detailed explanation of the weight of scientific evidence, uncertainties, and overall confidence for the 1,1-dichloroethane environmental hazard evidence is presented below in Section 4.2.4.1.1.

4.2.4.1.1 Confidence in the Environmental Hazard Data Set

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

For the acute aquatic assessment, the database consisted of four studies with overall quality determinations of high with both aquatic invertebrates and vertebrates represented. Data from three of these studies were supplemented using Web-ICE to generate a subsequent SSD output, therefore a robust confidence was assigned to quality of the database. Outcomes in the empirical hazard data and their corresponding ECOSAR toxicity predictions were generally consistent with the majority of toxicity values falling within a log scale of each other. For example, the ECOSAR acute toxicity daphnid prediction for 1,1-dichloroethane was in good agreement with the 1,1-dichloroethane empirical hazard value for *Daphnia magna* (69.9 vs. 34.3 mg/L, respectively) as was the analog 1,2-dichloropropane fish acute toxicity prediction in close agreement with the respective 1,2-dichloropropane empirical hazard value (94.8 vs. 133.34 mg/L, respectively). Although the ECOSAR 1,2-dichloropropane prediction for mysid shrimp was in less agreement with the 1,2-dichloropropane empirical toxicity value for mysid shrimp, the predictions were still within three to four-fold of the empirical hazard datapoint (89.3 mg/L vs. 24.79 mg/L, respectively). Therefore, a robust confidence was assigned to consistency of the acute aquatic assessment. The effects observed in the 1,1-dichloroethane and 1,2-dichloropropane empirical dataset for acute aquatic assessment were immobilization, abnormal swimming, and mortality, and EC50 (*Daphnia magna*) and LC50 (fathead minnow and mysid shrimp) values were reported in the three species utilized in the SSD analysis with additional predicted LC50 values reported from Web-ICE, therefore a robust confidence was assigned to the strength and precision consideration (Appendix Q).

For the acute benthic assessment, the database consisted of a measured 48-hour EC50 and LC50 in *C. riparius* exposed in water to 1,1-dichloroethane from a TSCA section 4(a)2 test order and 96-hour LC50 toxicity predictions for 36 benthic invertebrates based on empirical fish and aquatic invertebrate data for 1,1-dichloroethane and analog 1,2-dichloropropane (Table_Apx K-1). EPA determined this to be a sufficient number of benthic invertebrate predictions but acknowledging the fact that there were acute toxicity data for a single species of sediment-dwelling organisms exposed to 1,1-dichloroethane, a moderate confidence was assigned to quality of the database. Moderate confidence was assigned to the consistency consideration for the acute benthic assessment since the *C. riparius* empirical hazard data (150 mg/L or 150,000 ppb) were reasonably consistent with Web-ICE toxicity predictions of two species within the same genus; *C. tentans* (150,361 ppb) and *C. plumosus* (14,926 ppb). Similarly,

robust confidence was assigned to the strength and precision consideration as measured mortality and/or immobilization was indicated in one species and predicted mortality in 34 species.

For the chronic aquatic assessment, the database consisted of two studies with overall quality determinations of high (one study containing 1,1-dichloroethane hazard data obtained according to OECD Guideline for the Testing of Chemicals, 211 and the other study containing analog 1,2-dichloropropane hazard data), resulting in moderate confidence for quality of the database. Outcomes differed by taxa with mortality and growth effects observed in fathead minnow based on analog hazard data and reproductive effects observed in *Daphnia magna* based on 1,1-dichloroethane hazard data. 1,1-Dichloroethane and 1,2-dichloropropane ECOSAR chronic toxicity predictions were consistent with the 1,2-dichloropropane chronic fish toxicity hazard value (*e.g.*, ChV predictions of 12.0 mg/L 1,1-dichloroethane and 9.3 mg/L 1,2-dichloropropane compared to the empirical ChV 8.12 mg/L 1,2-dichloropropane), whereas the 1,1-dichloroethane chronic hazard prediction for daphnid was in less agreement but still within 10-fold of the 1,1-dichloroethane empirical hazard value for *Daphnia magna* utilized in setting the hazard threshold (6.5 vs. 0.93 mg/L, respectively). Therefore, a moderate confidence was assigned to the consistency consideration. In the two chronic studies, reproductive and growth effects were considered the most sensitive endpoints with high doses resulting in approximately 25 percent of control values for those endpoints. Therefore, a robust confidence was assigned to the strength and precision consideration for the chronic aquatic assessment (Appendix Q).

For the chronic benthic assessment, the database consisted of two studies with overall quality determinations of high or medium based on analog hazard data. One of the studies is a TSCA section 4(a)(2) test order report conducted according to OECD Guideline for the Testing of Chemicals, Guideline 233 ("Sediment-Water Chironomid Life-Cycle Toxicity Test Using Spiked Water or Spiked Sediment"); the second study was a high-rated exposure of *Ophryotrocha labronica* in water, resulting in moderate confidence for quality of the database. Outcomes occurred in offspring of both studies (% emerged or hatched) therefore a moderate confidence was assigned for consistency in chronic benthic assessment. Percent of *O. labronica* eggs hatched decreased to 0 percent at higher 1,1,2-trichloroethane concentrations, and emergence in the second-generation (F1) larvae in the 1,1,2-trichloroethane test order report was approximately 50 percent of the control treatment emergence. Additionally, the definitive chironomid emergence result is qualitatively supported by similar findings in the preliminary 2-generation screening study in the same study report where percent emergence at the high dose was less than 20 percent that of the control treatment; therefore, the strength and precision consideration was assigned robust confidence (Appendix Q).

For the algal assessment, the database consisted of one study with an overall quality determination of high containing 1,1-dichloroethane hazard data and three high- or medium-rated studies based on analog (1,2-dichloropropane) data resulting in a moderate confidence for quality of the database. Outcomes were consistent for two of the three algal species (*e.g.,* showing growth inhibition effects at comparable concentrations) whereas the third species showed no effect on growth to the highest concentrations tested across two studies, therefore a moderate confidence was assigned to the consistency consideration. The endpoints were based on growth reduction in algae, with 1,2-dichloropropane EC50 values achieved in two of the studies. Additionally, ECOSAR ChV predictions for 1,1-dichloroethane and 1,2-dichloropropane (12.1 and 10.4 mg/L, respectively) were closely aligned with the ChV utilized for the algal hazard threshold (10.0 mg/L); therefore, a robust confidence was assigned to the strength and precision consideration for the algal assessment (Appendix Q).

For terrestrial mammal assessment, no wildlife studies were available from systematic review; however, two studies with overall quality determinations of high representing two species (mice and rats), were

used from human health animal model studies. A TRV derived from the mammal studies was used to calculate the hazard threshold in mg/kg-bw. The terrestrial mammal data suggest potential trends (*e.g.*, species-specific growth effects, effects on survival); however, the ability to fully assess these trends for consistency is limited by the low number of studies. Regarding strength of the effect, mortality was substantial in the datum representing the TRV (≈40% reduction in survival) whereas reduction in growth, although significant, was smaller in magnitude. Moderate confidence was assigned to quality of the database, consistency, and strength and precision for the terrestrial mammalian assessment (Appendix Q).

For the terrestrial plant assessment, a single study with an overall quality determination of medium was available for the Canadian poplar resulting in slight confidence for the quality of the database. The terrestrial plant study measured growth inhibition and transpiration reduction effects. The single terrestrial plant study was insufficient to characterize consistency in the outcome resulting in slight confidence for consistency. For strength of effect in the terrestrial plant assessment, reduction in transpiration was substantial (50% reduction achieved); therefore, moderate confidence was assigned to this consideration.

Biological Gradient/Dose-Response

All studies used for calculating hazard thresholds contained multiple doses. For the acute aquatic and benthic assessments, measured effects were noted at increased doses and increased with duration (particularly for the fish data), therefore a robust confidence was assigned to this consideration for both assessments. For the chronic acute assessment, increase in effect was observed as chemical concentration increased, therefore a robust confidence was assigned to this consideration. For the chronic benthic assessment, decrease in percent eggs hatched and second-generation larval emergence was observed as chemical concentration increased, therefore a robust confidence was assigned to this consideration. For the algal assessment, when effects were noted, the effects increased as chemical dose and duration increased but was not demonstrated across species, therefore a moderate confidence was assigned to this consideration.

For terrestrial mammalian assessment, effects were generally noted at higher 1,1-dichloroethane concentrations and increased over duration, therefore robust confidence was assigned to this consideration. For the terrestrial plant assessment, there is evidence of dose-response with both reported endpoints (zero-growth and transpiration reduction); however, the zero-growth concentration was extrapolated outside the tested concentrations of 1,1-dichloroethane, therefore moderate confidence was assigned to this consideration (Appendix Q).

Relevance (Biological; Physical/Chemical; Environmental)

For the acute aquatic assessment, immobilization and mortality were noted in the empirical data for freshwater and saltwater aquatic invertebrates and a freshwater fish, all three of which are considered representative test species for aquatic assessments, and mortality was predicted in additional species. Although, modeled approaches such as Web-ICE can have more uncertainty than empirical data when determining the hazard or risk, the use of the probabilistic approach within this risk evaluation increases confidence compared to a deterministic approach and the use of the lower 95 percent CI instead of a fixed AF also increases confidence, as it is a more data-driven way of accounting for uncertainty. Two of the three species with empirical hazard data were exposed to 1,2-dichloropropane rather than 1,1-dichloroethane. Although EPA concludes that 1,2-dichloropropane is a robust analog for the environmental hazard read-across to 1,1-dichloroethane, the use of an analog still affects the physical and chemical relevance of the hazard confidence; therefore, a moderate confidence was assigned to the relevance consideration for the acute aquatic assessment (Appendix Q).

For the acute benthic assessment, mortality was measured in common test species *C. riparius* and mortality predictions were observed in 34 benthic invertebrates, including representative test species such as *Lumbriculus variegatus* and *Gammarus fasciatus*. The 1,1-dichloroethane exposure to *C. riparius* larvae took place in water with the assumption that this translates to exposure via sediment pore-water; however the exposure is not as relevant as if the exposures were administered in the sediment with 1,1-dichloroethane pore water concentration measurements taken. The empirical hazard data were derived from exposure to 1,1-dichloroethane; however, the predictions were based in part on empirical analog data (1,2-dichloropropane), therefore a moderate confidence was assigned to the relevance consideration for the acute benthic assessment (Appendix Q).

For the chronic aquatic assessment, ecologically relevant population level effects (reproductive, growth, mortality) were observed in two different species (*Daphnia magna* and fathead minnow), both of which are considered representative test species for aquatic toxicity tests. Although the *Daphnia magna* study utilized semi-static renewal, chemical measurements were obtained, and the fathead minnow study utilized flow-through conditions which is environmentally relevant for chronic exposure. In the case of the study on which the chronic aquatic threshold was based, the exposure was to 1,1-dichloroethane. Therefore, robust confidence was assigned to the relevance consideration for the chronic aquatic assessment.

For the chronic benthic assessment, an ecologically relevant population level effect (emergence) was observed in a representative species (*Chironomus riparius*) for benthic toxicity tests whereas *Ophryotrocha labronica*, a marine annelid, is less represented in the literature as a test species. Regarding physical and chemical relevance, the exposure was to 1,1,2-trichloroethane rather than 1,1-dichloroethane even though EPA concludes that 1,1,2-trichloroethane is an appropriate analog for environmental hazard read-across to 1,1-dichloroethane. Regarding environmental relevance, in the study exposing *C. riparius*, the test was conducted with sediment present in the system which is environmentally relevant for benthic exposure; however, the chemical exposure was administered at the beginning of each sediment exposure phase with 1,1,2-trichloroethane concentrations in sediment and benthic pore water significantly decreasing over the duration of the exposure phase (therefore not truly representative of chronic exposure in the benthic environment). The second study exposed *O. labronica* to 1,1,2-trichloroethane in aqueous conditions without sediment present in the system. Therefore, slight confidence is assigned to relevance.

For the algal assessment, similar effects were observed in two different species (a marine diatom and a green algae species), both of which are considered representative test species for algal toxicity tests, and the testing likely encompassed several generations of algae; however, a definitive approach was utilized with an AF of 10 to account for uncertainty when applying results from these two species to all algal species. The algal testing took place in aqueous growth medium which is considered environmentally relevant but was conducted with 1,2-dichloropropane rather than 1,1-dichloroethane. Therefore, a moderate confidence was assigned to the relevance consideration for the algal assessment (Appendix Q).

Regarding biological relevance and physical/chemical relevance for the terrestrial mammalian assessment, ecologically relevant population-level effects include behavior, growth, and mortality, and these data were on 1,1-dichloroethane. The TRV was established using a mortality endpoint in female mice; which is considered an ecologically relevant apical effect in mammalian receptors. It should be noted that the studies utilized gavage administration which could be considered less environmentally relevant than other methods of administration such as via drinking water or feed. Nevertheless, moderate confidence was assigned to the relevance consideration for the terrestrial mammal assessment (Appendix Q).

The ecologically relevant population level effects in the terrestrial plant assessment include lack of growth (zero-growth) and reduced transpiration (which would be a proxy for reduced growth/ Development, even though the endpoint is reported as respiratory) and the testing was performed with 1,1-dichloroethane. However, testing was performed in a single species in growth medium that could be considered less environmentally relevant than tests conducted in soil. Therefore, a slight confidence was assigned to the relevance consideration for the terrestrial plant assessment (Appendix Q).

4.2.5 Environmental Hazard Thresholds

EPA calculates hazard thresholds to identify potential concerns to aquatic and terrestrial species. For aquatic species, the hazard threshold is called a concentration of concern (COC), and for terrestrial species, the hazard threshold is called a hazard value or toxicity reference value (TRV). These terms (COC, TRV, and hazard value) describe how the values are derived and can encompass multiple taxa or ecologically relevant groups of taxa as the environmental risk characterization serves populations of organisms within a wide diversity of environments. See Section 4.2.4.1.1 and Appendices K and Q for more details on how EPA weighed the scientific evidence. After weighing the scientific evidence, EPA selects the appropriate toxicity value from the integrated data to use for hazard thresholds.

For aquatic species, EPA estimates hazard by calculating a COC for a hazard threshold. COCs can be calculated using a deterministic method by dividing a hazard value by an AF according to EPA methods as defined in Equation 4-1 (U.S. EPA, 2016b, 2013b, 2012b).

Equation 4-1.

$$COC = toxicity\ value \div AF$$

COCs can also be calculated using probabilistic methods. For example, an SSD can be used to calculate a hazardous concentration for 5 percent of species (HC05). The HC05 estimates the concentration of 1,1-dichloroethane that is expected to be protective for 95 percent of species. This HC05 can then be used to derive a COC, and the lower bound of the 95 percent confidence interval of the HC05 can be used to account for uncertainty instead of applying an AF. EPA has more confidence in the probabilistic approach when enough data are available because an HC05 is representative of a larger portion of species in the environment. The use of the lower 95 percent CI instead of a fixed AF of 5 also increases confidence as it is a more data-driven way of accounting for uncertainty (EPA-HQ-OPPT-2023-0265).

For terrestrial species, EPA estimates hazard by calculating a TRV, in the case of terrestrial mammals and birds, or by assigning the hazard value as the hazard threshold in the case of terrestrial plants and soil invertebrates. EPA prefers to derive the TRV by calculating the geometric mean of the NOAELs across sensitive endpoints (growth and reproduction) rather than using a single endpoint. The TRV method is preferred because the geometric mean of NOAELs across studies, species, and endpoints provides greater representation of environmental hazard to terrestrial mammals and/or birds. However, when the criteria for using the geometric mean of the NOAELs as the TRV are not met (according to methodology described in Appendix K.2.2), the TRVs for terrestrial mammals and birds are derived using a single endpoint.

4.2.5.1 Aquatic Species COCs

EPA derived two acute COCs, two chronic COCs, and an aquatic plant COC using a combination of probabilistic and deterministic approaches with 1,1-dichloroethane hazard data supplemented with read-across from 1,2-dichloropropane and 1,1,2-trichloroethane. Algae was assessed separately and not incorporated into acute or chronic COCs, because durations normally considered acute for other species

(*e.g.*, up to 96 hours) can encompass several generations of algae. See Appendix K for additional information on methods used to derive COCs. Table 4-9 summarizes the aquatic hazard thresholds.

Acute Aquatic Threshold

Due to few reasonably available acute toxicity data for aquatic organisms exposed to 1,1-dichloroethane, for the acute aquatic COC, EPA used the 48-hour 1,1-dichloroethane EC50 immobilization data from *Daphnia magna* and the 96-hour 1,2-dichloropropane LC50 toxicity data from mysid shrimp and fathead minnow (Table 4-7) as surrogate species to predict LC50 toxicity values for 75 additional aquatic organisms (25 fish, 2 amphibian, and 48 aquatic invertebrate species) using the Web-ICE application as described in Appendix K.2.1.1 (Raimondo and Barron, 2010). *Chironomus riparius* was not available as a surrogate species in Web-ICE however the *C. riparius* immobilization and mortality data from 48-hour exposure to 1,1-dichloroethane were included in the SSD. The test species (n = 4) and predicted species (n = 76) toxicity data were then used to calculate the distribution of species sensitivity to 1,1-dichloroethane and 1,2-dichloropropane exposure (as read-across to 1,1-dichloroethane) through the SSD toolbox as shown in Appendix K.2.1.2 (Etterson, 2020). The calculated HC05 was 11,170 µg/L (95 percent CI = 8,931 to 14,370 µg/L) (Figure_Apx K-1). The lower 95 percent CI of the HC05, 8,931 µg/L, was then used as the acute aquatic COC.

Acute Benthic Threshold

Due to few reasonably available acute toxicity data for benthic organisms exposed to 1,1-dichloroethane for the benthic acute COC, EPA used empirical hazard data for water-column and benthic organisms in combination with modeled data from the Web-ICE application (Raimondo and Barron, 2010) to assess acute hazard to sediment-dwelling organisms as described above. In addition to the single measured hazard value for benthic species exposed to 1,1-dichloroethane, predicted 96-hour LC50 values were generated for 34 benthic invertebrates based on empirical data for 1,1-dichloroethane and the analog 1,2-dichloropropane (Table_Apx K-1). Because the benthic invertebrate predicted hazard values were represented relatively equally in the low, middle, and high portions of the SSD (Figure_Apx K-4), EPA used the lower 95 percent CI of the calculated HC05 resulting from the above SSD analysis to represent the acute COC for sediment-dwelling organisms. This resulted in an acute benthic COC of 8,931 µg/L or ppb to be compared to benthic pore water exposures.

Chronic Aquatic Threshold

The chronic aquatic COC was derived from the 1,1-dichloroethane ChV of the 21-day LOEC/NOEC of 0.93 mg/L for the aquatic invertebrate *Daphnia magna* with the application of an AF of 10. The ChV for *Daphnia magna* was the most sensitive chronic endpoint represented in Table 4-7 for aquatic vertebrates and invertebrates representing effects of reproductive inhibition of adult *Daphnia magna* (Mitsubishi Chemical Medience Corporation, 2009d).

Chronic Benthic Thresholds

Due to the lack of reasonably available chronic toxicity data for benthic organisms exposed to 1,1-dichloroethane and the chronic benthic COCs were derived from the 1,1,2-trichloroethane 15-day EC10 of 68 mg/L for *Ophryotrocha labronica* with the application of an AF of 10 and from the 1,1,2-trichloroethane ChV of the 2-generation LOEC/NOEC of 29 mg/kg for *Chironomus riparius* with the application of an AF of 10. The EC10 for *O. labronica* was the most sensitive hazard value for benthic species exposed to 1,1,2-trichloroethane and represents reproductive effects on hatching (Rosenberg et al., 1975), and the ChV for *C. riparius* was the single sediment hazard value for benthic species representing growth and development effects for second generation larvae (Smithers, 2023).

Aquatic Plant Threshold

Due to the lack of reasonably available toxicity data with definitive hazard for aquatic plants exposed to 1,1-dichloroethane, the algal COC was derived from the 1,2-dichloropropane ChV of the 72-120 hour NOEC/LOEC of 10.0 mg/L for *Skeletonema costatum* with the application of an AF of 10. The ChV for *S. costatum* was carefully recalculated in Dow Chemical (2010) from data in a robust study (Dow Chemical, 1988) and represents growth and development effects over multiple generations.

Table 4-9. Environmental Hazard Thresholds for Aquatic Environmental Toxicity

Environmental Aquatic Toxicity	Analog	Hazard Value (ppb)	Assessment Factor (AF)	COC (ppb)	Assessment Medium
Acute aquatic exposure: Lower 95% CI of HC05 from SSD	1,1-dichloroethane and 1,2-dichloropropane	8,931	NA[a]	8,931	Water column
Acute benthic exposure: Lower 95% CI of HC05 from SSD	1,1-dichloroethane and 1,2-dichloropropane	8,931	NA[a]	8,931	Benthic pore water
Chronic aquatic exposure: based on aquatic invertebrate ChV	1,1-dichloroethane	930	10	93	Water column
Chronic benthic exposure: based on benthic invertebrate EC10	1,1,2-trichloroethane	68,000	10	6,800	Benthic pore water
Chronic benthic exposure: based on benthic invertebrate ChV	1,1,2-trichloroethane	29,000[b]	10	2,900[b]	Sediment
Aquatic plant exposure: based on algae ChV	1,2-dichloropropane	10,000	10	1,000	Water column

[a] EPA used the lower 95% confidence interval (CI) of the HC05 to account for uncertainties rather than an AF.
[b] Values in mg/kg, otherwise, hazard values in mg/L.

4.2.5.2 Terrestrial Species Hazard Values

For terrestrial species exposed to 1,1-dichloroethane EPA identified hazard values (thresholds) for terrestrial vertebrates and plants. Table 4-10 summarizes the environmental hazard thresholds for terrestrial species.

Terrestrial Vertebrate Threshold

EPA estimated hazard for terrestrial vertebrates by calculating a toxicity reference value (TRV), for mammals (Figure 4-2). For terrestrial mammals, the TRV is expressed as doses in units of mg/kg-bw/day. Although the TRV for 1,1-dichloroethane is derived from laboratory mice and rat studies, body weight is normalized, therefore the TRV can be used with ecologically relevant wildlife species to evaluate chronic dietary exposure to 1,1-dichloroethane. Representative wildlife species chronic hazard threshold will be evaluated in the trophic transfer assessments using the TRV. The following criteria were used to select the data to calculate the TRV with NOAEL and/or LOAEL data (U.S. EPA, 2007). For more details see Appendix K.2.2.

Step 1: The minimum data set required to derive either a mammalian or avian TRV consists of three results (NOAEL or LOAEL values) for reproduction, growth, or mortality for at least two mammalian or avian species.

- Because this condition was met, proceed to step 2.

Step 2: Calculation of a geometric mean requires at least three NOAEL results from the reproduction and growth effect groups.

- Because this condition was met, then proceed to step 4.

Proceed from Step 2 to Step 4[17]: When the geometric mean of the NOAEL for reproduction and growth is higher than the lowest bounded LOAEL for reproduction, growth, or mortality,

- Then the TRV is equal to the highest bounded NOAEL below the lowest bounded LOAEL.

For 1,1-dichloroethane, the geometric mean of NOAELs for growth endpoints is 2,240 mg/kg-bw/day which is higher than the lowest bounded LOAEL for reproduction, growth, or mortality (1,429 mg/kg-bw/day, growth). Therefore, according to the decision flowchart in Appendix K.2.2, the TRV was set as the highest bounded NOAEL below the lowest bounded LOAEL for reproduction, growth, or mortality, resulting in a TRV of 1,189 mg/kg-bw/day (mortality in female mice) (Figure 4-2). The TRV is representative of various exposure durations (*e.g.*, chronic [>90 days], subchronic [>30–90 days], short-term [>3 to 30 days]) with the exception of an acute exposure duration. This is reflective of the COUs where dietary exposure by trophic transfer is assessed from releases to surface water and daily maximum deposition and/or annual land application of 1,1-dichloroethane to soil.

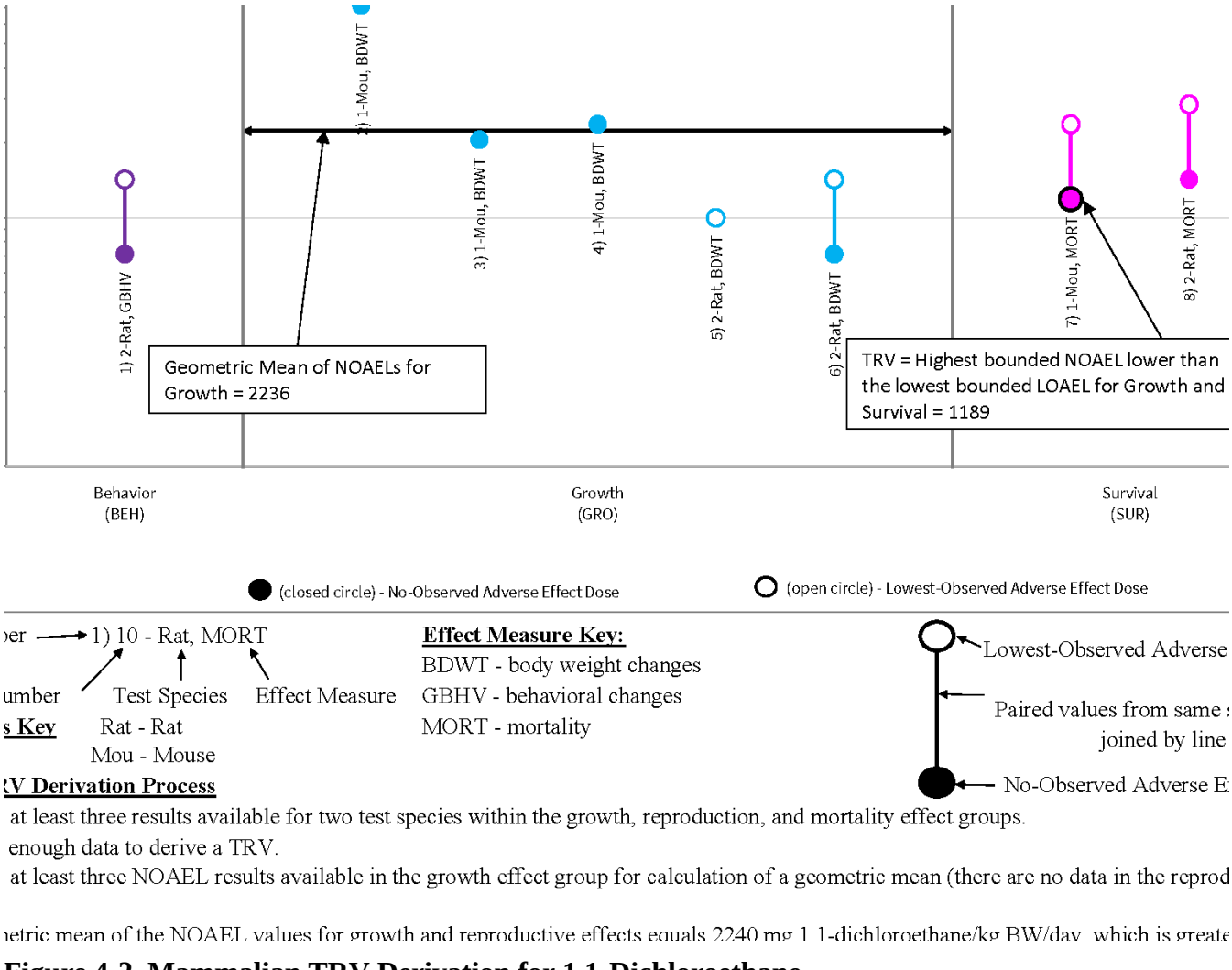


Figure 4-2. Mammalian TRV Derivation for 1,1-Dichloroethane

Terrestrial Plant Threshold

The terrestrial plant hazard threshold was derived from the 1,1-dichloroethane 2-week EC50 of 802 mg/L for *Populus x canadensis* (Canadian poplar). The EC50 for *Populus x canadensis* was the most

[17] Steps of the selection process are detailed in Appendix K.2.2.

sensitive hazard value in the single terrestrial plant reference representing transpiration effects for seedlings (Dietz and Schnoor, 2001).

Table 4-10. Environmental Hazard Thresholds for Terrestrial Environmental Toxicity

Environmental Terrestrial Toxicity	Analog	Hazard Value or TRV	Assessment Medium
Mammal: TRV	NA	1,189 mg/kg-bw/day	Dietary (trophic transfer)
Avian	NA	No data	No data
Soil invertebrate	NA	No data	No data
Terrestrial plant (*Populus x canadensis*): based on EC50	NA	802 mg/L	Soil porewater
NA = not applicable, data derived from 1,1-dichloroethane			

4.3 Environmental Risk Characterization

1,1-Dichloroethane – Environmental Risk Characterization (Section 4.3): Key Points

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

- RQs equal to 1 indicate that environmental exposures are the same as the hazard threshold. If the RQ exceeds 1, the exposure is greater than the hazard threshold. If the RQ is below 1, the exposure is less than the hazard threshold.
- For aquatic species in the water column, chronic RQs based on a hazard-based 21-day release to surface waters are above 1 and have corresponding days of exceedance equal to or greater than 21 days for five out of seven COUs evaluated quantitatively for risk to aquatic species from surface water releases. For algal species, an RQ based on a 21-day release to surface water is above 1 and has corresponding days of exceedance equal to or greater than 4 days for the Manufacturing COU.
    - No acute RQs exceeded 1 for aquatic species in the water column or sediment compartment for seven COUs evaluated quantitatively for risk to aquatic species from surface water releases. Chronic RQs based on total number of operating days are below 1 for aquatic species in the water column or sediment compartment for all seven COUs.
    - Because EPA lacked information on estimated days of release to surface waters, exposure durations are based on a hazard-based release duration or the total number of operating days.
    - Analog data were used to assess hazard in the water column—specifically, algal hazard and partial use in acute hazard—and in the sediment compartment. 1,1-Dichloroethane data were used to determine the exposure. The methodology demonstrating robustness of the analog selection is described in Section 4.2.1.1.
    - Because of 1,1-dichloroethane's high water solubility and releases to surface water, biota in the water column are particularly susceptible to 1,1-dichloroethane exposure. This could have potential community-level impacts from chronic 1,1-dichloroethane exposures in the water column.
    - EPA has robust confidence in the RQ inputs for the acute and chronic aquatic assessments and moderate confidence in the RQ inputs for the algal and benthic assessments.
- RQs were below 1 for five COUs evaluated quantitatively and expected to be below 1 for eight COUs evaluated qualitatively for risk to terrestrial species from air deposition and biosolids land application.
    - EPA has slight confidence in the RQ inputs for the terrestrial plant assessments.
    - EPA has moderate confidence in the RQ inputs for the screening level trophic transfer assessment.
    - RQs calculated for five COUs were below 1 for dietary exposure of 1,1-dichloroethane to representative insectivorous (shrew) and herbivorous (vole) mammals via trophic transfer using calculated soil and soil pore water concentrations resulting from air deposition or biosolid land application.
    - RQs for five COUs were below 1 for semi-aquatic terrestrial receptors (mink) via trophic transfer from fish and crayfish using the highest modeled 1,1-dichloroethane surface water concentrations and corresponding benthic pore water concentrations.

EPA considered fate, exposure, and environmental hazard to characterize the environmental risk of 1,1-dichloroethane. For environmental receptors, the Agency quantitatively estimated risks to aquatic species via water and sediment (including benthic pore water and sediment), and to terrestrial species via exposure to soil and soil pore water by air deposition and biosolids land application, and diet through

trophic transfer. Risk estimates to aquatic-dependent terrestrial species were conducted to include exposures to 1,1-dichloroethane via diet, water, and incidental ingestion of sediment. As described in Section 2.2.2, when released to the environment, 1,1-dichloroethane is expected to partition primarily to air (85%) with lesser amounts to water (15%), sediment (<1%) and soil (<1%). Based on its physical chemical properties, 1,1-dichloroethane is not likely to accumulate in sediment, soil, wastewater biosolids or biota and is not described as persistent and bioaccumulative. Direct exposure of 1,1-dichloroethane to terrestrial receptors via air was not assessed quantitatively, because dietary exposure was determined to be the driver of exposure to wildlife. In general, for terrestrial mammals and birds, relative contribution to total exposure associated with inhalation is secondary in comparison to exposures by diet and indirect ingestion. EPA has quantitatively evaluated the relative contribution of inhalation exposures for terrestrial mammals and birds in previous peer-reviewed *Guidance of Ecological Soil Screening Levels (Eco-SSL)* (U.S. EPA, 2003a, b).

Section 4.2 details reasonably available environmental hazard data and indicated that 1,1-dichloroethane presents hazard to aquatic and terrestrial organisms. For acute exposures, 1,1-dichloroethane, supplemented with analog 1,2-dichloropropane data, is a hazard to aquatic animals in the water-column and sediment at 8,931 ppb based on the lower 95 percent CI of the HC05 resulting from an SSD utilizing EPA's Web-ICE (Raimondo and Barron, 2010) and SSD toolbox applications (Etterson, 2020). For chronic exposures, 1,1-dichloroethane is a hazard to aquatic organisms in the water column with a ChV of 930 ppb for aquatic invertebrates. For exposures to algal species, 1,1-dichloroethane, based on analog 1,2-dichloropropane, is a hazard to algae in the water column with a ChV of 10,000 ppb. For chronic exposures to sediment-dwelling organisms, 1,1-dichloroethane, based on analog 1,1,2-trichloroethane, is a hazard with ChVs of 68,000 and 29,000 ppb in benthic pore water and sediment, respectively, for sediment-dwelling invertebrates. For terrestrial exposures, 1,1-dichloroethane is a hazard to mammals at 1,189 mg/kg-bw/day and a hazard to terrestrial plants with a hazard value of 802,000 ppb. As detailed in Section 4.2.4.1.1, EPA considers the evidence for aquatic hazard thresholds robust, algal thresholds as moderate, benthic/sediment thresholds as moderate, terrestrial mammalian threshold moderate, and the evidence for terrestrial plants threshold slight.

For the 1,1-dichloroethane risk evaluation, facility emissions data were obtained from databases such as TRI, DMR, and the NEI. The emissions data from these sources are the facility-specific releases of 1,1-dichloroethane to air, water and land on an annual basis (lb/site-yr or kg/site-yr). The total number of operating days/year for these facilities can be estimated with good confidence. For example, manufacturing processes are typically continuous process that run year-round with maybe some brief shut-down periods. The total number of operating days/year for these types of processes can be reliably estimated as 350. However, the number of days/year that the site manufactures, process or uses releases the chemical is uncertain. The number of release days/year may be less than the total number of operating days for the facility. To address this uncertainty, EPA has modeled two distinct "what-if" scenarios for releases to surface water to cover a range of possible release days at the facility. One scenario assumes the number of release days is equivalent to the hazard duration from which the chronic COCs were derived (Table 4-7). A second scenario assumes that the release is averaged out over the total number of operating days (Table 3-3), so an equal average daily release occurs on each of the operating days. Exposure concentrations from both scenarios were compared to the acute, algal, and chronic COCs.

4.3.1 Risk Characterization Approach

EPA characterized the environmental risk of 1,1-dichloroethane using risk quotients (RQs) (U.S. EPA, 1998; Barnthouse et al., 1982). The RQ is defined in Equation 4-2 as

Equation 4-2.

$$RQ = Predicted\ Environmental\ Concentration \div Hazard\ Threshold$$

Environmental concentrations for each compartment (*i.e.,* wastewater, surface water, sediment, and soil) were based on reported facility-specific releases of 1,1-dichloroethane to each media and modeled (*i.e.,* surface water, benthic pore water, and sediment estimated from VVMW-PSC) and/or calculated (*i.e.,* soil and soil pore water concentrations estimated from AERMOD-modeled air deposition rates) concentrations of 1,1-dichloroethane from Sections 3.3 and 4.1. EPA calculates hazard thresholds to identify potential concerns to aquatic and terrestrial species. These terms describe how the values are derived and can encompass multiple taxa or ecologically relevant groups of taxa as the environmental risk characterization serves populations of organisms within a wide diversity of environments. For hazard thresholds, EPA used the COCs calculated for aquatic organisms, and the hazard values or TRVs calculated for terrestrial organisms as detailed within Section 4.2.4.1.1.

RQs equal to 1 indicate that environmental exposures are the same as the hazard threshold. If the RQ is above 1, the exposure is greater than the hazard threshold. If the RQ is below 1, the exposure is less than the hazard threshold. RQs derived from modeled data for 1,1-dichloroethane are described in Section 4.3.2 for aquatic organisms and Sections 4.3.3 and 4.3.4 for terrestrial organisms. Although exposure concentrations in the water column, benthic porewater, and sediment were determined according to two different release scenarios (*e.g.,* the first is a hazard based-release duration and the second is based on total number of operating days); whereas days of exceedance information was used to determine whether the exposure concentrations resulting from these release scenarios exceeded the COCs for a relevant length of time. For aquatic species in the water column, acute RQ days of exceedance were determined as equal to or greater than one day, whereas for chronic RQs days of exceedance are equal to or greater than 21 days. RQs for algal species are presented separately and neither described as acute or chronic due to the relatively rapid replication time of most algal species. Algal RQs days of exceedance are equal to or greater than 4 days. For sediment-dwelling species exposed to benthic pore water, acute RQs days of exceedance are equal to or greater than 1 day, and days of exceedance for chronic RQs are equal to or greater than 15 days. For sediment-dwelling species exposed to sediment, chronic RQs days of exceedance are equal to or greater than 35 days. Acute RQs for exposure to 1,1-dichloroethane in sediment (mg/kg) were not calculated due to lack of hazard data. Exposure to the benthic compartment is represented by acute RQs calculated for exposure to 1,1-dichloroethane in benthic pore water (mg/L).

EPA used modeled (*e.g.,* PSC, AERMOD, SimpleTreat) data to characterize environmental concentrations for 1,1-dichloroethane and to calculate the RQ. Table 3-1 describes the COUs and OESs which result in environmental releases of 1,1-dichloroethane.

Aquatic Risk Characterization Approach; Surface Water, Benthic Pore Water, and Sediment
Risk estimates for seven COUs were developed for releases of 1,1-dichloroethane to surface water. Within the aquatic environment, a modeling approach was employed to predict surface water, benthic pore water, and sediment 1,1-dichloroethane concentrations. PSC considers model inputs of physical and chemical properties of 1,1-dichloroethane (*i.e.,* K_{OW}, K_{OC}, water column half-life, photolysis half-life, hydrolysis half-life, and benthic half-life) allowing EPA to model predicted benthic pore water and sediment concentrations. The PSC modeled 7Q10 surface water concentrations from facility-specific release pollutant loads. If the 7Q10 surface water concentrations corresponding to the respective exposure durations represented by the various COCs were greater than the acute, chronic, or algal COCs in the water column, the PSC Model was then used to confirm the modeled surface water concentration days of exceedance as determined by the respective COCs. For example, for 1,1-dichloroethane, five COUs modeled in PSC produced aquatic chronic RQ values greater than or equal to 1 based on the

number of release days based on chronic hazard studies, prompting the days of exceedance analysis in PSC. Similarly, if modeled benthic pore water and sediment concentrations corresponding to the respective exposure durations exceeded the benthic COCs, the PSC Model was used to confirm the modeled benthic pore water and sediment concentration days of exceedance as determined by those COCs. In cases of highly effluent-dominated release sites where facility discharge flow is considerably greater than the 7Q10 flow of the receiving water body, the facility discharge flow was substituted in place of the receiving water body flow as an input in PSC. This scenario can occur when *e.g.*, a facility produces high effluent discharge into a concrete basin with intermittent stream flow. This modification was applied only to the COU/OES Disposal/Disposal/Disposal/Waste handling, treatment, and disposal (remediation), where the highest-releasing facility discharge flow was approximately three times the 7Q10 flow of the receiving stream. The plant flow is 0.416 MLD and was taken from the discharge permit.

Releases of 1,1-dichloroethane to surface water were assessed quantitatively whereas air deposition of 1,1-dichloroethane to surface water from releasing facilities of TRI-reported fugitive emissions was assessed qualitatively. As described in Section 3.3.3.2.3, EPA does not expect 1,1-dichloroethane surface water concentrations modeled from air deposition to streams 100 m from releasing facilities of fugitive and/or stack air emissions to exceed the hazard thresholds for aquatic organisms. The analysis in Section 3.3.3.2.3 was based on the air deposition rates from the Manufacturing COU/OES, which had the highest maximum and mean deposition rates by over an order of magnitude in comparison to the maximum and mean air deposition rates of the other COU/OESs at 100 m based on TRI fugitive emissions. Because the nearest body of water from the manufacturing facility with the highest daily air deposition rate was approximately 340 m from facility, EPA does not expect risk estimates greater than or equal to 1 for aquatic receptors exposed to 1,1-dichloroethane in surface water resulting from air deposition.

EPA considers the biological relevance of species that COCs or hazard values are based on when integrating these values with the location of the surface water, pore water, and sediment concentration data to produce RQs. Life-history and habitat of aquatic organisms influence the likelihood of exposure above the hazard threshold in an aquatic environment. EPA has identified COC values associated with aquatic hazard values and include acute aquatic COC, chronic aquatic COC, acute benthic COC, two chronic benthic COCs, and algal COC. The acute aquatic COC and acute benthic COC are the lower 95 percent CI of the HC05 of an SSD—a modeled probability distribution of toxicity values from multiple taxa (including but not limited to *Chironomus riparius*, *Daphnia magna*, mysid shrimp, and fathead minnow) inhabiting the water column and benthic pore water. The chronic COC is represented by a reproductive endpoint from a 21-day exposure of *Daphnia magna* to 1,1-dichloroethane within the water column. The chronic benthic COC compared to benthic pore water is represented by a reproductive endpoint from a 15-day exposure of *Ophryotrocha labronica* to analog 1,1,2-trichloroethane within benthic pore water. A second chronic benthic COC compared to sediment is represented by an emergence endpoint from a 2-generation exposure of *Chironomus riparius* to analog 1,1,2-trichloroethane within sediment. The algal COC is represented by growth and development endpoints from 72 to 120-hour exposures to analog 1,2-dichloropropane within the water column.

Environmental RQ values by exposure scenario with 1,1-dichloroethane surface water concentrations (µg/L) were modeled by PSC and are presented in Table 4-11. The max daily average concentrations produced by PSC represent the maximum concentration (µg/L) over a 21-day (Scenario 1) or total number of operating days (Scenario 2) average period corresponding with the acute or chronic aquatic COC used for the RQ estimate. Max daily average surface water concentrations were also produced by PSC over a 21-day (Scenario 1) or total number of operating days (Scenario 2, Table 3-3) average period

corresponding with the algal COC used for the RQ estimate, as presented in Table 4-12. Environmental RQ values by exposure scenario with 1,1-dichloroethane benthic pore water concentrations (ppb) were modeled by PSC and are presented in Table 4-13. The benthic pore water concentrations produced by PSC represent the maximum concentration (ppb) over a 15-day (Scenario 1) or total number of operating days (Scenario 2, Table 3-3) average period corresponding with the acute or chronic benthic COC used for the RQ estimate. Environmental RQ values by exposure scenario with 1,1-dichloroethane sediment concentrations (mg/kg) were modeled by PSC and are presented in Table 4-14. The sediment concentrations produced by PSC represent the maximum concentration (mg/kg) over a 35-day (Scenario 1) or total number of operating days (Scenario 2, Table 3-3) average period corresponding with the chronic benthic COC. Use of surface water and benthic pore water concentrations in trophic transfer is described in Section 4.3.1.1.

Terrestrial Risk Characterization Approach; Air Deposition and Biosolids

As described in Section 3.3, IIOAC and subsequently AERMOD were used to estimate the release of 1,1-dichloroethane to soil via air deposition from specific exposure scenarios. Estimated concentrations of 1,1-dichloroethane that could be in soil via air deposition near-facility sources (10 m from the source) have been calculated for 1,1-dichloroethane releases reported to TRI in fugitive emissions, encompassing five COUs. EPA selected a distance of 10 m for evaluating 1,1-dichloroethane exposure to terrestrial organisms that could result from air deposition since this was the distance that resulted in the highest average daily deposition rate of 1,1-dichloroethane (Table 3-10). Soil and soil pore water concentrations were obtained using maximum 95th percentile daily air deposition rates of 1,1-dichloroethane (Table 4-7). EPA calculated RQs for exposure of terrestrial plants to 1,1-dichloroethane by directly comparing the 1,1-dichloroethane soil pore water concentrations to the terrestrial plant hazard value for 1,1-dichloroethane (Table_Apx L-1). Releases of 1,1-dichloroethane in fugitive and/or stack emissions modeled by Monte Carlo simulation (2 COUs) or reported to NEI (8 COUs), which could result in exposure to terrestrial receptors were assessed qualitatively for air deposition to soil due to the modeled maximum 95th percentile (NEI) or high-end (Monte-Carlo) air concentrations at 10 m from these sources being comparable or lower than modeled maximum 95th percentile air concentrations from fugitive emissions reported to TRI (see Table 3-9, Table 3-13, Table 3-13).

EPA also estimated soil and soil pore water concentrations of 1,1-dichloroethane from annual application of biosolids to tilled agricultural soil and pastureland (Table 4-8) as described in Appendix J to calculate RQs for terrestrial plants (Table_Apx L-2). Briefly, SimpleTreat was used to predict 1,1-dichloroethane concentrations in biosolids, and an EU/REACH screening method and modified Equilibrium Partitioning methodology to estimate soil and soil pore water concentrations, respectively, from biosolid application. Use of 1,1-dichloroethane soil and soil pore water concentrations in trophic transfer is described in Section 4.3.1.1.

In general, for terrestrial mammals and birds, relative contribution to total exposure associated with inhalation is secondary in comparison to exposures by diet and indirect ingestion (U.S. EPA, 2003a, b). For 1,1-dichloroethane, other factors that guided EPA's decision to qualitatively assess 1,1-dichloroethane inhalation exposure to terrestrial receptors were: limited facility releases and the lack of 1,1-dichloroethane inhalation hazard data in terrestrial mammals for ecologically relevant endpoints. Therefore, direct exposure of 1,1-dichloroethane to terrestrial receptors via air was not assessed quantitatively.

4.3.1.1 Risk Characterization Approach for Trophic Transfer

Trophic transfer is the process by which chemical contaminants can be taken up by organisms through dietary and media exposures and transfer from one trophic level to another. Chemicals can be transferred

from contaminated media and diet to biological tissue and accumulate throughout an organisms' lifespan (bioaccumulation) if they are not readily excreted or metabolized. Through dietary consumption of prey, a chemical can subsequently be transferred from one trophic level to another. If biomagnification occurs, higher trophic level predators will contain greater body burdens of a contaminant compared to lower trophic level organisms. Although 1,1-dichloroethane is not expected to be bioaccumulative, it is continuously released to the environment. When continuous releases occur, dietary exposure to wildlife is possible.

EPA conducted screening level approaches for aquatic and terrestrial risk estimation based on exposure via trophic transfer using conservative assumptions for factors such as: area use factor, 1,1-dichloroethane absorption from diet, soil, sediment, and water. A screening level analysis was conducted for trophic transfer and formulation of RQ values for aquatic and terrestrial pathways to representative mammalian species. If RQ values were greater than or equal to 1, further refined analysis is warranted. If an RQ value is less than 1, no further assessment is necessary. The screening level approach employs a combination of conservative assumptions (*i.e.*, conditions for several exposure factors included within Equation 4-3 below) and utilization of the maximum values obtained from modeled and/or monitoring data from relevant environmental compartments.

Equation 4-3.

$$RQ_j = DE_j \div HT_j$$

Where:

RQ_j	=	Risk quotient for contaminant (j) (unitless)
DE_j	=	Dietary exposure for contaminant (j) (mg/kg-BW/day)
HT_j	=	Hazard threshold (mg/kg-BW/day)

Dietary exposure estimates are presented in Appendix J.3.2. Terrestrial hazard data are available for mammals using hazard values detailed in Section 4.2.4. As described in Appendix J.3.1, representative mammal species were chosen to connect the 1,1-dichloroethane transport exposure pathway via trophic transfer of 1,1-dichloroethane uptake from contaminated soil and soil pore water to earthworm followed by consumption by an insectivorous mammal (short-tailed shrew), 1,1-dichloroethane uptake from contaminated soil pore water to plant (*Trifolium* sp.) followed by consumption by an herbivorous mammal (meadow vole). For semi-aquatic terrestrial species, a representative mammal (American mink) was chosen to connect the 1,1-dichloroethane transport exposure pathway via trophic transfer from fish or crayfish uptake of 1,1-dichloroethane from contaminated surface water and benthic pore water modeled from 1,1-dichloroethane surface water releases. As mentioned above, trophic transfer of 1,1-dichloroethane to semi-aquatic terrestrial species from air deposition to surface water is not anticipated due to low maximum daily air deposition rates of 1,1-dichloroethane to streams at distances equal to or exceeding 100 m from releasing facilities of fugitive emissions (Section 3.3.3.2.3). Therefore, EPA does not expect that risk estimates for trophic transfer of 1,1-dichloroethane to a semi-aquatic terrestrial mammal from air deposition to surface water would be equal to or greater than 1.

4.3.2 Risk Characterization for Aquatic Receptors

Because of 1,1-dichloroethane's high water solubility (Table 2-1), low log K_{OC} (Table 2-2), and known releases to surface water (Table 3-6), biota in the water column are more likely to be exposed to 1,1-dichloroethane than biota in the sediment. For example, surface water RQs for chronic exposures exceeded 1 for five COUs evaluated for 1,1-dichloroethane surface water releases based on a hazard guideline-based, 21-day release scenario with days of exceedance equal to or greater than the corresponding hazard duration (21 days) and approaching 1 (>0.9) for the Manufacturing COU when the

release is based on the total number of operating days (Table 3-3, Table 4-11). In contrast, none of the seven COUs evaluated quantitatively for surface water release resulted in RQs greater than or equal to 1 for chronic exposure to benthic pore water or sediment (Table 4-13, Table 4-14). No RQs were greater than 1 for acute exposures to biota in the water column or sediment for the seven COUs evaluated for surface water releases (Table 4-11, Table 4-13).

Exposures to algal species in the water column resulted an RQ greater than 1 for only the Manufacturing COU when based on a hazard guideline-based 21-day release scenario with days of exceedance equal to or greater than the corresponding hazard duration (4 days) and RQs less than 1 for all COUs evaluated for surface water releases based on total number of operating days (Table 4-12). The observation of surface water RQs greater than 1 for a hazard guideline-based release scenario (*e.g.*, hypothetical hazard-based release duration shorter than the number of operating days) indicate potential community-level impacts (*e.g.*, decline in aquatic invertebrate and algal populations leading to impacts on fish populations which depend on these species as food sources) for biota in the water column from surface water releases of 1,1-dichloroethane—particularly for the COUs manufacturing of 1,1-dichloroethane and remediation of waste handling, treatment, and disposal of 1,1-dichloroethane.

Releases of 1,1-dichloroethane to surface water were identified for seven COUs (Life cycle stage/ Category/Sub-category with their respective OES) with three COUs (Processing/As a reactant/ Intermediate in all other basic organic chemical manufacture; Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing; and Processing/recycling/recycling) represented by a single OES (Processing as a reactive intermediate) and COU (Disposal of 1,1-dichloroethane) represented by three OESs (General waste handling, POTW, and Remediation), as described below. As described in Section 3.3.3.2.1, the highest facility-specific release data reported between 2015 to 2020 was utilized for individual facility modeling, with the exception for the release data of the manufacturing COU facility where the next highest release data that occurred in 2016 was used in lieu of the highest release data corresponding with a hurricane event in 2020 (U.S. EPA, 2025f).

Manufacture/Domestic Manufacturing/Domestic Manufacturing/Manufacturing of 1,1-Dichloroethane as an Isolated Intermediate

Surface water: Surface water acute aquatic RQ values for manufacturing 1,1-dichloroethane were less than 1. The chronic aquatic RQ value based on a hazard guideline-based release duration (21 days) for manufacturing 1,1-dichloroethane was greater than 1 at 15.38 with 21 days of exceedance for the chronic aquatic COC which is equal to or greater than the 21-day duration of the chronic aquatic hazard data (Table 4-11). The surface water chronic aquatic RQ value based on total number of operating days (350 days) for manufacturing 1,1-dichloroethane was less than 1 at 0.91 (Table 4-11). The surface water algal RQ value based on a hazard guideline-based release duration (21 days) for manufacturing 1,1-dichloroethane was greater than 1 for the algal COC at 1.4, with 13 days of exceedance for the algal COC, which is greater than or equal to the 4-day duration of the algal hazard data, whereas the surface water algal RQ value based on the total number of operating days (350 days) for manufacturing 1,1-dichloroethane was less than 1 at 0.08 (Table 4-12).

Benthic Pore Water: The benthic pore water acute and chronic RQ values for manufacturing 1,1-dichloroethane were less than 1 for the acute benthic and chronic benthic COCs (Table 4-13).

Sediment: The sediment chronic RQs based on a hazard guideline-based release duration (35 days) or the total number of operating days (350 days) for manufacturing 1,1-dichloroethane were less than 1 for the chronic benthic COC (Table 4-14).

Processing/As a Reactant/ Intermediate in All Other Basic Organic Chemical Manufacture/Processing as a Reactive Intermediate; ***Processing/as a Reactant/Intermediate in All Other Chemical Product and Preparation Manufacturing/Processing as a Reactive Intermediate***; ***Processing/Recycling/Recycling/Processing as a Reactive Intermediate***

Surface water: The surface water acute RQ for processing 1,1-dichloroethane as a reactive intermediate represented by three COUs (Processing/As a reactant/Intermediate in all other basic organic chemical manufacture, Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing, and Processing/Recycling/Recycling) was less than 1 for the acute aquatic COC. The surface water chronic RQ value based on a hazard guideline-based release duration (21 days) for processing 1,1-dichloroethane as a reactant was greater than 1 at 2.54, with 21 days of exceedance for the chronic aquatic COC, whereas the surface water chronic RQ value based on the total number of operating days (350 days) for processing 1,1-dichloroethane as a reactant was less than 1 at 0.14 (Table 4-11). The surface water algal RQ values for processing 1,1-dichloroethane as a reactant were less than 1 for the algal COC (Table 4-12).

Benthic Pore Water: The benthic pore water acute and chronic RQ values for processing 1,1-dichloroethane as a reactive intermediate were less than 1 for the acute benthic COC and chronic benthic COC (Table 4-13).

Sediment: The sediment chronic RQs for processing 1,1-dichloroethane as a reactive intermediate were less than 1 for the chronic benthic COC (Table 4-14).

Processing/Processing – Repackaging/Processing – Repackaging/Processing – Repackaging

Surface water: The surface water acute and chronic RQ values for repackaging 1,1-dichloroethane were less than 1 for the acute aquatic COC, chronic aquatic COC, and algal COC (Table 4-11, Table 4-12).

Benthic Pore Water: The benthic pore water acute and chronic RQ values for repackaging 1,1-dichloroethane were less than 1 for the acute benthic COC and chronic benthic COC (Table 4-13).

Sediment: The sediment chronic RQs for repackaging 1,1-dichloroethane were less than 1 for the chronic benthic COC (Table 4-14).

Commercial Use/Other Uses/Laboratory Chemicals/Commercial Use as a Laboratory Chemical

Surface Water: The surface water acute and chronic RQ values for commercial use of 1,1-dichloroethane as a laboratory chemical were less than 1 for the acute aquatic COC, chronic aquatic COC, and algal COC (Table 4-11, Table 4-12).

Benthic Pore Water: The benthic pore water acute and chronic RQ values for commercial use of 1,1-dichloroethane as a laboratory chemical were less than 1 for the acute benthic COC and chronic benthic COC (Table 4-13).

Sediment: The sediment chronic RQs for commercial use of 1,1-dichloroethane as a laboratory chemical were less than 1 for the chronic benthic COC (Table 4-14).

Disposal/Disposal/Disposal/General Waste Handling, Treatment and Disposal

Surface Water: The surface water acute RQ values for general waste handling, treatment, and disposal of 1,1-dichloroethane were less than 1 for the acute aquatic COC. The surface water chronic RQ value based on a hazard guideline-based release duration (21 days) for waste handling, treatment, and disposal of 1,1-dichloroethane at a non-POTW facility was greater than 1 at 2.34, with 21 days of exceedance for

the chronic aquatic COC, whereas the surface water chronic RQ value based on the total number of operating days (250 days) for general waste handling, treatment, and disposal of 1,1-dichloroethane was less than 1 at 0.13 (Table 4-11). The surface water algal RQ values for general waste handling, treatment, and disposal of 1,1-dichloroethane were less than 1 (Table 4-12).

Benthic Pore Water: The benthic pore water acute and chronic RQ values for general waste handling, treatment, and disposal of 1,1-dichloroethane were less than 1 for the acute benthic COC and chronic benthic COC (Table 4-13).

Sediment: The sediment chronic RQs for general waste handling, treatment, and disposal of 1,1-dichloroethane were less than 1 for the chronic benthic COC (Table 4-14).

Disposal/Disposal/Disposal/Waste Handling, Treatment and Disposal (POTW)
Surface Water: The surface water acute and algal RQ values for waste handling, treatment, and disposal of 1,1-dichloroethane at POTW facilities were less than 1 for the acute aquatic COC and the algal COC (Table 4-11 and Table 4-12). The surface water chronic RQ value based on a hazard guideline-based release duration (21 days) for remediation of waste handling, treatment, and disposal of 1,1-dichloroethane was greater than 1 at 1.5 with 21 days of exceedance for the chronic aquatic COC, the surface water chronic RQ value based on the total number of operating days (365 days) for waste handling, treatment, and disposal of 1,1-dichloroethane at POTW facilities was less than 1 at 0.09 (Table 4-11).

Benthic Pore Water: The benthic pore water acute and chronic RQ values for waste handling, treatment, and disposal of 1,1-dichloroethane at POTW facilities were less than 1 for the acute benthic COC and chronic benthic COC (Table 4-13).

Sediment: The sediment chronic RQ for waste handling, treatment, and disposal of 1,1-dichloroethane at POTW facilities was less than 1 for the chronic benthic COC (Table 4-14).

Disposal/Disposal/Disposal/Waste Handling, Treatment and Disposal (Remediation)
Surface Water: The surface water acute and algal RQ values for remediation of waste handling, treatment, and disposal of 1,1-dichloroethane were less than 1 (Table 4-11 and Table 4-12). The surface water chronic RQ value based on a hazard guideline-based release duration (21 days) for remediation of waste handling, treatment, and disposal of 1,1-dichloroethane was greater than 1 at 6.2 with 35 days of exceedance for the chronic aquatic COC, whereas the surface water chronic aquatic RQ value based on total number of operating days (365 days) for remediation of waste handling, treatment, and disposal of 1,1-dichloroethane was less than 1 at 0.33 (Table 4-11).

Benthic Pore Water: The benthic pore water acute RQ and chronic values for remediation of waste handling, treatment, and disposal of 1,1-dichloroethane were less than 1 for the acute benthic and chronic benthic COCs (Table 4-13).

Sediment: The sediment chronic RQs for remediation of waste handling, treatment, and disposal of 1,1-dichloroethane were less than 1 for the chronic benthic COC (Table 4-14).

Distribution in Commerce/Distribution in Commerce/Distribution in commerce/Distribution in Commerce
Distribution of 1,1-dichloroethane in commerce does not result in surface water releases (Table 3-6) therefore RQs were not generated for this COU/OES.

Table 4-11. Environmental Risk Quotients (RQs) by COU for Aquatic Organisms with 1,1-Dichloroethane Surface Water Concentration (µg/L) Modeled by PSC

COU (Life Cycle Stage/Category/Subcategory)	OES	Number of Facilities [a]	Days of Release	Pollutant Load (kg/day) [b]	Max Daily Average (µg/L) [c]	COC Type	COC (µg/L) [d]	Days of Exceedance (days per year) [d]	RQ
Manufacture/Domestic manufacturing/Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	1/1	21	5.79	1,430	Acute	8,931	0	0.16
			350[e]	0.347	84.7	Acute	8,931	0	9.48E−03
			21	5.79	1,430	Chronic	93	21	15
			350[e]	0.347	84.7	Chronic	93	0	0.91
Processing/As a reactant/ Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	2/58	21	1.06	236	Acute	8,931	0	2.64E−02
			350[e]	6.34E−02	12.9	Acute	8,931	0	1.44E−03
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing			21	1.06	236	Chronic	93	21	2.5
Processing/Recycling/Recycling			350[e]	6.34E−02	12.9	Chronic	93	0	0.14
Processing/Processing – repackaging/Processing – repackaging	Processing – repackaging	0/3	21	5.51E−03	8.67	Acute	8,931	0	9.71E−04
			260[e]	4.45E−04	0.702	Acute	8,931	0	7.86E−05
			21	5.51E−03	8.67	Chronic	93	0	9.3E−02
			260[e]	4.45E−04	0.702	Chronic	93	0	7.6E−03
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	0/2	21	2.27E−03	7.78	Acute	8,931	0	8.71E−04
			260[e]	1.83E−04	0.638	Acute	8,931	0	7.14E−05
			21	2.27E−03	7.78	Chronic	93	0	8.4E−02
			260[e]	1.83E−04	0.638	Chronic	93	0	6.9E−03
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	1/22	21	2.37	218	Acute	8,931	0	2.44E−02
			250[e]	0.199	12.4	Acute	8,931	0	1.39E−03
			21	2.37	218	Chronic	93	21	2.3
			250[e]	0.199	12.4	Chronic	93	0	0.13
Disposal/Disposal/Disposal		1/125	21	3.88	143	Acute	8,931	0	1.60E−02
			365[e]	0.233	8.16	Acute	8,931	0	9.14E−04

COU (Life Cycle Stage/Category/Subcategory)	OES	Number of Facilities [a]	Days of Release	Pollutant Load (kg/day) [b]	Max Daily Average (µg/L) [c]	COC Type	COC (µg/L) [d]	Days of Exceedance (days per year) [d]	RQ
	Waste handling, treatment, and disposal (POTW)		21	3.88	143	Chronic	93	21	1.5
			365[e]	0.223	8.16	Chronic	93	0	8.8E−02
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (Remediation)	2/42	21	0.243	580	Acute	8,931	0	6.49E−02
			365[e]	1.40E−02	30.7	Acute	8,931	0	3.44E−03
			21	0.243	580	Chronic	93	35	6.2
			365[e]	1.40E−02	30.7	Chronic	93	0	0.33
Distribution in Commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce		N/A [f]						

[a] Number of facilities for a given OES with RQ > 1 and days of exceedance (DOE) ≥ 21 days
[b] Based on facility release data.
[c] Max daily average represents the maximum surface water concentration over a 21-day or total number of operating day average period corresponding with the acute aquatic or chronic aquatic COC used for the RQ estimate.
[d] Based on (acute) the lower 95% CI of the SSD HC_{05} based on empirical hazard data from *Daphnia magna* exposed to 1,1-dichloroethane in water and mysid shrimp and fathead minnow (*Pimephales promelas*) exposed to 1,2-dichloropropane in water and Web-ICE predictions or (chronic) 21-day hazard data from *Daphnia magna* exposed to 1,1-dichloroethane in water.
[e] Highest days of release based on total number of operating days (Table 3-3).
[f] Distribution in Commerce does not result in surface water releases (Table 3-6).

Table 4-12. Environmental Risk Quotients (RQs) by COU for Aquatic Non-vascular Plants with 1,1-Dichloroethane Surface Water Concentration (µg/L) Modeled by PSC

COU (Life Cycle Stage/Category/Subcategory)	OES	Number of Facilities [a]	Days of Release	Pollutant Load (kg/day) [b]	Max Daily Average (µg/L) [c]	COC Type	COC (µg/L) [d]	Days of Exceedance (days per year) [d]	RQ
Manufacture/ Domestic manufacturing/ Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	1/1	21	5.79	1,430	Algal	1,000	13	1.4
			350[e]	0.347	84.7			0	8.5E−02
Processing/As a reactant/ Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	0/58	21	1.06	236	Algal	1,000	0	0.24
Processing/As a Reactant/Intermediate in all other chemical product and preparation manufacturing									
Processing/Recycling/Recycling			350[e]	6.34E−02	12.9			0	1.3E−02
Processing/Processing – repackaging/Processing – repackaging	Processing – repackaging	0/3	21	5.51E−03	8.67	Algal	1,000	0	8.7E−03
			260[e]	4.45E−04	0.702			0	7.0E−04
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	0/2	21	2.27E−03	7.78	Algal	1,000	0	7.8E−03
			260[e]	1.83E−04	0.638			0	6.4E−04
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	0/22	21	2.37	218	Algal	1,000	0	0.22
			250[e]	0.199	12.4			0	1.2E−02
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	0/125	21	3.88	143	Algal	1,000	0	0.14
			365[e]	0.223	8.16			0	8.2E−03
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (remediation)	0/42	21	0.243	580	Algal	1,000	0	0.58
			365[e]	1.40E−02	30.7			0	3.1E−02
Distribution in Commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce		N/A[f]						

[a] Number of facilities for a given OES with RQ > 1 and DOE ≥ 4 days
[b] Based on facility release data.
[c] Max daily average represents the maximum surface water concentration over a 21-day or total number of operating day average period corresponding with the algal COC used for the RQ estimate.

COU (Life Cycle Stage/Category/Subcategory)	OES	Number of Facilities [a]	Days of Release	Pollutant Load (kg/day) [b]	Max Daily Average (µg/L) [c]	COC Type	COC (µg/L) [d]	Days of Exceedance (days per year) [d]	RQ

[d] Based on 4-day hazard data from diatom *Skeletonema costatum* exposed to 1,2-dichloropropane in water.
[e] Highest days of release based on total number of operating days (see Table 3-3).
[f] Distribution in Commerce does not result in surface water releases (see Table 3-6).

Table 4-13. Environmental Risk Quotients (RQs) by COU for Aquatic Organisms with 1,1-Dichloroethane Benthic Pore Water Concentration (µg/L) Modeled by PSC

COU (Life Cycle Stage/Category/Subcategory)	OES	Number of Facilities [a]	Days of Release	Pollutant Load (kg/day) [b]	Benthic Pore Water Concentration (µg/L) [c]	COC Type	COC (µg/L) [d]	Days of Exceedance (days per year) [d]	RQ
Manufacture/ Domestic manufacturing/Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	0/1	15	8.10	413	Acute	8,931	0	4.62E−02
			350[e]	0.347	78	Acute	8,931	0	8.73E−03
			15	8.10	413	Chronic	6,800	0	6.1E−02
			350[e]	0.347	78	Chronic	6,800	0	1.1E−02
Processing/As a reactant/ intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	0/58	15	1.48	66.5	Acute	8,931	0	7.45E−03
			350[e]	6.34E−02	12.4	Acute	8,931	0	1.39E−03
Processing/As a reactant/intermediate in all other chemical product and preparation manufacturing			15	1.48	66.5	Chronic	6,800	0	9.8E−03
Processing/Recycling/Recycling			350[e]	6.34E−02	12.4	Chronic	6,800	0	1.8E−03
Processing/Processing – repackaging/Processing – repackaging	Processing – repackaging	0/3	15	7.71E−03	2.51	Acute	8,931	0	2.81E−04
			260[e]	4.45E−04	0.61	Acute	8,931	0	6.83E−05
			15	7.71E−03	2.51	Chronic	6,800	0	3.7E−04
			260[e]	4.45E−04	0.61	Chronic	6,800	0	9.0E−05
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	0/2	15	3.18E−03	2.28	Acute	8,931	0	2.55E−04
			260[e]	1.83E−04	0.546	Acute	8,931	0	6.11E−05
			15	3.18E−03	2.28	Chronic	6,800	0	3.4E−04
			260[e]	1.83E−04	0.546	Chronic	6,800	0	8.0E−05
Disposal/Disposal/Disposal		0/22	15	3.32	62	Acute	8,931	0	6.94E−03

COU (Life Cycle Stage/Category/Subcategory)	OES	Number of Facilities [a]	Days of Release	Pollutant Load (kg/day) [b]	Benthic Pore Water Concentration (µg/L) [c]	COC Type	COC (µg/L) [d]	Days of Exceedance (days per year) [d]	RQ
	General waste handling, treatment, and disposal		250[e]	0.199	11.8	Acute	8,931	0	1.32E−03
			15	3.32	62	Chronic	6,800	0	9.1E−03
			250[e]	0.199	11.8	Chronic	6,800	0	1.7E−03
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	0/125	15	5.43	40.8	Acute	8,931	0	4.57E−03
			365[e]	0.223	7.85	Acute	8,931	0	8.79E−04
			15	5.43	40.8	Chronic	6,800	0	6.0E−03
			365[e]	0.223	7.85	Chronic	6,800	0	1.2E−03
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (remediation)	0/42	15	0.34	168	Acute	8,931	0	1.88E−02
			365[e]	1.40E−02	29.3	Acute	8,931	0	3.28E−03
			15	0.34	168	Chronic	6,800	0	2.5E−02
			365[e]	1.40E−02	29.3	Chronic	6,800	0	4.3E−03
Distribution in Commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce	N/A[f]							

[a] Number of facilities for a given OES with RQ > 1 and DOE ≥ 15 days
[b] Highest days of release based on total number of operating days (Table 3-3).
[c] Based on facility release data.
[d] Max daily average of benthic pore water concentration represents the maximum benthic pore water concentration over a 15-day or total number of operating day average period corresponding with the acute benthic or chronic benthic COC used for the RQ estimate.
[e] Based on (acute) probabilistic hazard threshold (*e.g.*, lower bound of the 95th confidence interval of the HC05) which included empirical hazard data for *Chironomus riparius* exposed to 1,1-dichloroethane in water and hazard predictions of sediment-dwelling organisms exposed to 1,1-dichloroethane and analog 1,2-dichloropropane or (chronic) 15-day hazard data from sediment-dwelling *Ophryotrocha labronica* exposed to analog 1,1,2-trichloroethane in water.
[f] Distribution in Commerce does not result in surface water releases (Table 3-6).

Table 4-14. Environmental Risk Quotients (RQs) by COU for Aquatic Organisms with 1,1-Dichloroethane Sediment Concentration (µg/kg) Modeled by PSC

COU (Life Cycle/Stage/Category/ Subcategory)	Occupational Exposure Scenario	Number of Facilities [a]	Days of Release	Pollutant Load (kg/day) [b]	Sediment Concentration (µg/kg) [c]	COC Type	COC (µg/kg) [d]	Days of Exceedance (days per year) [d]	RQ
Manufacture/ Domestic manufacturing/ Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	0/1	35	3.47	519	Chronic	2,900	0	0.18
			350[e]	0.347	124			0	4.3E−02
Processing/As a reactant/ intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	0/58	35	0.634	77.4	Chronic	2,900	0	2.7E−02
Processing/As a reactant/ intermediate in all other chemical product and preparation manufacturing									
Processing/Recycling/Recycling			350[e]	6.34E−02	19.6			0	6.8E−03
Processing/Processing – repackaging/Processing – repackaging	Processing – Repackaging	0/3	35	3.30E−03	3.13	Chronic	2,900	0	1.1E−03
			260[e]	4.45E−04	0.962			0	3.3E−04
Commercial use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	0/2	35	1.36E−03	2.84	Chronic	2,900	0	9.8E−04
			260[e]	1.83E−04	0.854			0	2.9E−04
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	0/22	35	1.42	76.5	Chronic	2,900	0	2.6E−02
			250[e]	0.199	18.6			0	6.4E−03
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	0/125	35	2.33	50.5	Chronic	2,900	0	1.7E−02
			365[e]	0.223	12.4			0	4.3E−03
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (remediation)	0/42	35	0.146	211	Chronic	2,900	0	7.3E−02
			365[e]	1.40E−02	46.3			0	1.6E−02
Distribution in commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce	N/A[f]							

COU (Life Cycle/Stage/Category/ Subcategory)	Occupational Exposure Scenario	Number of Facilities [a]	Days of Release	Pollutant Load (kg/day) [b]	Sediment Concentration (µg/kg) [c]	COC Type	COC (µg/kg) [d]	Days of Exceedance (days per year) [d]	RQ

[a] Number of facilities for a given OES with RQ > 1 and DOE ≥ 35 days
[b] Based on facility release data.
[c] Max daily average of sediment concentration represents the maximum sediment concentration over a 35-day or total number of operating day average period corresponding with the chronic benthic COC used for the RQ estimate.
[d] Based on 35-day hazard data from *Chironomus riparius* exposed to 1,1,2-trichloroethane in sediment.
[e] Highest days of release based on total number of operating days (Table 3-3).
[f] Distribution in Commerce does not result in surface water releases (Table 3-6).

4.3.3 Risk Characterization for Terrestrial Organisms

Risk was evaluated for terrestrial plants with direct exposure to 1,1-dichloroethane via air deposition to soil or from land application of biosolids. None of the pathways analyzed showed expected exposure of 1,1-dichloroethane high enough to result in RQs greater than one. See Appendix L.2 for additional detail.

4.3.4 Risk Characterization Based on Trophic Transfer in the Environment

Trophic transfer of 1,1-dichloroethane and risk to terrestrial species was evaluated using a screening level approach conducted as described in EPA's *Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a). None of the pathways analyzed showed expected exposure of 1,1-dichloroethane high enough to result in RQs greater than one. Details of the analysis can be found in Appendix L.3.

4.3.5 Overall Confidence and Remaining Uncertainties Confidence in Environmental Risk Characterization

4.3.5.1 Risk Characterization Confidence

The overall confidence in the risk characterization combines the confidence from the environmental exposure, hazard threshold, and trophic transfer sections. This approach aligns with the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) and *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025z). In the environmental risk characterization, confidence was evaluated from environmental exposures and environmental hazards. Hazard confidence was represented by evidence type as reported previously in Section 4.2.4.1.1 and again in Appendix K and Appendix Q. Trophic transfer confidence was represented by evidence type as reported in Appendix J.3.4 in Table_Apx J-16. Exposure confidence has been synthesized from Section 3 and is further detailed within Section 4.1.5. Synthesis of confidence for exposure, hazard, and trophic transfer (when applicable) resulted in the following confidence ranks for risk characterization RQ inputs: robust for acute and chronic aquatic evidence, moderate for algal evidence, moderate for acute and chronic benthic evidence, moderate for mammalian evidence, slight for terrestrial plant evidence based on air deposition, slight for terrestrial plant evidence based on biosolid land application, indeterminate for soil invertebrate evidence, and indeterminate for avian evidence (Appendix Q).

RQ Inputs for Aquatic, Algal, Benthic, and Semi-Aquatic Mammalian Assessments

Uncertainties and confidence in modeled exposure estimates from PSC have been described in Section 4.1.5. A robust confidence has been assigned to the exposure component of the RQ input for the aquatic, algal, and benthic assessments as well as the mammalian assessments based on consumption of fish or crayfish by a semi-aquatic terrestrial mammal (Appendix Q). Combining the robust exposure confidence for the PSC-modeled surface water, benthic pore water, and sediment 1,1-dichloroethane concentrations with the hazard confidences for aquatic, algal, and benthic assessments (robust, moderate, and moderate, respectively) resulted in overall confidences of robust, moderate, and moderate in the RQ inputs for the aquatic (acute and chronic), algal, and benthic (acute and chronic) assessments, respectively (Appendix Q).

Combining the moderate exposure confidence for the PSC-modeled surface water and benthic pore water 1,1-dichloroethane concentrations with the moderate hazard confidence for the mammalian assessments and moderate trophic transfer confidence based on the consumption of fish (surface water) or crayfish (benthic pore water) resulted in overall confidences of moderate in the RQ inputs for the mammalian assessments represented by a semi-aquatic terrestrial mammal (Appendix Q).

RQ Inputs for Terrestrial Mammalian and Terrestrial Plant Assessments
Uncertainties and confidence in air deposition from AERMOD have been described in Section 4.1.5. Calculations of soil and soil pore water concentrations from 1,1-dichloroethane daily air deposition rates may add further uncertainty from the robust confidence in the AERMOD air deposition, therefore resulting in a moderate confidence in the 1,1-dichloroethane soil and soil porewater concentrations from air deposition. The uncertainties in the soil and soil pore water concentrations resulting from land application of biosolids containing 1,1-dichloroethane have been described in Section 4.1.5, resulting in moderate confidence for 1,1-dichloroethane soil and soil pore water concentrations from biosolid land application.

Combining the moderate exposure confidence for the calculated soil and soil pore water concentrations based on AERMOD modeling of 1,1-dichloroethane air deposition from TRI-reported fugitive emissions with the respective hazard confidences for terrestrial mammalian and terrestrial plant assessments (moderate and slight, respectively) and trophic transfer confidence of moderate for the terrestrial mammalian assessment resulted in overall confidences of moderate and slight in the RQ inputs for the terrestrial mammalian and terrestrial plant assessments, respectively (Appendix Q). Although air deposition of 1,1-dichloroethane to soil from NEI-reported or environmental release-modeled fugitive and/or stack emissions (7 and 2 COUs, respectively) was assessed qualitatively, the same confidences of moderate and slight apply for the terrestrial mammal and terrestrial plant assessments, respectively. Combining the moderate exposure confidence for the calculated 1,1-dichloroethane soil and soil pore water concentrations based on biosolid land application with the respective hazard confidences for terrestrial mammalian and terrestrial plant assessments (moderate and slight, respectively) and trophic transfer confidence of moderate for the terrestrial mammalian assessment resulted in overall confidences of moderate and slight in the RQ inputs for the terrestrial mammalian and terrestrial plant assessments, respectively (Appendix Q).

4.3.6 Summary of Environmental Risk Characterization

Exposure concentrations were modeled based on COU-related releases to the aquatic and terrestrial environment. Table 4-15 displays RQ estimates for COU-related surface water releases to surface water, benthic pore water, and sediment (7 COUs):

- Manufacture/Domestic manufacturing/Domestic manufacturing
    - OES: Manufacturing of 1,1-dichloroethane as an isolated intermediate
- Processing/As a reactant/intermediate in all other basic organic chemical manufacture
- Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing
- Processing/Recycling/Recycling
    - OES: Processing as a reactive intermediate
- Processing/Processing – repackaging/Processing – repackaging
    - OES: Processing – repackaging
- Commercial Use/Other use/Laboratory chemicals
    - OES: Commercial use as a laboratory chemical
- Disposal/Disposal/Disposal
    - OES: General waste handling, treatment, and disposal
    - OES: Waste handling, treatment, and disposal (POTW)
    - OES: Waste handling, treatment, and disposal (remediation)

Table 4-16 displays RQ estimates and/or qualitative estimates of risk for COU-related releases resulting in air deposition to soil (8 COUs) and biosolid land application to soil (1 COU):

- Manufacture/Domestic manufacturing/Domestic manufacturing
    - OES: Manufacturing
- Processing/As a reactant/Intermediate in all other basic organic chemical manufacture
- Processing/As a reactant/Intermediate in all other chemical product and preparation Manufacturing
- Processing/Recycling/Recycling
    - OES: Processing as a reactive intermediate
- Processing/Processing – repackaging/Processing – repackaging
    - OES: Processing – repackaging
- Commercial Use/Other use/Laboratory chemicals
    - OES: Commercial use as a laboratory chemical
- Disposal/Disposal/Disposal
    - OES: General waste handling, treatment, and disposal
    - OES: Waste handling, treatment, and disposal (POTW)
- Distribution in Commerce/Distribution in commerce/Distribution in commerce
    - OES: Distribution in commerce

Table 4-15 displays RQ estimates for seven COUs in modeled 1,1-dichloroethane concentrations in surface water, benthic pore water, and sediment. Within the water column, acute RQs were below 1 for all seven COUs. Although chronic RQs based on a 21-day (hazard-based) release for aquatic receptors are above 1 for five COUs, with days of exceedance equal to or greater than the duration of exposure, the corresponding chronic RQs based on total number of operating days were below 1. Since EPA lacks information on estimated days of 1,1-dichloroethane release to surface waters for each COU/OES, total number of operating days was assumed as the maximum release duration and a chronic hazard-based duration was assumed as a lower-end release duration. However, it is likely that actual days of release of 1,1-dichloroethane to surface waters (and thereby refined RQ values) for each COU/OES falls somewhere in between these two durations. The Manufacturing COU/OES had the highest chronic and algal RQ values based on the hazard-based duration (RQs = 15 and 1.4, respectively) and total number of operating days (RQs = 0.91 and 0.085, respectively). The estimated exposure concentrations in water for the manufacturing COU/OES are based on TRI data from a single facility. The confidence in the acute and chronic aquatic RQ inputs were rated as robust and confidence in the algal RQ inputs rated as moderate as described in Section 4.3.5.1. Benthic pore water and sediment RQs were below 1 for all seven COUs. The confidence in the benthic RQ inputs were rated as moderate as described in Section 4.3.5.1. Because of 1,1-dichloroethane's high water solubility and relatively low log K_{OC}, EPA expects 1,1-dichloroethane to partition more to water than to sediment.

Table 4-16 displays RQ estimates for five COUs in calculated 1,1-dichloroethane concentrations in soil and soil pore water from air deposition of fugitive emissions (5 COUs) or biosolid land application (1 COU). Risk was also qualitatively estimated for eight COUs for air deposition of 1,1-dichloroethane to soil and soil pore water. RQs for terrestrial plants from 1,1-dichloroethane exposure in soil pore water were below 1 for all five COUs and expected to be below 1 for the remaining three COUs from air deposition and below 1 for the one COU from biosolids land application. The confidence in these RQ inputs were rated as slight as described in Section 4.3.5.1. RQ estimates for the trophic transfer of 1,1-dichloroethane to insectivorous (short-tailed shrew) or herbivorous (meadow vole) terrestrial mammals were below 1 for five COUs and expected to be below 1 for eight COUs based on NEI release data for air deposition to soil and soil pore water and below 1 for the one COU in soil and soil pore water from biosolids land application. The confidence in these RQ inputs were rated as moderate as described in Section 4.3.5.1. Additionally, Table 4-16 displays RQ estimates for seven COUs for trophic transfer of 1,1-dichloroethane from biota in surface water and sediment to semi-aquatic terrestrial mammals. RQ

estimates for trophic transfer of 1,1-dichloroethane to semi-aquatic terrestrial mammals based on fish consumption or crayfish consumption were below 1 for all seven COUs in surface water and benthic pore water, respectively. The confidence in these RQ inputs were rated as “moderate” as described in Section 4.3.5.1. Avian and soil invertebrate assessments are not reflected in Table 4-16 due to lack of reasonably available hazard evidence.

Table 4-15. COUs and Corresponding Environmental Risk for Aquatic Receptors Exposed to 1,1-Dichloroethane in Surface Water, Benthic Pore Water, and Sediment

COU (Life Cycle Stage/Category/ Subcategory)	OES	Aquatic Receptors [a] [b]											
		Surface Water						Benthic Pore Water				Sediment	
		Acute (Robust) [e]		Chronic (Robust) [e]		Algal (Moderate) [e]		Acute (Moderate) [e]		Chronic (Moderate) [e]		Chronic (Moderate) [e]	
		RQ [c]	DOE [d]	RQ [c]	DOE [d]	RQ [c]	DOE [d]	RQ [c]	DOE [d]	RQ [c]	DOE [d]	RQ [c]	DOE [d]
Manufacture/ Domestic manufacturing/ Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	0.048–0.81	0	0.91 to 15	0–21	0.085–1.4	0–13	4.4E−02 to 0.23	0	1.1E−02 to 6.1E−02	0	0.043–0.18	0
Processing/As a reactant/ Intermediate in all other basic organic chemical manufacture	Processing as a reactant	7.3E−03 to 0.13	0	0.14–2.5	0–21	0.013–0.24	0	7.0E−03 to 3.8E−02	0	1.8E−03 to 9.8E−03	0	6.8E−03 to 2.7E−02	0
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing													
Processing/Recycling/ Recycling													
Processing/Processing – repackaging/Processing – repackaging	Processing – repackaging	4.0E−04 to 4.9E−03	0	7.6E−03 to 9.3E−02	0	7.0E−04 to 8.7−03	0	3.4E−04 to 1.4E−03	0	9.0E−05 to 3.7E−04	0	3.3E−04 to 1.1E−03	0
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	3.6E−04 to 4.4E−03	0	6.9E−03 to 8.4E−02	0	6.4E−04 to 7.8E−03	0	3.1E−04 to 1.3E−03	0	8.0E−05 to 3.4E−04	0	2.9E−04 to 9.8E−04	0
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	7E−03 to 0.12	0	0.13 to 2.3	0–21	0.012 to 0.022	0	6.7E−03 to 3.5E−02	0	1.7E−03 to 9.1E−03	0	6.4E−03 to 2.6E−02	0
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	4.6E−03 to 8.1E−02	0	0.088 to 1.5	0–21	0.0082 to 0.14	0	4.4E−03 to 2.3E−02	0	1.2E−03 to 6.0E−03	0	4.3E−03 to 1.7E−02	0
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (remediation)	1.7E−02 to 0.33	0	0.33 to 6.2	0–35	0.031– to 0.58	0	1.7E−02 to 9.5E−02	0	4.3E−03 to 2.5E−02	0	1.6E−02 to 7.3E−02	0
Distribution in Commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce	N/A [k]											

COU (Life Cycle Stage/Category/ Subcategory)	OES	Aquatic Receptors [a] [b]											
		Surface Water						Benthic Pore Water				Sediment	
		Acute (Robust) [e]		Chronic (Robust) [e]		Algal (Moderate) [e]		Acute (Moderate) [e]		Chronic (Moderate) [e]		Chronic (Moderate) [e]	
		RQ [c]	DOE [d]	RQ [c]	DOE [d]	RQ [c]	DOE [d]	RQ [c]	DOE [d]	RQ [c]	DOE [d]	RQ [c]	DOE [d]

Modeled 1,1-dichloroethane concentrations and RQ values for all relevant COUs are available in Table 4-11, Table 4-12, Table 4-13, and Table 4-14.
[a] Risk assessed to aquatic receptors based on 1,1-dichloroethane releases to surface waters.
[b] All exposure values and DOEs modeled using PSC.
[c] Acute Risk Quotient (ARQ) derived using an acute Concentration of Concern of 1,769 ppb.
[d] DOE modeled using PSC.
[e] Confidence in Acute Risk Quotient (ARQ), Chronic Risk Quotient (CRQ), or Algal Risk Quotient inputs is detailed in Section 4.3.4.
[f] Chronic Risk Quotient (CRQ) derived using a chronic Concentration of Concern of 93 ppb and presented as a range based on 21-day release or total number of operating days (Table 3-3).
[g] Algal Risk Quotient derived using an algal Concentration of Concern of 1,000 ppb and presented as a range based on a 4-day release or total number of operating days (Table 3-3).
[h] Chronic Risk Quotient (CRQ) for sediment derived using benthic chronic Concentration of Concern of 2,900 ppb and presented as a range based on a 15-day release or total number of operating days (Table 3-3).
[i] ARQ for benthic pore water derived using benthic acute Concentration of Concern of 1,769 ppb.
[j] CRQ for benthic pore water derived using benthic chronic Concentration of Concern of 6,800 ppb and presented as a range based on a 35-day release or total number of operating days (Table 3-3).
[k] Distribution in Commerce does not result in surface water releases (Table 3-6).

Table 4-16. COUs and Corresponding Environmental Risk for Terrestrial Receptors Exposed to 1,1-Dichloroethane in Soil Pore Water (Plants) and Trophic Transfer

COU (Life Cycle Stage/Category/	OES	Terrestrial Receptors[a]									
		Soil Pore Water (Plants)		Trophic Transfer (Soil and Soil Pore Water) [b]				Trophic Transfer (Water) [c]		Trophic Transfer (Sediment) [c]	
		Plant RQ	Conf. in RQ Inputs [d]	Shrew RQ	Conf. in RQ Inputs [d]	Vole RQ	Conf. in RQ Inputs [d]	Mink RQ	Conf. in RQ Inputs [d]	Mink RQ	Conf. in RQ Inputs [d]
Manufacture/Domestic manufacturing/Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	3.3E−06	Slight	3.9E−06	Moderate	1.3E−06	Moderate	1.2E−04[e]	Moderate	1.1E−04 [f]	Moderate
Processing/As a reactant/ Intermediate in all other basic organic chemical manufacture	Processing as a reactant	1.8E−04	Slight	2.1E−04	Moderate	6.9E−05	Moderate	1.8E−05 [e]	Moderate	1.7E−05 [f]	Moderate
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing											
Processing/Recycling/Recycling											
Processing/Processing – repackaging/Processing – repackaging	Processing – repackaging	Risk estimates for air deposition to soil expected to be less than those generated based on TRI-fugitive emissions						9.7E−07	Moderate	8.5E−07	Moderate
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	Risk estimates for air deposition to soil expected to be less than those generated based on TRI-fugitive emissions						8.8E−07	Moderate	7.6E−07	Moderate
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	5.0E−07	Slight	5.8E−07	Moderate	1.9E−07	Moderate	1.7E−05 [e]	Moderate	1.6E−05 [f]	Moderate
	Waste handling, treatment, and disposal (POTW)	2.3E−05 [g]	Slight	2.6E−05 [g]	Moderate	8.7E−06 [g]	Moderate	1.1E−05 [e]	Moderate	1.1E−05 [f]	Moderate
		4.6E−05 [h]	Slight	5.3E−05 [h]	Moderate	1.7E−05 [h]	Moderate				
	Waste handling, treatment, and disposal (remediation)	N/A						1.2E−04 [e]	Moderate	1.2E−04 [f]	Moderate
Distribution in Commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce	Risk estimates for air deposition to soil expected to be less than those generated based on TRI-fugitive emissions						N/A[i]			

[a] Exposure to terrestrial receptors based on 1,1-dichloroethane releases as fugitive air and stack air deposition to soil, biosolids land application, and trophic transfer. RQs generated for air deposition to soil based on TRI-fugitive emissions of 1,1-dichloroethane.
[b] Estimated concentrations of 1,1-dichloroethane (95th percentile) that could be in soil via daily air deposition at a conservative (10 m from the source) exposure scenario.
[c] Fish and crayfish concentrations (mg/kg) were calculated using surface water and benthic pore water concentrations of 1,1-dichloroethane, respectively, from PSC assuming a BCF of 7 as estimated by EPI Suite™ (U.S. EPA, 2012c).
[d] Conf. = Confidence; Confidence in Risk Quotient (RQ) inputs are detailed in Section 4.3.4.
[e] Mink RQ based on fish concentrations of 1,1-dichloroethane.
[f] Mink RQ based on crayfish concentrations of 1,1-dichloroethane.

COU (Life Cycle Stage/Category/	OES	Terrestrial Receptors[a]									
		Soil Pore Water (Plants)		Trophic Transfer (Soil and Soil Pore Water) [b]				Trophic Transfer (Water) [c]		Trophic Transfer (Sediment) [c]	
		Plant RQ	Conf. in RQ Inputs [d]	Shrew RQ	Conf. in RQ Inputs [d]	Vole RQ	Conf. in RQ Inputs [d]	Mink RQ	Conf. in RQ Inputs [d]	Mink RQ	Conf. in RQ Inputs [d]

[g] Tilled agricultural soil type.
[h] Pastureland soil type.
[i] Distribution in Commerce does not result in surface water releases (Table 3-6).

5 HUMAN HEALTH RISK ASSESSMENT

5.1 Human Exposures

EPA evaluated all reasonably available information for occupational and general population human exposures, including consideration of increased exposure or susceptibility across PESS considerations (see Section 5.3.2). Exposures for consumers are not evaluated as no consumer use of 1,1-dichloroethane was identified in Section 1.1.3, Populations Assessed.

5.1.1 Occupational Exposures

1,1-Dichloroethane – Occupational Exposures (Section 5.1.1): Key Points

EPA evaluated the reasonably available information for occupational exposures. The following bullets summarize the key points of this section of the risk evaluation:

- EPA identified OESs for each COU of 1,1-dichloroethane.
- EPA assessed occupational exposures for each OES.
- The objective was to identify exposure groups for each OES and assess their exposure.
- For each OES, central tendency and high-end doses were estimated. Estimates based on monitoring data used the 50th and 95th percentile of the datasets for the central tendency and high-end exposures. Estimates based on modeling used probabilistic modeling approaches with Monte Carlo to identify the 50th and 95th percentile for central tendency and high-end exposures.
- EPA estimated occupational inhalation exposure (in ppm as an 8-hour TWA) and dermal exposures (in mg/day) to 1,1-dichloroethane and provided both high-end and central tendency exposures for occupational exposure scenarios associated with each OES.
    - Monitoring data for 1,1-dichloroethane was available for the Manufacturing OES. For the remaining OESs, exposures were estimated using the 1,1-dichloroethane manufacturing exposure data as well as surrogate exposure data for 1,2-dichloroethane and other solvents assessed in previous EPA risk evaluations and modeling.
    - High-end inhalation exposures range from 2.4×10^{-2} ppm to 13 ppm. High-end dermal exposures are 0.11 mg/day for the dilute waste disposal scenario and 6.7 mg/day for all other OESs.
    - Central tendency inhalation exposures range from 1.1×10^{-3} ppm to 3.5 ppm. Central tendency dermal exposures are 6.5×10^{-2} mg/day for the dilute waste disposal scenario and 2.2 mg/day for the other OESs.
- EPA also evaluated the weight of scientific evidence for the exposure assessment of each OES.

Where there was sufficient detail in the monitoring data, EPA assessed exposure to Similar Exposure Groups (SEGs). For example, EPA received inhalation monitoring data for 1,1-dichloroethane manufacturing where an SEG for occupational non-users (ONUs; see more below) was identified and monitored. If SEGs were not available from the monitoring data or were not able to be assessed from the modeling approach used, EPA followed its standard practice to assess exposure to (1) a generic exposure group of "workers" (*i.e.*, workers who may handle the chemical and have direct contact); and (2) a generic exposure group (ONUs; workers who work in the general vicinity of the "workers" but do not

handle the chemical and have direct contact). Where possible, for each OES, EPA identified job types and categories for workers and ONUs.

1,1-Dichloroethane has a vapor pressure of approximately 228 mmHg at 25 °C. Based on this high volatility, EPA anticipates that workers and ONUs will be exposed to vapor via the inhalation route. Based on the physical state, the Agency does not expect particulate or mist inhalation. EPA expects worker exposure to liquids via the dermal route but does not expect dermal exposure for ONUs because they do not directly handle 1,1-dichloroethane.

The United States has several regulatory and non-regulatory exposure limits for 1,1-dichloroethane, including the Occupational Safety and Health Administration (OSHA) permissible exposure limit (PEL) (29 CFR 1910.1000) is 100 ppm or 400 mg/m^3 over an 8-hour work day, time-weighted average (TWA) (OSHA, 2019). 1,1-Dichloroethane has a National Institute for Occupational Safety and Health (NIOSH) recommended exposure limit (REL) of 100 ppm (400 mg/m^3) TWA (NIOSH, 2018). Furthermore, the American Conference of Governmental Industrial Hygienists (ACGIH) set a threshold limit value (TLV) at 100 ppm TWA for 1,1-dichloroethane.

The following subsections briefly describe EPA's approach to assessing occupational exposures and results for each COU assessed. For additional details on development of approaches and results refer to *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

5.1.1.1 Approach and Methodology

EPA's approach for assessing occupational exposure to 1,1-dichloroethane is illustrated below in Figure 5-1.

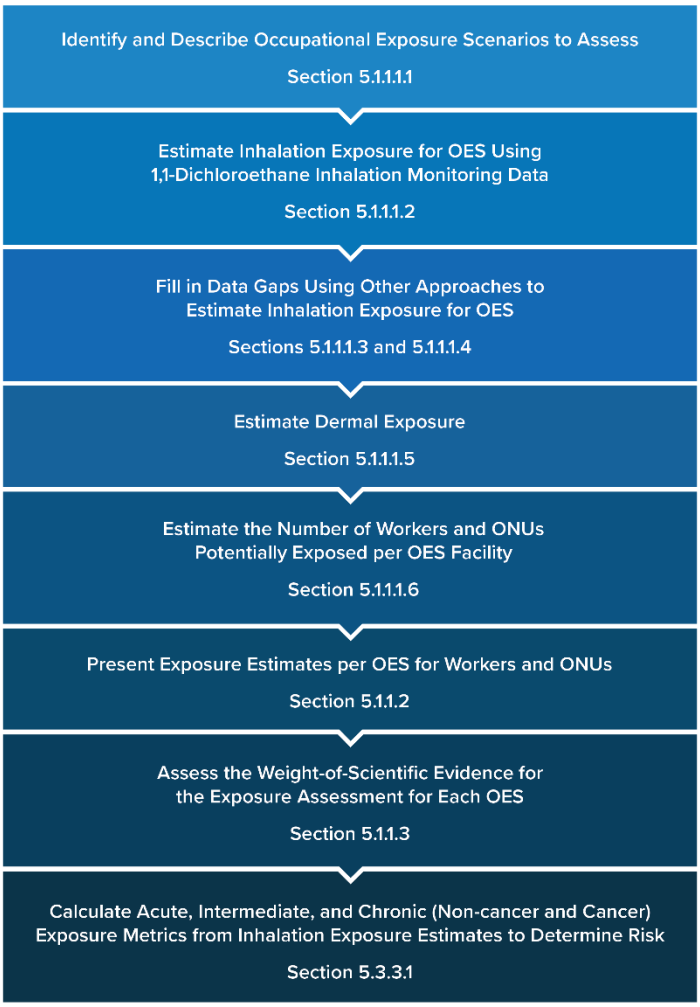


Figure 5-1. Overview of EPA's Approach to Estimate Occupational Exposures for 1,1-Dichloroethane

EPA follows the hierarchy established in Table 5-1 in selecting data and approaches for assessing occupational exposures. The basis of this hierarchy is from the *1991 CEB Manual* (CEB, 1991).

Table 5-1. Data and Approaches for Assessing Occupational Exposures to 1,1-Dichloroethane

Type of Approach	Description
1. Monitoring data	a) Personal and directly applicable
	b) Area and directly applicable
	c) Personal and potentially applicable or similar
	d) Area and potentially applicable or similar
2. Modeling approaches	a) Surrogate monitoring data
	b) Fundamental modeling approaches
	c) Statistical regression modeling approaches
3. Occupational exposure limits	a) Company-specific occupational exposure limits (OELs) (for site-specific exposure assessments; for example, there is only one manufacturer who provided their internal OEL to EPA but did not provide monitoring data)
	b) OSHA PELs
	c) Voluntary limits: ACGIH TLVs, NIOSH RELs, Occupational Alliance for Risk Science (OARS) workplace environmental exposure level (WEELs; formerly by AIHA)

EPA received inhalation monitoring data from the test order submission for both 1,1-dichloroethane manufactured as an isolated intermediate and 1,1-dichloroethane manufactured as a byproduct in the manufacture of 1,2-dichloroethane (Stantec ChemRisk, 2023). The OES of 1,1-Dichloroethane manufactured as an isolated intermediate is assessed in this risk evaluation. In accordance with an EPA decision made during scoping, the OES of 1,1-Dichloroethane manufactured as a byproduct will be assessed in the risk evaluation for 1,2-dichloroethane.

For additional information regarding the approaches taken to estimate occupational exposures, refer to Sections 5.1.1.1.1 through 5.1.1.1.5.

5.1.1.1.1 Identify and Describe Occupational Exposure Scenarios to Assess

As discussed in Section 3.1.1.1, EPA has identified seven OESs from the COUs to group scenarios with similar sources of exposure at industrial and commercial workplaces within the scope of the risk evaluation. EPA assessed occupational exposures during the Distribution in commerce of 1,1-dichloroethane qualitatively. Under the Waste handling, treatment, and disposal COU, EPA assessed occupational exposures for the OES of General disposal and POTW (Table 5-2).

Table 5-2. Similar Exposure Groups (SEGs) for 1,1-Dichloroethane

OES	Similar Exposure Groups (SEGs) for 1,1-Dichloroethane
Manufacturing of 1,1-dichloroethane as an isolated intermediate	Operators/Process technicians operate production control panels, record process parameters, conduct walk-throughs of production areas, perform equipment checks, and collect process samples. Logistic or distribution technician duties may include sampling, moving, spotting, and loading or unloading of bulk chemicals in storage tanks, railcars, barges, and/or tanker trucks, which requires connection and disconnection of loading hoses. Maintenance technicians install equipment, troubleshoot problems, diagnose issues, repair equipment or machinery in process areas of maintenance shops. Laboratory technicians conduct laboratory tests to assist with quality control, perform chemical experimentation, testing and analyses. ONUs perform office work, control board operations, production area walk-throughs. The test order summary report included durations of individual tasks performed by the worker with the majority being 1 hour or less which is inclusive of maintenance tasks such as leak repair.
Processing as a reactive intermediate	SEGs are expected to be similar as for Manufacturing.
Processing – repackaging	EPA assessed the general SEG categories of workers and ONUs. Workers are potentially exposed to 1,1-dichloroethane when transferring 1,1-dichloroethane from bulk containers into smaller containers. Workers may also be exposed via inhalation of vapor or dermal contact with liquids when cleaning transport containers following emptying. ONUs are expected to have lower inhalation exposures, lower vapor-through-skin uptake, and no dermal exposure.
Distribution in commerce	Occupational exposure was not assessed. The activities of loading 1,1-dichloroethane product into transport containers and unloading at receiving sites as well as repackaging into smaller containers can be considered part of Distribution in commerce and these are assessed but under those OESs. Cleanup of accidents/spills that may occur during transport are not within the scope of this risk evaluation.
Commercial use as a laboratory chemical	Laboratory technicians conduct laboratory tests to assist with quality control, perform chemical experimentation, testing and analyses. During these activities workers may be exposed via inhalation of vapor or dermal contact with 1,1-dichloroethane. EPA also assessed the general SEG of ONU. ONUs are expected to have lower inhalation exposures, lower vapor-through-skin uptake, and no dermal exposure.
General waste handling, treatment, and disposal	EPA assessed the general SEG categories of workers and ONUs. Workers are potentially exposed to 1,1-dichloroethane during the unloading and cleaning of transport containers. Workers may experience inhalation of vapor or dermal contact with liquids during the unloading process. ONUs are expected to have lower inhalation exposures, lower vapor-through-skin uptake, and no dermal exposure.
Waste handling, treatment, and disposal (POTW)	EPA assessed the general SEG categories of workers and ONUs. Workers are potentially exposed to 1,1-dichloroethane during the unloading and cleaning of transport containers. Workers may experience inhalation of vapor or dermal contact with liquids during the unloading process. ONUs are expected to have lower inhalation exposures, lower vapor-through-skin uptake, and no dermal exposure.
Waste handling, treatment, and disposal (remediation)	EPA did not assess occupational exposures during remediation of 1,1-dichloroethane. 1,1-dichloroethane is a contaminant removed by a remediation process. EPA did not find evidence that 1,1-dichloroethane is used for remediation.

5.1.1.1.2 Estimate Inhalation Exposure for OES Using 1,1-Dichloroethane Inhalation Monitoring Data

EPA used the evaluation strategies described in the *Application of Systematic Review in TSCA Risk Evaluations* (U.S. EPA, 2018) to collect inhalation exposure monitoring data. The Agency's approach is to collect inhalation monitoring data from literature sources and then evaluate the quality of the data. Data having high-, medium-, or low-quality ratings would then be used in the risk evaluation for estimating exposures. In general, higher ratings are given preference over lower ratings; however, lower ranked data may be used over higher ranked data when specific aspects of the data are carefully examined and compared. For example, a lower ranked data set that precisely matches the OES of interest may be used over a higher ranked study that does not as closely match the OES of interest.

EPA reviewed workplace inhalation monitoring data for 1,1-dichloroethane that was collected through test orders and also searched for 1,1-dichloroethane 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). EPA considered 8-hour TWA personal breathing zone (PBZ) monitoring data first.

1,1-Dichloroethane Test Order Monitoring Data

Occupational inhalation data for 1,1-dichloroethane during manufacturing as an isolated intermediate were provided via a test order submission from the Vinyl Institute, which includes manufacturers and processors of 1,1-dichloroethane (Stantec ChemRisk, 2023). The Vinyl Institute prepared a study plan for inhalation monitoring to collect inhalation monitoring data, including identification of representative sites for sampling. This testing consortium provided information on 12 total sites from their members that manufacture 1,1-dichloroethane as an isolated intermediate and/or non-isolated byproduct and selected 4 representative sites for sampling. EPA reviewed and approved the monitoring study plan.

These data were used to estimate inhalation exposures for the following OESs: Manufacturing of 1,1-dichloroethane as an isolated intermediate, Processing as a reactive intermediate, and Commercial use of laboratory chemicals. The test order submission also included inhalation data for the unintentional manufacture of 1,1-dichloroethane as a byproduct during the manufacture of 1,2-dichloroethane. Although this scenario is not included in this risk evaluation, it will be addressed in the risk evaluation for 1,2-dichloroethane.

Engineering Controls: Information on engineering controls at the facilities where 1,1-dichloroethane was monitored were provided in the test order submission. Production, logistics, and maintenance activities primarily occurred outdoors. Indoor work areas ranged in size and included process labs and control rooms. The type of engineering controls differed by process area. For example, in production areas, facilities utilized a closed loop sampling system to collect process samples. Outdoor process areas were equipped with alarms to monitor for leaks or emissions. Nitrogen purges were utilized before accessing opened process lines. If a portion of a process line was required to be isolated, the flow of product to the area was blocked by valves. In logistics areas, one outdoor shipping unit implemented a vapor recovery system to remove vapors from storage tanks and other storage vessels. Laboratories were all indoors. Engineering controls present in the laboratories included fume hoods, enclosed GC analyses and additional fans and local exhaust ventilation above lab processes.

Administrative Controls: Information on administrative controls at the facilities where 1,1-dichloroethane was monitored was also provided in the test order submission. Facilities implemented restricted entry procedures, ensuring that only employees which had been granted access and received

appropriate training were allowed to enter process areas, control rooms, and laboratories. Every facility used written SOPs for all job tasks and required employee training on appropriate conduct of the SOPs. Additional administrative controls included PPE matrices and hazard assessments posted in control rooms for operators and other process technicians to refer to regularly.

Personal Protective Equipment (PPE): Information on PPE requirements at the facilities monitored is provided in Section 5.3.3.1.

Manufacturing of 1,1-Dichloroethane as an Isolated Intermediate

EPA identified 55 worker and 7 ONU full-shift PBZ samples from the test order data to estimate inhalation exposures during the manufacturing process. The worker samples collected were from operators/process technicians, maintenance technicians, and laboratory technicians. From this monitoring data, EPA calculated the 50th and 95th percentile, 8-hour TWA concentrations to represent a central tendency and high-end estimate of potential occupational inhalation exposures, respectively, for this OES for the five SEGs identified. In addition, 36 task-length samples were collected for these workers. These samples were shorter in duration, ranging from 15 to 176 minutes. For further discussion of the task length samples, refer to Section 5.1.4.3 in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

Table 5-3. Summary of Manufacturing Inhalation Exposures to 1,1-Dichloroethane

Occupational Exposure Scenario (OES)	Type of Data	Vapor Pressure (mmHg)	Similar Exposure Group (SEG)	# of Data Points	# of Non Detects	Worker Inhalation Estimates (8-hour TWA ppm)	
						Central Tendency	High-End
Manufacturing of 1,1-dichloroethane as an isolated intermediate	1,1-Dichloroethane test order data (full-shift PBZ samples)	228	Operator/Process Technician	27	1	7.8E−03	0.73
			Logistics Technician	9	0	2.8E−03	5.3E−03
			Maintenance Technician	8	2	7.9E−02	0.41
			Laboratory Technician	9	0	1.1E−03	2.4E−02
			ONU	7	0	1.8E−03	1.8E−02

ONU was defined in the test order as workers performing office work, control board operations, production area walk-throughs.
Source of data was test order submission from the Vinyl Institute, which includes manufacturers and processors of 1,1-dichloroethane (Stantec ChemRisk, 2023). Data were rated "high" in systematic review.
Sample durations were often longer than 8 hours; 8-hour TWAs were calculated from the full shift results by multiplying the full shift exposure (ppm) × (sample duration [hours]/8-hour)
For the non-detects, data sets for both the operator/process technician had a geometric standard deviation >3, so ND values were divided by 2.
The Operator/Process Technician SEG does not include 2 data points that were identified as outliers by the text submitter—these were full shift sample data points (J-FS-OP-31, and J-FS-OP-33). These outliers were separated and included in a separate SEG in Table 5-4 for Operator/Process Technician and Emergency SEGs.

Among the operator/process technician samples, two full-shift samples were specifically related to responding to line leaks and identified by the submitter in the inhalation test order report to be outliers for the Operator/process technician SEG. These data were characterized in the summary report as abnormal plant conditions and emergency response. As a result, per SACC recommendation, EPA created a new SEG to characterize exposures for these activities. These exposures occur at a lower

exposure frequency and were not evaluated for chronic health effects. The results for are presented in Table 5-4.

Table 5-4. Inhalation Exposure Results for Operator/Process Technician and Emergency SEGs

Occupation Exposure Scenario (OES)	Type of Data	Vapor Pressure (mmHg)	Similar Exposure Group (SEG)	# of Data Points	Worker Inhalation Estimates (8-hour TWA, ppm)
					High-End
Manufacturing of 1,1-dichloroethane as an isolated intermediate	1,1-Dichloroethane test order data	228	Operator/Process Technician (responding to line leaks)	2[a]	1.9

[a] Prior to calculating summary statistics for each COU, Rosner's outlier test was used to detect outliers at a 5% significance level (Rosner, 1983). Two full shift sample data points (J-FS-OP-31, J-FS-OP-33) collected during the sampling effort were determined to be outliers in the dataset.
There were no non-detects for this SEG.

Processing as a Reactive Intermediate

EPA did not identify monitoring data for the processing as a reactive intermediate OES; however, the Agency assumed the exposures to be similar to manufacturing due to similar worker activities and the use of primarily closed systems during processing. Therefore, EPA incorporated the manufacturing data into the processing as a reactive intermediate exposure estimates as "analogous data." EPA refers to analogous monitoring data as monitoring data for the same chemical and similar OES. The Agency has used this assessment approach in previous risk evaluations, such as the *Risk Evaluation for Perchloroethylene (PCE)* (U.S. EPA, 2020g).

Table 5-5. Summary of Processing as a Reactive Intermediate Inhalation Exposure Estimates

OES	Type of Data	Vapor Pressure (mmHg)	Similar Exposure Group (SEG)	# of Data Points	# of Non-Detects	Worker Inhalation Estimates (ppm)	
						Central Tendency	High-End
Processing as a reactive intermediate	1,1-Dichloroethane test order data	228	Operator/Process Technician	27	1	7.8E−03	0.73
			Logistics Technician	9	0	2.8E−03	5.3E−03
			Maintenance Technician	8	2	7.9E−02	0.41
			Laboratory Technician	9	0	1.1E−03	2.4E−02
			ONU	7	0	1.8E−03	1.8E−02

Source of data was test order submission from the Vinyl Institute, which includes manufacturers and processors of 1,1-dichloroethane (Stantec ChemRisk, 2023).
The Operator/Process Technician SEG does not include 2 data points that were identified as outliers by the text submitter these were full shift sample data points (J-FS-OP-31, and J-FS-OP-33). These outliers were separated and included in a separate SEG in Table 5-4 for Operator/Process Technician and Emergency SEGs.

Commercial Use as a Laboratory Chemical

During the manufacturing process, EPA identified nine worker full-shift samples for laboratory technicians. The test order provided exposure data where a laboratory at the manufacturing facility was used for analysis of samples from the manufacturing process. The test order provided detail on room dimensions and ventilation rates for the laboratory. This data is used as analogous data for occupational exposure for the laboratory chemical use. EPA did not find data from its data search procedures of 1,1-dichloroethane use in other types of laboratory settings. The Agency is assuming the test order data is applicable to other types of laboratories associated with commercial use of 1,1-dichloroethane as a laboratory chemical.

Table 5-6. Summary of Commercial Use as a Laboratory Chemical Inhalation Exposure Estimates

OES	Type of Data	Vapor Pressure (mmHg)	Similar Exposure Group (SEG)	# of Data Points	Worker Inhalation Estimates (ppm)	
					Central Tendency	High-End
Commercial use as a laboratory chemical	1,1-Dichloroethane test order data	228	Laboratory Technician	9	1.1E−03	2.4E−02

Table 5-7. Summary of Approaches for the Occupational Exposure Scenarios Using 1,1-Dichloroethane Monitoring Data

OES	1,1-Dichloroethane Monitoring Data Approach
Manufacturing of 1,1-dichloroethane as an isolated intermediate	For the purposes of this risk evaluation, EPA used 1,1-dichloroethane test order data from the Vinyl Institute during the manufacturing of 1,1-dichloroethane as an isolated intermediate.
Processing as a reactive intermediate	EPA used 1,1-dichloroethane test order data from the Vinyl Institute during the manufacturing of 1,1-dichloroethane as an isolated intermediate due to expected similarities in exposure points.
Commercial use as a laboratory chemical	EPA used 1,1-dichloroethane test order data from the Vinyl Institute for laboratory technicians during manufacturing process. The Agency expects that laboratory exposures during manufacturing would be similar to exposures during commercial use.

For the remaining OESs, occupational inhalation exposure monitoring data for 1,1-dichloroethane were not available from the sources investigated. Therefore, EPA considered other assessment approaches as described in Sections 5.1.1.1.3 and 5.1.1.1.5, respectively.

5.1.1.1.3 Estimate Inhalation Exposure for OES Using Surrogate Monitoring Data

As described in Section 5.1.1.2, inhalation exposure monitoring data were not available for 1,1-dichloroethane for several of the OES. Therefore, EPA used monitoring data from 1,2-dichloroethane and methylene chloride to use as surrogate monitoring data for the same OES. EPA refers to "surrogate monitoring data" as monitoring data for a different chemical but the same (or similar) COU. Surrogate monitoring data are used when there are similarities in chemical properties, nature of workplace environment, and worker activities associated with the use of the chemical. Although PBZ surrogate data is not chemical-specific, it does provide evidence of exposure levels in actual workplaces and is generally preferred over the use of models, which may rely on conservative assumptions for parameter

values. Although using surrogate data can be a refinement over modeling, it can introduce some uncertainty in OES-specific estimates.

EPA determined exposure estimates using surrogate monitoring data for the following OESs: Waste handling, treatment, and disposal (general), and Waste handling, treatment, and disposal (specifically for POTWs). In both cases, the OESs are directly analogous; therefore, EPA expects the process and associated exposure points to be the same or similar. The Agency applied a vapor pressure correction factor when determining the exposure estimates for these OESs. For details on the application of vapor pressure correction factors to surrogate data, refer to *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

For the General waste handling, treatment, and disposal OES, EPA identified 22 full-shift worker samples from methylene chloride that were presented in the *Risk Evaluation for Methylene Chloride*. The inhalation exposure estimates for this OES are presented in Table 5-8.

Table 5-8. Summary of General Waste Handling, Treatment, and Disposal Inhalation Exposure Estimates

OES	Type of Data	Vapor Pressure (mmHg)	Worker Description	# of Data Points	Worker Inhalation Estimates (ppm)	
					Central Tendency	High-End
General waste handling, treatment, and disposal	Methylene chloride surrogate data	435	Worker	22	0.3	10
Source: Data from the *Risk Evaluation for Methylene Chloride* Data analysis for 1,1-dichloroethane and application of vapor pressure correction factor in *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).						

The high-end and central tendency exposures differ by a factor of 30. One possible explanation is that the data represent multiple SEGs with different exposure potentials and perhaps the majority of the data is for a worker SEG with a lower potential of exposure.

For the Waste handling, treatment, and disposal (POTW) OES, EPA identified from a search of the existing data for three full-shift worker samples from 1,2-dichloroethane. The inhalation exposure estimates for this OES are presented in Table 5-9.

Table 5-9. Summary of Waste Handling, Treatment, and Disposal (POTW) Inhalation Exposure Estimates

OES	Type of Data	Vapor Pressure (mmHg)	Worker Description	# of Data Points	Worker Inhalation Estimates (ppm)	
					Central Tendency	High-End
General waste handling, treatment, and disposal	1,2-dichloroethane surrogate data	79	Worker	3	0.25	0.68
For data sets of 3–5 data points, EPA's practice is to present the median value as the central tendency and the maximum value as the high-end.						

Table 5-10. Approach for the Occupational Exposure Scenarios Using Surrogate Monitoring Data

OES	Surrogate Monitoring Data Approach
General waste handling, treatment, and disposal	EPA used surrogate monitoring data from methylene chloride.
Waste handling, treatment, and disposal (POTW)	EPA used surrogate monitoring data from 1,2-dichloroethane.

For additional details on the use of surrogate monitoring data, refer to *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

5.1.1.1.4 Approaches for Estimating Inhalation Exposure for Remaining OESs and ONU Exposures

This section outlines the method for estimating inhalation exposures for the remaining OES lacking chemical-specific, analogous, or surrogate monitoring data, as well as the approach for estimating ONU exposures in the absence of data.

EPA did not identify inhalation monitoring data from 1,1-dichloroethane or surrogate data from other chemicals to assess exposures during the Processing – repackaging of 1,1-dichloroethane OES. Therefore, EPA estimated inhalation exposures using a Monte Carlo simulation with 100,000 iterations and the Latin Hypercube sampling method using the models and approaches described in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

For this OES, the Agency applied the EPA Mass Balance Inhalation Model to exposure points described in the July 2022 Chemical Repackaging GS (U.S. EPA, 2022a)—particularly for the unloading of drums, loading of containers, and cleaning of drums process. The EPA Mass Balance Inhalation Model estimates the concentration of the chemical in the breathing zone of the worker based on a vapor generation rate (*G*). An 8-hour TWA is then estimated and averaged over eight hours assuming no exposure occurs outside of those activities.

EPA used the vapor generation rate and exposure duration parameters from the *1991 CEB Manual* (CEB, 1991) in addition to those used in the EPA Mass Balance Inhalation Model to determine a time-weighted exposure for each exposure point. EPA estimated the time-weighted average inhalation exposure for a full work-shift (EPA assumed an 8-hour work-shift) as an output of the Monte Carlo simulation by summing the time-weighted inhalation exposures for each of the exposure points and assuming 1,1-dichloroethane exposures were zero outside these activities. EPA modeled three exposure scenarios: one where the worker performs all three repackaging activities (unloading, loading, and cleaning) on the same day, one where the worker performs unloading and cleaning on the same day, and one where the worker performs only loading activities. EPA did not have 1,1-dichloroethane specific information for repackaging. EPA's Chemical Repackaging GS provides estimates of daily throughput (kg/site-day) repackaged at a generic site. The model then calculates the time needed for unloading and loading. The model has a standard default for the amount of time for cleaning. The inhalation exposure estimates for this OES are presented in Table 5-11.

Table 5-11. Summary of Processing – Repackaging Inhalation Exposure Estimates

OES	Type of Data	Worker Description	Worker Inhalation Estimates (ppm)	
			Central Tendency	High-End
Processing – repackaging	1,1-dichloroethane modeled data	All Activities (Loading, Unloading, and Cleaning)	3.5	13
		Unloading and Cleaning	1.7	6.6
		Loading	1.7	6.6

Table 5-12. Approach for the Occupational Exposure Scenarios Using Modeling

OES	Inhalation Exposure Modeling Approach
Processing – repackaging	EPA used assumptions and values from the July 2022 Chemical Repackaging GS (U.S. EPA, 2022a) and applied the EPA Mass Balance Inhalation Model to exposure points listed in that GS.

Where EPA is unable to estimate inhalation exposure to ONUs, either because the source of monitoring data lacks sufficient detail or the modeling approach does not estimate the exposures that an ONU receives, EPA's general practice in conducting risk evaluations is to make an assumption that the ONU exposure is equivalent to the workers central tendency exposure. Table 5-3 provides a summary of inhalation monitoring data at a manufacturing facility which shows that the central tendency exposure for the operator/process technician exposure group falls between the high-end and central tendency exposures for the ONU exposure group. This was done for the ONU exposures in the following OESs: Processing – repackaging; Commercial use as a laboratory chemical; General waste handling, treatment, and disposal; and Waste handling, treatment, and disposal (POTW). It should be noted that this approach has more uncertainty than the cases where EPA was able to estimate exposures to ONUs from 1,1-dichloroethane inhalation monitoring data where ONUs were monitored.

5.1.1.1.5 Estimate Dermal Exposure to 1,1-Dichloroethane

Dermal exposure monitoring data were not available for the OES in the assessment from systematic review of the literature and not collected in the dermal test order. Therefore, to assess dermal exposure, the Agency used the EPA Dermal Exposure to Volatile Liquids Model (DEVL) to calculate the dermal retained dose for each OES. For details on workers activities that could potentially result in dermal exposure, refer to Table 5-2.

DEVL Model

The DEVL Model (see Equation 5-1) modifies EPA/OPPT Dermal Exposure to Liquids Model (peer-reviewed) by incorporating a "fraction absorbed (f_{abs})" parameter to account for the evaporation of volatile chemicals:

Equation 5-1.

$$D_{exp} = (S \times Qu \times f_{abs} \times Y_{derm} \times FT)/BW$$

Where:

D_{exp} = Dermal retained dose (mg/kg-day)
S = Surface area of contact (cm^2)
Qu = Quantity remaining on the skin after an exposure event (high-end: 2.1 mg/cm^2-event, central tendency 1.4 mg/cm^2-event (U.S. EPA, 1992))

Y_{derm} = Weight fraction of the chemical of interest in the liquid (wt %)
FT = Frequency of events (default: 1)
f_{abs} = Fraction of applied mass that is absorbed (%)
BW = Body weight (kg)

This model determines an acute potential dose rate (APDR) based on an assumed amount of liquid on skin during contact event per day and the theoretical steady-state fractional absorption for 1,1-dichloroethane. The exposure concentration is determined based on EPA's review of currently available products and formulations containing 1,1-dichloroethane. The dose estimates assume one dermal exposure event (applied dose) per work day and approximately 0.30 percent of the applied dose is absorbed through the skin, for 1,1-dichloroethane in neat form and at 50 percent concentration in the 1,2-dichloroethane vehicle. This absorption value is very similar to the IH SkinPerm Model value of 0.285%. Chemical dermal absorption behavior is predicted in IH SkinPerm model based on the physical-chemical properties of the chemical. Based on concordance between the adjusted Test Order data for dermal absorption and IH SkinPerm, EPA is assuming that a portion of the missing mass is absorbed. For the Waste handling and disposal COU, the dermal absorption value of 0.06 percent was utilized tested at 10 percent 1,1-dichloroethane in 1,2-dichloroethane as the vehicle. Both of these values are based on empirical test order data utilizing human skin following the guideline dermal absorption study OECD 428.

DEVL Model Parameters

The standard model considers an assumed amount of liquid on skin during one contact event per day (*Qu*), an absorption factor (f_{abs}), surface area of the hands (*S*) and the weight fraction of 1,1-dichloroethane (Y_{derm}) in the formulation to calculate a dermal dose. The model reduces to an assumed amount of liquid on the skin during one contact event per day adjusted by the weight fraction of 1,1-dichloroethane in the liquid to which the worker is exposed. For the deterministic approach, EPA assumed the worker would be handling neat 1,1-dichloroethane therefore, EPA assessed all exposure scenarios at a 100 percent weight fraction. This is a realistic assumption as the workers are manufacturing neat 1,1-dichloroethane for subsequent use as a reactive intermediate. In addition, workers perform daily tasks such as product sampling that can results in dermal exposures. Per SACC recommendation, EPA assessed a dilute scenario for the Waste handling and disposal OES in which the worker would handle 1,1-dichloroethane at a 10 percent weight fraction. EPA also used the 10 percent dilute fractional absorption value as reported in the dermal test order. Table 5-13 summarizes the model parameters and their values for estimating dermal exposures.

Table 5-13. Summary of Dermal Model Input Values

Input Parameter	Symbol	Value(s)	Unit	Reference
Dermal load	Q_u	1.4 (central tendency) 2.1 (high-end)	mg/cm^2-event	(U.S. EPA, 1992)
Surface area	S	535 (central tendency) 1,070 (high-end)	cm^2	(U.S. EPA, 2011a)
Weight fraction of chemical	Y_{derm}	1	Unitless	–
	$Y_{derm, dilute}$	0.1	Unitless	–
Frequency of events	FT	1	events/day	–
Fractional absorption	f_{abs}	0.003 (neat 1,1-dichloroethane)	Unitless	(Labcorp Early Development, 2024)
	$f_{abs, dilute}$	0.0006 (10% 1,1-dichloroethane)	Unitless	
Body weight	BW	80	Kg	(U.S. EPA, 2011a)

The values of the dermal load (Q_u) were based on experimental studies of non-aqueous liquids to measure the quantity remaining on the skin after contact. In the study, an initial wipe test was performed that consisted of the subjects wiping their hands with a cloth saturated in the liquid. The amount of liquid retained on the hands was measured immediately after the application. The high-end (2.1 mg/cm^2) and central tendency (1.4 mg/cm^2) dermal load values used in the 1,1-dichloroethane dermal exposure estimates were based on this study's data (U.S. EPA, 1992). The liquids used in the study were non-volatile so these values may be conservative for 1,1-dichloroethane. This is why the fraction absorbed term is part of the DEVL Model.

Data on dermal exposure measurements at facilities that manufacture, process, and use chemicals is limited. Table 5-14 below includes measured data that can be used for comparison with the dermal loading values used in the DEVL Model and the 1,1-dichloroethane dermal exposure model estimates provided in Table 5-15. The experimental dermal loading values in the DEVL Model are comparable to measured values recorded in the Pesticide Handlers Exposure Database (PHED) (per SAIC, 1996).

Table 5-14. Comparison of Dermal Exposure Values

Dermal Exposure Value	Type of Data	Notes	Reference(s)
1.4 mg/cm^2-event (central tendency) 2.1 mg/cm^2-event (high-end)	Experimental data	Used in EPA/OPPT Dermal Contact with Liquids Models	(U.S. EPA, 1992).
2.9 mg metalworking fluid/cm²-hr (geometric mean)	Measured data	Study of dermal exposures to electroplating and metalworking fluids during metal shaping operations	Roff, 2004 (as reported in OECD ESD on Metalworking Fluids)
0.5–1.8 mg/cm^2	Measured data	Dermal exposure data for workers involved in pesticide mixing and loading. The data included various combinations of formulation type and mixing/loading methods.	1992 Pesticide Handlers Exposure Database (PEHD), as reported in (SAIC, 1996)
0.0081–505.4 mg/day	Measured data	PMN manufacturer study of unprotected dermal exposures to trichloroketone for maintenance workers	Anonymous, 1996 (as reported in (SAIC, 1996)
0.0071–2.457 mg/day	Measured data	PMN manufacturer study of unprotected dermal exposures to trichloroketone for process operators	Anonymous, 1996 (as reported in (SAIC, 1996)
0.0105–0.0337 mg/day	Measured data	PMN manufacturer study of protected dermal exposures to trichloroketone for maintenance workers	Anonymous, 1996 (as reported in (SAIC, 1996)
0.0098–0.2417 mg/day	Measured data	PMN manufacturer study of protected dermal exposures to trichloroketone for process operators	Anonymous, 1996 (as reported in (SAIC, 1996)

A test order for an *in vitro* dermal absorption study (conducted per OECD 428 guideline) for 1,1-dichloroethane was issued and data received (Labcorp Early Development, 2024). The study was accepted by EPA for use in the 1,1-dichloroethane risk evaluation. The SACC reviewed the study and EPA has followed the SACC recommendation on how to use the data. The guideline study used human

skin from 92 percent female and 8 percent male samples. The dermal fractional absorption of 0.3 percent (100% or "neat" 1,1-dichloroethane) is used as one of the parameters in the DEVL model to estimate dermal retained dose as described in Equation 5-1 (*Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: in vitro Dermal Absorption Study Analysis* (U.S. EPA, 2025h) and *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: in vitro Dermal Absorption Study Calculation Sheet*) (U.S. EPA, 2025i). As recommended by the SACC, as an additional comparison, EPA ran the American Industrial Hygiene Association (AIHA) skin permeation model, IH SkinPerm, which predicts absorption based on chemical physical-chemical properties and obtained a result of 0.285 percent which is comparable to the 0.3 percent from experimental data.

For details on EPA's calculations of the dermal absorption factor from the dermal test order including addressing missing mass balance and high data variability based on OECD GD156 guidance (accessed June 16, 2025) and EFSA 2017 guidance (accessed June 16, 2025) refer to details in two supplemental files: *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: in vitro Dermal Absorption Study Analysis* (U.S. EPA, 2025h) and *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: in vitro Dermal Absorption Study Calculation Sheet* (U.S. EPA, 2025i)

EPA used a high-end exposed skin surface area (S) for workers of 1,070 cm^2 based on the mean two-hand surface area for adult males ages 21 or older from Chapter 7 of EPA's *Exposure Factors Handbook* (U.S. EPA, 2011a). For central tendency estimates, EPA assumed the exposure surface area was equivalent to only a single hand (or one side of two hands) and used half the mean values for two-hand surface areas (*i.e.*, 535 cm^2 for workers). The model estimates dermal exposure to the hands and does not account for dermal exposures to other parts of the body.

Though the exposed skin surface area may vary between the equivalent surface area of one and two hands, EPA does not assume skin is submerged in liquid 1,1-dichloroethane. "Submersion" would assume an unlimited supply of chemical so that dermal load is no longer limited and instead chemical flux would limit the absorbed dose. For volatile chemicals in a manufacturing setting, dermal dose is best characterized using fraction absorbed which accounts for volatility. Duration is captured by the 0.3% absorption of the entire dermal load across the surface area.

DEVL Model Execution – Deterministic

For the deterministic calculation, EPA used a single set of parameter values representing the central tendency and high-end cases. The Agency applied high-end and central tendency values for skin surface area and dermal loading, while using single values for the other parameters listed in Table 5-13. The Agency estimated dermal exposure for each OES based on these selected parameter values.

DEVL Model Execution – Probabilistic

For the probabilistic (stochastic) calculation approach, EPA used Monte Carlo simulations, utilizing the full distribution of each parameter listed in Table 5-13 except for fraction absorbed and event frequency, for which single parameter values were used. This approach generated a distribution of final exposure metric results, from which the 50th and 95th percentiles were selected to represent the central tendency and high-end exposure estimates.

For a summary of the dermal exposure results, refer to Table 5-17.

Other Notes
For further rationale on the dermal exposure assessment and parameters, refer to *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Laboratory Chemical Occupational Exposure and Environmental Release Modeling Results* (U.S. EPA, 2025j).

5.1.1.1.6 Estimate the Number of Workers and Occupational Non-Users Potentially Exposed

An assessment objective is to estimate the number of workers and ONUs potentially exposed. Normally, a primary difference between workers and ONUs is that workers may handle 1,1-dichloroethane and have direct contact with the chemical, while ONUs are working in the general vicinity of workers but do not handle 1,1-dichloroethane and do not have direct contact with 1,1-dichloroethane being handled by the workers. The size of the area that ONUs may work can vary across each OES and across facilities within the same OES and will depend on the facility configuration, building and room sizes, presence of vapor barrier, and worker activity pattern. Where possible, for each COU, EPA identified job types and categories for workers and ONUs. The Agency evaluated 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. Depending on the condition of use, 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.

Methodology
Where available, EPA used CDR data to provide a basis to estimate the number of workers and ONUs. Data were available from the 2016 and 2020 CDR for manufacturing sites; however, EPA determined this was not sufficient to determine the total number of workers for that OES. EPA supplemented the available CDR data using available market data; NAICS and SIC code data from TRI, DMR, and NEI sites identified for each condition of use (for number of sites estimated see Section 3.2.1.1); and analyzing U.S. Bureau of Labor Statistics (BLS) and U.S. Census data using the methodology described in the Environmental Releases and Occupational Exposure Assessment. Where market penetration data and site-specific NAICS/SIC codes from TRI/DMR/NEI were not available, EPA estimated the number of workers using data from GSs and ESDs. For additional details on development of estimates of number of workers refer to *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

EPA also determined the number of days per year that workers are potentially exposed to 1,1-dichloroethane. In general, the exposure frequency is the same as the number of operating days per year for a given OES (see Section 3.1.1.5). However, if the number of operating days is greater than 250 days per year, EPA assumed that a single worker would not work more than 250 days per year such that the maximum exposure days per year was still 250.

Results
Table 5-15 provides a summary for the number of workers and ONUs potentially exposed to 1,1-dichloroethane per facility. The estimates are provided for a facility within each OES.

Table 5-15. Total Number of Workers and ONUs Potentially Exposed to 1,1-Dichloroethane for Each OES

OES	Exposure Days per Year	Potential Number of Sites	Potential Number of Workers per Site	Potential Number of ONUs per Site	Notes
Manufacturing of 1,1-dichloroethane as an isolated intermediate	250	10	33	16	Number of workers and ONU estimates based on U.S. Census Bureau data, U.S. Bureau of Labor Statistics data, CDR, DMR, TRI, and NEI (U.S. BLS, 2023; U.S. Census Bureau, 2017).
Processing as a reactive intermediate	250	90	27	15	Number of workers and ONU estimates based on U.S. Census Bureau data, U.S. Bureau of Labor Statistics data, DMR, TRI, and NEI (U.S. BLS, 2023; U.S. Census Bureau, 2017).
Processing – repackaging	128	2	1	1	Exposure days per year and number of sites is based on the July 2022 Chemical Repackaging GS (U.S. EPA, 2022a). Number of workers and ONU estimates are based on U.S. Census Bureau data and U.S. Bureau of Labor Statistics data (U.S. BLS, 2023; U.S. Census Bureau, 2017).
Commercial use as a laboratory chemical	250	43–138	6	10	Exposure days per year and number of sites is based on the 2022 Draft GS on the Use of Laboratory Chemicals (U.S. EPA, 2023b). Number of workers and ONU estimates are based on U.S. Census Bureau data and U.S. Bureau of Labor Statistics Data (U.S. BLS, 2023; U.S. Census Bureau, 2017).
Waste handling, treatment, and disposal	250	672	14	12	Number of workers and ONU estimates based on U.S. Census Bureau data, U.S. Bureau of Labor Statistics data, DMR, TRI, and NEI (U.S. BLS, 2023; U.S. Census Bureau, 2017).
Waste handling, treatment, and disposal (POTW)	250	125	1	1	Number of workers and ONU estimates based on U.S. Census Bureau data, U.S. Bureau of Labor Statistics data, DMR, TRI, and NEI (U.S. BLS, 2023; U.S. Census Bureau, 2017).

5.1.1.2 Estimates of Occupational Exposure (ppm) and Dermal Exposure (mg/day)

Table 5-16 provides a summary for each of the OES by indicating whether monitoring data were used, how many data points were identified, the quality of the data, and also whether EPA used modeling to estimate inhalation and dermal exposures for workers and ONUs.

A summary of inhalation and dermal exposure estimates for each OES is presented in Table 5-17.

Table 5-16. Summary of Assessment Methods for Each Occupational Exposure Scenario

OES	Inhalation Exposure												Dermal Exposure		
	1,1-Dichloroethane Monitoring					Surrogate Monitoring					Modeling		Monitoring		Modeling
	Worker	# Data Points	ONU	# Data Points	Data Quality Ratings	Worker	# Data Points	ONU	# Data Points	Data Quality Ratings	Worker	ONU	Worker	Data Quality Rating	Worker
Manufacturing of 1,1-dichloroethane as an isolated intermediate	✓	55	✓	7	H	✓	172	O	N/A	H	O	O	O	N/A	✓
Processing as a reactive intermediate	✓	55	✓	7	H	✓	46	O	N/A	M	O	O	O	N/A	✓
Processing – repackaging	O	N/A	O	N/A	N/A	O	N/A	O	N/A	N/A	✓	O	O	N/A	✓
Commercial use as a laboratory chemical	✓	9	O	N/A	H	✓	76	O	N/A	H	O	O	O	N/A	✓
Distribution in commerce	Not estimated														
Waste handling, treatment, and disposal (POTW)	O	N/A	O	N/A	N/A	✓	3	O	N/A	M	O	O	O	N/A	✓
General waste handling, treatment, and disposal	O	N/A	O	N/A	N/A	✓	22	O	N/A	M	O	O	O	N/A	✓

O = no data available; ✓ = data available
Where EPA was not able to estimate ONU inhalation exposure from monitoring data or models, this was assumed equivalent to the central tendency experienced by workers for the corresponding OES; dermal exposure for ONUs was not evaluated because they are not expected to be in direct contact with 1,1-dichloroethane.

Table 5-17. Summary of Inhalation and Dermal Exposure Estimates for Each OES

OES	Worker Description	Exposure Days (day/year)	Worker Inhalation Estimates (ppm)		ONU Inhalation Estimates (ppm)		Worker Dermal Exposure Deterministic Estimates (mg/day)		Worker Dermal Exposure Probabilistic Estimates (mg/day)	
			Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End
Manufacturing of 1,1-dichloroethane as an isolated intermediate	Operator/process technician (non-emergency)	250	7.8E−03	0.73	3.2E−03	2.0E−02	2.2	6.7	3.2	5.5
	Operator/process technician (responding to line leaks)	Less than chronic	1.9	1.9						
	Maintenance technician	250	7.9E−02	0.41						
	Laboratory technician	250	1.1E−03	2.4E−02						
	Logistics Technician	250	2.8E−03	5.3E−03						
Processing as a reactive intermediate	Operator/process technician	250	7.8E−03	0.73	3.2E−03	2.0E−02	2.2	6.7	3.2	5.5
	Logistics Technician	250	2.8E−03	5.3E−03						
	Maintenance technician	250	7.9E−02	0.41						
	Laboratory technician	250	1.1E−03	2.4E−02						
Processing – repackaging	Loading, unloading, and cleaning	128	3.5	13	3.5		2.2	6.7	3.2	5.5
	Unloading and cleaning	128	1.7	6.6						
	Loading	128	1.7	6.6						
Commercial use as a laboratory chemical	Laboratory technician	250	1.1E−03	2.4E−02	1.1E−03		2.2	6.7	3.2	5.5
Distribution in commerce	Not estimated									
General waste handling, treatment, and disposal	–	250	0.30	10	0.30		2.2	6.7	3.2	5.5
General waste handling, treatment, and disposal (Dilute Scenario)	Not estimated								6.5E−02	0.11
Waste handling, treatment, and disposal (POTW)	–	250	0.25	0.68	0.25		2.2	6.7	3.2	5.5

OES	Worker Description	Exposure Days (day/year)	Worker Inhalation Estimates (ppm)		ONU Inhalation Estimates (ppm)		Worker Dermal Exposure Deterministic Estimates (mg/day)		Worker Dermal Exposure Probabilistic Estimates (mg/day)	
			Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End
Waste handling, treatment, and disposal (POTW) (Dilute Scenario)	Not estimated								6.5E−02	0.11
Where EPA was not able to estimate ONU inhalation exposure from monitoring data or models, this was assumed equivalent to the central tendency experienced by workers for the corresponding OES; dermal exposure for ONUs was not evaluated because they are not expected to be in direct contact with 1,1-dichloroethane.										

Using these 8-hour TWA exposure concentrations, EPA then calculated acute, intermediate, and chronic (non-cancer and cancer) exposures. These exposure metrics are then used to determine risk, as described in Section 5.3.3.1.

5.1.1.3 Weight of Scientific Evidence for the Estimates of Occupational Exposures from Industrial and Commercial Sources

EPA's conclusion on the weight of 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 exposure estimate—including quality of the data/information, applicability of the exposure data to the COU (including considerations of temporal relevance, locational relevance) and the representativeness of the estimate for the whole industry.

The best professional conclusion is summarized using the descriptors of robust, moderate, slight, or indeterminant, according to EPA's 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). For example, a conclusion of moderate weight of scientific evidence is appropriate where there is measured exposure data from a limited number of sources such that there is a limited number of data points that may not be representative of the worker activities or potential exposures. A conclusion of slight weight of scientific evidence is appropriate where there is limited information that does not sufficiently cover all potential exposures within the COU, and the assumptions and uncertainties are not fully known or documented. See EPA's 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) for additional information on weight of scientific evidence conclusions. A summary of the weight of scientific evidence conclusions for the inhalation estimates is provided below in Table 5-18.

Table 5-18. Weight of Scientific Evidence Conclusions for the Inhalation Exposure Assessment

OES	Weight of Scientific Evidence Conclusion	Overall Confidence in Inhalation Exposure Assessment Rationale
Manufacturing of 1,1-dichloroethane as an isolated intermediate	Moderate to Robust	EPA considered the assessment approach, the quality of the data, and uncertainties in assessment results to determine a weight of scientific evidence conclusion for the 8-hour TWA inhalation exposure estimates. The Agency used 1,1-dichloroethane test order inhalation data to assess inhalation exposures. The primary strength of these data is the use of personal and directly applicable data, and the number of samples available for workers and ONUs. Additionally, EPA assumed 250 exposure days per year based on 1,1-dichloroethane exposure each working day for a typical worker schedule; it is uncertain whether this captures actual worker schedules and exposures. Based on these strengths and limitations, EPA has concluded that the weight of 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.
Processing as a reactive intermediate	Moderate	1,1-Dichloroethane monitoring data for this scenario was not available. EPA used 1,1-dichloroethane test order data from the Manufacturing OES to assess inhalation exposures. The primary strength of this data is the use of personal and potentially applicable data. The primary limitations of these data include the uncertainty of the representativeness of these data toward the true distribution of inhalation concentrations in this scenario since the data was analogous from the Manufacturing OES. EPA also assumed 250 exposure days per year based on 1,1-dichloroethane exposure each working day for a typical worker schedule; it is uncertain whether this captures actual worker schedules and exposures. Based on these strengths and limitations, EPA has concluded that the weight of 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.
Processing – repackaging	Slight	EPA did not find any 1,1-Dichloroethane-specific information on a repackaging operation involving 1,1-dichloroethane. The basis for including this OES was the identification of Commercial use as a laboratory chemical as a COU for 1,1-dichloroethane and EPA's assumption that this would first necessitate a repackaging step into smaller containers suitable for use in commercial laboratory settings. This is a key uncertainty in the exposure estimates for this OES. 1,1-Dichloroethane monitoring data were not available for this scenario. Additionally, the Agency did not identify relevant monitoring data from other scenarios or chemicals assessed in previous EPA Risk Evaluations. Therefore, EPA modeled inhalation exposures. The Agency used assumptions and values from the July 2022 Chemical Repackaging GS (U.S. EPA, 2022a), which the systematic review process rated high for data quality, to assess inhalation exposures (OECD, 2009). The Agency used EPA/OPPT models combined with Monte Carlo modeling to estimate inhalation exposures. A strength of the Monte Carlo modeling approach is that variation in model input values and a range of potential exposure values is more likely than a

OES	Weight of Scientific Evidence Conclusion	Overall Confidence in Inhalation Exposure Assessment Rationale
		discrete value to capture actual exposure at sites. The primary limitation is the uncertainty in the representativeness of values toward the true distribution of potential inhalation exposures. In addition, EPA lacks 1,1-dichloroethane facility production volume data; and therefore, throughput estimates are based on CDR reporting thresholds. Also, because EPA could not estimate the number of exposure days per year associated with repackaging operations, the exposure days per year estimates are based on an assumed site throughput of imported containers. The estimates of inhalation exposure to ONUs have more uncertainty because the modeling approach used did not estimate exposure to workers in the "far-field," which is what EPA would normally use to estimate exposures to ONUs from modeling. A limitation in the modeling approach is that EPA did not find any specific information on 1,1-dichloroethane going through a repackaging step. Due to this lack of information on production volume and how 1,1-dichloroethane is handled and repackaged, EPA used default values for the models, thus potentially over-estimating exposures, especially for activities that handle a small portion of the manufactured volume. The OES for repackaging was based on an EPA assumption that a repackaging step would need to take place prior to the use of 1,1-dichloroethane as a laboratory chemical. There were two manufacturing sites identified for 1,1-dichloroethane, which EPA assumed also conducts repackaging activities. Repackaging, however, may also occur at the 12 sites which process 1,1-dichloroethane. For modeling purposes, EPA used the most conservative input parameter of repackaging at 2 sites that which equates to 25,000 lbs/site/yr. EPA then used a Monte Carlo modeling approach which included varying parameters such as container size to generate a distribution of estimates for exposure days and exposure concentrations to 1,1-dichloroethane. The parameters of lbs/yr for repackaging, number of sites and the daily amount handled are foundational parameters in the model and drive the estimated daily exposure levels and resulting risk estimates. The lack of information to support any of the foundational parameters for 1,1-dichloroethane is a major uncertainty in the assessment of this OES and EPA therefore assigns a slight confidence rating for this OES.
Commercial use as a laboratory chemical	Moderate	1,1-Dichloroethane monitoring data for this scenario was not available. EPA used 1,1-dichloroethane test order data for laboratory technicians from the manufacturing OES to assess inhalation exposures. The Agency considered the assessment approach, the quality of the data, and uncertainties in assessment results to determine a weight of scientific evidence conclusion for the 8-hour TWA inhalation exposure estimates. EPA used inhalation data to assess inhalation exposures. The primary strength of these data is the use of personal and potentially applicable data. The primary limitation is the number of samples available for workers. Data was not available for ONUs and an assumption that ONU exposure was equal to the laboratory technicians central tendency exposure was made. Additionally, there is uncertainty in the representativeness of these data toward the true distribution of inhalation concentrations in this scenario since the laboratory use occurred in a manufacturing setting. EPA assumed 250 exposure days per year based on 1,1-dichloroethane exposure each working day for a typical worker schedule; it is uncertain whether this captures actual worker schedules and exposures.

OES	Weight of Scientific Evidence Conclusion	Overall Confidence in Inhalation Exposure Assessment Rationale
		Based on these strengths and limitations, EPA has concluded that the weight of 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.
Waste handling, treatment, and disposal (general)	Slight	EPA does not currently have an Emission Scenario Document (ESD) or Generic Scenario (GS) that characterizes worker exposure potential at waste handling, treatment and disposal sites. EPA also did not identify any specific information on 1,1-dichloroethane pertaining to waste handling, treatment and disposal. This creates an uncertainty if chemical-specific monitoring data is not available for this OES. 1,1-Dichloroethane monitoring data was not available for this scenario. Additionally, EPA did not identify 1,1-dichloroethane monitoring data from other scenarios. Therefore, the Agency used surrogate inhalation data from methylene chloride to assess inhalation exposures. The primary limitations of these data include the uncertainty of the representativeness of these data toward the true distribution of inhalation concentrations in this scenario since the data were surrogate from methylene chloride, which results in a moderate confidence rating. EPA also assumed 250 exposure days per year based on 1,1-dichloroethane exposure each working day for a typical worker schedule; it is uncertain whether this captures actual worker schedules and exposures. The surrogate monitoring data used did not have sufficient metadata to identify an ONU exposure group within the data set. EPA made an assumption that ONU exposure was equal to central tendency of the data set. Because 1,1-dichloroethane-specific information (*e.g.*, inhalation monitoring data) on this OES is not available and a generic modeling scenario was also not available, EPA's confidence in this assessment is slight.
Waste handling, treatment, and disposal (POTW)	Slight	EPA does not currently have an ESD or GS that characterizes worker exposure potential at waste handling, treatment and disposal sites. EPA also did not identify any specific information on 1,1-dichloroethane pertaining to waste handling, treatment and disposal. This creates an uncertainty if chemical-specific monitoring data is not available for this OES. 1,1-Dichoroethane monitoring data was not available for this scenario. Additionally, EPA did not identify 1,1-dichloroethane monitoring data from other scenarios. Therefore, the Agency used surrogate inhalation data from 1,2-dichloroethane to assess inhalation exposures. The primary limitations of these data include the uncertainty of the representativeness of these data toward the true distribution of inhalation concentrations in this scenario since the data were surrogate from 1,2-dichloroethane, which results in a low confidence rating. In addition, the available surrogate data only provided 3 worker inhalation monitoring data samples for wastewater treatment. EPA also assumed 250 exposure days per year based on 1,1-dichloroethane exposure each working day for a typical worker schedule; it is uncertain whether this captures actual worker schedules and exposures. The surrogate monitoring data used did not have sufficient metadata to identify an ONU

OES	Weight of Scientific Evidence Conclusion	Overall Confidence in Inhalation Exposure Assessment Rationale
		exposure group within the data set. EPA made an assumption that ONU exposure was equal to central tendency of the data set. Because 1,1-dichloroethane-specific information (*e.g.,* inhalation monitoring data) on this OES is not available and a generic modeling scenario was also not available, EPA's confidence in this assessment is slight.

EPA estimated dermal exposures using modeling methodologies, which are supported by moderate evidence. The Agency used the EPA Dermal Exposure to Volatile Liquids Model to calculate the dermal retained dose. 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. These modifications improve the modeling methodology; however, the modeling approach is still limited by the low variability for different worker activities/exposure scenarios. Therefore, the weight of scientific evidence for the modeling methodologies is moderate. The exposure scenarios and exposure factors underlying the dermal assessment are supported by moderate to robust evidence.

Dermal exposure scenarios were informed by moderate to robust process information and GS/ESD. Exposure factors for occupational dermal exposure include amount of material on the skin, surface area of skin exposed, and absorption of 1,1-dichloroethane through the skin. These exposure factors were informed by literature sources, the *ChemSTEER User Guide* (U.S. EPA, 2015) for standard exposure parameters, and a European model, with ratings from moderate to robust. Based on these strengths and limitations, EPA concluded that the weight of scientific evidence for the dermal exposure assessment is moderate to robust for all OESs.

5.1.2 General Population Exposures

1,1-Dichloroethane – General Population Exposures (Section 5.1.2): Key Points

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

- Inhalation exposure is the major general population exposure pathway. EPA evaluated acute, chronic and lifetime general population exposures to 1,1-dichloroethane in ambient air, indoor air, and population in proximity to air emissions.
    - For exposures through ambient air, EPA considered potential exposures for communities within 10 km of a release site.
    - EPA estimated inhalation exposures at various distances from a release facility using AERMOD TRI and NEI modeled air concentrations (Section 3.3.1) and equations and exposure factors described in Appendix D.2.
- Dermal exposures from the exposure scenario of swimming in receiving water from 1,1-dichloroethane releases were estimated to result in low exposures.
- Oral exposures to 1,1-dichloroethane from ingestion of drinking water were estimated to result in low exposures.
- Oral exposures to 1,1-dichloroethane from ingestion of fish-containing 1,1-dichloroethane were estimated for adults, children and for subsistence and tribal fishers. Low bioaccumulation potential in fish results in low exposures.
- Oral exposures to 1,1-dichloroethane by children playing with and ingestion of 1,1-dichloroethane containing biosolids as applied to land were expected to result in low exposures.
- For each exposure pathway, central tendency and high-end doses were estimated. EPA's *Guidelines for Human Exposure Assessment* defined central tendency exposures as "an estimate of individuals in the middle of the distribution." It is anticipated that these estimates apply to populations exposed to reported releases of 1,1-dichloroethane to ambient air, surface water and land. High-end exposure estimates are defined as "plausible estimate of individual exposure for those individuals at the upper end of an exposure distribution, the intent of which is to convey an estimate of exposure in the upper range of the distribution while avoiding estimates that are beyond the true distribution."

General population exposures occur when 1,1-dichloroethane is released into the environment and the media is then a pathway for exposure. Section 3.3 provides a summary of the monitoring, database, and modeled data on concentrations of 1,1-dichloroethane in the environment. Figure 5-2 provides a graphic representation of where and in which media 1,1-dichloroethane is estimated to be found and the corresponding route of exposure.

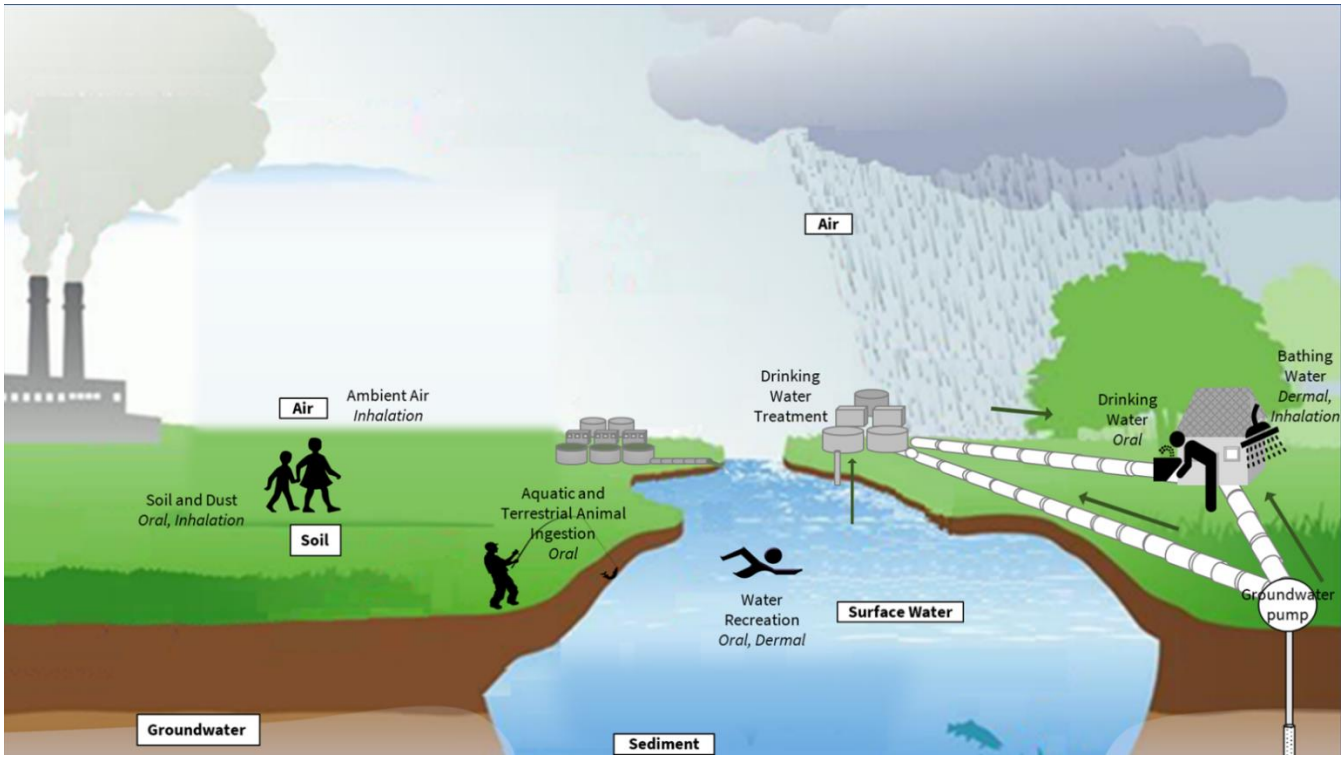


Figure 5-2. Potential Human Exposure Pathways to 1,1-Dichloroethane for the General Population[a]

[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 is depicted with grey arrows. This diagram pairs with Figure 2-1 and Figure_Apx J-1 depicting the fate and transport of the subject chemical in the environment.

5.1.2.1 Approach and Methodology

Exposure to 1,1-dichloroethane results from direct releases to ambient air and surface water resulting from its use in the chemical manufacturing processes. 1,1-Dichloroethane has been detected in the indoor and outdoor environment although exposures likely vary across the general population. See tornado plots and associated tables in the *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025z) for a summary of the various environmental media1,1-dichloroethane has been detected.

Releases of 1,1-dichloroethane are likely to occur through the direct release to air, water, and soil, with partitioning between the environmental compartments. Most 1,1-dichloroethane releases will ultimately partition to air based on its vapor pressure; however, a smaller amount will remain in water due to its water solubility. For a more detailed discussion about 1,1-dichloroethane environmental partitioning, please see Section 2.2.2. and Appendix C.2.1.2.

Exposure to the general population was estimated for the industrial and commercial releases per OES. Table 3-4 illustrates how the industrial and commercial releases to the environmental media varies by OES.

For the ambient air assessment, EPA first estimated 1,1-dichloroethane concentrations at various distances from facilities reporting releases to TRI and NEI. These modeled concentrations (Sections 3.3.1 and 3.3.2) were used to estimate inhalation exposures (5.1.2.2) and potential associated risks. For facilities where estimated cancer risks exceeded the 1 in 1,000,000 (1×10^{-6}) lifetime cancer benchmark, EPA used 2020 U.S. Census block data to conduct a population analysis within a Census block,

including an evaluation of PESS as well as metrics associated with the population characteristics. Proximity of general population to community infrastructures was also evaluated, such as parks, schools, places of worship, childcare centers, and hospitals (Section 5.1.2.2.3). Finally, EPA refined residential locations within Census blocks using land use analysis to determine the likelihood of actual lifetime exposures. The Agency concluded based on the results of the ambient air exposure analyses and the land use analyses that risks to the general population are not supported.

Modeled surface water concentrations (Section 3.3.3.2) were utilized to estimate oral drinking water exposures (Section 5.1.2.4.1) oral fish ingestions exposures (Section 5.1.2.4.2), incidental oral exposures (Sections 5.1.2.4.3 and 5.1.2.4.4), and incidental dermal exposures (Section 5.1.2.3) for the general population. Modeled groundwater concentrations (Appendix G.1.2.3), resulting from 5 lb of 1,1-dichloroethane TSCA land disposal were estimated but due to the low estimated exposure concentration, drinking water was not evaluated as a potential pathway of concern. Although 1,1-dichloroethane has been detected in groundwater as drinking water monitoring data, the low 1,1-dichloroethane concentrations confirmed low oral drinking water exposures (Section 5.1.2.4.1) to the general population. Modeled (Appendix G.1.1) soil concentrations via deposition were used to estimate dermal exposures (Sections 5.1.2.4.4) to children who play in mud and other activities with soil.

Exposures estimates from industrial and commercial releases of 1,1-dichloroethane were compared to exposure estimates from non-scenario specific monitoring data to ground truth the results (*e.g.*, ambient air exposures). Figure 3-5 and Table 3-8 summarize the environmental media monitoring data that were available in the United States. For a description of statistical methods, methodology of data integration and treatment of non-detects and outliers used to generate the AMTIC estimates please reference the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Ambient Monitoring Technology Information Center (AMTIC), 1,1-Dichloroethane Monitoring Data 2015 to 2020* (U.S. EPA, 2025d).

Exposure to general population per conditions of use were estimated for emissions to water and air, as depicted in Figure 5-3.

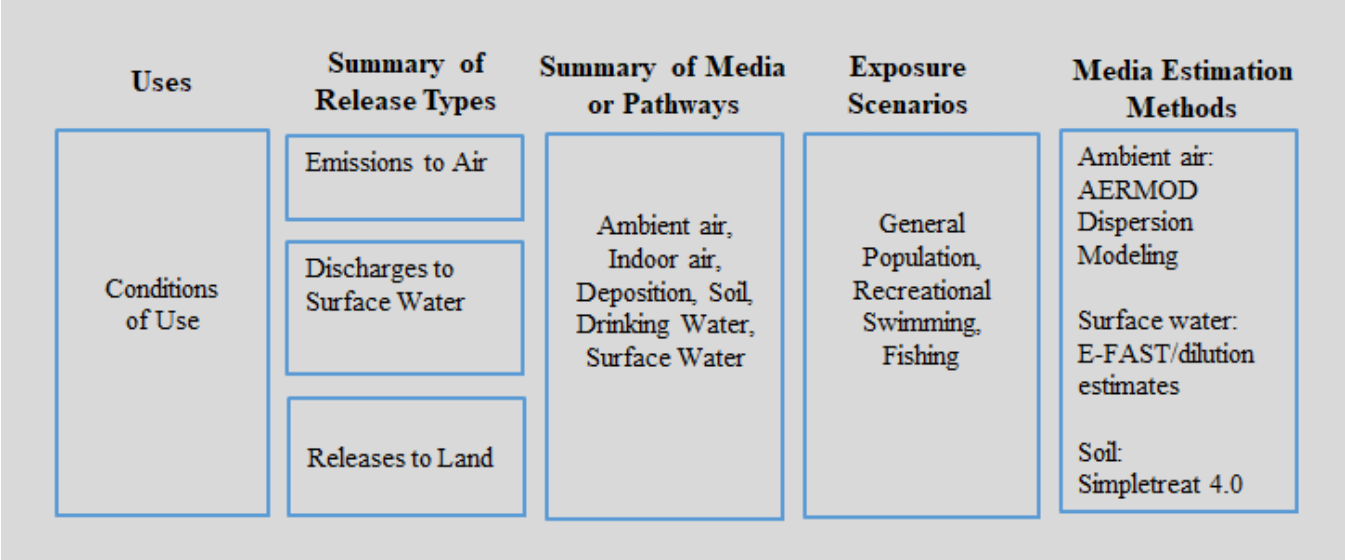


Figure 5-3. Overview of General Population Exposure Assessment for 1,1-Dichloroethane

For each exposure pathway, central tendency and high-end doses were estimated. EPA's *Guidelines for Human Exposure Assessment* (accessed June 16, 2025) defined central tendency exposures as "an estimate of individuals in the middle of the distribution." It is anticipated that these estimates apply to most individuals exposed to facility releases of 1,1-dichloroethane to ambient air, surface water and land. High-end exposure estimates are defined as plausible estimate of individual exposure for those individuals at the upper end of an exposure distribution, the intent of which is to convey an estimate of exposure in the upper range of the distribution while avoiding estimates that are beyond the true distribution." It is anticipated that these estimates apply to some individuals, particularly those who may live near facilities with elevated concentrations.

General Population Exposure Scenarios

Figure 5-2 provides an illustration of the exposure scenarios considered for general population exposure.

Ambient Air Exposure Scenarios: The Multi-Year Methodology AERMOD using TRI or NEI release data evaluated exposures to members of the general population at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and two area distances (30–60 m and 100–1,000 m) from each TRI and NEI releasing facility for each OES (or generic facility for alternative release estimates). EPA presents in subsequent sections modelled annual averages that represent the 95th percentile exposure scenario. Human populations for each of the eight finite distances were placed in a polar grid every 22.5 degrees around the respective distance ring. This results in a total of 16 modeled exposure points around each finite distance ring for which exposures are modeled. Figure 5-4 provides a visual depiction of the placement of exposure points around a finite distance ring. Although the visual depiction only shows exposure point locations around a single finite distance ring, the same placement occurred for all eight finite distance rings.

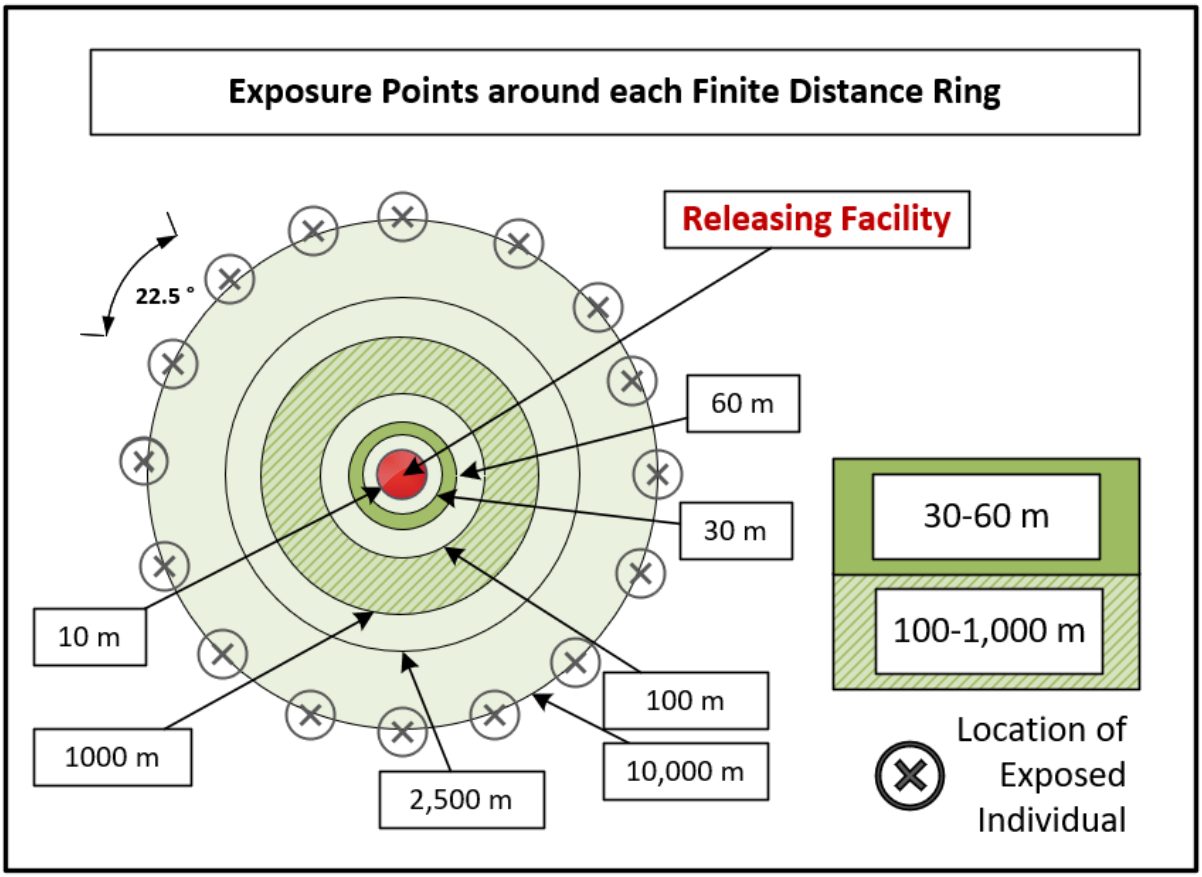


Figure 5-4. Modeled Exposure Points for Finite Distance Rings for Ambient Air Modeling (AERMOD)

Modeled exposure points for the area distance 30 to 60 m evaluated were placed in a cartesian grid at equal distances between 30 and 60 m around each releasing facility. Exposure points were placed at 10-meter increments. This results in a total of 80 points for which exposures are modeled. Modeled exposure points for the area distance 100 to 1,000 m evaluated were placed in a cartesian grid at equal distances between 100 and 1,000 m around each releasing facility. Exposure points were placed at 100-meter increments. This results in a total of 300 points for which exposures are modeled. provides a visual depiction of the placement of exposure points (each dot) around the 100 to 1,000 m area distance ring. All exposure points were at 1.8 m above ground, as a proximation for breathing height for ambient air concentration estimations. A duplicate set of exposure points was at ground level (0 m) for deposition estimations.

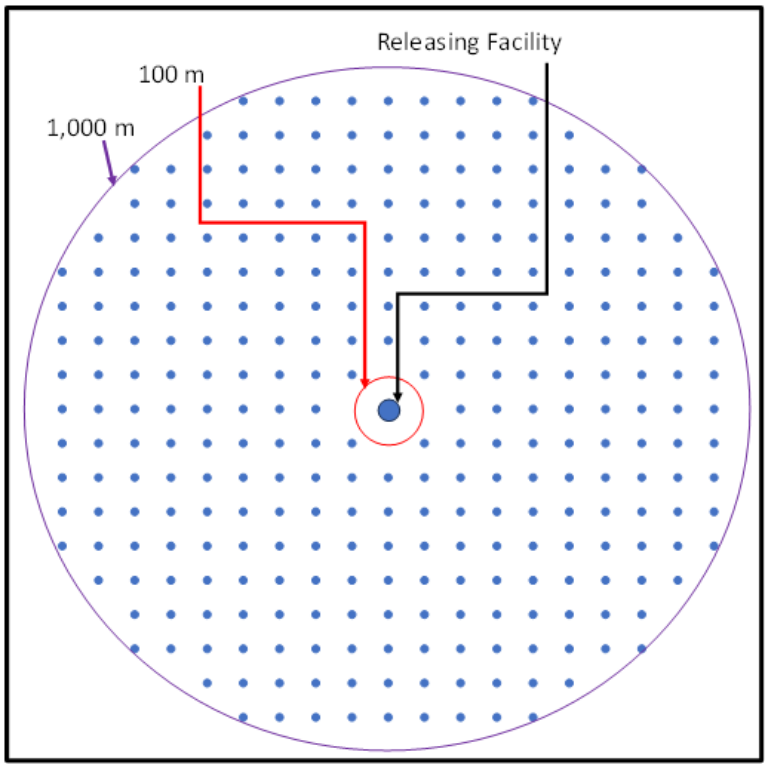


Figure 5-5. Modeled Exposure Point Locations for Area Distance for Ambient Air Modeling (AERMOD)

The ambient air is a major pathway for 1,1-dichloroethane and the general population may be exposed to ambient air concentrations and air deposition because of 1,1-dichloroethane releases. Relevant exposures scenarios considered in this risk evaluation include ambient air inhalation for populations living nearby releasing facilities, and ingestion exposure of soil to children resulting from ambient air deposition from a nearby facility. These scenarios are described in more detail here.

Soil Exposure Scenarios: 1,1-Dichloroethane can also be present in the biosolids resulting from the 125 POTWs treating effluent containing 1,1-dichloroethane from releases associated with TSCA COUs (see Table 3-4). These 1,1-dichloroethane-containing biosolids may be spread onto soils as a common biosolids disposal method. EPA considered exposure pathway via children playing in soil where biosolids were spread. Given pica behavior of children where soil is ingested, EPA's *Exposure Factors Handbook* (U.S. EPA, 2011a) recommends a 3 to 6 year old ingestion rate to estimate the possible ingestion of 1,1-dichloroethane in soil via the biosolids pathway. EPA acknowledges this exposure pathway would represent an upper bound exposure to children; however, this would also ensure that risks to children would not be missed in the 1,1-dichloroethane exposure assessment.

As mentioned above, air deposition fluxes from AERMOD were used to estimate soil concentrations at various distances from the largest emitting facility for each OES. Oral ingestion exposure estimates of soil were calculated for children aged 3 to 6 years using EPA's *Exposure Factors Handbook* (U.S. EPA, 2011a) recommended ingestion rate for that age group.

Water Exposure Scenarios: 1,1-Dichloroethane is expected to be found in surface waters through the direct facility release of the chemical into receiving water bodies. Section 3.3.3.2 provides modeled estimates of 1,1-dichloroethane in surface water at the site of release and Section 3.3.3.6 presents modeled estimates in downstream locations that are expected to supply PWS and become a source of

drinking water for the general public. Section 3.3.3.4 provides model estimates of 1,1-dichloroethane in benthic pore waters and benthic sediment, but these scenarios are not expected to lead to general population exposure. Likewise, surface water concentrations of 1,1-dichloroethane resulting from air deposition were estimated for the ecological assessment but are not expected to result in any significant exposure to the general population. Appendix G.1.2.3 provides modeled estimates of 1,1-dichloroethane in groundwater due to estimated migration from landfill leachate, although groundwater estimates are very low and so do not expect to result in a general population exposure. The relevant surface water estimates at PWS locations were used to calculate an exposure dose from drinking water for the general population. Additionally, modeled surface water concentrations (see Section 3.3.3.6) were used to calculate a dermal exposure estimate from swimming, incidental ingestion estimates from swimming, fish ingestion exposure at the site of facility release of 1,1-dichloroethane.

5.1.2.2 Summary of Inhalation Exposure Assessment

EPA evaluated acute, chronic and lifetime general population exposures to 1,1-dichloroethane in ambient air, indoor air, and population in proximity to air emissions. In this analysis, EPA evaluated lifetime cancer, chronic non-cancer, and acute non-cancer exposures based on the 10th, 50th, and 95th percentile air concentrations estimated for ambient air (Section 3.3.1.2) for all the facilities within each OES reporting to TRI and NEI. In the following sections EPA is presenting exposures to ambient and indoor air based on the maximum 95th percentile estimated air concentrations for the facilities within each OES to characterize high-end exposures. For the two OESs where there was no site-specific data available for estimating ambient air concentrations, exposures are shown for high-end modeled releases (Section 3.2.1.4). The complete set of inhalation exposure estimates are presented in the supplemental files cited below. Additionally, EPA presented data on populations living in proximity to air releasing facilities to characterize PESS exposures.

5.1.2.2.1 Ambient Air Exposure

To evaluate human inhalation exposures from industrial and commercial fugitive and stack emissions, EPA calculated ACs, ADCs, and LADCs based on IIOAC- and AERMOD-modeled air concentrations estimated in Section 3.3.1. The LADCs presented in Table 5-19 are based on the maximum 95th percentile air concentrations estimated for the facilities within each OES reporting to TRI. The complete set of inhalation exposure estimates are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025o). LADCs within 10 km of release types considered here range from 0 to 232 µg/m^3. The LADCs presented in Table 5-20 are based on the maximum 95th percentile air concentrations estimated for the facilities within each OES reporting to NEI. The complete set of inhalation exposure estimates are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD NEI Exposure and Risk Analysis* (U.S. EPA, 2025n). LADCs within 10 km of release types considered here range from 0 to 32 µg/m^3, which is within a similar range to LDACs estimated from TRI air releases. These lifetime exposure estimates are based on 78 years of exposure over a 78-year lifetime and are relevant to all lifestages. These lifetime exposures were estimated from TRI air releases as shown in Figure 3-3, and from NEI air releases as show in Figure 3-4. As mentioned in Section 3.3.1, approximately 30 percent of the facilities reporting 1,1-dichloroethane releases to TRI (7 out of 23 facilities) are in the State of Texas and approximately 40 percent of them (9 out of 23 facilities) are in the State of Louisiana.

Table 5-21 provides a summary of the LADCs for the Commercial use as a laboratory chemical, and Processing – repackaging OESs where there was no site-specific data available for modeling. These lifetime exposure estimates are presented for high-end modeled releases, high-end meteorology (Lake

Charles, Louisiana[18]), both rural and urban setting, and the maximum 95th percentile air concentrations estimated for each OES. The complete set of inhalation exposure estimates are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD Generic Releases Exposure and Risk Analysis* (U.S. EPA, 2025m). The LADCs are based on 78 years of exposure over a 78-year lifetime and are relevant to all lifestages. LADCs within 10 km of release types presented here range from 4.7×10^{-4} to 1.5 µg/m^3.

[18] The high-end meteorological station used represents meteorological datasets that tended to provide high-end concentration estimates relative to the other stations within IIOAC (see Appendix D.1.2.4).

Table 5-19. Lifetime Average Daily Concentrations (LADC) Estimated Within 10,000 m of 1,1-Dichloroethane TRI Releases to Air

OES	# Facilities Evaluated in OES[a]	Maximum 95th Percentile[b] LADCs Estimated Within 10–10,000 m of Facilities (µg/m^3)									
		10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	2,500 m	5,000 m	10,000 m
Manufacturing	9	2.3E02	9.0E01	6.9E01	3.7E01	1.8E01	2.5	4.1E−01	9.3E−02	3.0E−02	1.0E−02
Processing as a reactive intermediate	6	1.5E01	6.4	4.3	2.5	1.2	1.6E−01	2.7E−02	1.3E−02	6.8E−03	2.9E−03
General waste handling, treatment, and disposal	8	1.9E01	9.3	6.1	3.9	1.9	1.4E−01	4.8E−02	1.1E−02	3.4E−03	1.1E−03

[a] For each OES, EPA modeled all TRI-reported releases considering source attribution (fugitive and stack releases) for each facility from 2015–2020. Not all facilities reported releases for all 6 years.
[b] EPA is only presenting exposures based on the maximum 95th percentile estimated air concentrations. The complete set of inhalation exposure estimates at the 10th, 50th and 95th percentiles are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025o).

Table 5-20. Lifetime Average Daily Concentrations (LADC) Estimated Within 10,000 m of 1,1-Dichloroethane Releases to Air Reported to NEI

OES	# Releases Evaluated in OES[a]	Maximum 95th Percentile[b] LADCs Estimated Within 10–10,000 m of Facilities (µg/m^3)									
		10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	2,500 m	5,000 m	10,000 m
Commercial use as a laboratory chemical	2	3.7E−02	1.2E−02	7.2E−03	4.2E−03	1.9E−03	1.9E−04	3.8E−05	8.2E−06	2.6E−06	8.4E−07
Manufacturing	9	2.1E01	6.1	6.1	6.1	5.7	1.0	1.2E−01	2.6E−02	8.3E−03	2.6E−03
Processing as a reactive intermediate	50	3.2E01	1.2E01	8.2	4.9	2.2	2.7E−01	4.8E−02	1.7E−02	6.7E−03	2.4E−03
General waste handling, treatment, and disposal	102	1.3E01	8.2	6.5	4.1	2.1	2.1E−01	5.2E−02	1.1E−02	3.4E−03	1.0E−03
Facilities not mapped to an OES	59	9.2	3.7	2.8	1.5	7.3E−01	1.2E−01	1.8E−02	3.9E−03	1.3E−03	4.0E−04

[a] For each OES, EPA modeled all NEI-reported releases considering source attribution (fugitive and stack releases) for each facility for 2014 and 2017 reported data. Not all facilities reported releases for both years.
[b] EPA is only presenting exposures based on the maximum 95th percentile estimated air concentrations. The complete set of inhalation exposure estimates at the 10th, 50th and 95th percentiles are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD NEI Exposure and Risk Analysis* (U.S. EPA, 2025n).

Table 5-21. Lifetime Average Daily Concentrations Estimated Within 10,000 m of 1,1-Dichloroethane Releases to Air for the Commercial Use as a Laboratory Chemical, and Processing – Repackaging for Laboratory Chemicals OESs

OES[a]	Meteorology[b]	Source	Land	Maximum 95th Percentile[c] LADCs Estimated Within 10–10,000 m of Facilities (µg/m^3)									
				10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	2,500 m	5,000 m	10,000 m
Processing – repackaging for laboratory chemicals	High	Stack and Fugitive	Urban	9.3E−01	2.6E−01	2.1E−01	1.5E−01	1.4E−01	3.8E−02	1.3E−02	3.8E−03	1.3E−03	4.7E−04
	High	Stack and Fugitive	Rural	9.3E−01	2.6E−01	2.0E−01	1.2E−01	1.0E−01	3.4E−02	1.5E−02	4.5E−03	1.9E−03	9.8E−04
Commercial use as a laboratory chemical	High	Stack and Fugitive	Urban	1.5	4.4E−01	3.9E−01	3.1E−01	3.5E−01	1.0E−01	3.4E−02	1.0E−02	3.7E−03	1.3E−03
	High	Stack and Fugitive	Rural	1.5	4.3E−01	3.5E−01	2.5E−01	2.4E−01	9.0E−02	4.0E−02	1.3E−02	5.1E−03	2.5E−03

[a] EPA modeled releases for these OESs (Section 3.1.1.4).
[b] There are no site-specific data available for air modeling for these OESs. The high-end meteorological station used (Lake Charles, LA) represents meteorological datasets that tended to provide high-end concentration estimates relative to the other stations within IIOAC (Appendix D).
[c] EPA is only presenting exposures based on the maximum 95th percentile estimated air concentrations in this table. The complete set of inhalation exposure estimates are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD Generic Releases Exposure and Risk Analysis* (U.S. EPA, 2025m).

5.1.2.2.2 Indoor Air Exposure

EPA calculated LADCs for indoor air exposure based on the IIOAC modeled indoor air concentrations in Section 3.3.2.2. Table 5-22 shows LADCs based on the maximum 95th percentile air concentrations estimated for the facilities within each OES reporting to TRI. EPA is presenting exposures to indoor air based on the maximum 95th percentile estimated air concentrations to characterize high-end exposures. The complete set of inhalation exposure estimates are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis* (U.S. EPA, 2025p). LADCs from 100 to 1,000 m of release types considered here range from 1.3×10^{-2} to 7.4 µg/m^3. These lifetime exposure estimates are based on 78 years of exposure over a 78-year lifetime and are relevant to all lifestages.

Table 5-22. Indoor Air Lifetime Average Daily Concentrations (LADCs) Estimated Within 1,000 m of 1,1-Dichloroethane Releases to Air Reported to TRI[a]

OES[b]	# Facilities Evaluated in OES	Maximum LADCs[c] Estimated Within 100 to 1,000 m of Facilities (µg/m^3)		
		100 m	100–1,000 m	1,000 m
Manufacturing	9	1.8E01	2.0	8.3E−01
Processing as a reactive intermediate	6	9.5E−01	1.1E−01	4.5E−02
General waste handling, treatment, and disposal	8	6.4E−01	7.5E−02	3.0E−02

[a] EPA calculated LADCs based on the IIOAC modeled indoor air concentrations.

[b] For each OES, EPA modeled all TRI-reported releases considering source attribution (fugitive and stack releases) for each facility from 2015–2020. Not all facilities reported releases for all 6 years.

[c] EPA is only presenting exposures based on the maximum 95th percentile estimated air concentrations in this table. The complete set of inhalation exposure are presented in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis* (U.S. EPA, 2025p).

5.1.2.2.3 Populations in Proximity to Air Emissions

EPA reviewed the 95th percentile LADC (lifetime average daily concentration) as a basis for selecting AERMOD TRI sites that reflect high-end exposures. Of the 23 TRI facility releases that were modeled using AERMOD, a subset of 10 AERMOD TRI release sites with the highest LADC were the focus of the population evaluation. The goal of this evaluation was to characterize the general population, the population that comprises PESS groups (see Section 5.3.2), and the population with respect to age/lifestage, and other characteristics that surround this subset of high-end exposure sites at relevant distances. Nearby environments and community infrastructure of interest were also examined to further understand exposure to these groups and the general public in locations outside their residence. Census block level information that captures residential areas were used to estimate population numbers and metrics. Distance estimates between AERMOD TRI release sites, census block centroids, and community locations of interest were compared with modeled AERMOD distances to evaluate the degree of exposure possible. A full description of the purpose, methods, and uncertainties of this evaluation can be found in Appendix C.3.

Based on use of U.S. Census location data, of 10 AERMOD TRI release sites, four (three in Louisiana and one in Texas) were estimated to have populations within a census block living within 1,000 m of the source of emissions (see Table 5-23) and the presence of general population living anywhere within a census block that was within 1,000 m was considered relevant for high-end exposure characterization.

Table 5-23. Population Density Estimates Within 1,000 m of a Subset of AERMOD TRI Air Release Sites that Reflect High-End Exposures

OES	TRIFID	Facility Location	Highest LADC AERMOD Modeled Distance (m)	Next AERMOD Modeled Distance (m)	Distance to Closest Census Block (m)
Manufacturing	70734VLCNMASHLA	Geismar, LA	30	60	1,599
	77571LPRTC2400M	La Porte, TX	100	1,000	N/A
	70734BRDNCLOUIS	Geismar, LA	100	1,000	1,300
	70669GRGGL1600V	Westlake, LA	60	100	890
	70669PPGNDCOLUM	Westlake, LA	1,000	2,500	1,391
	7076WBLCBP21255	Plaquemine, LA	100	1,000	505
	7754WBLCBP231NB	Freeport, TX	10	30	267
	70765GRGGLHIGHW	Plaquemine, LA	30	60	2,139
Processing as a reactant	70764LLMNXHWY40	Plaquemine, LA	100	1,000	975
Waste handling, disposal, treatment, and recycling	71836SHGRVPOBOX	Foreman, AR	100	1,000	1,371
N/A = Not assessed as there was not a census block at this location					

Although the results from Table 5-23 provide an understanding of the size of the general population in the areas surrounding high-end exposures, EPA also evaluated the modeled AERMOD TRI distances where high-end exposures are expected with respect to where these populations are anticipated to live. Table 5-23 shows the greatest discrete AERMOD modeled distance from the emission source where a high-end exposure has been identified and includes the next discrete AERMOD modeled distance, where high-end exposure was not identified. Both modeled distances were evaluated since in some cases the area in between is lacking modeled results, and so it is possible a population can experience a high-end exposure in between the "highest" and the "next" AERMOD modeled distances. The last column in Table 5-23 includes the estimated distance between the AERMOD TRI release site and the nearest census block with an expected population. Of the 10 subset AERMOD TRI release sites, 4 have populations within proximity to the release sites that may potentially experience high-end exposures. It is important to note that there is a degree of uncertainty in distance estimates for reasons outlined in Appendix C.3. Thus, these results should be interpreted with caution; distances that overlap within a few hundred meters may be within the error bound surrounding the distance estimates and comparisons.

The population of targeted PESS groups were estimated based on a weighted approach that scales census information at the block group level to individual census blocks. The results from individual census blocks within 1,000 and 2,600 m of the AERMOD TRI release sites were then evaluated. The PESS groups included children under 5 and 18 years because childcare centers and public schools were observed near several of the ARMOD TRI release sites and children could be susceptible to lifetime exposures and potential cancer risks. Women of childbearing age were identified as a potential PESS group in Section 5.3.2; however, the census information does not include pregnancy data explicitly. In turn, the population of females of reproductive age (15–50 years old; per the American Community Survey and U.S. Census Bureau[19] data on fertility) was used to characterize women of childbearing age.

[19] US Census Bureau reference to the American Community Survey at: https://www.census.gov/topics/health/fertility/about.html (accessed June 16, 2025).

The population aged over 65 was also estimated, although this age range was not explicitly identified as a PESS group for 1,1-dichloroethane.

The populations that make up these age groups within 1,000 m of the subset of AERMOD TRI release sites are shown in Table 5-24. It shows that there are children, females ages 15 to 50, and adults older than 65 living within or near areas of high-end exposures to 1,1-dichloroethane. Of the 4 sites with estimated populations living within or near high-end exposure areas, almost 500 females of reproductive age were estimated to live within 1,000 m of the source of emission, or approximately 30 percent of the total general population within 1,000 m. Although the population of women of reproductive age may be greater than the population of pregnant women, these results indicate that the number of women of childbearing age within or near areas of high-end exposures to 1,1-dichloroethane are still considerable.

Table 5-24. Population Density Estimates by Age Groups Within 1,000 m of the Subset of AERMOD TRI Air Release Sites[a]

OES	TRIFID	Facility Location	Total Population	Children Under 5	Children Under 18	Females 15–49	Population 65+
Manufacturing	70734VLCNMASHLA	Geismar, LA	0	0	0	0	0
	77571LPRTC2400M	La Porte, TX	0	0	0	0	0
	70734BRDNCLOUIS	Geismar, LA	0	0	0	0	0
	70669GRGGL1600V	Westlake, LA	135	0	8	62	17
	70669PPGNDCOLUM	Westlake, LA	0	0	0	0	0
	7076WBLCBP21255	Plaquemine, LA	128	9	17	33	24
	7754WBLCBP231NB	Freeport, TX	1,378	60	446	392	116
	70765GRGGLHIGHW	Plaquemine, LA	0	0	0	0	0
Processing as a reactant	70764LLMNXHWY40	Plaquemine, LA	21	1	5	5	3
Waste handling, disposal, treatment and recycling	71836SHGRVPOBOX	Foreman, AR	0	0	0	0	0

[a] Population data associated with census block groups were gathered from 2021 census data and the American Community Survey 2017 to 2021.

Although exposures to maximum 1,1-dichloroethane concentrations resulting in risk are not expected, the PESS populations within 1,000 m represent an exposure to high-end ambient air concentrations to 1,1-dichloroethane.

The locations of childcare centers, schools, places of worship, and healthcare facilities were also identified within 1,000 m of the subset of AERMOD TRI release sites. No private schools, colleges or universities, hospitals, urgent care centers, VA health facilities, or dialysis clinics were located even out to within 2,600 m of any of the subset of AERMOD TRI release sites. One childcare center and two places of worship were located within 1,000 m of the subset of AERMOD TRI release sites. Collectively these results do indicate that other PESS groups that attend, work, or frequent these community locations might be exposed to 1,1-dichloroethane due to proximity to the subset of TRI release sites evaluated in this section.

5.1.2.3 Summary of Dermal Exposure Assessment

Incidental Dermal Exposure from Swimming

The general population may potentially swim in surface waters that are affected by 1,1-dichloroethane contamination. Modeled surface water concentrations assuming the facility release annual load was over the number of facility operating days. The surface water concentrations were used to estimate acute doses and average daily doses from dermal exposure while swimming.

The following equations from EPA's Office of Pesticide Program Swimmer Exposure Assessment Model (SWIMODEL) (accessed June 16, 2025) were used to calculate incidental dermal (swimming) acute (ADR) and chronic (ADD) doses for all COUs, for adults, youth, and children:

Equation 5-2.

$$ADR = (SWC \times K\_p \times SA \times ET \times CF1 \times CF2) / BW$$

Equation 5-3.

$$ADD = (SWC \times K\_p \times SA \times ET \times RD \times ET \times CF1 \times CF2) / (BW \times AT \times CF3)$$

Where:

ADR	=	Acute Dose Rate (mg/kg-day)
ADD	=	Average Daily Dose (mg/kg-day)
SWC	=	Chemical concentration in water (µg/L)
K_p	=	Permeability coefficient (cm/hour)
SA	=	Skin surface area exposed (cm^2)
ET	=	Exposure time (hours/day)
RD	=	Release days (days/year)
ED	=	Exposure duration (years)
BW	=	Body weight (kg)
AT	=	Averaging time (years)
CF1	=	Conversion factor (1.0×10^{-3} mg/µg)
CF2	=	Conversion factor (1.0×10^{-3} L/cm^3)
CF3	=	Conversion factor (365 days/year)

The 1,1-dichloroethane skin permeability coefficient used in the equation above was the predicted K_p value presented in the EPA Risk Assessment Guidance for Superfund for organic contaminants in water ($K_p = 6.7\times10^{-3}$ cm/hour). This K_p was chosen instead of the permeability coefficient received from submitted 1,1-dichloroethane dermal absorption test order study ($K_p = 0.3\times10^{-3}$ cm/hour for 10% 1,1-dichloroethane in IPM). The K_p from the 1,1-dichloroethane test order dermal absorption study measured was diluted in a solvent instead of in an aqueous solution as would be appropriate to estimate exposures from a swimming scenario (see dermal test order data description Section 5.1.1.1.5). The K_p diluted in a solvent as provided by the test order study has a lower permeability than the estimate provided in the EPA risk Assessment Guidance for Superfund but given this swimming scenario represents a high-end exposure scenario; the appropriateness of the aqueous solution was the overriding factor in the choosing the aqueous K_p.

Other inputs for the above dose estimate varied per lifestage. For adults, per the U.S.EPA SWIMODEL, the skin surface area was 19,500 cm^2, the exposure time was 3 hours per day and the exposure duration was 1 day for acute exposures and 57 years for chronic exposures. Some of these assumptions may be conservative but were appropriate to represent a high-end exposure scenario used to capture the possible high-end risks from 1,1-dichloroethane releases to surface waters.

Table 5-25 presents a summary of the estimated dermal exposures from facility releases to surface waters. The table lists the facility corresponding to the maximum 1,1-dichloroethane surface water concentrations per OES and the highest resultant dermal exposures from swimming as a high-end estimate of exposures. Though dermal exposures were estimated for all facilities within each COU reporting 1,1-dichloroethane releases, the facility with the maximum levels of exposure as reported in Table 5-25 provides information regarding which receiving water bodies are sources of higher exposures and ensures that EPA has captured acute and chronic risks associated with high-end exposures (see Section 5.3.3.2.2 for more detailed discussion).

Table 5-25. Highest Modeled Incidental Dermal (Swimming) Doses for all COUs, for Adults, Youth, and Children[a]

OES	Facility[b]	Receiving Waterbody[c]	Surface Water Concentration[d]		Adult (21+ years)		Youth (11–15 years)		Child (6–10 years)	
			30Q5 Conc. (µg/L)	Harmonic Mean Conc. (µg/L)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)
Manufacturing	LA0000761	Bayou D'Inde & Bayou Verdine	1.7E04	9.7E03	8.4E−02	1.3E−04	6.4E−02	1.0E−04	3.9E−02	6.1E−05
Processing as a reactant intermediate	TX0119792	Unnamed, San Jacinto Bay	4.8E03	4.8E03	2.3-02	6.4E−05	1.8E−02	4.9E−05	1.1E−02	3.0E−05
Processing – repackaging	IL0064564	Rock River	1.8E02	1.82E02	8.9E−04	2.4E−06	6.8E−04	1.9E−06	4.2E−04	1.1E−06
Commercial use as a laboratory chemical	IL0034592	Sawmill Creek	8.7E01	5.1E01	4.2E−04	6.8E−07	3.2E−04	5.2E−07	2.0E−04	3.2E−07
Waste handling, treatment, and disposal (non-POTW)	NN0021610	Little Colorado River	7.3E02	7.3E02	3.6E−03	9.8E−06	2.7E−03	7.5E−06	1.7E−03	4.6E−06
Waste handling, treatment, and disposal (POTW)	NE0043371	Stevens Creek	2.7E03	1.7E03	1.3E−02	2.3E−05	1.0E−02	1.7E−05	6.1E−03	1.1E−05
Waste handling, treatment, and disposal (remediation)	CA0064599	South Fork of Arroyo Conejo Creek	4.1E04	4.1E04	2.0E−01	5.5E−04	2.0E−01	4.2E−04	9.3E−02	2.5E−04
Unknown	OH0143880	Spring Creek	7.2E03	7.2E03	3.5E−02	9.7E−05	2.7E−02	7.4E−05	1.6E−02	4.5E−05

OES	Facility[b]	Receiving Waterbody[c]	Surface Water Concentration[d]		Adult (21+ years)		Youth (11–15 years)		Child (6–10 years)	
			30Q5 Conc. (µg/L)	Harmonic Mean Conc. (µg/L)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)

[a] Modeled 1,1-dichloroethane surface water concentrations are at the facility's point of discharge. See *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming Central Tendency Exposure Estimates* (U.S. EPA, 2025q) and *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming High-End Exposure Estimates* (U.S. EPA, 2025r) for additional details.
[b] Facility data, including NPDES ID, are from DMRs, as reported in the EPA Pollutant Loading Tool.
[c] The receiving water body was identified from NPDES permit information of the releasing facility for the 2015–2020 reporting period.
[d] Modeled hydrologic flow data (*i.e.*, stream flow) are associated with the facility's receiving water body at the point of release). Receiving water flow data (30Q5 and harmonic mean) were retrieved from the NHDPlus V2.1 Dataset (U.S. EPA, 2016c).

5.1.2.4 Summary of Oral Exposure Assessment

5.1.2.4.1 Drinking Water Exposure

EPA estimated drinking water exposures for those facility effluents containing 1,1-dichloroethane discharged to receiving water bodies upstream of drinking water intakes. The Manufacturing and Commercial use as a laboratory chemical COUs/OES did not have downstream drinking water intakes and were not included in the drinking water exposure estimates. The surface water exposures presented in Table 5-26 are the maximum acute dose rate (ADR) and average daily dose (ADD) for adults and infants (using drinking water for formula) at the calculated drinking water intake after dilution from the point of release. The point of release concentrations were based on the 30Q5 flow of each of the corresponding receiving water bodies and the annual effluent discharges occurring over the facility operating days (see Table 3-3).

Table 5-26. Highest Drinking Water Exposures from Surface Water Releases[a]

OES	Facility[b]	Surface Water Concentration[c]	Adult (21+ years)		Infant (Birth to <1 year)	
		30Q5 Conc. (µg/L)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)
Manufacturing	–	–	–	–	–	–
Processing as a reactant intermediate	IL0000141	8.7E−04	3.5E−08	1.1E−11	1.2E−07	2.9E−11
Processing – repackaging	LA0124583	1.3E−04	5.4E−09	1.7E−12	1.9E−08	4.4E−12
Commercial use as a laboratory chemical	–	–	–	–	–	–
Waste handling, treatment, and disposal (non-POTW)	MI0044130	2.5E−01	1.0E−05	7.5E−09	3.5E−05	1.9E−08
Waste handling, treatment, and disposal (POTW)	CA0048194	1.1E−06	4.4E−11	1.8E−14	1.5E−10	4.7E−14
Waste handling, treatment, and disposal (remediation)	MI0042994	2.6E−04	1.0E−08	3.6E−12	3.7E−08	9.3E−12
Unknown	MI00004057	5.2E−04	2.1E−08	6.4E−12	7.3E−08	1.6E−11

[a] Facilities presented per OES are those with the highest 1,1-dichloroethane surface water concentrations. Modeled 1,1-dichloroethane drinking water concentration is at the point of drinking water facility (public water system) intake. Estimate considers dilution from the point of discharge and does not consider drinking water treatment removal.
[b] Facility data, including NPDES ID, are from DMRs, as reported in the EPA Pollutant Loading Tool.
[c] Modeled hydrologic flow data (*i.e.*, stream flow) are associated with the facility's receiving water body at the point of release (facility 30Q5 flow). The point of drinking water intake (intake 30Q5) was retrieved from the NHDPlus V2.1 dataset (U.S. EPA, 2016c). The receiving water body was identified from NPDES permit information of the releasing facility for the 2015–2020 reporting period.

1,1-Dichloroethane concentrations in drinking water and population exposures have also been evaluated through the EPA Office of Water, Office of Ground Water and Drinking Water and described in the Final Regulatory Determination 4 Support Document (accessed June 16, 2025) (January 2021, EPA 815-R-21-001). 1,1-Dichloroethane was evaluated as a candidate for regulation under SDWA as a drinking water contaminant under the fourth Contaminant Candidate List (CCL 4) Regulatory

Determination process. In 2021, 1,1-Dichloroethane was determined to not satisfy the criteria required under SDWA and did not warrant regulation. Maximum 1,1-dichloroethane concentrations among sampled large, medium, and small PWSs were 1.5 µg/L, and none of the detections exceeded the health reference level of 1,000 µg/L. Based on the data indicating that 1,1-dichloroethane was not occurring in drinking water at levels of public health concern, the EPA Office of Water made a determination not to regulate 1,1-dichloroethane under SDWA. The estimated drinking water concentrations presented in Table 5-26 from TSCA releases represent estimates of water concentrations near the discharge sites, well below those reported in the Office of Water PWS monitoring data of finished drinking water data at public water systems.

5.1.2.4.2 Fish Ingestion Exposure

EPA calculated fish ingestion exposure using modeled surface water concentrations for 1,1-dichloroethane per corresponding COU using the release pattern of facility discharges equal to the facilities' operating days (see Table 3-3) and both a high-end and a central tendency ingestion rates for adults and children and a high-end ingestion rate characterizing adult subsistence fisher ingestion rate of 142.40 g/day (see Table 5-27). To further characterize potential tribal exposures, EPA considered and included two facilities releasing in tribal lands (Navajo Nation: NN0021610 and NN0020265). Lifeways and practices of members of tribal nations may result in their higher exposures from fish consumption. Concentrations of 1,1-dichloroethane in fish were calculated by multiplying the maximum modeled surface water concentrations based on the number of operating days per year for each industrial and commercial release scenario (Table 3-3) by the EPI Suite™-generated BCF of 7 (Table 2-2). EPA estimated exposure from fish consumption using an adult ingestion rate, for 6 to less than 11 and 11 to less than 16 years according to the following equation (Equation 5-4):

Equation 5-4.

$$Exposure\ Estimate = (SWC \times BAF \times IR \times CF1 \times CF2 \times ED)/(AT \times BW)$$

Where:

SWC	=	Surface water (dissolved) concentration (µg/L)
BAF	=	Bioaccumulation factor (L/kg wet weight)
IR	=	Fish ingestion rate (g/day)
CF1	=	Conversion factor (0.001 mg/µg)
CF2	=	Conversion factor for kg/g (0.001 kg/g)
ED	=	Exposure duration (year)
AT	=	Averaging time (year)
BW	=	Body weight (80 kg)

A BCF is preferred in estimating exposure because it considers the animal's uptake of a chemical from both diet and the water column. For 1,1-dichloroethane, the BCF value (see Table 2-2) was estimated as 7 using EPI Suite™ (U.S. EPA, 2012c). The modeled surface water concentrations were converted to fish tissue concentrations using the estimated BCF.

The years within an age group (*i.e.,* 33 years for adults) was used for the exposure duration and averaging time for adult chronic exposures. Table 5-27 presents the summary of the highest fish ingestion dose resulting from the corresponding highest receiving water concentration and facility release per COU/OES. This represents the high-end exposures to 1,1-dichloroethane via fish ingestion. Fish ingestion exposures however, were estimated for all facilities releasing 1,1-dichloroethane to receiving water bodies and are presented in supplemental files (see *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming Central Tendency Exposure Estimates* (U.S. EPA, 2025q) and *Risk Evaluation for 1,1-*

Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming High-End Exposure Estimates (U.S. EPA, 2025r).)

Table 5-27. Summary of Fish Ingestion Exposures[a]

OES	Facility [a]	Receiving Waterbody	Surface Water Conc.	Adult (21+ years) High-End/ Subsistence [b]		Small Child (1–2 years) High-End/90th Percentile [c]	
			7Q10 (µg/L)	Acute (mg/kg-day)	Chronic (mg/kg-day)	Acute (mg/kg-day)	Chronic (mg/kg-day)
Manufacturing	LA0000761	Bayou D'Inde & Bayou Verdine	85.7	1.1E−03	2.9E−06	2.5E−04	6.8E−07
Processing as a reactant intermediate	TX0119792	Unnamed Ditch, San Jacinto Bay	13.6	1.7E−04	4.6E−07	3.9E−05	1.1E−07
Processing – repackaging	IL0064564	Rock River	0.7	8.7E−06	2.4E−08	2.0E−06	5.5E−09
Commercial use as a laboratory chemical	IL0034592	Sawmill Creek	0.6	8.0E−06	2.2E−08	1.8E−06	5.0E−09
Waste handling, treatment, and disposal (non-POTW)	NE0043371	Steven's Creek	18.1	2.3E−04	6.2E−07	5.2E−05	1.4E−07
	NN0021610	Little Colorado River, AZ [d]	2.9	3.6E−05	1.0E−07	8.4E−06	2.3E−08
Waste handling, treatment, and disposal (POTW)	KY0022039	Valley Creek	8.2	1.0E−04	2.8E−07	2.4E−05	1.4E−07
	NN0020265	Chinle Wash, AZ [d]	5.0	6.2E−05	1.7E−07	1.4E−05	4.0E−08
Waste handling, treatment, and disposal (remediation)	CA0064599	South Fork of Arroyo Conejo Creek	30.7	1.4E−03	3.8E−06	3.2E−04	8.8E−07
Unknown	OH0143880	Spring Creek	20.6	2.6E−04	7.0E−07	5.9E−05	1.6E−07

[a] Facilities presented per OES are those with the highest 1,1-dichloroethane surface water and fish tissue concentrations. Surface water, fish ingestion exposures were estimated for all facilities releasing 1,1-dichloroethane to receiving water bodies. See *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming Central Tendency Exposure Estimates* (U.S. EPA, 2025q) and *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming High-End Exposure Estimates* (U.S. EPA, 2025r) for additional details.

[b] High-end assumes subsistence fish ingestion rate: 142.4g/day (U.S. EPA, 2011a)

[c] High-end child 90th percentile fish ingestion rate: 7.7g/day (U.S. EPA, 2011a)

[d] Tribal fish ingestion rate: 216 g/day (U.S. EPA, 2011a)

5.1.2.4.3 Incidental Oral Ingestion from Swimming

The general population may swim in surfaces waters (streams and lakes) that are affected by 1,1-dichloroethane contamination. Modeled surface water concentrations where discharges occur were used to estimate acute doses and average daily doses due to ingestion exposure while swimming. EPA estimated the annual load from facility releases occurred over the number of facility operating days in modeling surface water concentrations.

The following equations (Equation 5-5 and Equation 5-6) were used to calculate incidental oral (swimming) doses for all COUs, for adults, youth, and children:

Equation 5-5.

$$ADR = \frac{SWC \times IR \times CF1}{BW}$$

Equation 5-6.

$$ADD = \frac{SWC \times IR \times ED \times RD \times CF1}{BW \times AT \times CF2}$$

Where:

ADR	=	Acute Dose Rate (mg/kg/day)
ADD	=	Average Daily Dose (mg/kg/day)
SWC	=	Surface water concentration (ppb or µg/L)
IR	=	Daily ingestion rate (L/day)
RD	=	Release days (days/year)
ED	=	Exposure duration (years)
BW	=	Body weight (kg)
AT	=	Averaging time (years)
CF1	=	Conversion factor (1.0×10^{-3} mg/µg)
CF2	=	Conversion factor (365 days/year)

Table 5-28 presents a summary of the estimated oral exposures from facility releases to surface waters. The table lists the facility corresponding to the maximum 1,1-dichloroethane surface water concentrations per OES and the highest resultant oral exposures from swimming. Because the acute dose of 1,1-dichloroethane is estimated to be very low compared to oral hazard values, acute and chronic risk estimates of oral exposures are only presented in the supplemental files and not in subsequent sections of this risk evaluation.

Table 5-28. Summary of Incidental Oral Exposures from Swimming[a]

OES	Facility [b]	Receiving Water Body [b]	Surface Water Concentration[c]		Adult (21+ years)		Youth (11–15 years)		Child (6–10 years)	
			30Q5 Conc. (µg/L)	Harmonic Mean Conc. (µg/L)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)	ADR_{POT} (mg/kg-day)	ADD (mg/kg-day)
Manufacturing	LA0000761	Bayou D'Inde & Bayou Verdine	1.7E04	9.7E03	5.9E−02	9.2E−05	9.2E−02	1.4E−04	5.2E−02	8.1E−05
Processing as a reactant intermediate	TX0119792	Unnamed Stream, San Jacinto Bay	4.8E03	4.8E03	1.6−02	4.5E−05	2.6E−02	7.0E−05	1.4E−02	3.9E−05
Processing – repackaging	IL0064564	Rock River	1.8E02	1.82E02	6.3E−04	1.7E−06	9.8E−04	2.7E−06	5.5E−04	1.5E−06
Commercial use as a laboratory chemical	IL0034592	Sawmill Creek	8.7E01	5.1E01	3.0E−04	4.8E−07	4.6E−04	7.4E−07	2.6E−04	4.2E−07
Waste handling, treatment, and disposal (non-POTW)	NN0021610	Little Colorado River	7.3E02	7.3E02	2.5E−03	6.9E−06	3.9E−03	1.1E−05	2.2E−03	6.0E−06
Waste handling, treatment, and disposal (POTW)	NE0043371	Stevens Creek	2.7E03	1.7E03	9.2E−03	1.6E−05	1.4E−02	2.5E−05	8.1E−03	1.4E−05
Waste handling, treatment, and disposal (remediation)	CA0064599	South Fork of Arroyo Conejo Creek	4.1E04	4.1E04	1.0E−01	3.9E−04	2.0E−01	6.0E−04	1.0E−01	3.4E−04
Unknown	OH0143880	Spring Creek	7.2E03	7.2E03	2.5E−02	6.8E−05	3.9E−02	1.1E−04	2.2E−02	6.0E−05

[a] Facilities presented per OES are those with the highest 1,1-dichloroethane surface water concentrations. See *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming Central Tendency Exposure Estimates* (U.S. EPA, 2025q) and *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming High-End Exposure Estimates* (U.S. EPA, 2025r) for additional details.
[b] Facility data, including NPDES ID, are from DMRs, as reported in the EPA Pollutant Loading Tool. The receiving water body was identified from NPDES permit information of the releasing facility for the 2015 to 2020 reporting period.
[c] Modeled hydrologic flow data (*i.e.*, stream flow) are associated with the facility's receiving water body at the point of release (facility 30Q5 flow). The point of drinking water intake (intake 30Q5) was retrieved from the NHDPlus V2.1 dataset (U.S. EPA, 2016c).

5.1.2.4.4 Incidental Oral Ingestion from Soil (Biosolids)

A full description of the modeling approach of exposures to children from incidental ingestion of soil from biosolids land application from facility-specific releases can be found in Appendix I. Below is a summary of the methodology and results. EPA assessed incidental ingestion of soil to be human health protective even though the likelihood of the occurrence of this exposure scenario to children is expected to be low.

Based on the estimation of biosolids concentration in Section G.1.2.6, the concentration of 1,1-dichloroethane in pastureland soil receiving an annual application of biosolids was estimated to be 58.8 µg/kg.

ADDs for children ingesting soil receiving biosolids (*e.g.*, agricultural land) were calculated for 1,1-dichloroethane using Equation 5-7 below.

Equation 5-7.

$$ADD = (C \times IR \times EF \times ED \times CF)/(BW \times AT)$$

Where:

ADD	=	Average Daily Dose (mg/kg/d)
C	=	Soil concentration (mg/kg)
IR	=	Intake rate of contaminated soil (mg/d)
EF	=	Exposure frequency (d)
CF	=	Conversion factor (1.0×10^{-6} kg/mg)
BW	=	Body weight (kg)
AT	=	Averaging time (non-cancer: ED × EF, cancer: 78 years × EF)

The recommended intake rate for children aged 3 to 6 years for soil pica (soil ingestion) is 1,000 mg/d. (U.S. EPA, 2017c). Mean body weight (18.6 kg) for 3- to 6-year-olds was taken from EPA's *Exposure Factors Handbook* (U.S. EPA, 2011a).

Table 5-29. Modeled Exposure to 1,1-Dichloroethane in Land Applied Biosolids for Children

OES	Average Daily Dose (mg/kg-day)
Disposal	3.16E−06

Thus, at the estimated 1,1-dichloroethane soil concentration of 58.8 ug/kg, the ADD for a 3- to 6-year-old child ingesting 1,000 mg/day of contaminated soil would be 3.16×10^{-6} mg/kg/day (Table 5-29).

Because this average daily dose estimate of 1,1-dichloroethane exposure is very low compared to oral hazard values, acute and chronic risk of oral exposures from ingestion of soil from land application of biosolids were not expected and were not estimated.

5.1.2.4.5 Incidental Oral Ingestion from Soil (Air Deposition)

A full description of the modeling approach and estimates of exposures to children from incidental ingestion of soil from air deposition of facility-specific ambient air releases can be found in Appendix I. Following is a summary of the methodology and results.

Modeled soil concentrations were calculated from 95th percentile air deposition concentrations for 100 and 1,000 m from a facility. These calculations were conducted for the Processing as a reactant OES (Table 5-30).

The recommended intake rate for children aged 3 to 6 years for soil pica is 1,000 mg/d (U.S. EPA, 2017c). Mean body weight (18.6 kg) for 3- to 6-year-olds was taken from the *Exposure Factors Handbook* (U.S. EPA, 2011a).

Table 5-30. Modeled Soil Ingestion Doses for the Processing as a Reactant OES, for Children

OES	Distance (m)	95th Percentile Soil Concentration (µg/kg)	Average Daily Dose (mg/kg-day)
Processing as a reactant	100	4.91E3	2.64E−04
	1,000	6.29E1	3.72E−06

Because this average daily dose estimate of 1,1-dichloroethane exposure is very low compared to oral hazard values, acute and chronic risk of oral exposures from ingestion of soil were not expected and were not estimated.

5.1.2.5 Weight of Scientific Evidence Conclusions for General Population Exposure

5.1.2.5.1 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the General Population Exposure Assessment

Except for two OESs, site-specific information was reasonably available when estimating releases of 1,1-dichloroethane to the environment. Thus, there is high confidence in the environmental release estimates and the resulting modeled exposure estimates for those COUs with reported releases. For scenarios using modeled releases, EPA has high confidence in the methods used to model exposures; however, there are uncertainties in the specific release volumes and locations that modeled releases represent. EPA therefore has overall moderate confidence for the COUs using modeled releases.

Ambient and Indoor Air Inhalation Exposures

EPA assessed several different evidence streams, including evidence supporting the exposure scenarios, the quality and representativeness of available monitoring data (Sections 3.3.1.1 and 3.3.2.1), evidence supporting modeling approaches and input data (Sections 3.3.1.2 and 3.3.2.2), evidence supporting release data used as model input data (Section 3.2.2), and concordance between modeled and monitored ambient air concentrations (Section 3.3.4).

Releases: 1,1-Dichloroethane concentrations in air were estimated for areas around industrial and commercial COUs/OESs reported to TRI and NEI, and for two COUs/OESs for which release estimates are based on modeled information (Sections 3.3.1.2 and 3.3.2.2). The associated strengths and limitations of these estimated environmental concentrations are described in Section 3.3.4. Industrial and commercial COUs/OESs that rely on release data reported to TRI and NEI, site-specific release estimates are supported by moderate to robust evidence. For COUs/OESs that rely primarily on generic scenarios, release estimates are supported by moderate evidence as described in Section 3.2.2.

Modeling Methodologies and Model Input Data: As stated in Section 3.3.4, the modeling methodology used to estimate exposure concentrations via the ambient air pathway is supported by robust evidence. Model input data on air releases are supported by moderate to robust evidence. The ability to locate

releases by location strengthens assumptions when selecting model input parameters that are typically informed by location (*e.g.*, meteorological data, land cover parameters). Thus, model input data on air releases are supported by moderate to robust evidence.

Comparison of Modeled and Monitored Data: Measured or monitored data were available for comparison. Comparison of estimated and measured exposures provide robust evidence (Section 3.3.4).

Exposure Scenarios and Exposure Factors: The general population air exposure scenarios and exposure factors used to estimate exposures are described in Section 5.1.2.1. The exposure factors used to build the exposure scenarios are directly relevant to general population exposures for communities living near releasing facilities. 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. For this analysis EPA minimizes that uncertainty by assuming exposed individuals live or work nearby facilities for 78 years (and have a 78-year life span). This period is within the range of potential habits and exposure patterns expected in the general population. Therefore, exposure scenarios underlying these exposure estimates are supported by robust evidence.

Overall Confidence in Exposure Estimates: The overall confidence in the air inhalation exposure estimates resulting from modeled air concentrations is based in part on the industrial and commercial releases reported to the TRI and NEI databases. The AERMOD modeling methodology used for this analysis is robust and considers contributions from both stack and fugitive emissions. High-end versus central tendency concentrations are statistically defined as 95th percentile and 50th percentile concentrations per releasing facility and the maximum 95th percentile per COU is the facility with the highest 95th percentile concentration among all facilities within a COU. Modeled releases do not have location data therefore there is a lower confidence in both the estimated concentrations and the associated exposure estimates. The exposure scenarios considered are most relevant to long-term residents in fenceline communities. Overall confidence varies due to variable levels of confidence in underlying release information used to the support the analysis.

Oral Exposures: Surface Water Concentrations

Facility-specific estimates of aqueous concentration (derived from facility annual loads and receiving water body hydrology) to the water column were estimated to evaluate human exposures via drinking water, oral ingestion, dermal contact, and via fish ingestion. In this first step, annual load estimates were acquired from the ECHO Pollutant Loading Tool for 6 years between 2015 to 2020. The Loading Tool uses facility reported data from DMRs to calculate and then extrapolate loads for the entire year. There are several hierarchically organized steps that the ECHO Loading Tool takes to prioritize reported data for the calculation inputs in order to ensure an annual load estimate is of the best quality possible. For example, reported measurements of the quantity (load) of a chemical in facility effluent is prioritized over measurements of concentration from grab samples that must be paired with an effluent hydrologic flow value. There are inherent uncertainties surrounding the annual load estimates based on the quality of the input data from DMRs, and thus could be several reasons why annual load estimates may be considered moderate-to-poor quality. For instance, too few periods of reported DMR data make extrapolation across the year unreasonable; concentration measurements from grab samples may not have been taken at the same time or location as measurements of effluent hydrologic flow; and detection limit reporting and usage may be inconsistent. While annual load estimates from the ECHO Loading Tool do lend themselves to more efficient national-scale evaluations, the quality of the annual loads are strongly linked to the quality of reported DMR data, which should be viewed with moderate confidence at best unless it can be demonstrated that high-quality input data from DMRs are being used.

The highest annual load across the 2015 to 2020 timeframe was identified and used to estimate aqueous (water column) concentrations within the receiving water body at the site of effluent release. Thus, these initial aqueous concentrations only account for the effect of dilution and do not include source/sink processes that may increase or decrease the concentration in the ambient environment. It is also important to note that the Loading Tool calculations replace non-detects with one-half the detection limit to ensure potentially non-zero concentration estimates were considered. This is a Loading Tool option that was discussed and selected. While using concentration estimates based on one-half the detection limit may overestimate concentration (and thus load) in some cases, this step was taken to likewise remain conservative with EPA methodology and assumptions.

Aqueous concentrations used for human exposure assessment were estimated using the highest 2015 to 2020 annual releases and estimates of 30Q5 and harmonic mean (HM) hydrologic flow data for the receiving water body that were derived from National Hydrography Dataset (NHD) modeled (EROM) flow data. NHD 14-digit HUC reach codes were obtained directly from the DMRs for the facilities (based on their NPDES codes), which was then used to obtain modeled NHD hydrologic flow values (*e.g.*, lowest monthly and annual means). These flow data were used to estimate 30Q5 and HM flow using a regression-based approach that is discussed in further detail in Appendix E. The confidence in these flow values should be considered moderate-to-robust provided modeled NHD flow data has been widely used and thoroughly vetted. However, a regression-based calculation as opposed to a modeling approach was used to estimate 30Q5 and HM from NHD-acquired flow data. The latter possibly yielding a more robust confidence level. Aqueous concentrations of 1,1-dichloroethane are based on simply flow dilution using this approach, while no other source/sink processes are included.

Aqueous concentrations for human exposure assessment were based on annual releases that occurred over facility operating days. This assumption is least conservative given that each facility's release pattern is not known and any fewer days of release would result in higher concentrations estimated in the receiving water body. Additional information surrounding the methods and uncertainties for the drinking water, oral ingestion, dermal contact, and fish ingestion can be found in Appendix E.

Oral Exposures: Fish Ingestion Estimates

To account for the variability in fish consumption across the United States, and from the geographically different receiving water bodies, fish intake estimates were considered for tribal fishers, subsistence fishing populations and the general population. Each of these populations is characterized by different fish consumption rates that affect exposures to possible 1,1-dichloroethane from fish ingestion with tribal consumption representing a high-end of the exposure distribution and general population consumption levels representing a central tendency of exposures (see EPA's *Exposure Factors Handbook* for consumption rates (U.S. EPA, 2011a)). In estimating fish concentrations for each releasing facility, surface water concentrations were at the point of 1,1-dichloroethane effluent release. It is unclear what level of dilution may occur between the surface water at the facility outfall and habitats where fish reside. A source of uncertainty in the fish ingestion estimates was the BAF estimate. No monitoring data were available indicating the consumption of fish containing 1,1-dichloroethane.

Oral Exposures: Soil and Swimming Ingestion Estimates

Land application of biosolids containing 1,1-dichloroethane and air deposition onto land represent two pathways where soils containing 1,1-dichloroethane could be a source of exposure to children who play and potentially ingest soils. EPA's *Exposure Factors Handbook* provided detailed information on the child skin surface areas and event per day of the various scenarios (U.S. EPA, 2017c). This represents a high-end exposure scenario as not all children exhibit pica. However, it is also not a rare syndrome therefore EPA is representing the soil ingestion estimates as reasonable high-end exposures for children

who are a PESS subgroup to consider in this risk evaluation. It is unclear how relevant dermal and ingestion estimates from soil exposure are as 1,1-dichloroethane is expected to either volatilize or migrate from surface soils to groundwater. Furthermore, there are inherent uncertainties associated with estimating exposures from the transport of chemicals through various media (*e.g.*, air to land and subsequent soil ingestion and dermal absorption).

All releasing facilities were included in estimating dermal and incidental oral ingestion from swimming. However, the highest receiving water body concentrations per OES was presented in the summary Table 5-26 and Table 5-31. In characterizing swimming exposures as a high-end exposure scenario, EPA assumed receiving water body concentrations at the point of facility effluent release. 1,1-dichloroethane concentrations will dilute when released to surface waters, but it is unclear what level of dilution will occur when the general population swims in waters containing a number of releases of 1,1-dichloroethane over a year.

Sections 5.1.2.2, 5.1.2.2.3, and 5.1.2.4 summarize exposure assessment approaches taken to estimate general population exposures. The weight of scientific evidence conclusions supporting the exposure estimate is decided based on the strengths, limitations, and uncertainties associated with the various lines of evidence and considerations used in estimating exposures. The conclusions are summarized using the following descriptors: robust, moderate, slight, or indeterminate.

EPA used general considerations (*i.e.*, relevance, data quality, representativeness, consistency, variability, and uncertainties) as well as chemical-specific considerations to characterize the confidence of each of the exposure scenarios.

Appendix Q presents the weight of scientific evidence summary tables for the routes exposures and corresponding exposure scenarios assessed for the general population exposed to 1,1-dichloroethane.

5.1.3 Aggregate Exposure Scenarios

Section 6(b)(4)(F)(ii) of amended TSCA requires EPA, as a part of the risk evaluation, to describe whether aggregate or sentinel exposures under the COUs were considered and the basis for their consideration.

EPA has defined aggregate exposure as "the combined exposures from a chemical substance across multiple routes and across multiple pathways" (40 CFR 702.33). The fenceline methodology, *Draft Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities Version 1.0* (accessed June 16, 2025), aggregated inhalation estimates and drinking water estimates from co-located facilities. In this risk evaluation, EPA employed this approach for the general population ambient air exposure scenarios and quantitatively evaluated combined exposure and risk across multiple TRI facilities in proximity releasing 1,1-dichloroethane to air. For inhalation, this aggregate screening analysis did not identify locations where the proximity and risk estimates of nearby facilities led to aggregate risk estimates greater than 1×10^{-6} and therefore did not have a substantial impact on the overall findings. Therefore, aggregate air inhalation was not further characterized. Details of the methods and results of this screening aggregate analysis are described in Appendix D.3.

5.1.4 Sentinel Exposures

EPA defines sentinel exposure as "the exposure from a chemical substance that represents the plausible upper bound of exposure relative to all other exposures within a broad category of similar or related exposures" (40 CFR 702.33). In terms of this risk evaluation, EPA considered sentinel exposures by considering risks to human populations who may have upper bound exposures; for example, workers

and ONUs who perform activities with higher exposure potential, or certain physical factors like body weight or skin surface area exposed. EPA characterized high-end exposures in evaluating exposure using both monitoring data and modeling approaches. Where statistical data are available, EPA typically uses the 95th percentile value of the available dataset to characterize high-end exposure for a given COU.

5.2 Human Health Hazard

1,1-Dichloroethane – Human Health Hazards
Key Points

EPA evaluated the reasonably available information for human health hazards and identified hazard points of departure (PODs) for adverse effects following acute, intermediate, and chronic exposures. Differences in endpoints used in past assessments have been identified. These differences are based on OPPT systematic review criteria. EPA requested the SACC to provide input on the selection of the non-cancer and cancer PODs in the draft 1,1-dichloroethane risk evaluation. These PODs represent the potential for greater biological susceptibility across subpopulations. The most biologically relevant and sensitive PODs for non-cancer and cancer effects for 1,1-dichloroethane from among the human health hazards identified—along with the corresponding human equivalent dose (HED), the human equivalent concentration (HEC), and the total combined uncertainty factors (UF) for each route and exposure duration—are summarized below. Based on the identified PODs for each exposure duration, the following HEDs and HECs were calculated to represent an 8-hour/day 40-hour work week occupational exposure duration (referred to as a *worker* exposure duration) and a 24-hour general population exposure duration (referred to as a *continuous* exposure duration) in the risk evaluation, respectively.

For non-cancer, the lack of adequate data by all routes and durations of exposure for 1,1-dichloroethane required the use of data from 1,2-dichloroethane as read-across. The lack of adequate non-cancer data by the dermal route for 1,2-dichloroethane required route-to-route extrapolation from oral PODs. Similarly for cancer, the lack of adequate cancer data for 1,1-dichloroethane by any route required data from 1,2-dichloroethane using read-across. The following bullets summarize the key points of this section of the risk evaluation.

Non-Cancer
The POD for the **acute** oral/dermal exposure routes is based on renal toxicity, specifically increased relative kidney weight ($BMDL_{10}$ =153); the POD for the acute inhalation exposure route is based on olfactory effects, specifically nasal necrosis ($BMCL_{10}$ = 48.9 mg/m^3).

- HED (worker) = 19.9 mg/kg; HED (continuous) = 19.9 mg/kg
- HEC (worker) = 10.14 ppm; HEC (continuous) = 2.42 ppm
- Total UF = 30 for oral, inhalation, and dermal

The POD for the **intermediate** oral/dermal exposure routes is based on renal toxicity, specifically increased relative kidney weight ($BMDL_{10}$ = 27 mg/kg); the POD for the **intermediate** inhalation exposure route based on male reproductive effects, specifically decreased sperm concentration ($BMCL_5$ = 21.2 mg/m^3).

- HED (worker) = 9.1 mg/kg; HED (continuous) = 6.5 mg/kg
- HEC (worker) = 22 ppm; HEC (continuous) = 5.2 ppm
- Total UF = 30 for oral, inhalation, and dermal

The POD for the **chronic** oral, inhalation and dermal exposure routes is based on the respective intermediate PODs with the total uncertainty factor including an additional subchronic-to-chronic duration extrapolation uncertainty factor of 10× to account for the duration adjustment

- HED (worker) = 9.1 mg/kg; HED (continuous) = 6.5 mg/kg
- HEC (worker) = 22 ppm; HEC (continuous) = 5.2 ppm
- Total UF = 300 for oral, inhalation, and dermal

Cancer
- Oral Slope Factor = was not derived quantitatively due to limitations in available data
- Dermal Slope Factor = was not derived quantitatively due to limitations in available data
- Inhalation Unit Risk (IUR) (continuous) = 7.1×10^{-6} per µg/m^3; IUR (worker) = 2.4×10^{-6} per µg/m^3

5.2.1 Approach and Methodology

EPA used the general approach described in Figure 5-6 to evaluate and extract evidence for 1,1-dichloroethane human health hazard and dose−response information. This approach is based on the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), updates to the systematic review processes presented in the 1,1-Dichloroethane Systematic Review Protocol (U.S. EPA, 2025z), and the *Framework for Human Health Risk Assessment to Inform Decision Making* (U.S. EPA, 2014c).

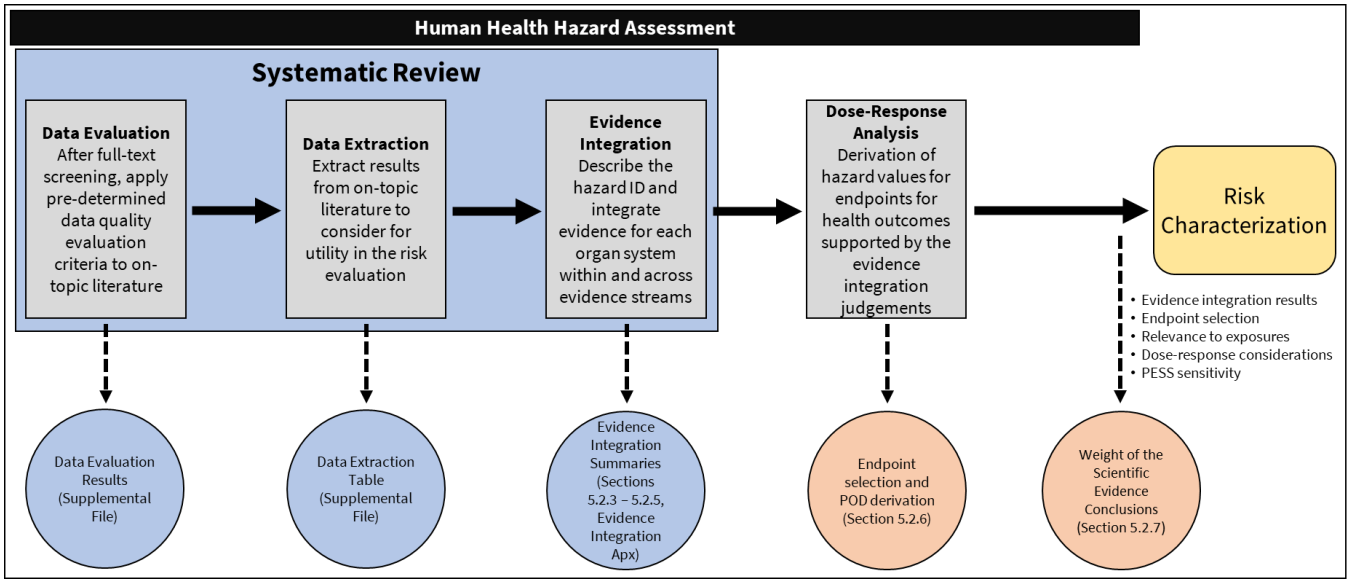


Figure 5-6. EPA Approach to Hazard Identification, Evidence Integration, and Dose-Response Analysis for Human Health Hazard

5.2.1.1 Identification and Evaluation of 1,1-Dichloroethane Hazard Data

For the human health hazard assessment, EPA used a systematic review (SR) approach described in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) to identify relevant studies of acceptable data quality and integrate the pertinent data while evaluating the weight of scientific evidence. For identified hazards and endpoints with the weight of scientific evidence supporting an adverse outcome, studies were considered for dose-response analysis. The 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) describes the general process of evidence evaluation and integration, with relevant updates to the process presented in the 1,1-Dichloroethane Systematic Review Protocol (U.S. EPA, 2025z).

For data quality evaluation, EPA systematically reviewed literature studies for 1,1-dichloroethane first by reviewing screened titles and abstracts and then full texts for relevancy using population, exposure, comparator, and outcome (PECO) screening criteria. Studies that met the PECO criteria were evaluated for data quality using pre-established metrics as specified in the 1,1-Dichloroethane Systematic Review Protocol (U.S. EPA, 2025z). Studies (based on the specified metrics) received overall data quality determinations of either Uninformative, Low, Medium, or High. The results and details of the data quality evaluation for 1,1-dichloroethane human health hazard epidemiology studies are included in the *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for Human Health Hazard Epidemiology* (U.S. EPA, 2025y). This supplemental file is hereafter referred to as the *1,1-Dichloroethane Data Quality Evaluation Information for Human Health Hazard Epidemiology* (U.S. EPA, 2025y). The results and details of the data quality evaluation for 1,1-dichloroethane animal toxicity studies are included in the *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for Human Health Hazard Animal Toxicology* (U.S. EPA, 2025x). This supplemental file is hereafter

referred to as *1,1-Dichloroethane Data Quality Evaluation Information for Human Health Hazard Animal Toxicology* (U.S. EPA, 2025x) or OPPT SR review (U.S. EPA, 2025x).

Following data quality evaluation, EPA completed data extraction of the toxicological information from each on topic study that met the PECO criteria. This data extraction included studies of all data quality determinations including "uninformative". The results of data extraction for human and animal for 1,1-dichloroethane toxicity studies are reported in the *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Extraction Information for Environmental Hazard and Human Health Hazard Animal Toxicology and Epidemiology* (U.S. EPA, 2025t). This supplemental file is hereafter referred to as the *1,1-Dichloroethane Data Extraction Information for Environmental Hazard and Human Health Hazard Animal Toxicology and Epidemiology* (U.S. EPA, 2025t).

EPA completed a hazard identification and evidence integration for 1,1-dichloroethane based on a review and evaluation of the results of the SR process including data quality evaluation and data extraction. The hazard identification and evidence integration completed for 1,1-dichloroethane are provided in Section 5.2.2 for toxicokinetics, Section 5.2.3 for non-cancer human and animal study data (stratified by organ system), Section 5.2.4 for genotoxicity and Section 5.2.5 for cancer. Details are provided in Appendix N.

Based on these hazard identification and evidence integration results, EPA completed a dose-response assessment for 1,1-dichloroethane in Section 5.2.6. These analyses of the 1,1-dichloroethane data resulted in the identification of data gaps that are summarized in Section 5.2.1.2.

5.2.1.2 1,1-Dichloroethane Data Gaps

EPA identified 3 community-based epidemiological studies, 1 occupational epidemiological study, and 16 animal toxicity studies for inclusion in the risk evaluation, and thereby, candidate studies to complete dose-response assessment and inform the identification of points of departure (PODs) for 1,1-dichloroethane. Excluding studies rated as uninformative for dose-response in the data quality evaluation left nine 1,1-dichloroethane animal toxicity studies and the three community-based epidemiological studies with acceptable study quality ratings for subsequent consideration as candidates for dose-response analysis. Each of these studies was evaluated in the dose-response assessment (Section 5.2.6) and none were identified as suitable for the identification of PODs for use in the risk evaluation. In short, the available toxicity database for 1,1-dichloroethane consists of a small number of animal studies evaluating a limited number of measured parameters.

In summary, EPA identified data gaps for 1,1-dichloroethane for non-cancer PODs by the acute, intermediate, and chronic oral, dermal, and inhalation routes; and cancer PODs by the oral, inhalation, and dermal routes (see Sections 5.2.1.2.1 and 5.2.1.2.2 for details).

A summary of the identified data gaps for 1,1-dichloroethane are provided in the following subsections for non-cancer and cancer, respectively.

5.2.1.2.1 Non-Cancer Data Gaps

Oral

EPA evaluated and extracted the data for human health hazard identification and evidence integration for oral exposures of 1,1-dichloroethane. In the dose-response assessment, EPA did not identify acceptable studies to inform the identification and derivation of PODs for 1,1-dichloroethane for acute, intermediate, and chronic oral exposures.

There were two acute-duration oral studies of 1,1-dichloroethane that were rated acceptable and were considered in the dose-response assessment for use in the risk evaluation. These studies included an acute lethality study in guinea pigs by Dow Chemical 1947) and a single-dose lethality study in rats by Muralidhara 2001). The limitations of these studies that preclude their use for POD derivation are described in Section 5.2.6.1.2.

There were three short-term (>1 to 30 days) and sub-chronic (>30 to 91 days) animal toxicology studies that were rated acceptable and were considered in the dose response assessment for use in the risk evaluation. These studies include a 10-day exposure in rats (Muralidhara et al., 2001), a 14-day exposure in rats (Ghanayem et al., 1986), and a 13-week exposure in rats (Muralidhara et al., 2001). The limitations of these studies that preclude their use for POD derivation are described in Section 5.2.6.1.3.

There was one chronic-duration oral study of 1,1-dichloroethane in mice that was rated acceptable and considered in the dose-response assessment for use in the risk evaluation. This study was a 52-week drinking water study in mice (Klaunig et al., 1986). The limitation of this study that precludes its use for POD derivation is described in detail in Section 0.

Inhalation

EPA evaluated and extracted the data for human health hazard identification and evidence integration for inhalation of 1,1-dichloroethane. EPA did not identify available or acceptable data for dose-response assessment to inform the identification of PODs for 1,1-dichloroethane for acute, intermediate, and chronic inhalation exposures.

There were no acute duration (≤24 hours) inhalation exposure studies of 1,1-dichloroethane identified as suitable for dose-response based on systematic review. One developmental inhalation toxicity study in rats for 1,1-dichloroethane by Schwetz et al. (1974) was identified as a candidate for dose-response analyses for use in the risk evaluation as an acute and/or intermediate inhalation POD. The limitation of this study that precludes its use for POD derivation is described in Sections 5.2.6.1.2 and 5.2.6.1.3.

There were two chronic inhalation studies of 1,1-dichloroethane that were identified as candidates for dose-response analyses for use in the risk evaluation. These studies included a 13-week exposure for rats, cats, guinea pigs, and rabbits (Hofmann et al., 1971)and a 6-month exposure for a single mongrel dog (Mellon Institute, 1947). The limitations of these studies that preclude their use for POD derivation are described in Section 0.

Dermal

EPA did not identify any non-cancer animal toxicological data for 1,1-dichloroethane by the dermal route.

5.2.1.2.2 Cancer Data Gaps

Oral

After data quality evaluation and data extraction as described in Section 5.2.1.1, EPA identified cancer data on 1,1-dichloroethane from one study with testing in both Osborne-Mendel rats and B6C3F1 mice. This study is a National Toxicological Program (NTP) study in rats and mice (NCI, 1978). The rat portion of this study was rated as uninformative for dose-response by systematic review (U.S. EPA, 2025x) based on a confounding health outcome unrelated to exposure. Specifically, rats from all study groups (including both sexes and controls) exhibited high incidences of pneumonia (up to 95%), indicating infections in these animals. This aspect was not discussed nor mentioned by the study authors and is unclear how these infections impacted study results. The mouse portion of this 1,1-dichloroethane

cancer bioassay revealed a statistically significant increase in benign uterine endometrial stromal polyps (4/46) in high-dose females, which were not observed in any other group. No other statistically significant evidence of cancer was observed. Additionally, pre-cancerous endometrial polyps are not a tissue growth amenable to calculate cancer slope factors. As a result, EPA did not use the NCI (1978) oral cancer study on 1,1-dichloroethane in Osborne-Mendel rats and B6C3F1 mice to calculate cancer slope factors for 1,1-dichloroethane.

Additional studies that were not identified nor classified as cancer bioassays but rather performed to determine the carcinogenic potential for 1,1-dichloroethane were evaluated.

The study by Klaunig et al. (1986) sought to examine the tumorgenicity of 1,1-dichloroethane based on the evaluating the ability of 1,1-dichloroethane to function as an initiator and/or promotor of carcinogenicity during a 52-week duration in B6C3F1 male administered 1,1-dichloroethane via drinking water. This study did not identify a significant increase in the incidences of either lung or liver tumors in neither the diethylnitrosamine initiated nor non-initiated mice treated dosed with 1,1-dichloroethane as compared to controls. Furthermore, as the measurement of water consumption was only performed on a weekly basis, there exists uncertainty to the exposure level as evaporation and spillage could attribute to 1,1-dichloroethane loss but were not assessed. Although methods were implemented to minimize these effects, the volatility of 1,1-dichloroethane when considering the drinking water approach to chemical administration presents limitations to the interpretation of the findings within this study due to the uncertainty of the doses.

In Milman et al. (1988), the carcinogenic potential of 1,1-dichloroethane was evaluated also based on initiation and promotion assays in partially hepatectomized Osborne-Mendel rats. For the initiation assay, rats were administered a single gavage dose of corn oil or 700 mg/kg-bw of 1,1-dichloroethane in corn oil, placed on either a control diet or that which contained phenobarbital for 7 weeks and the study was concluded with a final week of control diet for both groups. At necropsy, livers were examined for gamma-glutamyltranspeptidase (GGT)-positive foci with no increased incidence in the treated rats when compared to controls. In the promotion assay, rats were administered a single tumor initiating dose of diethylnitrosamine or water via intraperitoneal injection. Six days later, animals began receiving 1,1-dichloroethane at 700 mg/kg/day for 5 days/week for 7 weeks. One week later, rats were necropsied and livers were examined histopathologically for GGT-positive foci. A significant increase in liver foci were observed in animals that received 1,1-dichloroethane for 7 weeks in conjunction with a single dose of diethylnitrosamine, but not statistically significant in animals that received 1,1-dichloroethane without diethylnitrosamine.

Inhalation

EPA, after data quality evaluation and data extraction as described in Section 5.2.1.1, did not identify a cancer study via the inhalation exposure route for 1,1-dichloroethane.

Dermal

EPA, after data quality evaluation and data extraction as described in Section 5.2.1.1, did not identify a cancer study via the dermal exposure route for 1,1-dichloroethane.

5.2.1.3 Identification of an Analog and the Use of Read-Across from 1,2-Dichloroethane Hazard Data

Because acceptable human health hazard data were not available to assess risks for 1,1-dichloroethane, EPA was required to use a “read-across” approach using data available for a closely related chemical or analog to evaluate the human health hazard of 1,1-dichloroethane. An analysis of other chlorinated

solvents as potential analogs for read-across data was performed following the general principles for read-across as outlined in Lizarraga et al. (2019) and further refinements to the read-across framework presented in a subsequent publication by Lizarraga et al. (2023), taking into consideration structural similarities, physical-chemical properties, metabolism, and toxicological similarities. The analyses resulted in the identification of 1,2-dichloroethane (a close isomer of 1,1-dichloroethane) as the most appropriate analog to fill the identified data gaps for 1,1-dichloroethane. EPA has high confidence that the 1,2-dichloroethane data will be protective of hazards associated with 1,1-dichloroethane in light of the absence of chemical-specific data for 1,1-dichloroethane.

5.2.1.3.1 Structural Similarity

The first step in identification of possible analogs is to examine structural similarity. There are several different methods for determining structural similarity. The structural similarity analysis for human health hazard was equal to analyses for structural similarity for environmental hazard that is described in Section in 4.2.1.1.1 and outlined in Figure 4-1.

5.2.1.3.2 Physical and Chemical Similarities

The comparison of key physical and chemical properties of 1,1-dichloroethane and the three primary candidate analogs identified based on structural similarities (1,2-dichloroethane, 1,1,2-trichloroethane, and 1,2-dichloropropane) is shown in Table 5-31. Considering the common variability in physical and chemical results across methods and laboratories over time, 1,1-dichloroethane has similar values to 1,2-dichloroethane for water solubility, log K_{OW}, molecular weight, physical state, Henry's Law constant and vapor pressure, all of which can affect their absorption, distribution, metabolism, and excretion (ADME) and target tissue levels. For example, in Table 5-31, water solubility and K_{OW} between 1,1-dichloroethane and 1,2-dichloroethane appear to be different. However, in general, variability in physical and chemical properties results for the same chemical for water solubility and K_{OW} can differ by orders of magnitude; therefore, differences in reported physical and chemical values are not uncommon (Gigante et al., 2021; Pontolilloand and Eganhouse, 2001). In addition, the physical and chemical properties for 1,1,2-trichloroethane and 1,2-dichloropropane are also included in Table 5-31. For 1,1,2-trichloroethane, the vapor pressure is 10 times lower, the Henry's Law constant is 7 times lower, and the molecular weight is 35 percent higher than 1,1-dichloroethane, which may have ADME implications, and therefore was not considered as close of a chemical candidate analog for read-across compared to 1,2-dichloroethane.

Table 5-31. Comparison of 1,1-Dichloroethane and 1,2-Dichloroethane for Physical and Chemical Properties Relevant to Human Health Hazard

Chlorinated Solvent	Water Solubility (mg/L)	Log K_{OW}	Molecular Weight	Physical State	Henry's Law Constant (atm-m^3/mol)	Vapor Pressure (mmHg)
1,1-Dichloroethane	5,040	1.79	98.95	Liquid	0.00562	228
1,2-Dichloroethane	8,600	1.48	98.96	Liquid	0.00118	79
1,1,2-Trichloroethane	4,590	1.89	133.41	Liquid	0.00082	23
1,2-Dichloropropane	2,800	1.99	112.99	Liquid	0.00282	40

5.2.1.3.3 Metabolic Similarities

In Vitro Metabolism Studies – 1,1-Dichloroethane

The proposed metabolic pathways (see Figure_Apx N-1) for 1,1-dichloroethane have been elucidated from *in vitro* studies using rat hepatic microsomes (McCall et al., 1983; Sato et al., 1983; Van Dyke and Wineman, 1971). As outlined, the primary metabolic pathway involves oxidation of the C-1 carbon by cytochrome P450 (CYP450) resulting in an unstable alpha-haloalcohol followed by dechlorination to

produce acetyl chloride and acetic acid, which is the major metabolite. The alpha-haloalcohol may also undergo a chlorine shift to yield chloroacetyl chloride and monochloroacetic acid, although this reaction is not favored. CYP450 oxidation at the C-2 position results in the formation of 2,2-dichloroethanol, dichloroacetaldehyde, and dichloroacetic acid as minor metabolites. Metabolism of 1,1-dichloroethane can be increased by induction with phenobarbital and ethanol, but not β-naphthoflavone (McCall et al., 1983; Sato et al., 1983). Similarly, enzymatic dechlorination is inducible by phenobarbital, but not 3-methylcholanthrene (Van Dyke and Wineman, 1971).

In Vivo and In Vitro Metabolism Studies – 1,2-Dichloroethane

No human studies on the metabolism of 1,2-dichloroethane were located. Figure_Apx N-2 outlines the primary metabolic pathways for 1,2-dichloroethane, elucidated from *in vitro* studies and *in vivo* studies in rats and mice, include cytochrome P450 (CYP450) oxidation and glutathione (GSH) conjugation (IPCS, 1995). Metabolism by CYP450 results in an unstable gem-chlorohydrin that releases hydrochloric acid, resulting in the formation of 2-chloroacetaldehyde. 2-Chloroacetaldehyde is oxidized via aldehyde dehydrogenase to form chloroacetic acid or reduced to form 2-chloroethanol, and these metabolites are conjugated with GSH and excreted in the urine. Metabolism via glutathione-S-transferase results in formation of S-(2-chloroethyl)-glutathione, which rearranges to form a reactive episulfonium ion. The episulfonium ion can form adducts with protein, DNA or RNA, or interact further with GSH to produce water soluble metabolites that are excreted in the urine. Both 1,1-dichloroethane and 1,2-dichloroethane produce reactive HCl acid and reactive chloroaldehydes as metabolites.

As depicted in Figure_Apx N-1 and Figure_Apx N-2, in terms of metabolic similarities between 1,1- and 1,2-dichloroethane, both are directly reactive and both form chloroaldehydes, which can form persistent DNA crosslinks (OECD, 2015).

5.2.1.3.1 Toxicological Similarity – Non-Cancer

There are limited to no available non-cancer data available by the acute, intermediate and chronic oral, inhalation routes, and dermal routes for 1,1-dichloroethane. As a result, the 1,2-dichloroethane database was systematically reviewed and evaluated to identify non-cancer PODs to be used as read-across from 1,2-dichloroethane to fill in those 1,1-dichloroethane data gaps and calculate quantitative risk estimates.

Table 5-32 shows a qualitative comparison of common non-cancer findings between 1,1-dichloroethane and 1,2-dichloroethane, highlighting an overall similarity. The final non-cancer quantitative PODs selected for 1,1-dichloroethane (using 1,2-dichloroethane data as read-across) were based upon the strength of the evidence from data that ranked Moderate to High by systematic review, was of reliable and sufficient quality, and was the most biologically relevant and sensitive using the best available science.

Table 5-32. Qualitative Comparison of Non-Cancer Findings Between 1,1-Dichloroethane and 1,2-Dichloroethane

Effects	1,1-Dichloroethane	1,2-Dichloroethane
Reproductive/ developmental	Evidence is inadequate to assess whether 1,1-dichloroethane exposure may cause reproductive/ developmental toxicity under relevant exposure circumstances.	Evidence suggests, but is not sufficient to conclude, that 1,2-dichloroethane can cause effects on male reproductive structure and/or function under relevant exposure conditions. Evidence is inadequate to determine whether 1,2-dichloroethane can cause effects on the developing organism. There is no evidence that 1,2-dichloroethane causes effects on female reproductive structure and/or function.
Renal	Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes renal toxicity under relevant exposure circumstances.	Evidence indicates that 1,2-dichloroethane likely causes renal effects under relevant exposure circumstances.
Hepatic	Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes hepatic toxicity under relevant exposure circumstances.	Evidence suggests, but is not sufficient to conclude, that 1,2-dichloroethane can cause hepatic effects under relevant exposure conditions.
Nutritional/ metabolic	Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes body weight decrements under relevant exposure circumstances.	Evidence suggests that 1,2-dichloroethane can cause body weight decrements under relevant exposure circumstances.
Neurological/ behavioral	Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes neurological effects under relevant exposure circumstances.	Evidence indicates that 1,2-dichloroethane likely causes neurological/behavioral effects under relevant exposure circumstances.
Immune/ hematological	Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes immune system suppressions.	Evidence suggests, but is not sufficient to conclude, that 1,2-dichloroethane can cause immune system suppression under relevant exposure conditions.
Respiratory tract	–	Evidence suggests, but is not sufficient to conclude, that 1,2-dichloroethane can cause nasal effects under relevant exposure conditions.
Mortality	Evidence indicates that 1,1-dichloroethane exposure is likely to cause death under relevant exposure circumstances.	Evidence indicates that 1,2-dichloroethane can cause death under relevant exposure circumstances and lethal levels have been identified in animal studies.

5.2.1.3.2 Toxicological Similarity – Cancer

There are no adequate cancer data available for 1,1-dichloroethane. As a result, the 1,2-dichloroethane database was systematically reviewed and evaluated to identify cancer PODs to be used as read-across from 1,2-dichloroethane and calculate quantitative risk estimates.

Table 5-33 shows a qualitative comparison of common cancer findings between 1,1-dichloroethane and 1,2-dichloroethane, highlighting an overall similarity. Table 5-33 does not, however, reflect the full database for either chemical. The final cancer PODs selected for both chemicals were based upon the strength of the evidence from data that ranked “Medium” to “High” in systematic review, was of reliable

and sufficient quality, and was the most biologically relevant and sensitive using the best available science.

Table 5-33. Qualitative Comparison of Cancer Findings for 1,1-Dichloroethane Compared to 1,2-Dichloroethane

Study Type	1,1-Dichloroethane	1,2-Dichloroethane
Oral	Mammary gland adenocarcinoma, hemangiosarcoma in rats (NCI, 1978); uninformative rating for dose-response by systematic review	Mammary gland adenocarcinoma, hemangiosarcoma in rats (NTP, 1978); uninformative rating for dose-response by systematic review
	Endometrial stromal polyps (precursor) in mice (NCI, 1978); high rating for dose-response by systematic review	Endometrial stromal polyps (precursor) and hepatocarcinoma in mice (NTP, 1978); high rating for dose-response by systematic review
Inhalation	None identified	Mammary gland adenomas; fibroadenomas, adenocarcinomas; subcutaneous fibromas; bronchioalveolar adenoma and carcinoma; endometrial stromal polyps; hepatocellular adenoma, (Nagano et al., 2006), high rating for dose-response by systematic review
Dermal	None identified	Bronchioalveolar adenomas and adenocarcinomas (mice, 1 dose), (Suguro et al., 2017); high rating for dose-response by systematic review
Human studies	None identified	None identified

Table 5-34 provides a comparison of the cancer study findings between 1,1-dichloroethane and 1,2-dichloroethane.

Table 5-34. Summary of Cancer Study Findings for 1,1-Dichloroethane and 1,2-Dichloroethane

Chronic Study Finding	1,1-Dichloroethane	1,2-Dichloroethane
Endometrial polyps	+	+
Hepatocellular carcinomas	+	+
Hemangiosarcomas	+	+
Mammary gland tumors	+	+

[a] In general, similar tumor types or pre-cancerous lesions were observed with 1,1-dichloroethane as seen in the bioassays for 1,2- dichloroethane (*i.e.*, hepatocellular carcinomas, endometrial polyps, hemangiosarcomas, and mammary gland tumors) in F344 rats and/or BDF1 mice, (Nagano et al., 2006); high rating for dose-response by systematic review rating study.

Table 5-35 provides the results of the predicted carcinogenicity of 1,1-dichloroethane and 1,2-dichloroethante using the OncoLogic™ (accessed June 16, 2025) Model. This model was developed by EPA to evaluate the carcinogenic potential of chemicals following sets of knowledge rules based on studies of how chemicals cause cancer in animals and humans. Both 1,1-dichloroethane and 1,2-dichloroethane possessed similar results based on OncoLogic™ and similar genotoxicity profiles (see Appendix N.3).

Table 5-35. OncoLogic Carcinogenic Potential Results for 1,1-Dichloroethane and 1,2-Dichloroethane

Parameter	1,1-Dichloroethane	1,2-Dichloroethane
Classification for carcinogenicity	Low to Medium Concern	Medium Concern
Chemistry	Geminal alkyl dihalide	Vicinal alkyl dihalide
Chemical reactivity	Geminal alkyl dihalide < vicinal alkyl dihalide	

Animal studies provide limited evidence that 1,1-dichloroethane may cause cancer in rodents (NCI, 1978). Rats and mice exposed to 1,2-dichloroethane via gavage for 78 weeks exhibited a positive dose-related trend in the incidence of liver tumors in male mice and mammary gland tumors and hemangiosarcomas in female rats (NTP, 1978). Poor survival, however, in both control and treated animals limits the validity of these results. Cancer mode-of-action data for 1,1-dichloroethane are very limited and consist of a small number of genotoxicity experiments. show the results of *in vitro* and *in vivo* genotoxicity, respectively, and cell transformation assays of 1,1-dichloroethane (see Section 5.2.4 for further details).

5.2.1.3.3 Read-Across Conclusions

1,2-Dichloroethane was identified as the best available candidate chemical to fill the identified data gaps for 1,1-dichloroethane. Due to the limited data available for 1,1-dichloroethane, the selection of the more data comprehensive 1,2-dichloroethane as the analog was considered as "health protective" based on the overall comparison between the two chemicals on structural similarity, physical and chemical properties, ADME and toxicological similarity. As indicated in Section 5.2.1.3, identification of 1,2-dichloroethane through evaluation of the above parameters suggested that 1,2-dichloroethane is more reactive which may be an attributing factor as to the basis as to why the health effects identified as a result of 1,2-dichloroethane exposures seem to be elicited at relatively lower doses/concentrations as compared to those caused by 1,1-dichloroethane exposures. As a thorough evaluation into the potencies of 1,1- and 1,2-dichloroethane is limited based on data availability from 1,1-dichloroethane, some SACC members agreed that the more reactive 1,2-dichloroethane has reliable toxicity data. Thus, 1,2-dichloroethane may represent an acceptable worst-case analog and this approach was appropriate and could be used to support the evaluation of 1,1-dichloroethane. This approach could be considered as conservative and as a result be "health protective" to effects that could result from 1,1-dichloroethane exposures that were not otherwise captured in the 1,1-dichloroethane database.

Table 5-36 illustrates the many qualitative non-cancer and cancer toxicity endpoints and other chemical properties both 1,1-dichloroethane and 1,2-dichloroethane have in common and further discussed in Section 5.2.3.1. This comparison is based on the literature studies and the ATSDR reports for both isomers (ATSDR, 2024, 2015). Many of the identified endpoints for 1,1-dichloroethane and 1,2-dichloroethane were from studies evaluated by systematic review; however, not all studies were characterized as suitable for dose-response for non-cancer PODs or cancer slope factors to use for quantitative risk estimates.

Table 5-36. Common Hazards and Properties of 1,1-Dichloroethane and 1,2-Dichloroethane

1,1-Dichloroethane and 1,2-Dichloroethane Common Hazards and Properties		
Hazard-Property	**1,1-Dichloroethane**	**1,2-Dichloroethane**
Chemical reactivity	+	+
Dichloroethane isomers	+	+
Irritation	+	+
Narcosis	+	+
Genotoxicity without metabolic activation	+	+
Immunotoxicity	+	+
Endometrial polyps	+	+
Hepatocellular carcinoma	+	+
Hemangiosarcomas	+	+
Mammary gland tumors	+	+
Nephrotoxicity	+	+
Hepatoxicity	+	+
Metabolic toxicity	+	+
Cardiotoxicity	+	+

5.2.1.4 Identification and Evaluation of 1,2-Dichloroethane Hazard Data

The same process as described for 1,1-dichloroethane in Section 5.2.1 applies to the identification and evaluation of 1,2-dichloroethane hazard data. The results of the systematic review process (data quality evaluation and data extraction) for 1,2-dichloroethane are recorded in the same respective supplemental files for 1,1-dichloroethane including *1,1-Dichloroethane Data Quality Evaluation Information for Human Health Hazard Epidemiology* (U.S. EPA, 2025y), *1,1-Dichloroethane Data Quality Evaluation Information for Human Health Hazard Animal Toxicology* (U.S. EPA, 2025x), and *1,1-Dichloroethane Data Extraction Information for Environmental Hazard and Human Health Hazard Animal Toxicology and Epidemiology* (U.S. EPA, 2025t).

After EPA completed the data evaluation and data extraction for 1,2-dichloroethane, a hazard identification and evidence integration of the data were completed and the results are provided in Section 5.2.2 for toxicokinetics, Section 5.2.3 for non-cancer data stratified by organ system, Section 5.2.4 for genotoxicity, and Section 5.2.5 for cancer. Based on these hazard identification and evidence integration results, EPA completed a dose-response assessment for 1,2-dichloroethane in Section 5.2.6.

5.2.1.5 Structure of the Human Health Hazard Assessment

Appendix N provides the details of the human health hazard assessment for 1,1-dichloroethane and the identified analog 1,2-dichloroethane. Appendix N.1 provides a summary of toxicokinetics for both 1,1-dichloroethane and 1,2-dichloroethane. Appendix N.2 provides a non-cancer dose response assessment for both chemicals and summarizes the non-cancer POD derivation for acute, intermediate, and chronic durations. Appendix N.3 describes evidence for genotoxicity and cancer for both chemicals. Appendix N.4 provides the equations used in derivation of non-cancer and cancer PODs for the 1,1-dichloroethane risk assessment. Appendix N.5 and Appendix N.11 provide summaries of continuous and worker non-cancer PODs. Appendix N.6 includes the non-cancer evidence integration tables for 1,1-dichloroethane. Appendix N.7 includes the non-cancer evidence integration tables for 1,2-dichloroethane. Appendix N.8 includes the cancer evidence integration tables for 1,1-dichloroethane. Appendix N.9 includes the cancer

evidence integration tables for 1,2-dichloroethane. Appendix N.10 provides the cancer dose-response assessment. Lastly, Appendix N.12 provides the human health hazard confidence summary.

5.2.2 Toxicokinetics Summary

This section provides a summary on the absorption, distribution, metabolism, and elimination (ADME) data available for 1,1-dichloroethane and 1,2-dichloroethane. For full details on toxicokinetics see Appendix N.1. which provides details on the toxicokinetics of 1,1- and 1,2-dichloroethane including absorption (Appendix N.1.1), distribution (Appendix N.1.2), metabolism (Appendix N.1.3) and excretion (Appendix N.1.4).

5.2.2.1 1,1-Dichloroethane

The pulmonary absorption of 1,1-dichloroethane is likely to occur since previous use of 1,1-dichloroethane as a gaseous anesthetic in humans provides evidence of systemic absorption and distribution to the CNS by the inhalation route (ATSDR, 2015). Evidence of dermal absorption and penetration is supported in a study by Reid and Muianga (2012) in rabbits, fitted with masks to prevent inhalation of dermally applied 1,1,-dichloroethane to shaved abdominal skin. Halogen ions were detected in exhaled breathe passed through pure alcohol by flaming a copper wire that resulted in a green flame one hour after collection indicating absorption into the bloodstream; however, the level or rate of absorption was not quantified but only described qualitatively. Tissue:air partition coefficients calculated using a vial equilibration method on tissues obtained from male Fischer 344 rats also suggest that 1,1-dichloroethane is likely distributed to highly perfused tissues (*i.e.*, liver, muscle) and will accumulate in fat (Gargas and Andersen, 1989).

The metabolic pathways for 1,1-dichloroethane have been elucidated from *in vitro* studies using rat hepatic microsomes (McCall et al., 1983; Sato et al., 1983; Van Dyke and Wineman, 1971). The primary metabolic pathway involves oxidation by cytochrome P450 to give an unstable alpha-haloalcohol followed by dechlorination to produce acetyl chloride and acetic acid, which is the major metabolite. Cytochrome P450 oxidation results in the formation of 2,2-dichloroethanol, reactive dichloroacetaldehyde, and dichloroacetic acid as minor metabolites, under *in vitro* conditions.

Via inhalation, the metabolic rate constants for 1,1-dichloroethane were estimated for male Fischer 344 rats using a gas uptake method in rats exposed to initial concentrations of 360, 1,980, 4,500, or 8,804 mg/m^3, from which was concluded that the liver metabolism of 1,1-dichloroethane is saturable process at high concentrations (Gargas et al., 1990).

The extent of oral metabolism was evaluated in Osborne-Mendel rats and B6C3F1 mice administered 700 or 1,800 mg/kg-bw/day 1,1-dichloroethane, respectively, by gavage for 4 weeks (Mitoma et al., 1985). The total percentages of administered dose found in exhaled CO_2, excreta, and body carcass 48 hours after the administration of the radiolabeled dose were 7.45 percent in rats and 29.3 percent in mice. The 1,1-dichloroethane is highly absorbed orally. Within 48 hours in rats, 91 percent of the administered dose was eliminated in expired air (86 percent unchanged, 5 percent as CO_2). In mice, 95 percent of the administered dose was eliminated in expired air (70 percent unchanged, 25 percent as CO_2) within 48 hours.

EPA did not identify *in vivo* animal data that evaluated elimination following exposure to 1,1-dichloroethane by the dermal route nor inhalation routes and PBPK models were not identified. The highest dermal absorption value reported in the 1,1-dichloroethane OECD 428 study was 0.27 percent at 50 percent concentration in 1,2-dichloroethane as the COU vehicle. The mass balance corrected mean dermal absorption for neat 1,1-dichloroethane was 0.22 percent and the 95 percent upper confidence

limit for the neat chemical was 0.29 percent dermal absorption, or similar to the dermal absorption reported for the identified analog 1,2-dichloroethane at 0.21 percent. The IH SkinPerm model produced a very similar dermal absorption for 1,1-dichloroethane at 0.285 percent, so the value of 0.3 percent dermal absorption was utilized for risk calculations. The mean K*p* value and the 95 percent upper confidence limit K_p value for neat 1,1-dichloroethane were 0.00229 and 0.00371 cm/hour, respectively. The reported *in vitro* mean K_p value and 95 percent upper confidence limit K_p value for the analog 1,2-dichloroethane were similar at 0.00109 and 0.00137 cm/hour, respectively for the neat chemical (Schenk, 2018, 4940676).

5.2.2.2 1,2-Dichloroethane

Following oral administration in rats the elimination of 1,2-dichloroethane was rapid and occurred primarily via unchanged parent compound and carbon dioxide in the expired air and via excretion of soluble metabolites in the urine. Women inhaling 1,2-dichloroethane present in the workplace air eliminated the compound unchanged in the expired air with similar observations in women exposed via dermal contact to liquid 1,2-dichloroethane. It should be noted that in female workers exposed dermally to 1,2-dichloroethane, the breast milk levels were considerable at 283 micromolar and that similar concentrations caused cytotoxicity to human immune T cells *in vitro* at 5 and 10 percent cell death at concentrations of 157 and 379 micromolar, respectively (McDermott and Heffron, 2013). The 26-week 1,2-dichloroethane dermal study in mice produced lung tumors supporting that long term dermal exposure can produce serious systemic effects despite low dermal absorption levels (exposures 3 times/week induced 100 percent lung tumor incidence in female mice (Suguro et al., 2017).

1,1-Dichloroethane generates reactive 2,2-dichloroacetaldehyde during its metabolism and 1,2—dichloroethane generates the DNA crosslinker 2-chloroacetaldehyde during its metabolism. The metabolism of these chloroacetaldehyde by mitochondrial aldehyde dehydrogenase (ALDH) was investigated by Sharpe and Carter with 2,2-dichloroacetaldehyde being metabolized at a rate 16- to 36-fold slower than 2-chloroacetaldehyde (Sharpe and Carter, 1993). These data suggests that the reactive chloroacetaldehyde from 1,1-dichloroethane is cleared far slower by mitochondrial ALDH than the reactive chloroacetaldehyde from 1,2-dichloroethane with relevance to the hazard outcomes. In support, within the Milman et al. (1988) study, 1,1-dichloroethane was positive in the *in vivo* rat liver tumor promoter assay and 1,2-dichloroethane was not. In a study by Cheever et al. (1990), the oral administration of the ALDH inhibitor disulfiram increased the blood levels of 1,2-dichloroethane in rats by five-fold when administered via inhalation and significantly increased the incidences of testes tumors and mammary gland adenocarcinomas, which also indicates that ALDH activity is important for 1,2-dichloroethane clearance *in vivo* to protect from adverse outcomes. This data has relevance to PESS for people with the mitochondrial ALDH gene polymorphism having decreased aldehyde clearance activity with increased rates of multiple diseases such as cancer, heart disease and neurological diseases.

5.2.3 Non-Cancer Hazard Identification and Evidence Integration

The sections below describe adverse outcome and mechanistic data available as well as evidence integration conclusions for each human health hazard outcome observed in 1,1- and 1,2-dichloroethane toxicity studies. EPA identified very few epidemiological studies relevant to non-cancer endpoints. Therefore, evidence is primarily based on available laboratory animal toxicity studies—exclusively via the oral and inhalation routes.

The 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) describes the general process of evidence evaluation and integration, with relevant updates to the process presented in the 1,1-Dichloroethane Systematic Review Protocol (U.S. EPA, 2025z). Section 5.2.7 provides a detailed evaluation of the 1,1- and 1,2-dichloroethane hazard outcomes and evidence integration conclusions.

The analyses are presented as a series of evidence integration tables in Appendix N.6 for 1,1-dichloroethane (non-cancer), Appendix N.7 for 1,2-dichloroethane (non-cancer), Appendix N.8 for 1,1-dichloroethane (cancer), and Appendix N.9 for 1,2-dichloroethane (cancer).

5.2.3.1 Critical Human Health Hazard Outcomes

The sections below focus on hazard identification and evidence integration of kidney toxicity, olfactory tissue toxicity, and sperm effects, which are the most sensitive critical human health hazard outcomes associated with 1,2-dichloroethane. These hazard outcome categories received *likely* evidence integration conclusions, and sensitive health effects were identified for these hazard outcomes. In the risk evaluation, renal toxicity forms the basis of the POD used for acute, intermediate, and chronic oral exposure scenarios. Olfactory effects, presented as degeneration and necrosis of the olfactory neuroepithelium, is the basis of the POD used for acute inhalation exposure and sperm effects is the basis for intermediate and chronic inhalation exposure scenarios. Due to a lack of adequate dermal studies, dermal hazard was based on route-to-route extrapolation from oral exposure, based on ADME properties (see Appendix N.1). Additionally, hazard identification and evidence integration of other toxicity outcomes are also outlined to emphasize the integration of the identified health outcomes of both 1,1- and 1,2-dichloroethane.

5.2.3.1.1 Renal Toxicity

Humans

EPA did not identify epidemiological studies that evaluated any potential renal hazards for 1,1- or 1,2-dichloroethane.

Laboratory Animals

A review of acute, intermediate, and chronic studies identified renal effects following 1,1-dichloroethane exposure and studies were also identified that demonstrate renal effects following 1,2-dichloroethane exposure.

Oral

In the short-term Muralidhara et al. (2001) 10-day single oral gavage study, male Sprague-Dawley rats, administered 1,1-dichloroethane at a dose of 0, 1,000, 2,000, 4,000 or 8,000 mg/kg-bw/day resulted in a significantly reduced absolute kidney weights and nonprotein sulfhydryl (NPSH) content in the 2,000 and 4,000 mg/kg-bw/day dose groups on day 10. In addition, slightly elevated renal nonprotein sulfhydryl (NPSH) content in the 2,000 and 4,000 mg/kg-bw/day dose groups were also seen on day 5 and 10 of the study. BUN levels were not significantly affected by 1,1-dichloroethane in the short-term 10-day study nor did 1,1-dichloroethane cause proteinuria, glycosuria or enzymuria. Additionally, protein and glucose levels along with N-acetylglucosaminidase (NAG), acid phosphatase (ACP), alkaline phosphatase (ALP) and maltase (MAL) activities were indicated in the study to not be altered at any dose level of 1,1-dichloroethane during the 10-day study though these data were not presented in the study report. Relative kidney weights were not affected in animals treated with 1,1-dichloroethane. Additionally, gross morphological changes and chemically associated lesions as evaluated by H&E-stained sections were not identified in the kidney of dosed rats.

In the subchronic study by Muralidhara et al. (2001), male Sprague-Dawley rats, administered 1,1-dichloroethane via oral gavage for 5 days/week for 13 weeks at a dose of 0, 500, 1,000, 2,000, or 4,000 mg/kg-bw/day did not indicate a significant elevation of BUN at any dose level nor increase in urinary protein nor glucose excretion as compared to controls, though this data was not presented. In contrast, elevated acid phosphatase (ACP) in the 2,000 and 4,000 mg/kg-bw groups at 6 weeks, and ACP and N-acetylglucosaminidase (NAG) were elevated in the 1,000, 2,000, and 4,000 mg/kg-bw/day groups at 8

weeks. At 12 weeks, however, urinary excretion of ACP was significantly decreased at all doses after 12 weeks of the gavage exposure and urinary NAG in treated rats was also not different from the controls at this time point. In addition, histopathological effects on the kidney did not identify chemically induced changes as incidences of mild nephropathy were high in the control group (7/10 animals) as well as the rats examined from the treated groups (10/15, 8/15, 6/14, and 2/4 rats in the 500, 1,000, 2,000, or 4,000 mg/kg-bw/day dosage groups, respectively). Slight degeneration of the tubular epithelium, reactive hyperplasia, tubular dilation, and the presence of casts were the findings associated with the characterization of the nephropathy.

In Cheever et al. (1990), it was noted that in a preliminary study on 4-month-old Osborne-Mendel rats dosed with 150 mg/kg-bw by oral gavage of ^{14}C radiolabeled 1,2-dichloroethane it was identified that the ^{14}C was almost completely eliminated within 24 hours after administration. Elimination of the ^{14}C was found primarily in the urine (49.7-51.5 percent, primarily as thiodiglycolic acid, thiodiglycolic acid sulfoxide and chloroacetic acid), in expired air (35.5-39.6 percent) and only a small portion in the feces as detected as ^{14}C. This data suggested that the kidneys and the lungs are potential targets due to oral exposure to 1,2-dichloroethane.

B6C3F1 mice in the Storer et al. (1984) study that were administered a single oral gavage dose of 1,2-dichloroethane at 0, 100, 200, 300, 400, 500, 600 mg/kg-bw resulted in relative kidney weights increased at 300 mg/kg-bw doses and greater (13 percent increase as compared to controls at 300 mg/kg dose). In support, L-iditol dehydrogenase (IDH, 9-fold increase) and blood urea nitrogen (BUN) indicated a trend increase at 200 mg/kg-bw and greater doses but was not statistically significant due to the low number of animals tested (N=5).

In the Morel et al. (1999) acute single exposure oral gavage study in male Swiss OF1 mice treated with 0, 1,000, or 1,500 mg/kg-bw of 1,2-dichloroethane, a significant increase in damaged renal tubules (7.66% vs. 0.32% in controls) was seen only seen in the highest dose group with the lowest dose already above the limit dose.

In the 10-day short-term oral gavage Daniel et al. (1994) study, male and female Sprague-Dawley rats were treated with 0, 10, 30, 100, or 300 mg/kg-bw/day of 1,2-dichloroethane. In this study, 10 females and 8 males died at the highest dose concentration of 300 mg/kg-bw/day precluding statistical comparison of this group with controls, with no deaths occurring in the other treatment groups. No apparent relative kidney weights were identified across treatment groups.

In the subchronic 90 day (7 day/week for 13 weeks) oral gavage study by Daniel et al. (1994), male and female Sprague-Dawley rats treated with 0, 37.5, 75, or 150 mg/kg-bw/day of 1,2-dichloroethane resulted in increased relative kidney weights in both males and females (18 and 15 percent higher than controls, respectively) at the 75 and (35 and 22 percent higher than controls, respectively) at the 150 mg/kg-bw/day dose.

The subchronic 90-day oral gavage study in Wistar rats by van Esch et al. (1977) dosed at 0, 10, 30 or 90 mg/kg-bw/day resulted in a significantly increase in relative kidney weight of 17 and 16 percent higher than controls in males and females in the 90 mg/kg-bw/day, respectively. No histopathological abnormalities related with the administration of 1,2-dichloroethane were observed in the tissues examined.

In the subchronic study by NTP (1991), oral gavage of 1,2-dichloroethane at the dosages of 0, 30, 60, 120, 240 or 480 mg/kg-bw/day for 13 weeks in male F344 rats, resulted in significant increases in

absolute kidney weights at 30, 60, and 120 mg/kg/day (9, 21 and 25 percent, respectively) and significant increases in relative kidney weights at 60 and 120 mg/kg-bw/day doses (15 and 26 percent, respectively). Female F344 rats dosed at 0, 18, 37, 75, 150, or 300 mg/kg/day at 5 days/week via oral gavage for 13 weeks caused significant increases in absolute kidney weights (12 and 23 percent) and relative kidney weights (10 and 21 percent) at 75 and 150 mg/kg-bw/day, respectively.

Additionally, the subchronic study by NTP (1991) also included administration of 1,2-dichloroethane via drinking water to male and female F344, Sprague Dawley and Osborne-Mendel rats as well as male and female B6C3F1 mice. In this portion of the study animals were administered the target concentrations of 0, 500, 1000, 2000, 4000, or 8000 ppm for 13 weeks. The drinking water portion of this study was concluded to be uninformative for dose response due to uncertainties in the doses due to evaporation and spillage.

The estimated consumed concentration of 1,2-dichloroethane was 0, 49, 86, 147, 259, or 515 mg/kg-day in male F344 rats and 0, 58, 102, 182, 320, or 601 mg/kg-day in female F344 rats based on water consumption. Although the authors did not identify a NOAEL or LOAEL, a LOAEL of 58 mg/kg-day based on kidney weights in females can be proposed. Renal tubular regeneration was observed in all dosed and control male rats as indicated by one or more foci of basophilic-stained tubules lined by tubular epithelium in the cortex or outer medulla of the kidney. The lesion was minimal to mild and occurred in 9/10 rats in each group. No difference in severity was seen between groups. The incidence of renal tubular regeneration in females; however, appeared to be dose related and was observed in 9/10 at 8,000 ppm, 3/10 at 4,000 ppm, 2/10 at 2,000 ppm, 1/10 at 1,000 ppm, 0/10 at 500ppm, and in 0/10 controls. This lesion was also of minimal severity in all affected female rats (NTP, 1991).

Regarding the Sprague Dawley rats, the estimated consumed concentration of 1,2-dichloroethane was 0, 60, 99, 165, 276, or 518 mg/kg-day in males and 0, 76, 106, 172, 311, or 531 mg/kg-day in females based on water consumption. Again, the authors did not identify a NOAEL or LOAEL, a LOAEL of 76 mg/kg-day based on kidney weights in females can be proposed (NTP, 1991).

In Osborne-Mendel rats, the estimated consumed concentration of 1,2-dichloroethane was 0, 54, 88, 146, 266, or 492 mg/kg-day in males and 0, 82, 126, 213, 428, or 727 mg/kg-day in females based on water consumption. Although the authors did not identify a NOAEL or LOAEL for this group of rats either, a LOAEL of 82 mg/kg-day based on kidney weights in females can be proposed (NTP, 1991).

In the B6C3F1 mice, the estimated consumed dose of 1,2-dichloroethane was 0, 249, 448, 781, 2710, or 4207 mg/kg-day in males and 0, 244, 647, 1182, 2478, or 4926 mg/kg-day in females based on water consumption. The authors identified a NOAEL of 781 mg/kg-day (2000 ppm; based on reported calculated doses) in males based on kidney lesions and a NOAEL in females at 2478 mg/kg-day (4000 ppm; based on reported calculated doses) on the basis of mortality. Further evaluation of the study indicated a significant increase in absolute and relative kidney weight in female mice at 500 ppm, a LOAEL of 244 mg/kg-day (based on reported calculated doses) in female mice was thus proposed (NTP, 1991).

NCI (1978), a chronic gavage study in male and female Osborne-Mendel rats administered 1,1-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 382, or 764 mg/kg-day for males and 0, 475, or 950 mg/kg-day for females, evaluated gross and microscopic pathology of the kidney and urinary bladder. This study did not indicate specific adverse histopathological changes in treated rats as compared to controls as both male and female rats exhibited chronic inflammation in the kidney within all treatment groups. These observations, however, were

confounded due to high incidence of pneumonia that resulted in low survival of control and treated groups.

NCI (1978) included a chronic gavage study in male and female B6C3F1 mice administered 1,1-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 1442, or 2885 mg/kg-day for males and 0, 1665, or 3331 mg/kg-day for females. An evaluation of gross and microscopic pathology of the kidney and urinary bladder did not indicate specific adverse histopathological changes in treated mice as compared to controls as both male and female mice exhibited chronic inflammation in the kidney within all treatment groups.

Inhalation

In the Hofmann et al. (1971) 1,1-dichloroethane inhalation study, there was kidney damage in cats exposed to 1000 ppm (4047 mg/m^3) 1,1-dichloroethane for 10 weeks (6 hours/day), as indicated in histopathology analysis but limited information regarding these effects were provided in the report.

Storer et al. (1984) identified increased serum BUN (85%) and relative kidney weight (12%) in B6C3F1 male mice as compared to controls after a 4-hour exposure to 1,2-dichloroethane vapor of 499 ppm (2020 mg/m^3). Increased mortality at concentrations greater than 499 ppm precluded a more thorough evaluation of these effects in this study and subsequent dose -response analysis.

The subchronic inhalation study by Mellon Institute (1947) in a single mongrel dog exposed to air or 1000 ppm (1,067 ppm analytically) of 1,1-dichloroethane for 7 hours/day every other day for 6 months, although limited in its data reporting, did not identify effects on BUN or kidney histology. Mellon Institute (1947) also performed a subchronic inhalation study in a single mongrel dog exposed to 200 ppm (243 ppm analytically) of 1,2-dichloroethane for 7 hours/day every other day for 6 months that indicated an increase in kidney weight and marked cloudy swelling of the convoluted tubules with attendant desquamation and cast formation in the kidney. These findings were stated without the corresponding data provided in the study report.

Additionally, Mellon Institute (1947) also performed a subchronic inhalation study on male and female Albino rats also exposed to 1000 ppm (1,067 ppm analytically) of 1,1-dichloroethane for 7 hours/day every other day for 6 months. Additionally, Mellon Institute (1947) also performed a subchronic inhalation study on male and female Albino rats also exposed to 200 ppm (243 ppm analytically) of 1,2-dichloroethane for 7 hours/day every other day for 6 months to evaluate the kidney. An increase in kidney weight was identified in treated rats as compared to controls. An evaluation of kidney weights did not identify a difference as compared to controls. This study, however, experienced several limitations such as lung infections were identified in rats from all groups, resulting in a high mortality rate including in the control group. As animals died, attempts were made to replace them; however, the specifics such as the number of replacement animals were poorly described. Additionally, data from replacement animals (which received a maximum of 45 exposures) were included in the weight curves as if they had started with the original group which further confounds data interpretation. Individual animal data were; however, provided in the study.

Mechanistic

EPA did not identify mechanistic studies that evaluated any potential renal hazards for 1,1- or 1,2-dichloroethane.

Evidence Integration Summary
There were no human epidemiological nor mechanistic studies available for either 1,1- or 1,2-dichloroethane and therefore, there is *indeterminate* human evidence and mechanistic support to assess whether 1,1-dichloroethane or 1,2-dichloroethane may cause renal changes in humans.

The evidence in animals is *indeterminate* based on studies on 1,1-dichloroethane on the magnitude and severity of histological changes in the kidney and clinical signs of renal toxicity. Available toxicological studies showed changes in kidney weight, clinical chemistry, urinary excretion, and/or kidney histology; however, many of the studies that observed effects had limitations, and kidney effects were not seen consistently across studies using different species, exposure routes, or study durations. In contrast, evidence in animal studies for 1,2-dichloroethane is *moderate* based on several high- and medium-quality studies that found associations between 1,2-dichloroethane exposure and increased kidney weights, blood urea nitrogen (BUN), and/or renal tubular histopathology in rats (both sexes) and mice following inhalation, oral, dermal, and intraperitoneal injection exposures.

Overall, EPA concluded that while evidence is inadequate to assess whether 1,1-dichloroethane exposure may cause renal toxicity under relevant exposure circumstances, evidence indicates that 1,2-dichloroethane likely causes renal effects under relevant exposure circumstances.

5.2.3.1.1 Respiratory

Humans
EPA did not identify epidemiological studies that evaluated any potential respiratory hazards for 1,1- or 1,2-dichloroethane.

Laboratory Animals
A review of acute, intermediate, and chronic studies did not identify studies that indicated respiratory effects following 1,1-dichloroethane exposure and studies were identified that demonstrate respiratory effects following 1,2-dichloroethane exposure.

Oral
In the short-term Muralidhara et al. (2001) 10-day single oral gavage study, male Sprague-Dawley rats, administered 1,1-dichloroethane at a dose of 0, 1000, 2,000, 4,000 or 8,000 mg/kg-bw/day gross morphological changes and chemically associated lesions as evaluated by H&E-stained sections were not identified in the lungs of dosed rats. Mild focal pneumonitis was seen in the lungs of animals though was particularly evident in control animals as well.

In the subchronic study by Muralidhara et al. (2001), male Sprague-Dawley rats, administered 1,1-dichloroethane via oral gavage for 5 days/week for 13 weeks at a dose of 0, 500, 1,000, 2,000, or 4,000 mg/kg-bw/day showed pulmonary congestion in moribund and dead rats. Pulmonary inflammation was characterized by interstitial infiltration of inflammatory cells and a thickening of the alveolar septa which was, as per the authors, considered a frequent finding in male rats of this age in their lifecycle, though the study did not indicate what age the rats were at the initiation of the study.

In the study by Salovsky et al. (2002), a single oral dose of 136 mg/kg-bw 1,2-dichloroethane in male Wistar rats resulted in increased total number of cells in the bronchioalveolar lavage fluid (BALF) of male Wister rats at 30 days after dosing. Non-inflammatory histological changes such as cyanosis, interstitial edema, vacuolar changes, desquamative changes, atelectasis and alveolar macrophage proliferation were also seen in the lungs. Inflammatory histological such as macrophage proliferation that was mixed with a small number of neutrophils and eosinophils) occurred in the peribronchial (mild

degree on day 5 and mild-moderate on days 15 and 30), interstitial (mild-moderate on days 5 and 30 and moderate on day 15), and interbronchial (mild on day 1, mild-moderate on day 5) regions. These histological data were only presented qualitatively.

In the 10 day short-term oral gavage Daniel et al. (1994) study, male and female Sprague-Dawley rats were treated with 0, 10, 30, 100, or 300 mg/kg-bw/day of 1,2-dichloroethane. In this study, 10 females and 8 males died at the highest dose concentration of 300 mg/kg-bw/day precluding statistical comparison of this group with controls, with no deaths occurring in the other treatment groups. Gross and histopathological evaluation identified diffuse reddening in the lungs concurrent with the mortality exhibited in animals within the 300 mg/kg-bw-day group. Additional histopathological evaluation of other animals in this group of the study was not performed.

Inhalation

In the acute Dow Chemical (2006) inhalation study, histological changes and injury were identified in the olfactory mucosa of F344/DUCRL rats exposed for 4 or 8 hours to 1,2-dichloroethane vapor at 100 and 200 ppm, respectively.

In the study by Mellon Institute (1947), a single mongrel dog was exposed to air or 1000 ppm (1,067 ppm analytically) of 1,1-dichloroethane 7 hours/day, every other day for 6 months. The dog exposed to 1,1-dichloroethane exhibited lung congestion and mortality was not observed due to this exposure.

Additionally, Mellon Institute (1947) also performed a subchronic inhalation study on male and female Albino rats also exposed to 1000 ppm (1,067 ppm analytically) of 1,1-dichloroethane for 7 hours/day every other day for 6 months. This study, however, experienced several limitations such as lung infections were identified in rats from all groups, resulting in a high mortality rate including in the control group. As animals died, attempts were made to replace them; however, the specifics such as the number of replacement animals were poorly described. Additionally, data from replacement animals (which received a maximum of 45 exposures) were included in the weight curves as if they had started with the original group which further confounds data interpretation. Individual animal data were; however, provided in the study.

Mechanistic

EPA did not identify mechanistic studies that evaluated any potential respiratory hazards for 1,1- or 1,2-dichloroethane.

Evidence Integration Summary

There were no human epidemiological nor mechanistic studies available for 1,1-dichloroethane and therefore, there is *indeterminate* human evidence and mechanistic support to assess whether 1,1-dichloroethane may cause respiratory tract changes in humans. Additionally, there were no human epidemiological nor mechanistic studies identified for 1,2-dichloroethane and therefore, there is *indeterminate* human evidence to assess whether 1,2-dichloroethane may cause respiratory tract changes in humans.

Evidence based on animal studies was *indeterminate* as no studies were identified that indicated an association between respiratory tract effects and 1,1-dichloroethane exposure.

In a high-quality study, an association between 1,2-dichloroethane inhalation exposure and nasal lesions was observed in rats exposed to concentrations ≥ 435 mg/m^3 ($\geq$107.5 ppm). Among high- and medium-quality studies of longer duration and higher doses, as well as a high-quality study of acute inhalation

exposure, did not show effects of 1,2-dichloroethane on lower respiratory tract tissues of rats. Additionally, one medium-quality study reported lung lesions in rats after a single gavage dose so an association to lower respiratory effects. Based on this, evidence from animal studies was thus considered *slight to moderate*.

Overall, EPA concluded that the evidence is inadequate to assess whether 1,1-dichloroethane exposure may cause respiratory tract toxicity under relevant exposure circumstances. EPA also concluded that the evidence suggests, but is not sufficient to conclude, that 1,2-dichloroethane may cause lower respiratory tract effects under relevant exposure conditions but result in upper respiratory effects.

5.2.3.1.2 Reproductive/Developmental

Humans

EPA did not locate any human epidemiology studies for 1,1-dichloroethane that could be utilized for a non-cancer dose response analysis and the overall non-cancer 1,1-dichloroethane epidemiology literature is considered indeterminate for non-cancer health effects. A case-control study relating birth defects to exposure to various chlorinated solvents as estimated by maternal residential proximity to industrial point sources of emissions found that exposure risk values greater than zero were associated with increased odds of spina bifida and septal heart defects (Brender et al., 2014). This study also found that low exposure risk for 1,1-dichloroethane was associated with increased odds of septal heart defects, but medium and high exposure risk for 1,1-dichloroethane were not (Brender et al., 2014). This was the only acceptable study located in the literature that evaluated the relationship between 1,1-dichloroethane and any non-cancer health outcome in humans.

Evidence from the 1,2-dichloroethane literature is similarly indeterminate. The aforementioned Brender et al. (2014) study found associations between any exposure to 1,2-dichloroethane and neural tube defects and spina bifida; however as previously mentioned exposure was estimated based on maternal residential proximity to industrial point sources of emissions rather than using a measured level of exposure. Additionally, two studies of 1,2-dichloroethane presence in drinking water and congenital anomalies found a relationship between 1,2-dichloroethane detection and major cardiac defects in newborns, but the same relationship was not significant when comparing odds of major cardiac defects between newborns with 1,2-dichloroethane water concentrations above 1 ppb versus equal to or below 1 ppb (Bove, 1996; Bove et al., 1995).

Laboratory Animals

A review of acute, intermediate, and chronic studies identified studies that indicated reproductive/developmental effects following 1,1-dichloroethane exposure and studies were also identified that demonstrate reproductive/developmental effects following 1,2-dichloroethane exposure.

Oral

In the short-term Muralidhara et al. (2001) 10-day oral gavage study, male Sprague-Dawley rats, administered 1,1-dichloroethane at a dose of 0, 1.000, 2,000, 4,000 or 8,000 mg/kg-bw/day did not develop chemically associated lesions as examined by H&E-stained sections of the testis, or epididymis of rats sacrificed at 1, 5, or 10 days.

In the subchronic study by Muralidhara et al. (2001), male Sprague-Dawley rats, administered 1,1-dichloroethane via oral gavage for 5 days/week for 13 weeks at a dose of 0, 500, 1,000, 2,000, or 4,000 mg/kg-bw/day showed that histopathological evaluation of testis and epididymis sections was considered normal, though details of this evaluation were limited.

NCI (1978), a chronic gavage study in male and female Osborne-Mendel rats administered 1,1-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 382, or 764 mg/kg-day for males and 0, 475, or 950 mg/kg-day for females, indicated no changes in reproductive histopathology that was statistically different in treated animals as compared to controls based on evaluation of the testes, prostate, uterus and ovaries.

NCI (1978) also included a chronic gavage study in male and female B6C3F1 mice administered 1,1-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 1442, or 2885 mg/kg-day for males and 0, 1665, or 3331 mg/kg-day for females. An evaluation of gross and microscopic pathology of the prostate, testes, and epididymis did not suggest chemical-induced effects associated with 1,1-dichloethane exposure. In female, evaluation of uterus, endometrium and ovary revealed histopathological findings that were prevalent in control and treated animals and thus not considered chemically associated.

In a study by Payan et al. (1995), Sprague-Dawley dams were administered 1,2-dichloroethane by gavage at doses of 0, 1.2, 1.6, 2.0, and 2.4 mmol/kg (corresponding to 0, 119, 158, 198, and 238 mg/kg-bw/day) during gestation day (GD) 6 to GD 21. Evaluation of developmental endpoints included the number of resorptions, the number of live and dead fetuses, live fetal weights, sex ratios, and examinations of external, visceral, and skeletal anomalies and malformations. Due to premature delivery of three dams in the 238 mg/kg-day dosage group a day before scheduled euthanasia, these litters were excluded from the final analysis of the reproductive of the study due to potential cannibalization. No significant effect was identified regarding the mean number of implantation sites and live fetuses, fetal sex ratio, nor male and female fetal weights. At the 198 mg/kg-day dose, however, significant increases in the mean percentage of non-surviving implants/litter (resorptions and dead fetuses) and resorption sites/litter were seen relative to controls. This observation was not seen in any other dose group. Fetal skeletal anomalies and malformations were seen in all dose groups as well as controls and did not indicate 1,2-dichloroethane treatment-related effect. The overall GD 6 to 21 changes in maternal body weight gain were not statistically significant at any dose nor consistently dose responsive. These data and the increases in non-implants and resorptions are difficult to interpret given the significant maternal toxicity (decreases in maternal body weight gain) observed at corresponding doses (30 and 49% at 200 and 240 mg/kg/day, respectively), and that there was no effect on the number of live fetuses per litter despite the changes in non-surviving implants/litter and resorption sites/litter.

In an OECD 443 extended one-generation study by (WIL Research, 2015), male and female Crl:CD (SD) rats (27/sex/group) were administered 1,2-dichloroethane daily via drinking water at target exposures of 0, 50, 150 or 300 mg/kg/day for at least 28 consecutive days prior to mating with the concurrent control groups administered reverse osmosis-treated water. The F_0 male rats received continued administration of 1,2-dichloroethane throughout mating and until the day of euthanasia at day 92 or 93 of the study. A continuation of treatment of 1,2-dichloroethane for the F_0 females occurred throughout mating, gestation, and lactation through the day of euthanasia on lactation day 22. Offspring (3 pups/sex/litter from all available litters) were selected at PND 21 and identified as the F_1 generation. The administration of 1,2-dichloroethane to the F_1 offspring also began on PND 21 with the animals divided into either a cohort to evaluate reproductive/developmental toxicity or developmental neurotoxicity until the day of euthanasia.

The average quantities of 1,2-dichloroethane consumed during the F_0 and F_1 generations were determined to be below the target exposure levels assigned due to identification of a concentration-dependent decrease in water consumption attributed to decreased palatability of 1,2-dichloroethane in the drinking water making the study uninformative for dose response. In context, multiple NOAELs

were far higher than the 100 mg/kg/day maximum tolerated dose for 1,2-dichloroethane reported by (Milman et al., 1988).

The target exposure level of 50 mg/kg/day (mean calculated levels of 31 for F_0 males and 40 mg/kg/day for F_0 females) was considered to be the no-observed-adverse-effect level (NOAEL) for F_0 male and female systemic toxicity based on mean body weight losses and/or lower body weight gains observed in the 150 and 300 mg/kg/day dose groups during the pre-mating period and the NOAEL for the entire generation in F_0 males. During gestation (days 0 to 20) and lactation (days 1 to 21), mean body weight gains in the F_0 females dosed within the 150 and 300 mg/kg/day groups were similar to the control group though mean body weights remained lower in these groups as this was a continuation of the lowered mean body weights observed during the pre-mating period.

In the study by (WIL Research, 2015), administration of 1,2-dichloroethane via drinking water did not indicate evidence of reproductive toxicity at any exposure level based on reproductive performance in the F_0 generation based on male and female mating and fertility, male copulation, and female conception indices. Additionally, changes to the number of days between pairing and coitus, mean gestational lengths nor the process of parturition were shown to be a result of 1,2-dichloroethane exposure. Furthermore, no affects to spermatogenic parameters such as testicular and epididymal sperm concentrations, sperm production rate, sperm motility, and sperm morphology at any dosage level were identified nor were changes to the estrous cycle duration observed in the F_0 and F_1 generations.

Based on lower F_1 male and female offspring body weights and body weight gains in the 300 mg/kg/day group as compared to controls throughout the postnatal period, the NOAEL for neonatal toxicity was considered to be a target exposure level of 150 mg/kg/day (mean calculated exposure level of 97 mg/kg/day for males and 93 mg/kg/day for females). Although lower body weight gains were observed at the 300 mg/kg/day dose, no affects regarding the number of F_1 pups born, live litter size, percentage of male at birth, F_1 postnatal survival, clinical observations anogenital distance, necropsy findings or developmental landmarks were seen and be attributed to 1,2-dichloroethane exposure. Based on these results a NOAEL for F_0 and F_1 reproductive toxicity at the target exposure level of 300 mg/kg/day (mean calculated levels of 155 mg/kg/day for F_0 males, 182 mg/kg/day for F_0 females, 184 mg/kg/day for F_1 males, and 169 mg/kg/day for F_1 females) was assigned. Additionally, no effects on F_1 behavioral development, mean brain weights and measurements, macroscopic and microscopic findings, and brain morphometry were observed. Based on these results, the NOAEL for F_1 developmental neurotoxicity was considered to be at the target exposure level of 300 mg/kg/day.

An evaluation of the post-weaning period indicated that 1,2-dichloroethane lowered mean body weights and/or body weight gains for F_1 males in the 50, 150, and 300 mg/kg/day groups and females exhibited these affects in the 300 mg/kg/day group only as compared to the control group. Lower mean water consumption was noted for F_1 males and females at all exposure levels generally throughout the post-weaning period which further implies that 1,2-dichloroethane exposure via drinking water may result in decreased palatability and lower doses to those administered.

As the (WIL Research, 2015) study evaluated developmental neurotoxicity due to 1,2-dichloroethane exposure, a cohort of animals underwent a Functional Observational Battery (FOB) at PND 65 that did not indicate any 1,2-dichloroethane associated effects based on an evaluation of handling, open field, sensorimotor, neuromuscular, or physiological parameters. No effects were observed based on auditory startle responsiveness on PND 20 and 60 nor for motor activity on PND 65 for F1 males and females at any exposure level. These findings are inconsistent with the neurotoxicity effects, neuropathology and altered motor activity via inhalation dosing in other studies (Zhong et al., 2022).

In summary of the findings from the (WIL Research, 2015) study, the target exposure level of 50 mg/kg/day (mean calculated exposure of 31 for F_0 males, 40 for F_0 females, and 37 mg/kg/day for F_1 males) was considered to be the NOAEL for F_0 male and female and F_1 male systemic toxicity. The target exposure level of 150 mg/kg/day (mean calculated exposure level of 93 mg/kg/day) was considered the NOAEL for F_1 female systemic toxicity. Due to lower F_1 male and female offspring body weights and body weight gains observed in the 300 mg/kg/day group throughout the postnatal period, the NOAEL for neonatal toxicity was considered to be at 150 mg/kg/day (mean calculated exposure of 97 and 93 mg/kg/day for males and females, respectively). Furthermore, as no evidence of reproductive toxicity at any exposure level based on the evaluation of reproductive performance in the F_0 generation and sperm measurements nor estrous cyclicity in the F_0 and F_1 generations, the NOAEL for F_0 and F_1 reproductive toxicity was considered at the highest target exposure level of 300 mg/kg/day (mean calculated exposure levels of 155 mg/kg/day for F_0 males, 182 mg/kg/day for F_0 females, 184 mg/kg/day for F_1 males, and 169 mg/kg/day for F_1 females). Additionally, due to no observed effects on F_1 behavioral development, mean brain weights and measurements, macroscopic and microscopic findings, and brain morphometry, the NOAEL for F_1 neurotoxicity was considered to be the target exposure level of 300 mg/kg/day (mean calculated exposure levels of 184 mg/kg/day for F_1 males, and 169 mg/kg/day for F_1 females) as well.

In the reproductive/developmental inhalation study in female rats by Vozovaia (1977), non-pedigreed female rats were exposure to 15±3 mg/m^3 at 4 hours/day for 6 day/week to dichloroethane (chemical identity unclear) for 4 months to evaluate mortality, body weight, immunological effects, neurological effects, muscular activity, liver effects and effects on the estrous cycle and reproductive organ pathology. The study indicated that only changes in estrous cycle duration occurred that manifested at month 2 of the exposure period with the lengthening of estrous at 1.5 ± 0.11 days as compared to 1.01 ± 0.03 days in the control with no substantial change of the resting stage or the length of the entire cycle. The greatest deviations were observed at month 3 of the experiment, when lengthening occurred in the entire cycle at 6.1 ± 0.19 days as compared to 4.81 ± 0.15 days in the control animals with estrous lasting 1.9 ± 0.34 days vs. 1.02 ± 0.8 days in the controls and the resting stage of 3.3 ± 0.20 days compared to 2.61 ± 0.09 days in the controls. The estrous cycle, however, as reported by the authors normalized by month 4. At the conclusion of the 4 months, rats were subsequently mated and half of the pregnant animals were exposed during gestation while a cohort of pregnant animals were not subjected to inhalation exposure. This was done to evaluate whether developmental effects were the result of exposure before or during pregnancy to elucidate if dichloroethane affects the fetus directly or the ovaries in the females.

The authors indicated that no systemic effects were observed in pregnant rats and the ovaries did not display any histopathological findings. Prenatal development, however, according to the authors, significantly perturbed with increased embryonic mortality at 27.9 ± 5.74 percent as compared to 11.0 ± 3.12 percent in the control group. Preimplantation death occurred at a ratio of 1.5 ± 0.4 per female in treated females as compared to 0.3 ± 0.3 in the control. Additionally, hematomas in the head, neck and front limbs were detected in some fetuses. The authors also indicate that rats not exposed during pregnancy did not exhibit changes in fertility and no overall embryonic mortality as compared to control. Thus, the authors suggest that the effects associated with dichloroethane may be attributed to the distribution of the chemical via the placenta. In a parallel experiment, Vozovaia (1977) dichloroethane concentrations were measured in the blood and uterus of pregnant rats, the placenta, the amniotic fluid, and the total fetal tissues. In some fetuses, liver concentrations were also measured. Pregnant females were exposed to 1,000 ± 42 mg/m^3 4 hours/day for 3 and 7 days, respectively, during GD10 to GD13 and from GD10 to GD17. When pregnant rats were challenged with dichloroethane at 1000 mg/m^3 for 3 days, it was observed not only in the blood of the females (83.8 ± 20.2 mg%), but also in the tissues of

the placenta (43.0 ± 9.6 mg%), the amniotic fluid (55.5 ± 11.1 mg%), and the fetal tissues (50.6 ± 11.5 mg%). When the experiment was extended to 7 days, the amount of dichloroethane in the blood of the females did not substantially increase, whereas its concentration increased by a factor of 2.5 in the placenta, by a factor of 3.7 in the liver of the fetus, and by a factor of 4 in the fetal tissues in comparison with the values in the 3-day experiment. As this study provides data that would be useful for hazard identification, the specific chemical identity of the test chemical was not provided. Additionally, methodological details regarding exposure, no reporting of data for the endpoints evaluated and no description of statistical analyses used in the study prevent the use of this study for dose-response.

Inhalation

The inhalation study by Schwetz et al. (1974) that exposed nonpregnant female rats for 7 hours/day for 10 days or pregnant rats on GD 6 to15 to 1,1-dichloroethane at concentrations of 0, 3,800 or 6,000 ppm identified increased incidence of delayed ossification of fetal sternabrae at 6,000 ppm (24,300 mg/m^3); however, no effect on the incidence of fetal resorptions, fetal body measurements nor gross or soft tissue anomalies in the fetuses of treated pregnant rats. Furthermore, it is to be noted that the incidences of vertebrae with bipartite centra were also shown to be significantly lower in fetuses of rats exposed to 3800 ppm than in controls. Exposure to 1,1-dichloroethane did not result in alterations in conception rate nor the number of implantations or litter size in treated pregnant rats as compared to the controls. A key limitation of the study was that the treatment concentrations were not conducted within the same experiment but rather two separate experiments with an unknown time between them. Additionally, control data from the two experiments were pooled for all endpoints except one which showed a difference among control groups and the incidence of a specific skeletal variation was high in one of the control groups with greater than 60 percent of litters affected.

Rao et al. (1980), a reproductive/developmental study in pregnant SD rats exposed to 1,2-dichloroethnae vapor at 0, 100, or 300 ppm during GD 6 to 15 identified a significant decrease in bilobed thoracic centra incidences; however, due to increased incidence in maternal mortality a dose-response evaluation could not be performed on this effect. Additionally, a multi-generational evaluation by Rao et al. (1980) also identified decreased body weight of F_{1B} male weanlings as a result of exposure to 150 ppm (613 mg/m^3) for 6 hours/day for 7 weeks *in utero*.

Exposure to pregnant SD rats to 1,2-dichloroethane in Payan et al. (1995) indicated a significant decrease in pregnancy rate at 250 ppm (1,000 mg/m^3); however, this effect was not seen at the highest concentration of 300 ppm (1200 mg/m^3).

Zhang et al. (2017), a reproductive study, that evaluated the effects of 1,2-dichloroethane on male Swiss mice due to a 4-week exposure resulted in changes is sperm morphology and concentration along with decreased seminiferous tubules and the height of germinal epithelium at 25 ppm (102 mg/m^3).

Mechanistic

EPA did not identify mechanistic studies that evaluated any potential reproductive/developmental hazards for 1,1-dichloroethane. Male mice treated with 86 ppm or 173 ppm (350 or 700 mg/m^3, respectively) for 4 weeks resulted in an inhibition of the cyclic adenosine monophosphate (cAMP)-response element binding (CREB) protein and the cAMP-response element modulator (CREM), subsequently inducing apoptosis, and resulting in reproductive toxicity in male mice as indicated by a decrease in sperm concentration of greater than 25 percent (4.65 ± 0.52 vs. 3.30 ± 0.57 M/g), in the control vs. 700 mg/m^3 treated animals, respectively (Zhang et al., 2017).

In the study by Payan et al. (1995), an experiment in pregnant Sprague Dawley rats exposed to a single dose of 160 mg/kg ^{14}C-1,2-dichloroethane (the maximum dose that did not produce maternal nor embryo/fetal toxicity) on GD 12 or 18, was performed to assess placental transfer, tissue distribution, and metabolic tissue profiles of the chemical during fetal development. At administration at GD 12 and 18, the highest initial tissue concentrations were found in the maternal stomach and intestine, with radiolabeled 1,2-dichloroethane also detected in the maternal liver, kidney, and ovary. Although gastrointestinal absorption and distribution were similar at GD 12 and 18, maximum radioactivity levels in maternal plasma, kidneys and liver were lower in the GD 18 treated rats as compared to those treated at GD 12. Additionally, urinary excretion of 1,2-dichloroethane in the first 48 hours was lower in the GD 18 treated rats as compared to those treated at GD 12.

Distribution of radioactivity in all maternal tissues and the conceptus showed increases between 1- and 4-hours post-administration with rapid declined after 48 hours with maternal kidney and liver showing the highest concentrations of radiolabeled 1,2-dichloroethnae at all time-points (1, 2, 4, 24, and 48 hours post-treatment). Maternal plasma, uterus, and conceptuses contained low levels of radioactivity with the conceptuses accounting for less than 0.06 percent of the administered dose at all time points and a maximum of 0.11 µmol-eq/g 1,2-dichloroethane at 4 hours. A greater proportion of radioactivity in the maternal kidneys, liver, and ovaries was seen in relation to the maternal plasma. Distribution across the placenta was demonstrated by detection of radiolabeled 1,2-dichloroethane in the developing fetus within 1 hour with the maximum concentration detected 4 hours after exposure. Administration of 160 mg/kg ^{14}C-1,2-dichloroethane on GD 18 showed a greater degree of accumulation in the developing fetuses and the placenta as compared to pregnant rats dosed at GD 12. Fetal and placental radioactivity levels were comparable up to 24 hours post-treatment, at which concentrations peaked at 2 hours but began to decline at 24 and subsequently at 48 hours. Levels of radioactivity in the fetuses and placenta of rats dosed at GD 18 were at approximately 80 percent to that seen in maternal plasma, in contrast to the levels of radioactivity in the conceptuses at 60 to 67 percent of that found in maternal plasma of rats dosed at GD12 for up to 4 hours post-administration and remained at this level after 24 and 48 hours. Furthermore, radioactivity in the amniotic fluid at 48 hours post-administration at GD 18 was also higher as compared to maternal plasma levels.

Evidence Integration Summary

Due to limited and inconclusive epidemiological as well as a lack of mechanistic studies, there is *indeterminate* human evidence and mechanistic support to assess whether 1,1-dichloroethane may cause reproductive/developmental changes in humans. Additionally, the available animal toxicological studies were also limited and inconclusive and thus provided evidence that was identified as *indeterminate* for reproductive/developmental effects due to 1,1-dichloroethane.

In high- and medium-quality studies, associations were observed between 1,2-dichloroethane exposure and various birth defects (neural tube defects including spina bifida and heart defects of different types). However, the effect sizes were small with associations that were weak and, in some cases, based on very low group sizes. Results of the two available epidemiological studies were also not consistent (neural tube defects/spina bifida in one study but not the other; different types of cardiac defects in the two studies), and both studies were limited in various ways (*e.g.,* incomplete data on neural tube defects, potential exposure misclassification, questionable temporality, co-exposures to other chemicals that were also associated with the same defects). Based on these evaluations, the evidence of reproductive/developmental effects due to 1,2-dichloroethnae was considered *indeterminate* for these effects.

In high-quality studies, mice exposed to 1,2-dichloroethane by inhalation or intraperitoneal injection, but not by drinking water, exhibited effects on testicular pathology and sperm parameters. Most of the data in rats indicated no effect on the testes (or other reproductive organs); however, sperm parameters were not evaluated in rats. Thus, the evidence for effects on the male reproductive tract was considered *moderate*. Evidence was considered *moderate* based on inhalation studies in rats, oral studies in rats and mice, and a dermal study in mice that all indicated no effects of 1,2-dichloroethane on female reproductive organ weights or histopathology.

With regard to developmental effects, a high-quality study on 1,2-dichloroethane indicated sterility in male mice exposed by intraperitoneal injection (Daigle et al., 2009). In a short-term study by Daigle et al. (2009), sexually mature male C57BL/6 mice were administered 0, or 10 mg/kg/day of 1,2-dichloroethane in corn oil via intraperitoneal injection once a day for 5 days. Mice were sacrificed at 8-, 15-, 31-, and 46-days post-injection (1–3 mice/timepoint) and testicular pathology was assessed. Adverse pathology as exhibited by tubular damage, marked vacuolization of cells and loss of spermatogonia was evident 8 days after exposure which deteriorated with time and plateaued from days 15 to 46. Subsequently, Daigle et al. (2009), also evaluated reproductive capability of dosed mice where sexually mature male C57BL/6 mice (3/group) were administered 0, 5, 10, 20 or 40 mg/kg/day of 1,2-dichloroethane in corn oil via intraperitoneal injection once a day for 5 days. Forty-five days after the last injection (thus to allow for complete turnover of spermatogenesis) males were paired with female Balb/c females. Males were classified as permanently sterile if found to be infertile for 6 months or longer. Those permanently sterile males and male mice that recovered to fertility were sacrificed only after siring two consecutive litters. Temporary sterility at 3 to 5 weeks was seen in 2 of 3 mice and permanent sterility in 1 of 3 mice exposed to 5 mg/kg/day. Permanent sterility was seen in all mice exposed to at least (≥) 10 mg/kg/day. Testicular pathology was significantly increased at 5 and 10 mg/kg/day based on a significant reduction in spermatogenesis—a significant increase in the percentage of tubules that only contained Sertoli cells and histological changes as compared to control. Testes of sterile mice were atrophic and the epididymides were shrunken and deflated. Fertility-recovered males (2/3 in the 5 mg/kg/day group) displayed both active spermatogenesis and disruptions of spermatogenesis among the tubules. Preservation of the Leydig cells was observed after exposure. Due to laboratory processing error; however, the excised testes from the 20 and 40 mg/kg dose mice were destroyed and unavailable for complete histological analyses across dosage groups.

In addition, evidence for effects on weanling pup body weight after 1,2-dchloroethane inhalation exposure was considered weak and inconsistent. Thus, evidence was considered *slight* for developmental effects due to 1,2-dichloroethane.

Mechanistic evidence for reproductive/developmental effects based on inhibition of CREM/CREB signaling and the occurrence of apoptosis in testes of male mice exposed to 1,2-dichloroethane *in vivo* to support observed effects on testes pathology, sperm morphology, and fertility in this species was considered *moderate*.

Overall, EPA concluded that the evidence is inadequate to assess whether 1,1-dichloroethane exposure can cause reproductive/developmental toxicity under relevant exposure circumstances; the evidence indicates that 1,2-dichloroethane likely causes effects on male reproductive structure and/or function under relevant exposure conditions. The nature of the effect chosen for calculating risks—changes in sperm morphology and concentration identified by Zhang et al. (2017)—is considered adverse and the fertility of human males is known to be sensitive to changes in sperm numbers and quality (U.S. EPA, 1996). The evidence is inadequate to determine whether 1,2-dichloroethane can cause effects on the

developing organism and there is no evidence that 1,2-dichloroethane causes effects on female reproductive structure and/or function.

5.2.3.1.3 Neurological/Behavioral

Humans

EPA did not identify any epidemiological studies that evaluated potential neurological hazards for 1,1-dichloroethane. The clinical use of 1,1-dichloroethane as an anesthetic supports narcotic effects on the human nervous system and this clinical use was discontinued due to cardiac arrythmias (Reid and Muianga, 2012). Chlorinated aliphatic solvents are known to cause central nervous system depression, and respiratory tract and dermal irritation in humans (ATSDR, 2015). Case reports of human exposure to 1,2-dichloroethane by inhalation or ingestion indicated clinical signs of neurotoxicity (dizziness, tremors, paralysis, coma) as well as histopathology changes in the brain at autopsy (ATSDR, 2024). Workers exposed to 1,2-dichloroethane for extended periods were shown to develop cerebral edema and toxic encephalopathy (ATSDR, 2024). A single study of Russian aircraft manufacturing workers reported decreased visual-motor reaction and decreased upper extremity motor function, as well as increased reaction making errors in workers exposed to 1,2-dichloroethane compared to those that were not; however the results were only described qualitatively and no statistical analyses were conducted, and the study was determined to be uninformative by systematic review (Kozik, 1957).

Laboratory Animals

A review of acute, intermediate, and chronic studies identified studies that indicated neurological/behavioral effects following 1,1-dichloroethane exposure and studies were also identified that demonstrate neurological/behavioral effects following 1,2-dichloroethane exposure.

Oral

In the acute Muralidhara et al. (2001) single dose oral gavage study, male Sprague-Dawley rats were administered a single dose of 0, 1,000, 2,000, 4,000, 8,000, 12,000, or 16,000 mg/kg bw and observed for 2 weeks. Rats initially exhibited excitation, followed by progressive motor impairment and sedation. CNS depression was observed in a dose-dependent manner at concentrations at or exceeding (≥) 2,000 mg/kg-bw. Methods for evaluating CNS depression; however, were not described and results are described qualitatively. Fatalities in all the animals within the highest dose occurred, with deaths occurring within 24 hours of dosing.

In the short-term Muralidhara et al. (2001) 10-day oral gavage study, male Sprague-Dawley rats, administered 1,1-dichloroethane at a dose of 0, 1,000, 2,000, 4,000 or 8,000 mg/kg-bw/day resulted in rats exhibiting excitations that subsequently progressed into motor impairment and CNS depression at dosages exceeding 2,000 mg/kg-bw/day. Gross morphological changes and chemically associated lesions as evaluated by H&E-stained sections were not identified in the brains of dosed rats. The authors indicate that CNS depression was a major adverse effect associated with high and repeated oral doses of 1,1-dichloroethane and that the magnitude and duration of sedation was dose-dependent; however, this was only described qualitatively in the study.

In the subchronic study by Muralidhara et al. (2001), male Sprague-Dawley rats, administered 1,1-dichloroethane via oral gavage for 5 days/week for 13 weeks at a dose of 0, 500, 1,000, 2,000, or 4,000 mg/kg-bw/day resulted in rats exhibiting excitations that subsequently progressed into motor impairment and moderate CNS depression at dosages greater or equal than 2,000 mg/kg-bw/day. Additionally, protracted narcosis after each day's dosing at the highest dose administered was also observed. Histopathological evaluation of brain sections was considered normal, though details of this evaluation

were limited. The methodology of how CNS depression was not defined, and results were only described qualitatively. Histopathology on the brain was also not observed.

In the subchronic 90 day (7 day/week for 13 weeks) oral gavage study by Daniel et al. (1994), male and female Sprague-Dawley rats treated with 0, 37.5, 75, or 150 mg/kg-bw/day of 1,2-dichloroethane resulted in significantly increased brain relative weights were by 7.5 and 23 percent at 75 and 150 mg/kg-bw/day in male rats, respectively.

The subchronic 90-day oral gavage study in male and female Wistar rats by van Esch et al. (1977) based on dosages of 0, 10, 30 or 90 mg/kg-bw/day that resulted in a 9 percent increase in relative brain weight was seen in female rats dosed at 90 mg/kg-bw/day as compared to controls. This effect was not seen in males. No histopathological abnormalities related with the administration of 1,2-dichloroethane were observed in the tissues examined.

Inhalation

Neurotoxicity and histological changes in the brains of SD rats exposed to 1,2-dichloroethane for 12 hours was seen in a study by Zhang et al. (2010) at a LOAEL of 5,000 mg/m^3 as indicated by abnormal behavior and edema; however, details regarding the histological severity of edema were not provided.

Male SD rats exposed to 1.5 hours of 1,2-dicloroethane in Zhou et al. (2016) were shown to develop histological changes in the brain as denoted by edema at 975.9 ppm (3,950 mg/m^3).

In a study by (Huang et al., 2020), male CD-1 mice were exposed to 1,2-dichloroethane via whole body inhalation at 0, 100, 350 or 700 mg/m^3 (mean measured concentrations of 0.25 (control), 114.02, 368.14, and 728.01 mg/m^3, respectively) for 6 hours/day for 28 consecutive days. An open-field study was performed to characterize mouse exploratory behavior based on distance traveled, distance in the central and peripheral zones, resting time in the zone, and locomotor activity. Data from the open-field test identified mice exhibiting slight body shaking, running in circles, decreased activity, slow movement and fatigue in the 350 mg/m^3 exposure group; however, neither incidences nor statistical significance were indicated. Additionally, mice in the 700 mg/m^3 exposure group displayed significant decreases in distance traveled, distance in the peripheral zone, average velocity and locomotor activity. Histopathological examination of mice cerebella indicated shrunken and hypereosinophilic cytoplasm accompanied by nuclear pyknosis in the 350 and 700 mg/m^3 exposure groups with only statistically significant incidences at 700 mg/m^3. Quantitative analyses of mouse cerebellar granular cell (CGC) apoptosis also indicated significantly increased levels of apoptosis-positives cells at 700 mg/m^3 as compared to controls.

In the study by (Zhong et al., 2020), male and female Sprague-Dawley rats were exposed to 0, 600, or 1,800 mg/m^3 (mean measured concentrations of 0.4, 555, and 1,699 mg/m^3, respectively) 1,2-dichloroethane for 8 hours/day for 7days via nose-only. Data indicated an increase the brain/body weight ratio. Additionally, histopathological evaluation indicated instances of brain edema with the presence of vacuolations at in female rats within the 600 mg/m^3 exposure group. Significant vacuolations in the cerebral cortex in both male and female rats were also observed in the 1,800 mg/m^3 exposure group as compared to controls.

The (Zhong et al., 2020) whole-body study exposed male CD-1 mice to 0,100, 350, or 700 mg/m^3 (mean measured concentrations of 0.35, 124.57, 388.11, and 781.47 mg/m^3, respectively) for 6 hours/day for 28 days. Data indicated an increase brain water content and significant vacuolations in the cerebral cortex in both male and female rats in the 700 mg/m^3 exposure group as compared to controls.

In a study by (Liang et al., 2021), male Swiss mice were exposed to 1,2-dichloroethane via whole body inhalation at 0, 100, 350 or 700 mg/m^3 for 6 hours/day for 28 consecutive days. Mean measured concentrations were indicated by the author to be in agreement with nominal values; however, limited details were provided. Relative brain weight was shown to have a slight though significant decreases in only the 700 mg/m^3 as compared to controls. Absolute brain weight, however, was not provide within the study. Histopathological examination identified significantly increased areas of vacuolization in the cerebral cortex of mice exposed to the 350 and 700 mg/m^3 of 1,2-dichloroethane (40 and 65%, respectively) as compared to controls. The percentage of TUNEL-positive apoptotic cells in the cerebral cortex were also significantly increased in the 350 and 700 mg/m^3 exposure groups by 2.5- and 8-fold, respectively as compared to controls. Additionally, the increased protein expression of caspase-3, cleaved caspase 3, cytochrome c and Bax along with downregulation of Bcl-2 at 300 and 700 mg/m^3 were also indicative of apoptosis.

In a study by (Zhong et al., 2022), male CD-1 mice were exposed to 1,2-dichloroethane via whole body inhalation at 0, 100, 350 or 700 mg/m^3 for 6 hours/day for 28 consecutive days. Mean measured concentrations were indicated by the author to be in agreement with nominal values; however, limited details were provided. Mice were evaluated in open field tests for total distance traveled, distance traveled in the central area relative to the total distance traveled, time spent in the central area, and for mean speed. Data indicated that mice in the 700 mg/m^3 exposure group exhibited significant reductions in total distance traveled. Additionally, mice also displayed significant decrease in the relative distance traveled in the central area relative to the total distance traveled and mean time in the central area in the 350 and 700 mg/m^3 exposure groups as compared to controls. Furthermore, histopathological examination of the cerebral cortex identified concentration-dependent increases in vacuolization and demyelination at 350 and 700 mg/m^3 as demonstrated by disordered nerve fiber arrangement and cavitation when compared to controls. Brain-water content was also shown to be increased in 700 mg/m^3 exposed mice as compared to controls though no significant changes were observed in relative brain weight in any 1,2-dichloroethane-treated groups as compared to controls.

Mechanistic

EPA did not identify mechanistic studies that evaluated any potential neurological hazards for 1,1-dichloroethane. EPA identified mechanistic studies that suggest 1,2-dichloroethane can result in brain edema due to a downregulation of tight junction proteins (occluding and ZO-1) and mRNA, increase of free calcium, decreased ATP content, and decrease ATPase activity in the brains of mice after an exposure of to 296 ppm (1,200 mg/m^3) for 3.5 hours/day for 3 days (Wang et al., 2018; Wang et al., 2014).

Evidence Integration Summary

There were no human epidemiological nor mechanistic studies available for 1,1-dichloroethane and therefore, there is *indeterminate* human evidence and mechanistic support to assess whether 1,1-dichloroethane may cause neurological/behavioral changes in humans.

Case reports document clinical signs of neurotoxicity and brain histopathology changes in humans exposed to 1,2-dichloroethane by inhalation or ingestion as well as the ability of 1,2-dichloroethane to downregulate tight junction proteins and energy production while also upregulating aquaporin and matrix metalloproteinase in the brains of exposed mice. Based on these human epidemiological and mechanistic data available for 1,2-dichloroethane, the evidence is *slight* for an association between 1,2-dichloroethane and adverse neurological effects.

Animal studies identified the capability of 1,1-dichloroethane to induce central nervous system depression in rats exposed by gavage, and this finding is consistent with its past use as a human anesthetic. Several high- and medium-quality studies using rats exposed to 1,2-dichloroethane by inhalation or gavage or mice exposed by intraperitoneal injection showed the occurrence of neurobehavioral changes, clinical signs of neurotoxicity, and/or changes in brain histopathology. Therefore, EPA determined that the animal evidence for adverse neurological/behavioral effects based on these data are *moderate* for the association between both 1,1- and 1,2-dichloroethane and adverse neurological/behavioral effects.

Overall, EPA concluded that while evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes neurological effects under relevant exposure circumstances. The evidence indicates that 1,2-dichloroethane likely causes neurological/ behavioral effects under relevant exposure circumstances.

5.2.3.1.1 Immunological/Hematological

Humans

EPA did not identify epidemiological studies that evaluated any potential immunological/hematological hazards for 1,1- or 1,2-dichloroethane.

Laboratory Animals

A review of acute, intermediate, and chronic studies identified studies that indicated immunological/hematological effects following 1,1-dichloroethane exposure and studies were also identified that demonstrate immunological/hematological effects following 1,2-dichloroethane exposure.

Oral

Only one study by Zabrodskii et al. (2004) was identified that involved random-bred male and female albino rats being administered inducers of the monooxygenase system (phenobarbital or benzenal) three days prior to a single gavage dose of dichloroethane at 930 mg/kg-bw. The effects included significant decreases in T-cell dependent (1.71-fold) and T-cell independent (1.54-fold) humoral responses 5 days after exposure as measured by the number of antibody-producing cells in the spleen, decreased natural cytotoxicity (1.91-fold) evaluated 48 hours after the exposure, decreased antibody-dependent cell cytotoxicity (1.64-fold) 5 days after immunization of the rats with 10^8 sheep erythrocytes and delayed hypersensitivity reactions (1.63-fold) that was evaluated 24 hours post-exposure as compared to control. Treatment with phenobarbital or benzenal resulted in greater immunosuppression as compared to controls and rats exposed to 1,2-dichloroethnae alone. Although the chemical identity was only identified as dichloroethane in the study, the metabolites of 2-chloroethanol, chloroacetic aldehyde, and chloroacetic acid listed in the study are indicative of 1,2-dichloroethane.
However, in perspective since 1,2-dichloroethane data is being utilized for read-across to 1,1-dichloroethane the study is still relevant for hazard identification.

Munson et al. (1982), a study in male CD-1 mice administered 1,2-dichloroethane by oral gavage for 14 days at doses of 0, 4.9, 49 mg/kg-bw/day resulted in decreased antibody-forming cells with immunosuppression at adverse 25 and 40 percent levels at the 4.9 and 49 mg/kg-bw/day dose groups, respectively. Suppression of cell-mediated immune responses were also indicated at both dosages. A decrease in leukocytes at approximately 30 percent was reported in the highest dosage group. No effects were observed regarding the organ weights of the liver, spleen, lungs, thymus, kidney, or brain. Additionally, hepatic clinical chemistry also remained unchanged.

Additionally in the Munson et al. (1982) study, male CD-1 mice were also administered 1,2-dichloroethane via drinking water for 90 consecutive days at the time-weighted concentrations of 0, 3, 24, or 189 mg/kg-bw/day yet in contrast to the 14 day gavage study, no alteration on hematological/immunological endpoints were observed base on evaluation of hemoglobin, hematocrit, erythrocytes, leukocytes or platelets. Furthermore, 1,2-dichloroethane did appear to result in a reduction in hemagglutination as well as the AFC/spleen and AFC/10^6 spleen cells, a statistically significant reduction was not observed. However, with drinking water studies there are concerns for accurate dosing due to evaporation and spillage.

In the 10-day short-term oral gavage Daniel et al. (1994) study, male and female Sprague-Dawley rats were treated with 0, 10, 30, 100, or 300 mg/kg-bw/day of 1,2-dichloroethane. In this study, 10 females and 8 males died at the highest dose concentration of 300 mg/kg-bw/day precluding statistical comparison of this group with controls, with no deaths occurring in the other treatment groups. Hematological analysis in the 10-day oral gavage study did not identify differences in white blood cell count, red blood cell count, hemoglobin, or hematocrit between treated and control animals. Serum clinical chemistry levels for glucose, blood urea nitrogen (BUN), creatinine, alkaline phosphatase, aspartate aminotransaminase (AST), alanine aminotransaminase (ALT), lactate dehydrogenase (LDH) and calcium did not show significant differences between treatment and control rats across dosages. Cholesterol, in contrast, was indicated to have increased in the 100 mg/kg-bw/day treatment group in males although the study did not provide the data in the report.

Hematological analyses within the subchronic 90 day oral gavage study by Daniel et al. (1994) in male and female Sprague-Dawley rats treated with 0, 37.5, 75, or 150 mg/kg-bw/day of 1,2-dichloroethane resulted in females with decreased red blood cell count (7 percent), lymphocytes (10 percent), hemoglobin (7 percent) and hematocrit (8 percent) that contrasted to increased platelets (26 percent), overall white blood cells count (58 percent), neutrophils (33 percent), and monocytes (25 percent) at 150 mg/kg-bw/day. Eosinophils exhibited a decrease to zero but was not statistically significant at 75 mg/kg-bw/day. In males, hemoglobin and hematocrit values were decreased at both 75 (5 and 2%, respectively) and at 150 mg/kg-bw/day (6 and 6.5%, respectively) with an 18 percent increase in platelets at only 150 mg/kg-bw/day. Clinical chemistry data was not provided in the report for the 90 oral gavage Daniel et al. (1994) study though the authors indicate that few clinical chemistry values showed statically significant changes due to 1,2-dichloroethane exposure such as increased potassium in females and alkaline phosphatase in males in the 75 and 150 mg/kg-bw/day treatment groups with decreased albumin levels also seen in females within these same groups as compared to control. Urinalysis, as per the authors, did not indicate any treatment-related alterations across treatment groups neither males nor females.

Inhalation

In the study by Sherwood et al. (1987), female CD-1 mice exposed to 1,2 dichloroethane for 3 hours at 5.4 ppm (22 mg/m^3) resulted in mortality following streptococcal challenge but it needs to be noted that the inoculation with the bacteria was unlikely representative of a human equivalent immunological challenge. However, the control mice without 1,2-dichloroethane exposure had greatly reduced mortality and mice are the preferred species for immunotoxicity testing. At 2.3 ppm, 1,2-dichloroethane did not increase mortality over controls. Male SD rats in the same study did not exhibit any effects to the streptococcal immunological challenge after exposures up to 200 ppm (801 mg/m^3). In addition, in Sherwood et al. (1987), identified no effects in female CD-1 mice or male SD rats due to streptococcal challenge after 1,2-dichloroethane inhalation exposure for 5 or 12 days in the mice or rats, respectively.

Mechanistic
EPA did not identify mechanistic studies that evaluated any potential immunological/hematological hazards for 1,1-dichloroethane. However, its analog 1,2-dichloroethane was cytotoxic to human Jurkat T lymphocyte cells *in vitro*. Human T cell death at 5 and 10 percent levels occurred at concentrations of 157 and 379 micromolar, respectively, or similar to milk levels in female workers and blood levels in rats both via dermal exposures (ATSDR, 2024; McDermott and Heffron, 2013). This study also reported increases in reactive oxygen species and increased cellular calcium levels by 1,2-dichloroethane and other similar chlorinated solvents (trichloroethylene, perchloroethylene and dichloromethane). The human T cell death caused by 1,2-dichloroethane was inhibited by the antioxidant N-acetylcysteine. Additionally, 1,2-dichloroethane possessing immunological/hematological effects is demonstrated in an *in vitro* study that identified reduced phagocytic activity of mouse peritoneal macrophages to 76 percent of control levels at a concentration of 200 mM (Utsumi et al., 1992).

Evidence Integration Summary
There were no human epidemiological nor mechanistic studies available for 1,1-dichloroethane and therefore, there is *indeterminate* human evidence and mechanistic support to assess whether 1,1-dichloroethane may cause immunological/hematological changes in humans. Additionally, there were no human epidemiological studies available for 1,2-dichloroethane and therefore, there is *indeterminate* human evidence to assess whether 1,2-dichloroethane may cause immunological/hematological changes in humans. Limited mechanistic evidence based on *in vitro* data that showed reductions in macrophage phagocytic activity and erythrocyte GST activity after exposure to 1,2-dichloroethane was also considered to be *indeterminate*.

The evidence in animals is *indeterminate* based on only one available study on 1,1-dichloroethane on the magnitude and severity of immunological/hematological effects in rats. Available toxicological studies based on high-quality inhalation and gavage studies of immune function in mice indicated an association between 1,2-dichloroethane exposure and immunosuppression was observed. A more limited inhalation study in rats and a longer-term drinking water study in mice that was rated uninformative did not show any effects. Evidence from other studies showed only small effects on hematology and no effects on relevant organ weights or histopathology. Based on this information, evidence based on animal studies for 1,2-dichloroethane, suggests the immunological/hematological effects as *slight*.

Overall, EPA concluded that evidence is inadequate to assess whether 1,1-dichloroethane exposure may cause immunological/hematological toxicity under relevant exposure circumstances. However slight weight of evidence information indicates that its isomer 1,2-dichloroethane causes immune system suppression under relevant exposure conditions to both animals and humans. This conclusion is suggested by multiple lines of evidence such as the cytotoxicity to human immune T cells *in vitro* at relevant human tissue levels, the cell mediated immunosuppression in mice at a LOAEL value of 4.89 mg/kg-day, decreased leukocytes count in mice.

5.2.3.1.2 Hepatic

Humans
EPA did not identify epidemiological studies that evaluated any potential hepatic hazards for 1,1-dichloroethane. A single study of liver damage markers in the blood of vinyl chloride workers showed abnormal levels of aspartate aminotransferase (AST) and alanine transaminase (ALT) in the moderate 1,2-dichloroethane exposure intensity group compared with the low 1,2-dichloroethane exposure intensity group; however, all participants were also exposed to low levels of vinyl chloride monomer, which may also affect liver enzyme levels (Cheng et al., 1999).

Laboratory Animals
A review of quality acute, intermediate, and chronic studies identified studies that indicated hepatic effects following 1,1-dichloroethane exposure and studies were also identified that demonstrate hepatic effects following 1,2-dichloroethane exposure.

Oral
In the short-term Muralidhara et al. (2001) 10-day single oral gavage study, male Sprague-Dawley rats, administered 1,1-dichloroethane at a dose of 0, 1000, 2,000, 4,000 or 8,000 mg/kg-bw/day resulted a decrease in absolute liver weights at days 5 and 10 of the study in the lower dosages as all rats died at the highest dose. A comparable decrease in relative liver weights was also seen within the same groups that exhibited decreased absolute liver weights at 5 days and remained only within the 2000 and 4000 mg/kg groups at the at the conclusion of the 10-day study evaluation. Serum sorbitol dehydrogenase (SDH) and ALT activities were not significantly different from controls at any dose level as evaluated throughout the 10-day dosing regimen, although the data was not presented in the study. Hepatic microsomal cytochrome P450 exhibited sporadic increases that were not identified as neither dose nor time dependent. Hepatic non-protein sulfhydryl (NPSH) levels were not significantly altered in treated rats as compared to controls during the duration of the study.

In the subchronic study by Muralidhara et al. (2001), male Sprague-Dawley rats, administered 1,1-dichloroethane via oral gavage for 5 days/week for 13 weeks at a dose of 0, 500, 1,000, 2,000, or 4,000 mg/kg-bw/day showed a histological finding in hepatocytes in the 4000 mg/kg group of animals that were euthanized at 11 weeks of a mild condensation and change in cytoplastic staining indicative of glycogen mobilization. Relative liver weights were not shown to be significantly different from controls with regard to the 4000 mg/kg dosed animals or at the conclusion of the 13 weeks for the other dosage groups. Additionally, no elevation in serum sorbitol dehydrogenase (SDH) or ornithine-carbamyl transferase (OCT) were observed at any dose after 4, 8 or 12 weeks of exposure, though the data was not presented in the study. Additionally, no elevation in serum sorbitol dehydrogenase (SDH) or ornithine-carbamyl transferase (OCT) were observed at any dose after 4, 8 or 12 weeks of exposure.

NCI (1978), a chronic gavage study in male and female Osborne-Mendel rats administered 1,1-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 382, or 764 mg/kg-day for males and 0, 475, or 950 mg/kg-day for females, indicated no changes in liver histopathology of statistical difference in treated animals as compared to controls. Additionally, clinical chemistry was not evaluated in this study to suggest alterations in hepatic status.

In NCI (1978), the chronic gavage of male and female B6C3F1 mice administered 1,1-dichloroethane did not appear to induce hepatotoxicity that can be considered chemically associated when treated males were compared to controls. Additionally, female mice histopathological findings associated with the liver were not provided in the study report.

In Cottalasso et al. (2002), a single gavage of 628 mg/kg-bw of 1,2-dichloroethane in female Sprague-Dawley rats after 16 hours of fasting resulted in increased alanine aminotransferase (ALT), aspartate aminotransferase (AST), and lactate dehydrogenase at 45, 44 and 67 percent as compared to controls, respectively. Histological examination also identified moderate steatosis.

In the 10-day oral gavage study by Daniel et al. (1994), male and female Sprague-Dawley rats administered 0, 10, 30, 100, or 300 mg/kg-bw/day of 1,2-dichloroethane exhibited significantly increased relative liver weights (14% relative to controls) and serum cholesterol levels in male rats alone at 100 mg/kg-bw/day.

The short-term 10-day oral gavage study in Wistar rats by van Esch et al. (1977) dosed at 0, 3, 10, 30, 100, or 300 mg/kg-bw/day 1,2-dichloroethane resulted in death of all animals in the 300 mg/kg-bw/day that upon subsequent histological evaluation showed extensive vacuolization in the liver with fat droplets that were indicative of fatty degeneration. Triglyceride content in the liver was also elevated in the 30 mg/kg-day group, but not at 100 mg/kg-day.

In the subchronic 90 day (7 day/week for 13 weeks) oral gavage study by Daniel et al. (1994), male and female Sprague-Dawley rats treated with 0, 37.5, 75, or 150 mg/kg-bw/day of 1,2-dichloroethane resulted in relative liver weights increased by 32 percent in female rats at 150 mg/kg-bw/day and increased in male rats by 20 and 31 percent at 75 and 150 mg/kg-bw/day, respectively.

The subchronic 90-day oral gavage study in male Wistar rats by van Esch et al. (1977) dosed at 0, 10, 30, 90 mg/kg-bw/day resulted in a significantly increase in relative liver weight of 13 percent higher than controls in females at the highest dose. This effect was not seen in males. No histopathological abnormalities related with the administration of 1,2-dichloroethane were observed in the tissues examined nor were any changes in liver enzymes levels.

Inhalation

An inhalation study that exposed nonpregnant female rats for 7 hours/day for 10 days or pregnant rats on GD 6 to15 to concentrations of 0, 3800 or 6000 ppm of 1,1-dichloroethane evaluated serum glutamic-pyruvic transaminase (SGPT; also known as alanine transaminase, ALT), liver weights, and gross liver pathology (Schwetz et al., 1974). This study identified increases in relative liver weight in the nonpregnant females at 6000 ppm (24,300 mg/m^3) 6 days after the 10th and last daily exposure but did not identify any other effects on nonpregnant rat liver parameters nor in the pregnant rats as compared to the pooled controls. A key limitation of the study was that the treatment concentrations were not conducted within the same experiment but rather two separate experiments with an unknown time between them. Additionally, control data from the two experiments were pooled for all endpoints except one which showed a difference among control groups and the incidence of a specific skeletal variation. Exposure to 1,2-dichloroethane for 4 hours at 499 ppm (2020 mg/m^3) via inhalation in Storer et al. (1984) identified increased serum ALT (2-fold) and SDH (11-fold) in B6C3F1 male mice as compared to controls.

Absolute and relative liver weights in male Swiss mice at ≥10% as compared to controls was indicated in a 6 hour/day for 28 days study by Zeng et al. (2018) at a concentration of 89.83 ppm (364 mg/m^3).

IRFMN (1978), in a chronic 12-month study in both male and female SD rats, resulted in an increase of ALT and LDH in both sexes when exposure to 50 ppm (200 mg/m^3).

Mechanistic

EPA did not identify mechanistic studies that evaluated any potential hepatic hazards for 1,1-dichloroethane. In the study by Storer et al. (1984), B6C3F1 mice were administered a single dose of 1,2-dichloroetane at 100, 200, 300, or 400 mg/kg via oral gavage in corn oil and euthanized 4 hours later. It was identified that a statistically significant increase in DNA damage in hepatic nuclei was present in all dose groups, as characterized by single-strand breaks, when compared to controls.

Evidence Integration Summary

There were no human epidemiological nor mechanistic studies available for either 1,1-dichloroethane and therefore, there is *indeterminate* human evidence and mechanistic support to assess whether 1,1-

dichloroethane may cause hepatic changes in humans. In addition, there is *indeterminate* human evidence as the only human epidemiological study was considered inadequate due to confounding associated with co-exposure to vinyl chloride. No adequate mechanistic studies were identified as hepatic enzyme induction was demonstrated by intraperitoneal injection in mice. Limited *in vitro* data indicate that 1,2-dichloroethane may increase oxidative stress or impair glucose and/or lipid metabolism in mice and in rat hepatocytes and liver slices; however, this information suggests that overall mechanistic evidence for hepatic effects is *indeterminate*.

Due to limitation in the availability of toxicological studies on 1,1-dichloroethane that showed changes in liver weight and/or histology in the absence of relevant clinical chemistry findings, EPA determined that the animal evidence for adverse effects on the liver are *slight* for the association between 1,1-dichloroethane and adverse hepatic effects. Several high- and medium-quality studies in rats and mice found associations between 1,2-dichloroethane exposure and increased liver weights, serum enzymes, and/or histopathology changes following inhalation, oral, and intraperitoneal injection exposures. Based on these studies, EPA determined that the animal evidence for adverse effects on the liver are *moderate* for the association between 1,2-dichloroethane and adverse hepatic effects.

Overall, EPA concluded that evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure or 1,2-dichloroethane cause hepatic toxicity under relevant exposure circumstances.

5.2.3.1.3 Nutritional/Metabolic

Humans

EPA did not identify epidemiological studies that evaluated any potential nutritional/metabolic hazards for 1,1- or 1,2-dichloroethane.

Laboratory Animals

A review of acute, intermediate, and chronic studies identified studies that indicated nutritional/metabolic effects following 1,1-dichloroethane exposure and studies were also identified that demonstrate nutritional/metabolic effects following 1,2-dichloroethane exposure.

Oral

In the short-term Muralidhara et al. (2001)10 day oral gavage study, male Sprague-Dawley rats, administered 1,1-dichloroethane at a dose of 0, 1,000, 2,000, 4,000 or 80,00 mg/kg-bw/day resulted in a dose-dependent decreases in body weight at doses ≥1000 mg/kg-bw/day with rats in the 2,000 and 4,000 mg/kg-bw/day dosage groups not gaining any weight during the 10 day exposure period. All rats in the 8000 mg/kg-bw/day exposure group died within 24 hours of dosing.

In the subchronic study by Muralidhara et al. (2001), male Sprague-Dawley rats, administered 1,1-dichloroethane via oral gavage for 5 days/week for 13 weeks at a dose of 0, 500, 1,000, 2,000, or 4,000 mg/kg-bw/day resulted in the rats receiving 4,000 mg/kg-bw/day, the highest dose, experienced body weight gain consistently lower than that of controls and the other treated groups. This effect was accompanied by a progressive increase in the number of deaths, from the initial week of exposure until week 11, when the seven surviving 4,000 mg/kg-bw/day treated rats were terminated. One death occurred in the 2,000 mg/kg-bw/day group during the sixth week of 1,1-dichloroethane treatment with body weight gain significantly lower than controls from the fourth week until the end of the 13-week study. There were no fatalities in the 500 or 1,000 mg/kg-bw/day groups were observed and no reductions in body weight gain were seen as compared to controls.

In the 10 day short-term oral gavage Daniel et al. (1994) study, male and female Sprague-Dawley rats were treated with 0, 10, 30, 100, or 300 mg/kg-bw/day of 1,2-dichloroethane. In this study, 10 females and 8 males died at the highest dose concentration of 300 mg/kg-bw/day precluding statistical comparison of this group with controls, with no deaths occurring in the other treatment groups. Final body weights did not indicate significant differences between treatment groups and controls.

In the subchronic 90 day (7 day/week for 13 weeks) oral gavage study by Daniel et al. (1994), male and female Sprague-Dawley rats treated with 0, 37.5, 75, or 150 mg/kg-bw/day of 1,2-dichloroethane that resulted in a significant decrease in final body weight (17 percent) in male rats in the 150 mg/kg-bw/day group with comparable weights among other groups and within females as compared to controls.

In a developmental study by Payan et al. (1995), Sprague-Dawley dams were administered 1,2-dichloroethane by gavage at doses of 0, 1.2, 1.6, 2.0, and 2.4 mmol/kg (corresponding to 0, 119, 158, 198, and 238 mg/kg-bw/day) during gestation day (GD) 6 to GD 20. Maternal weight change was significantly less than that of controls between GD 9 and 12 of gestation at 198 mg/kg-day, and between GD 6 and 9 and GD 9 and 12 of gestation at 238 mg/kg-day, respectively. However, maternal body weight gain was not statistically significant nor consistently dose responsive for the overall period of GD6-21 for any dose groups. Absolute weight gain was significantly reduced by 30 and 49 percent in the 198 and 238 mg/kg-day dosage groups, respectively, doses above the maximum tolerated dose.

The short-term NTP (1978) preliminary dose-range finding study in male and female Osborne-Mendel rats gavaged with 0, 40, 63, 100, 150 or 251 mg/kg-bw/day of 1,2-dichloroethane for 5 days/week for 6 weeks suggested body weight effects during exposure; however, due to the lack of quantitative data provided in the study report, a thorough evaluation of the data could not be performed.

Inhalation

The inhalation study by Schwetz et al. (1974) that exposed nonpregnant female rats for 7 hours/day for 10 days or pregnant rats on GD 6 to15 to 1,1-dichloroethane at concentrations of 0, 3,800 or 6,000 ppm identified decreased maternal food consumption and maternal body weight gains in rats treated with 3,800 and 6,000 ppm 1,1-dichloroethane as compared to pooled controls. A key limitation of the study was that the treatment concentrations were not conducted within the same experiment but rather two separate experiments with an unknown time between them. Additionally, control data from the two experiments were pooled for all endpoints except one which showed a difference among control groups and the incidence of a specific skeletal variation.

In the study by Mellon Institute (1947), a single mongrel dog was exposed to air or 1,000 ppm (1,067 ppm analytically) of 1,1-dichloroethane 7 hours/day, every other day for 6 months. Reporting for this study is very limited, but it appeared that there was a significant deviation in weight gain of the treated dog of 1.31 kg as compared to the control that showed a 3.66 kg gain at the end of the exposure duration. Additionally, in the study by Mellon Institute (1947), a single mongrel dog was exposed to air or 200 ppm (243 ppm analytically) of 1,2-dichloroethane 7 hours/day, every other day for 6 months. Reporting for this study is very limited, but it appeared that there was a significant decrease in the weight gain of the treated dog as compared to the control.

Mechanistic

EPA did not identify mechanistic studies that evaluated any potential nutritional/metabolic hazards for 1,1- or 1,2-dichloroethane.

Evidence Integration Summary
There were no human epidemiological nor mechanistic studies available for either 1,1- or 1,2-dichloroethane and therefore, there is *indeterminate* human evidence and mechanistic support to assess whether 1,1-dichloroethane or 1,2-dichloroethane may cause nutritional/metabolic changes in humans.

An evaluation of 1,1-dichloroethane animal studies identified an induction of body weight decrements in rats at high gavage exposures (≥2,000 mg/kg-bw/day) and in one dog exposed by inhalation (1,067 ppm). No body weight effects were seen; however, in mice or in rats at lower exposure levels. Thus, the evidence for nutritional/metabolic effects due to 1,1-dichloroethane is considered *moderate.*

The evidence is considered *slight* for animal studies for 1,2-dichloroethane based on decreased body weight as reported in mice and guinea pigs exposed by inhalation and rats and mice exposed orally to 1,2-dichloroethane in high- and medium-quality studies. Several high- and medium-quality studies in a few species via various routes of exposure also reported no effect on body weight, sometimes at lower exposure levels and/or shorter exposure durations to 1,2-dichloroethane.

Overall, EPA concluded that evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes body weight decrements under relevant exposure circumstances. EPA also concluded that the evidence suggests, that 1,2-dichloroethane may cause nutritional/ metabolic effects under relevant exposure conditions.

5.2.3.1.4 Mortality

Humans
EPA did not identify epidemiological studies that evaluated any potential mortality hazards for 1,1-dichloroethane. EPA identified two limited retrospective cohort studies that found no increase in mortality of workers from either petrochemical or herbicide manufacturing plants with presumed exposure to 1,2-dichloroethane relative to the general U.S. population (BASF, 2005; Teta et al., 1991).

Laboratory Animals
A review of acute, intermediate, and chronic studies identified studies that indicated mortality following 1,1-dichloroethane exposure and studies were also identified that demonstrate mortality following 1,2-dichloroethane exposure.

Oral
In Dow Chemical (1947), a study in guinea pigs of unspecified stain, sex, or number/group dosed with either a single dose of 300 or 1,000 mg/kg-bw of 1,1-dichloroethane resulted in a survival of all animals at the lower dose but mortality of all animals at the higher dose. It is unclear as to the method animals were fed or if a vehicle control was used due to limited details provided in the study report.

In the acute Muralidhara et al. (2001) single dose oral gavage study, male Sprague-Dawley rats were administered a single dose of 0, 1,000, 2,000, 4,000, 8,000, 12,000, or 16,000 mg/kg bw and observed for 2 weeks. Mortality was increased in a dose-dependent manner at concentrations ≥4000 mg/kg-bw.

In the short-term Muralidhara et al. (2001) 10-day oral gavage study, male Sprague-Dawley rats, administered 1,1-dichloroethane at a dose of 0, 1,000, 2,000, 4,000 or 8,000 mg/kg-bw/day resulted in all rats at the 8000 mg/kg-bw/day dose died within 24 hours of dosing.

In the subchronic study by Muralidhara et al. (2001), male Sprague-Dawley rats, administered 1,1-dichloroethane via oral gavage for 5 days/week for 13 weeks at a dose of 0, 500, 1,000, 2,000, or 4,000

mg/kg-bw/day resulted in 1/15 animals dying in the 2000 mg/kg bw dose group and 8/15 animals dying in the 4,000 mg/kg bw dose group, which resulted in early termination of the highest dose group at 11 weeks.

The short-term 10-day oral gavage study in male Wistar rats by van Esch et al. (1977) dosed at 0, 3, 10, 30, 100, or 300 mg/kg-bw/day 1,2-dichloroethane resulted in death of all animals in the 300 mg/kg-bw/day exposure group.

In the 10-day short-term oral gavage Daniel 1994) study, male and female Sprague-Dawley rats were treated with 0, 10, 30, 100, or 300 mg/kg-bw/day of 1,2-dichloroethane. In this study, 10 females and 8 males died at the highest dose concentration of 300 mg/kg-bw/day precluding statistical comparison of this group with controls, with no deaths occurring in the other treatment groups.

In a study by Payan et al. (1995), Sprague-Dawley dams were administered 1,2-dichloroethane by gavage at doses of 0, 1.2, 1.6, 2.0, and 2.4 mmol/kg (corresponding to 0, 119, 158, 198, and 238 mg/kg-bw/day) during gestation day (GD) 6 to GD 21 that did not result in maternal death in any of the dosage groups.

Inhalation

In the study by Francovitch et al. (1986), male CD-1 mice treated with 1,2-dichloroethane for 4 hours via inhalation resulted in a dose-related increase in mortality beginning at a concentration of 1000 ppm (4050 mg/m^3).

Male SD rats exposed via inhalation to 1,2-dichloroethane for 7 hours/day for 5 days/weeks resulted in the occurrence of mortality starting at 304 ppm (1,230 mg/m^3) (Igwe et al., 1986a).

Female SD rats exposed to 300 ppm (1210 mg/m^3) 1,2-dichloroethane resulted in increased incidences in mortality in dams when exposed for 10 days during GD 6 to 15 (Rao et al., 1980). Additionally, in Rao et al. (1980), New Zealand white rabbits treated with 1,2-dichloroethane for 7 hours/day during the 13 days of GD 6 to 18 also showed increased incidences of maternal mortality beginning at the exposure concentration of 100 ppm (405 mg/m^3).
In the study by Payan et al. (1995), female SD rats treated with 1,2-dichloroethnae resulted in increased incidence of maternal death at a LOAEL of 329 ppm (1,330 mg/m^3).

Mechanistic

EPA did not identify mechanistic studies that evaluated any potential mortality hazards for 1,1-or 1,2-dichloroethane.

Evidence Integration Summary

There were no human epidemiological nor mechanistic studies available for 1,1-dichloroethane and therefore, there is *indeterminate* human evidence and mechanistic support to assess whether 1,1-dichloroethane may cause mortality in humans. Limited epidemiological data show no increase in mortality among workers with presumed exposure to 1,2-dichloroethane but are insufficient to draw any broader conclusions. Therefore, there is *indeterminate* human evidence to assess whether 1,2-dichloroethane may cause mortality in humans. There were no mechanistic studies available for 1,2-dichloroethane and therefore, there is *indeterminate* mechanistic support to assess whether 1,2-dichloroethane may cause mortality in humans.

The evidence in laboratory animals is *robust* based on an evaluation of studies that identified the occurrence of mortalities in several species of animal exposed to 1,1-dichloroethane (≥1000 mg/kg-bw) via gavage in high-quality studies. Evidence was also considered *robust* with regard to animal studies of 1,2-dichloroethane as treatment-related increases in the incidence of mortality were observed in several animal species exposed to 1,2-dichloroethane via inhalation, oral, or dermal exposure for acute, intermediate, or chronic durations in multiple studies.

Overall, EPA concluded that the evidence indicates that 1,1-dichloroethane exposure is likely to cause death under relevant exposure circumstances in animals and the evidence also indicates that 1,2-dichloroethane may cause death under exposures levels higher than those where other health effects were identified with lethality observed in animal studies of varying species. Due to epidemiological studies not identifying nor characterizing death in humans exposed to known concentrations of 1,1- or 1,2-dichloroethane, the Agency was unable to conclude if death is a health concern under relevant exposure levels.

5.2.3.1.5 Other Effects

In the short-term Muralidhara et al. (2001) 10-day single oral gavage study, male Sprague-Dawley rats, administered 1,1-dichloroethane at a dose of 0, 1000, 2,000, 4,000 or 8,000 mg/kg-bw/day gross morphological changes and chemically associated lesions as evaluated by H&E-stained sections were not identified in the adrenals of dosed rats.

In the subchronic study by Muralidhara et al. (2001), male Sprague-Dawley rats, administered 1,1-dichloroethane via oral gavage for 5 days/week for 13 weeks at a dose of 0, 500, 1,000, 2,000, or 4,000 mg/kg-bw/day histopathological evaluation of stomach sections was considered normal, though details of this evaluation were limited. Additionally, histopathological evaluation of adrenal sections was considered normal, though details of this evaluation were limited.

In the oral gavage study by Ghanayem et al. (1986) male Fisher 344 rats were administered 1,1-dichloroethane at 0, 350 or 700 mg/kg-bw for 5 days/week for 2 weeks. Twenty-four hours following the final dose, animals were euthanized and evaluated histopathologically for forestomach lesions. The incidences of forestomach cell proliferation and hyperkeratosis did not occur in a greater frequency as compared to controls at either administered concentration. Due to limited endpoint evaluations, this study was not considered further for dose-response.

In the 10-day short-term oral gavage Daniel et al. (1994) study, male and female Sprague-Dawley rats were treated with 0, 10, 30, 100, or 300 mg/kg-bw/day of 1,2-dichloroethane. In this study, 10 females and 8 males died at the highest dose concentration of 300 mg/kg-bw/day precluding statistical comparison of this group with controls, with no deaths occurring in the other treatment groups. Histopathological evaluation of other animals in this group of the study was not performed. In the remaining dosage groups, inflammation of the mucosal and submucosal layer of the forestomach with minimal severity were identified in the 100 mg/kg-bw/day group in both males and females with an occurrence of 60 percent, although data was not provided in the study report.

5.2.4 Genotoxicity Hazard Identification and Evidence Integration

Genotoxicity hazard identification and evidence integration for 1,1-dichloroethane and the identified analog 1,2-dichloroethane can be found in Appendix N.3.1 and N.3.2. Mutagenicity and genotoxicity data for 1,1-dichloroethane are very limited and consist of a small number of genotoxicity experiments. Available information shows that 1,1-dichloroethane induces DNA repair and binds to DNA in liver cells, and that it induces chromosomal aberrations and micronuclei in bone marrow. Overall, the

available data provide limited support for the genotoxicity of 1,1-dichloroethane. For more details, see Table_Apx N-25 and Table_Apx N-26 showing the results of *in vitro* and *in vivo* genotoxicity, and cell transformation assays of 1,1-dichloroethane. However, the Milman et al. (1988) study with a High systematic review rating demonstrated positive findings in the Ames assay with and without metabolic activation.

In summary, mode-of-action information pertaining specifically to tissues susceptible to tumor formation after exposure to 1,1-dichloroethane (*e.g.*, liver, mammary, blood) is limited to studies showing that 1,1-dichloroethane induces DNA repair and binds to DNA in liver cells, and that it induces chromosomal aberrations and micronuclei in bone marrow. Bacterial mutagenicity findings, however, were not consistent. These data are not sufficient to determine the mode of action for any tumor type associated with exposure to 1,1-dichloroethane. Overall, the available data provide limited support for the genotoxicity of 1,1-dichloroethane, and no information on alternative modes of carcinogenic action. Thus, the weight of scientific evidence judgement for cancer effects based on mechanistic evidence for 1,1-dichloroethane are slight.

Evidence from *in vivo* studies using multiple animal species and routes of exposure and *in vitro* studies using multiple test systems indicates that 1,2-dichloroethane and/or its metabolites can induce mutations, chromosomal aberrations, DNA damage, and DNA adducts in certain test systems. The available data show that biotransformation of 1,2-dichloroethane to reactive metabolites via a major CYP450-mediated oxidative pathway and a minor glutathione conjugation pathway contributes to the observed effects. There are species-, sex-, tissue-, and dose-related differences in the interactions between 1,2-dichloroethane and/or its metabolites and DNA. For more details, see Appendix N.3 that provides a summary of the studies identified for *in vitro* and *in vivo* genotoxicity, and cell transformation assays of 1,2-dichloroethane.

Overall, evidence for 1,2-dichloroethane has shown its ability to induce mutations, cause clastogenic effects, result in DNA damage, and have the capability for DNA binding/adduct formation *in vitro* and *in vivo*. The preponderance of the substantial database consists of positive results. Although these effects could plausibly be related to formation of tumors, a direct connection between these events and 1,2 dichloroethane induced carcinogenesis has not been conclusively demonstrated. Few mechanistic data examining alternative modes of carcinogenic action are available. Thus, the overall weight of scientific evidence judgement for cancer effects based on mechanistic evidence for 1,2-dichloroethane are moderate.

5.2.5 Cancer Hazard Identification, Mode of Action (MOA) Summary, and Evidence Integration

5.2.5.1 Cancer Hazard Identification and Evidence Integration

Appendix N.3 provides hazard identification and evidence integration for cancer for 1,1-dichloroethane and the identified analog 1,2-dichloroethane.

5.2.5.1.1 Human Evidence

Human Evidence for 1,1-Dichloroethane

EPA did not locate any human epidemiology studies for 1,1-dichloroethane that could be utilized for a cancer dose response analysis, and the overall 1,1-dichloroethane cancer epidemiology literature is considered indeterminate. A study of ambient air concentration estimates of 1,1-dichloroethane and breast cancer in women in the United States did not find significantly increased risk in the upper four quintiles of exposure when compared individually to the first quintile, nor did the study find

significantly increased risk when the case definition of breast cancer only included those tumors that were estrogen-receptor positive (Niehoff et al., 2019). An additional study, Garcia et al. (2015) investigated cancer risk based on female teachers in California's exposure to ambient air concentrations of 1,1-dichloroethane broken into quintiles, and also generally did not provide adequate evidence of carcinogenicity. The study did not find evidence of increased risk of breast cancer in the upper four quintiles of exposure when compared individually to the first quintile in the full study population, but did find limited increased risk for breast cancer when defining cases of breast cancer as those with tumors that were either estrogen-receptor positive or progesterone-receptor positive (ER+/PR+), and when defining cases of breast cancer as only those cases that were not currently using hormone therapy. However, this increased risk was only observed in quintiles three and four of exposure but not quintile five for the ER+/PR+ case definition subset, and only observed in quintile three of exposure but not quintiles four or five for the subset not currently using hormone therapy. Therefore, the evidence of 1,1-dichloroethane carcinogenicity from the human study data is inadequate to draw definitive conclusions.

Human Evidence for 1,2-Dichloroethane

The 1,2-dichloroethane human epidemiology literature is similarly indeterminate as to whether 1,2-dichloroethane exposure causes cancer due to a lack of published studies. A few studies showed significant relationships between 1,2-dichloroethane and certain types of cancers; however, these relationships existed in very specific subgroups and were not consistent across exposure groups, which limits EPA's ability to draw conclusions from their results. For example, although Niehoff et al. (2019) found a slight increase in the risk for ER+ invasive breast cancer in the fourth quintile of exposure as compared with the first, this relationship was not significant in the fifth quintile of exposure as compared with the first. This study also did not find a significant relationship between 1,2-dichloroethane exposure and overall incidence of breast cancer, which was consistent with the only other study investigating this relationship (Garcia et al., 2015). Similarly, 1,2-dichloroethane exposure was associated with a borderline significant increase in pancreatic cancer, but only among Black females with low estimated exposure intensity (and not medium or high exposure intensity) (Kernan et al., 1999). Studies of brain cancer and kidney cancer showed no significant relationship with 1,2-dichloroethane exposure (Dosemeci et al., 1999; Austin and Schnatter, 1983).

Another study observed higher incidence of all-cause cancer than was expected in a cohort of workers when compared to the general population, but the statistical significance of this result was not reported, and the significance of all-cause cancer is not clear (BASF, 2005). This same study looked at many specific cancer standardized incidence ratios (SIRs) as well, but none were statistically significantly elevated except for prostate cancer, which no other studies in the literature reported observing. Sobel et al. (1987) did not show a statistically significant relationship between 1,2-dichloroethane exposure and soft-tissue sarcoma, but also had very low statistical power with a sample size of seven 1,2-dichloroethane exposed participants. In general, more studies would be needed to draw conclusions about the weight of evidence for the relationship between 1,2-dichloroethane exposure and cancer from the epidemiologic literature, and none of the existing studies measured exposure in a way that could be used to estimate a quantitative dose-response relationship.

5.2.5.1.2 Animal Evidence

Animal Evidence for 1,1-Dichloroethane

The NCI (1978) cancer study on 1,1-dichloroethane in Osborne-Mendel rats provides limited evidence of the carcinogenicity based on significant dose-related increases in the incidence of hemangiosarcomas at various sites and mammary carcinomas in female rats, neither of which were observed in male rats. However, the high incidence of pneumonia and deaths in all groups prevented the use of the data for calculation of oral slope factors. Technical grade 1,1-dichloroethane in corn oil was administered by

gavage 5 days/week for 78 weeks to groups of rats/sex/dose. In male rats, survival at 111 weeks was low at 30, 5, 4, and 8 percent (untreated control, the vehicle control, the low-dose, and the high- dose groups, respectively). In female rat groups survival was also low at 40, 20, 16, and 18 percent (untreated control, vehicle control, low- and high-dose groups, respectively). For hemangiosarcomas, the incidence in female rats there was a statistically significant positive dose-related trend at 0/19 for matched vehicle controls, 0/50 for the low-dose group, and 4/50 for the high-dose group. In the study, a Cochran-Armitage test indicated a significant positive association between dosage and the incidence of hemangiosarcoma when comparing both to the matched vehicle control ($p = 0.041$) and to the pooled vehicle control ($p = 0.021$). The Fisher exact tests, however, did not detect any significant differences between groups. In female rats, the incidence of mammary gland adenocarcinomas was 1/20 for the untreated group, 0/19 for the vehicle control group, 1/50 for low-dose, and 5/50 for high-dose groups, which showed a statistically significant dose-related positive trend in rats surviving at least 52 weeks.

The (NCI, 1978) cancer study on 1,1-dichloroethane in B6C3F1 mice revealed a statistically significant increase in benign uterine endometrial stromal polyps (4/46) in high-dose females, which were not observed in any other group. However, pre-cancerous endometrial polyps are not a tissue growth amenable to calculate cancer slope factors. In the study, groups of 50 B6C3F1 mice/sex/group were administered technical grade 1,1-dichloroethane in corn oil by gavage 5 days/week for 70 weeks with 20 mice/sex/group in the control groups. In female mice, survival at termination was 80, 80, 80, and 50 percent for the untreated control group, the vehicle control group, the low-, and high-dose groups, respectively. Survival in male mice was 35, 55, 62, and 32 percent in the untreated control group, the vehicle control group, the low-, and high-dose groups, respectively. Liver carcinomas were reported in only the vehicle control (1/19) and the low-dose groups (1/47) in female mice, no liver tumors were seen in the untreated controls or in the high-dose group. The incidence of hepatocellular carcinomas in male mice surviving at least 52 weeks was 1/19, 6/72, 8/48, and 8/32 in the matched vehicle control group with a statistically significant trend test, a pooled vehicle control group consisting of mice from this group and identical controls from other concurrent experiments, and the low-, and high- dose groups, respectively. However, an increased incidence of hepatocellular carcinoma in male mice was not statistically significant by either pairwise or trend test at 2/17 in the untreated control group, 1/19 in the vehicle control group, 8/49 in the low-dose, and 8/47 in the high-dose groups.

NIOSH considers the chloroethanes: ethylene dichloride (1,2-dichloroethane); hexachloroethane; 1,1,2,2-tetrachloroethane; and 1,1,2-trichloroethane; to be potential occupational carcinogens. Additionally, NIOSH recommends that the other five chloroethane compounds—1,1-dichloroethane, ethyl chloride, methyl chloroform, pentachloroethane, and 1,1,1,2-tetrachloroethane—be treated in the workplace with caution because of their structural similarity to the four chloroethanes shown to be carcinogenic in animals. In an EPA cancer study on the 1,1-dichloroethane metabolite dichloroacetic acid, it was concluded to cause liver cancer in as little as 4 weeks of exposure Wehmas et al. (2017).

Because the cancer studies for 1,1-dichloroethane were not usable for the cancer assessment nor was a study identified via the inhalation and dermal routes of exposure, the cancer data for the identified analog 1,2-dichloroethane was evaluated and an evidence integration of available data for 1,2-dichloroethane as presented in Appendix N.9 provides a more comprehensive evaluation of tumor types in multiple organ systems. There was no reliable cancer study via the inhalation route for 1,1-dichloroethane, so the cancer data for 1,2-dichloroethane was utilized for the inhalation route. For the oral route, cancer data for 1,1-and 1,2-dichloroethane were confounded by mortality and/or disease (pneumonia) and thus precluded there use for derivation of the oral cancer slope factor. The 1,2-dichloroethane inhalation cancer study from Nagano et al. (2006) did produce some of the same tumors as observed in the 1,2-dichloroethane oral cancer study. The highest estimated inhalation unit risk (IUR)

is 7.1×10^{-6} (per $\mu g/m^3$) for combined mammary gland adenomas, fibroadenomas, and adenocarcinomas and subcutaneous fibromas in female rats in the inhalation study by Nagano et al. (2006). Based on a qualitative comparison of 1,1 and 1,2-dichloroethane tumor types associated with the oral route, EPA is confident that cancer studies identified for 1,2-dichloroethane would be representative of those seen in 1,1-dichloroethane if data were available.

Animal Evidence for 1,2-Dichloroethane

The NTP (1978) cancer study for 1,2-dichloroethane in Osborne-Mendel rats and B6C3F1 mice provides evidence of the carcinogenicity treated by oral gavage for 78 weeks. Male rats had significantly increased incidence of forestomach squamous-cell carcinomas and circulatory system hemangiosarcomas. Significant increases in mammary adenocarcinoma incidence in female rats and mice were observed. Alveolar/bronchiolar adenomas developed in mice of both sexes and females developed endometrial stromal polyps and sarcomas, while males developed hepatocellular carcinomas. The high incidence of death in the rat study that was attributed to high incidence of pneumonia caused it to have an uninformative rating in systematic review, so cancer slope factors were not modeled from this data set. Additionally, the NTP (1978) cancer study for 1,2-dichloroethane in mice also contained a number of limitations such as the dosage adjustments performed during the study due to acute toxicological effects within high dose group, the occurrence of pneumonia (though in a lower incidence as compared to the rats) in the mice and potential confounding as a result of other volatile chlorinated solvents treatments occurring in the same dosing space.

Carcinogenicity associated with exposure to 1,2-dichloroethane via inhalation was investigated by (Maltoni et al., 1980) in Sprague-Dawley rats and Swiss mice at concentrations of 0, 5, 10, 50, or 150-250 ppm (initial exposure to 250 ppm resulted in acute toxicity and thus reduced to 150 ppm after unspecified number of weeks) 7 hours/day, 5 days/week, for 78 weeks that did not result in increases in tumor incidences that could be attributed to 1,2-dichloroethane exposure. Although control animals were housed in a separate room from treated groups, concentration monitoring was not performed during the study thus resulting uncertainties in the actual exposures levels. Additionally, a study by (Cheever et al., 1990) exposed Sprague-Dawley rats to 50 ppm of 1,2-dichloroethane 7 hours/day 5 days/week for 2 years and did not identify increased incidence of tumor in treated animals as compared to control. The low concentration of exposure may have attributed to the lack of an induction in tumor within the 1,2-dichloroethane treatment group.

In contrast to prior inhalation studies that sought to evaluate the carcinogenicity of 1,2-dichloroethane via inhalation, Nagano et al. (2006) treated F344 rats and BDF1 mice at concentrations of 0,10, 40, or 160 ppm or 0, 10, 30, or 90 ppm, respectively, for 6 hours/day 5 days/week for 104 weeks. In the F344 rats, increased incidences of subcutaneous fibromas along with the occurrence of mammary gland adenomas, fibroadenomas, and adenocarcinomas were identified at 160 ppm of 1,2-dichloroethane. Additionally, increased incidences of liver hemangiosarcomas were observed in male mice in the 30 and 90 ppm treatment groups for 1,2-dichloroethane. BMD modeling of the combined tumor incidences in female rats was performed as the incidences of the mammary tumors and subcutaneous fibromas showed a significant positive trend with increased concentration and were significantly different from the control group at 160 ppm (combined mammary tumors were also significantly different from controls at 40 ppm). The incidences of mammary tumors in the control group were at incidence rates that did not exceed the maximum tumor incidences when compared to historical controls and thus retained in the modeling.

5.2.5.1.3 Mode of Action (MOA) Summary

The U.S. EPA (2005b) *Guidelines for Carcinogen Risk Assessment* defines mode of action as "a sequence of key events and processes, starting with the interaction of an agent with a cell, proceeding through operational and anatomical changes and resulting in cancer formation."

Appendix N.3 provides hazard identification and evidence integration for cancer for 1,1-dichloroethane and the identified analog 1,2-dichloroethane. A limited number of *in vitro* and *in vivo* experiments on 1,1-dichloroethane genotoxicity are available. *In vitro* experiments include two bacterial mutagenicity studies, a study of chromosomal aberrations in mammalian cells, studies of DNA repair in mouse and rat, hepatocytes studies of mammalian cell transformation, a test of chromosome malsegregation in fungi, and a study of cell-free DNA binding. *In vivo* experiments include two DNA binding assays and a bone marrow chromosomal aberration assay. The Milman et al. (1988) study demonstrated positive findings in the Ames assay with and without metabolic activation. Immunotoxicity was also demonstrated for the identified analog 1,2-dichloroethane (Zabrodskii et al., 2004; Munson et al., 1982). Both mutagenicity and immunosuppression are accepted mechanisms for tumorigenesis (Hilton et al., 2022); however, as 1,2-dichloroethane has been shown to induce mutations, clastogenic effects, DNA damage, and DNA binding/adduct formation *in vitro* and *in vivo*, the direct association of these events to the formation of tumors and 1,2 dichloroethane-induced carcinogenesis has not been conclusively demonstrated. Few mechanistic data examining alternative modes of carcinogenic action are available as is the case with the limited immunotoxicity available to suggest immunotoxicity as an alternative mode of action.

Overall MOA Conclusions

Animal studies provide limited evidence that 1,1-dichloroethane may cause cancer in rodents. Rats and mice exposed via gavage for 78 weeks exhibited a positive dose-related trend in the incidence of liver tumors in male mice as well as mammary gland tumors and hemangiosarcomas in female rats. Poor survival in both control and treated rats limits the validity of these results. The mouse cancer study indicated that 1,1-dichloroethane produced pre-cancerous endometrial polyps. Cancer mode-of-action data for 1,1-dichloroethane are limited and consist of a small number of genotoxicity experiments. The Milman initiation-promotion study in rats indicated that 1,1-dichloroethane is a liver tumor promotor when dosed at 700 mg/kg/day for 7 weeks and it was positive in the Ames assay with and without metabolic activation (Milman et al., 1988).

In summary, MOA information pertaining specifically to tissues susceptible to tumor formation after exposure to 1,1-dichloroethane (*e.g.,* liver, mammary, blood) is limited to studies showing that 1,1-dichloroethane induces DNA repair and binds to DNA in liver cells, and that it induces chromosomal aberrations and micronuclei in bone marrow. These data are not sufficient to determine the mode of action for any tumor type associated with exposure to 1,1-dichloroethane as bacterial mutagenicity findings were not consistent. Alkyl halides such as 1,1-dichloroethane are known to be DNA alkylating agents; however, the available data *in vivo* suggests that adducts formed should be considered biomarkers of exposure, rather than mutagenic adducts. Overall, the available data provide limited support for the genotoxicity or mutagenicity of 1,1-dichloroethane or immunosuppression as an alternative mode of carcinogenic action.

5.2.5.1.4 Weight of Scientific Evidence

Weight of Scientific Evidence Conclusions

There are no human epidemiology studies that were amenable to dose-response analysis; however, studies in rats and mice were available for 1,1-dichloroethane and its analog 1,2-dichloroethane.

Chronic cancer studies performed by NCI (1978) on 1,1-dichloroethane qualitatively resulted in the same tumor types or pre-cancerous lesions as seen in the bioassays of the similar isomer 1,2-dichloroethane (*i.e.*, hepatocellular carcinomas, endometrial polyps, hemangiosarcomas, etc). However, the rat studies for both chemicals were not utilized for cancer slope factor derivation due to the excessive animal deaths and pre-cancerous endometrial polyps in mice for 1,1-dichloroethane are not considered for cancer slope factor analysis.

Additionally, U.S. EPA (1990) previously designated the cancer classification of 1,1-dichloroethane as a Group C, a possible human carcinogen, based on no human data and limited evidence of carcinogenicity in two animal species (rats and mice) as shown by an increased incidence of mammary gland adenocarcinomas and hemangiosarcomas in female rats and an increased incidence of hepatocellular carcinomas and benign uterine polyps in mice (NCI, 1978). As a comparison, NTP (1978) identified many of the same tumor types as seen in the bioassays of 1,1-dichloroethane. These included significant increases in the incidences of forestomach squamous cell carcinomas and hemangiosarcomas in male rats and an increased incidence of mammary adenocarcinomas in both female rats and mice. In addition, alveolar and bronchiolar adenomas were reported in male and female mice; endometrial stromal polyps and sarcomas in female mice; and hepatocellular carcinomas in male mice. In context, the oral slope factor for rats for 1,2-dichloroethane was calculated as 9.1×10^{-2} mg/kg-day based on hemangiosarcomas in rats. As indicated previously, due to confounding associated with disease occurrence and mortality neither study was used for derivation of the oral slope factor.

Based on the common tumor types observed between 1,1- and 1,2-dichloroethane and a lack of inhalation bioassay for 1,1-dichloroethane, EPA is confident that the inhalation bioassay by Nagano (2006) for 1,2-dichloroethane provides a reliable IUR value for the risk evaluation. Considering that 1,2-dichloroethane is categorized as a Group B2 (probable human carcinogen) by U.S. EPA (NCEA, 1987), identified as "reasonably anticipated to be a human carcinogen" by the National Toxicology Program (NTP, 2021), and being considered a more potent carcinogen than 1,1-dichloroethane by OncoLogic based on the greater reactivity of a vicinal dihalide (1,2-dichloroethane) than a geminal dihalide (1,1-dichloroethane), utilizing an oral slope factor and IUR value from 1,2-dichloroethane for the 1,1-dichloroethane risk evaluation is considered to be human health protective.

5.2.6 Dose-Response Assessment

According to the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a), hazard endpoints that receive evidence integration judgments of *demonstrates* and *likely* are considered for dose-response analysis. Endpoints with *suggestive* evidence can be considered on a case-by-case basis. Studies that received high or medium overall quality determinations (or low-quality studies if no other data are available) with adequate quantitative information and sufficient sensitivity can be compared.

The only hazard outcome category for which evidence *demonstrates* or is *likely* for 1,1-dichloroethane to cause the effect in humans was for mortality. Therefore, hazard outcomes that received *suggestive* judgements would then be the most robust evidence integration decisions in the case of 1,1-dichloroethane. These evidence, however, were identified as suggestive but not conclusive or inadequate regarding 1,1-dicholoethane. Due to the limitations associated with a lack of available studies to generate a robust weight of evidence specifically based on 1,1-dichloroethane data, EPA integrated data from both 1,1-dichloroethane and the identified analog 1,2-dichloroethane to provide a more adequate weight of evidence evaluation of comprehensive toxicological endpoints. As the health effect with the most robust and sensitive POD among these *suggestive* outcomes were derived from 1,2-dichloroethane, these data were used for risk characterization for each exposure scenario to be protective of other adverse effects as described in the sections below.

Data for the dose-response assessment were selected from oral and inhalation toxicity studies in animals specifically from 1,2-dichloroethane. Additionally, no usable PBPK models are available to extrapolate between animal and human doses or between routes of exposure using 1,1- or 1,2-dichloroethane-specific information. Two PBPK models were developed to describe the disposition of 1,2-dichloroethane. There are reliable inhalation studies for POD selections, so the PBPK models are not required to convert an oral dose for the inhalation route. The PBPK models are not needed, the standard RfC processes are reliable to calculate the HEC values from animal data. The D'Souza et al. (1988) model used five compartments (lung, liver, richly perfused tissues, slowly perfused tissues, and fat). Sweeney et al. (2008) extended and updated the D'Souza et al. (1988) model by adding two gastrointestinal compartments, a compartment for the kidney, and an additional metabolism pathway for extrahepatic enzymes. Reliable human data is needed to calibrate the PBPK model. Because the model has not been validated in humans, it is unclear whether this model would be useful for extrapolating between rats and humans.

The PODs estimated based on effects in animals were converted to HEDs or CSFs for the oral and dermal routes and HECs or IURs for the inhalation route. For this conversion, EPA used guidance from U.S. EPA (2011b) to allometrically scale oral data between animals and humans. Although the guidance is specific for the oral route, EPA used the same HEDs and CSFs for the dermal route of exposure as the oral route because the extrapolation from oral to dermal routes is done using the human oral doses, which do not need to be scaled across species. EPA accounts for dermal absorption in the dermal exposure estimates, which can then be directly compared to the dermal HEDs.

For consistency, all HEDs and the CSF are expressed as daily doses and all HECs are based on daily, continuous concentrations (24 hours per day) using a breathing rate for individuals at rest. Adjustments to exposure durations, exposure frequencies, and breathing rates are made in the exposure estimates used to calculate risks for individual exposure scenarios.

The endpoints of concern for 1,1-dichloroethane (based on read across from 1,2-dichloroethane includes renal/kidney, olfactory, reproductive effects and cancer. These data were used for risk characterization for each exposure scenario to be protective of other adverse effects as described in the sections below. The health effects identified as suggestive and evaluated for dose response were renal, olfactory and reproductive/developmental.

5.2.6.1 Selection of Studies and Endpoints for Non-Cancer Toxicity

The following subsections provide a description of the selection of critical non-cancer PODs for acute, intermediate and chronic exposures for 1,1-dichloroethane (using data for the analog 1,2-dichloroethane to fill data gaps). The sections provide a summary of the evaluation of the possible PODs and the rationale for selection of the critical study (and POD) in a series of tables. The tables are intended to streamline the text of this RE. Appendix N.2 provides the details of the non-cancer dose response assessment for 1,1-dichloroethane and the analog 1,2-dichloroethane.

For the 1,1-dichloroethane risk evaluation, all data considered for PODs are obtained from animal toxicity studies in rats or mice. EPA used dichotomous models to fit quantal data (*e.g.*, incidences of tumors) and continuous models to fit continuous data (*e.g.*, body and organ weights), as recommended by EPA's BMD Technical Guidance (U.S. EPA, 2012b). The BMDs/BMDLs (benchmark doses lower 95% CL) are provided based on a daily exposure (*i.e.*, 7 days per week) for easier comparison across all hazard endpoints and thus, doses were adjusted as needed before BMD modeling. EPA modeled endpoints that had statistically significant pairwise comparisons between individual doses and controls

or significant dose-response trends. EPA also considered potential biologically significant changes from controls where possible and/or that appeared to exhibit a dose-response relationship upon visual inspection. Multiple health endpoints may have been modeled from each study, depending on the relevance of the data to adverse health outcomes and to identify sensitive health endpoints for each domain.

EPA relied on the BMD guidance and other information to choose benchmark responses (BMRs) appropriate for each endpoint. Although the BMD Technical Guidance does not recommend default BMRs, it describes how various BMD modeling results compare with NOAEL values, and the guidance does recommend calculating 10 percent extra risk (ER) for quantal data and one standard deviation (SD) for continuous data, a conversion of the study duration to a 24-hour duration, to compare modeling results across endpoints. EPA also modeled percent relative deviations (RD) for certain continuous endpoints such as a BMR for decreased sperm concentration at 5 percent, as this was considered biologically relevant. EPA's choice of BMRs for the 1,1-dichloroethane health endpoints are described in more detail in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Benchmark Dose Modeling* (U.S. EPA, 2025e) that present BMD modeling results for each health domain.

5.2.6.1.1 Uncertainty Factors Used for Non-Cancer Endpoints

For the non-cancer health effects, the Agency applied specific uncertainty factors (UF) to identify benchmark MOEs for acute, intermediate, and chronic exposure durations for each exposure route among studies used to estimate risks. EPA guidance from (U.S. EPA, 2012a, 2002b, 1993) further discuss use of UFs in human health hazard dose-response assessment. A total uncertainty factor for each POD is calculated by multiplication of each of the applied UFs. The use of uncertainty factors in risk characterization is further described in Sections 5.3.1.1 and 5.3.1.1. Other potential uncertainty factors not relevant to this assessment that EPA may consider are described in Appendix N.2.3.

1. **Interspecies Uncertainty Factor (UF_A) of 3**
   EPA uses data from oral toxicity studies in animals to derive relevant HEDs, and (U.S. EPA, 2011a) recommends allometric scaling (using the ¾ power of body weight) to account for interspecies toxicokinetics differences for oral data. When applying allometric scaling, EPA guidance recommends reducing the UF_A from 10 to 3. The remaining uncertainty is associated with interspecies differences in toxicodynamics. EPA also uses a UF_A of 3 for the inhalation HEC that accounts for dosimetric adjustment and dermal HED values as these values are derived from the oral HED.

2. **Intraspecies Uncertainty Factor (UF_H) of 10**
   EPA used a default UF_H of 10 to account for variation in sensitivity within human populations due to limited information regarding the degree to which human variability may impact the disposition of or response to 1,2-dichloroethane.

3. **Subchronic-to-Chronic Duration Uncertainty Factor (UF_S) of 10**
   EPA used a UF_S of 10 to account for extrapolating from data obtained in a study with less-than-lifetime (short-term/subchronic) exposure to lifetime (chronic) exposure. This UF_S was applied to the BMDL/BMCL from the subchronic studies identified for the intermediate oral and inhalation PODs, respectively. As chronic studies from the 1,1- and 1,2-dichloroethane databases were not identified as suitable for the chronic duration POD, due to uncertainties and limitations identified within these studies, the assumption that effects from a given compound in a subchronic study occur at a 10-fold higher concentration than in a corresponding (but absent)

chronic study was applied. This assumption is based on the U.S EPA (2002b) guidance. The lack of suitable chronic studies prevented chronic POD selection as well as prevented the estimation of a refined UF_S. The uncertainties and limitations from the chronic studies evaluated for both 1,1- and 1,2-dichloroethane are described in Section 5.2.6.1.4. The uncertainties and limitations from the chronic studies evaluated for both 1,1 and 1,2-dichloroethane are described in Section 0.

As the intermediate oral study was based on a 90-day subchronic study, the application of the UF_S of 10 for the subchronic-to-chronic duration is considered reasonable and conservative. As the intermediate inhalation study was based on a 4-week short-term study the application of the UF_S of 10 for the subchronic-to-chronic duration is considered conservative as the duration adjustment is being applied to an exposure duration that is lower than the subchronic duration the uncertainty factor is intended for and thus a lowering of the uncertainty factor does not seem justifiable.

5.2.6.1.2 Non-Cancer PODs for Acute Exposures

Oral

1,1-Dichloroethane: Only the single-dose experiment by (Muralidhara et al., 2001) was considered as a potential study adequate for evaluation of 1,1-dichloroethane toxicity and POD derivation following acute oral exposures. A NOAEL of 1,000 mg/kg-bw and a LOAEL of 2,000 mg/kg-bw were identified based on clinical signs of neurotoxicity characterized by the authors as "excitation followed by progressive motor impairment and sedation." Although the acute-duration oral data are limited, the observation of central nervous system or CNS effects is consistent with the past use of 1,1-dichloroethane as a human anesthetic (ATSDR, 2015). This study, however, was not selected for the acute POD as this dose resulted in sedation/CNS depression but the methods that evaluated this endpoint were not provided. This effect was thus not considered a sensitive endpoint as the magnitude of this effect was also not quantitatively described in the study thus necessitating the integration of studies within the 1,2-dichloroethane database to identify a more sensitive endpoint.

The data available for 1,1-dichloroethane in Muralidhara et al. (2001) were near the LD_{50} value of 8,200 mg/kg-day and were not considered appropriate for use for POD identification. For 1,2-dichloroethane, a total of four oral animal toxicity studies are available, with three studies having medium or high data quality for dose-response analysis and identification of the intermediate oral duration POD.

There were two acute-duration oral studies of 1,1-dichloroethane that were rated acceptable based on systematic review evaluation: an acute lethality study in guinea pigs by (Dow Chemical, 1947) and a single-dose lethality study in rats by (Muralidhara et al., 2001). The study by (Dow Chemical, 1947), however, reported no details on the animal strain, sex, age, or condition; number of animals tested; method of administration; or duration of follow-up. These limitations in the study preclude its use for POD derivation.

1,2-Dichloroethane: When looking within the 1,2-dichloroethane study database, a greater number of toxicological endpoints were identified. These studies were evaluated by systematic review and only 4 studies were considered for the acute oral non-cancer dose assessment. In Cheever et al. (1990), it was noted that in a preliminary study on 4-month-old Osborne-Mendel rats dosed with 150 mg/kg-bw by oral gavage of radiolabeled 1,2-dichloroethane it was identified that the ^{14}C was almost completely eliminated within 24 hours after administration. Elimination of the ^{14}C was found primarily in the urine (49.7–51.5 percent), in expired air (35.5–39.6 percent) and only a small portion in the feces as detected

as $^{14}CO_2$. This suggested that the kidneys are a potential target due to oral exposure to 1,2-dichloroethane.

In the Morel et al. (1999) acute single exposure oral gavage study in male Swiss OF1 mice treated with 0, 1,000, or 1,500 mg/kg-bw of 1,2-dichloroethane, a significant increase in damaged renal tubules (7.66 vs. 0.32% in controls) was seen only seen in the highest dose group with the lowest dose already above the limit dose. B6C3F1 mice in the Storer et al. (1984) study that were administered a single oral gavage dose at 0, 100, 200, 300, 400, 500, 600 mg/kg-bw resulted in absolute kidney weights increased at 300 mg/kg-bw doses and greater. Relative kidney weights in Storer et al. (1984) were also increased in the 300 mg/kg and higher dose groups along with serum BUN (serum BUN showed a trend increase but the 300 mg/kg/day dose was not statistically significant to control (at n = 5); however, the benchmark dose (BMD) analysis using all data points together showed significance above 106 mg/kg/day). Thus, based on both histological and clinical chemistry parameters, the Storer et al. (1984) study based on mice kidney weight was identified as the recommended candidate for the acute oral POD. To calculate risks for the acute exposure duration in the risk evaluation, EPA used a daily HED of 19.9 mg/kg-bw (based on a $BMDL_{10\%}$ of 153 mg/kg-bw) from Storer et al. (1984) and based on a significant (13 percent) increase in relative kidney weight in male B6C3F1 mice administered a single dose of 1,2-dichloroetane at 100, 200, 300, or 400 mg/kg via oral gavage in corn oil. This study was given a high overall quality determination and a UF of 30 was used for the benchmark MOE during risk characterization (Table 5-43).

Evaluation of the 1,2-dichloroethane studies also suggest the liver and respiratory system as targets of oral 1-2-dichloroethane exposure. In the Munson et al. (1982) study, an acute single oral gavage to 1-2-dichloroethane in CD-1 mice identified a LD_{50} of 413 and 489 mg/kg for female and male mice, respectively. Upon necropsy of these animals, it was identified that the lungs and liver appeared to be the primary target organs.

In support of liver toxicity, in the study by Storer et al. (1984), B6C3F1 mice were administered a single dose of 1,2-dichloroethane at 100, 200, 300, or 400 mg/kg via oral gavage in corn oil and euthanized 4 hours later. It was identified that a statistically significant increase in DNA damage in hepatic nuclei was present in all dose groups, as characterized by single-strand breaks, when compared to controls. The study by Storer et al. (1984) also indicated increased IDH (also known as sorbitol dehydrogenase, SDH) and AAT (alanine aminotransferase) serum levels were also increased at the 200 mg/kg and higher doses in the B6C3F1 mice. In Cottalasso et al. (2002), a single gavage of 628 mg/kg of 1,2-dichloroethane in female Sprague-Dawley rats resulted in increased alanine aminotransferase (ALT), aspartate aminotransferase (AST), and lactate dehydrogenase as compared to controls. Additionally, histological evaluation of the liver showed moderate steatosis. Increased malondialdehyde (MDA), a marker of lipid peroxidation, was also seen in the treated animals when compared to controls. Although clinical chemistry for liver enzyme implicates liver injury due to 1,2-dichloroethane exposure, gross pathology (changes in liver weight or quantified histological changes) was not identified.

With regard to the respiratory system, only the study by Salovsky et al. (2002), a single oral dose of 136 mg/kg-bw 1,2-dichloroethane in male Wistar rats resulted in increased total number of cells in the bronchioalveolar lavage fluid (BALF) of male Wister rats at 30 days after dosing. Histological changes were only presented qualitatively. Thus, this study was not identified as the POD due to limited data that was quantitative.

The POD for the acute oral exposure route was thus based on renal toxicity, specifically increased relative kidney weight at a $BMDL_{10}$ of 153 based on Storer et al. (1984) with a benchmark MOE of 30.

The HED for the occupational (worker) scenario was then calculated as 19.9 mg/kg while the HED for the general population (continuous) scenario was calculated as 19.9 mg/kg.

Table 5-37 and Figure 5-7 show the recommended acute oral study and POD (in consideration of both 1,1-dichloroethane and 1,2-dichloroethane toxicity data) followed by co-critical endpoints (PODs within the range of the recommended study) and other studies considered in support of the recommended POD presented within exposure response arrays.

Inhalation

No acute PODs were identified from studies for inhalation exposures to 1,1-dichloroethane. The 10-day inhalation study by Schwetz et al. (1974) was not used because the effects on developing fetuses and/or offspring are limited and inconclusive and were considered inadequate for derivation of an acute inhalation POD, and because the only effect reported were decreases in maternal body weight which occurred following 10-days of exposure. Likewise, a route-to-route extrapolation from the acute Storer et al. (1984) oral study was not conducted given the differences in absorption rates across routes, method of dosing effects on blood levels and hazards (*i.e.*, gavage bolus dose vs. slower inhalation dosing), the lack of a PBPK model, and the inherent uncertainties when performing oral-to-inhalation route extrapolations for a volatile solvent (*i.e.*, most of the oral dose is eliminated in expired air). Therefore, there is inadequate data to identify an inhalation POD for the acute duration scenario. An 8-hour inhalation study in male and female rats exposed to 1,2-dichloroethane by Dow Chemical (2006) was used based on read-across to 1,1-dichloroethane. A $BMCL_{10}$ of 48.9 mg/m^3 and BMD of 81.4 mg/m^3 were identified based on degeneration with necrosis of the olfactory mucosa. The acute inhalation HEC for occupational and continuous exposure of 10.14 ppm (41.1 mg/m^3) and 2.42 ppm (9.78 mg/m^3), respectively, with a benchmark MOE of 30, was used for risk assessment of acute inhalation exposure (Table 5-43). The resulting RGDR value of 0.2 is the combined value for male (0.25) and female (0.16) F344 rats used to calculate HEC continuous (U.S. EPA, 2012a).

The POD for the acute inhalation exposure route was thus based on olfactory effects, specifically nasal necrosis at a $BMCL_{10}$ of 48.9 mg/m^3 based on Dow Chemical (2006) with a benchmark MOE of 30. The HEC for the occupational (worker) scenario was then calculated as 10.14 ppm while the HEC for the general population (continuous) scenario was calculated as 2.42 ppm.

Table 5-38 and Figure 5-8 show the recommended acute inhalation study and POD (in consideration of both 1,1-dichloroethane and 1,2-dichloroethane toxicity data) followed by co-critical endpoints (PODs within the range of the recommended study) and other studies considered in support of the recommended POD presented within exposure response arrays.

Dermal

No acute exposure studies on 1,1- or 1,2-dichloroethane via the dermal route were identified. Therefore, the acute oral HED of 19.9 mg/kg-bw/day was extrapolated for the dermal route, with a benchmark MOE of 30, and was used for risk assessment of acute dermal exposures (Table 5-43).

Table 5-37. Acute Oral Non-Cancer POD-Endpoint Selection Table

Chemical/Endpoint	POD (mg/kg/day)	Study Parameters	Comments
POD selected for risk evaluation of non-cancer for acute oral exposures			
1,2-Dichloroethane, kidney weight	BMDL = 153 BMD = 270 NOAEL = 200 mg/kg; LOAEL = 300 mg/kg	Storer et al. (1984), Gavage, SR High B6C3F1 Mice – Male: Single exposure: 0, 200, 300, 400, 500, or 600 mg/kg	Single exposure study with a POD dose virtually identical to the POD dose where resorptions were observed. This POD is protective for other endpoints such as narcosis, BUN, IDH, resorptions, etc. Death started at 400 mg/kg; LD_{50} (males) = 450 mg/kg).
Co-critical studies			
1,2-Dichloroethane, blood urea nitrogen (BUN)	NOAEL = 200 LOAEL = 300	Storer et al. (1984), Gavage, SR High B6C3F1 Mice – Male: Single exposure: 0, 200, 300, 400, 500, or 600 mg/kg	Adverse increase in BUN supporting kidney effects, not statistically significant due to low sample size (n = 5). The BMD_{10} for BUN was 55, which is far lower than the BUN NOAEL value of 200 mg/kg; thus, the BMD_{10} value is not representative of the BUN data. Also, none of the models derived goodness-of-fit p-values for the means.
1,2-Dichloroethane, L-iditol dehydrogenase (IDH)	NOAEL = 200 LOAEL = 300	Storer et al. (1984), Gavage, SR High B6C3F1 Mice – Male: Single exposure: 0, 200, 300, 400, 500, or 600 mg/kg	Nine-fold adverse increase in IDH marker of tissue damage (associated mostly with kidney and liver damage), not statistically significant due to low sample size (n = 5). Neither the constant nor nonconstant variance models provided adequate fit to the variance data. No model selected.
Other studies/endpoints considered			
1,1-Dichloroethane, CNS depression/sedation	NOAEL = 1,000 LOAEL = 2,000	Muralidhara et al. (2001), Gavage, SR Medium SD Rats – Male: Single exposure: 0, 1,000, 2,000, 4,000, or 8,000 mg/kg	1,2-Dichloroethane oral LD_{50} is 725 mg/kg (PubChem), so POD too near lethal doses. Narcosis is not a sensitive endpoint in the database. This is the only 1,2-dichloroethane study that passed SR with an acute oral POD.
1,2-Dichloroethane, kidney histopathology	NOAEL = 1,000 LOAEL = 1,500	Morel et al. (1999), Gavage, SR High Swiss OF1 Mice – Male: 0, 1,000, 1,500 mg/kg	Significant increase in damaged renal tubules but lowest dose above the limit dose.
1,2-Dichloroethane, liver weight	LOAEL = 625	Moody et al. (1981), Gavage, SR Medium SD Rats – Male: Single exposure: 0, 625 mg/kg	Increased liver weight. Dose is not a sensitive POD.
1,2-Dichloroethane, liver clinical chemistry	NOAEL = 134	Kitchin et al. (1993), Gavage, SR High SD Rats – Female:	No effects reported. Inadequate dosing (too low).

Chemical/Endpoint	POD (mg/kg/day)	Study Parameters	Comments
		Single exposure: 0, 134 mg/kg	
1,2-Dichloroethane, fetal resorptions	NOAEL = 160 LOAEL = 200 (data not amenable for BMD modeling)	Payan et al. (1995), Gavage Pre-Natal Developmental, SR High SD Rats – Female: Dosing GD6–20: 0, 120, 160, 200, or 240 mg/kg	The increases in non-implants and resorptions are difficult to interpret given the significant maternal toxicity at corresponding doses (30 and 49% at 200 and 240 mg/kg/day, respectively) consisting of decreases in maternal bw gain, and the fact that there was no effect on the number of live fetuses per litter despite the changes in non-surviving implants/litter and resorption sites/litter. Therefore, cannot be used as POD.

Table 5-38. Acute Inhalation Non-Cancer POD-Endpoint Selection Table

Chemical/ Endpoint	POD (mg/m^3)	Study Parameters	Comments
POD selected for non-cancer risk evaluation for acute inhalation exposures			
1,2-Dichloroethane, respiratory	$BMCL_{10}$ = 48.9 mg/m^3 or 12.1 ppm NOAEC = 202 LOAEC = 405	Dow Chemical (2006), SR High F344 Rats – Male: 8 hours/day 1 days: 0, 50, 100, 150, 200, 600, 2,000 ppm; 0, 202, 405, 607, 809, 2,428, 8,095 mg/m^3	Degeneration with necrosis of the olfactory neuroepithelial mucosa.
Co-critical endpoints			
1,2-Dichloroethane, reproductive toxicity/fetal development	Reproductive/ Developmental $BMCL_5$ = 25 pup BW decreased at 613 $BMCL_{10}$ = 50 mg/m^3 NOAEC = 305 LOAEC = 613	Rao et al. (1980), Vapor, SR Medium SD Rats – Both sexes: Inhalation. Prior to mating, during gestation, and post-natally for 2 F1 generations: 0, 25, 75, 150 ppm; 0, 102, 305 or 613 mg/m^3	Decreased body weight of selected F1B male weanlings at 150 ppm. Study used for co-critical endpoints with $BMDL_{10}$ very close to the recommended endpoint. Considering NOAECs/LOAECs, using the recommended nasal necrosis endpoint will be protective of the decreases in pup body weight.
Other studies/endpoints considered			
1,2-Dichloroethane, prenatal developmental	Reproductive/ Developmental Toxicity: NOAEL = 1,200 Maternal Toxicity: NOAEC = 1,000 LOAEC = 1,200	Payan et al. (1995), Vapor, SR High SD Rats – Both sexes: Inhalation exposure for 2 weeks. GD 6–20. 6 hours/day 7 days/week: 0, 150, 200, 250, 300 ppm; 0, 610, 820, 1,000, 1,200 mg/m^3	Repro./Dev. Toxicity: Pregnancy rate among females at 250 ppm was significantly lower ($p < 0.05$). This effect was not observed at the highest concentration of 300 ppm. No other significant effects reported. Maternal Toxicity: 2/26 dams died at 300 ppm (highest dose). Maternal body weight gain at GD 6–21 was significantly decreased at 300 ppm. No mention of food consumption. NOAEC/LOAEC higher than recommended endpoint. Not amenable to BMD modeling.

Chemical/ Endpoint	POD (mg/m^3)	Study Parameters	Comments
1,2-Dichloroethane, prenatal developmental	Reproductive/ Developmental LOAEL = 405 Maternal Toxicity: NOAEL = 405 LOAEL = 1,214	Rao et al. (1980), Vapor, SR Medium SD Rats – Female: Inhalation exposure for 10 days. GD 6–15. 7 hours/day: 0, 100, 300 ppm (0, 405, 1,214 mg/m^3)	Developmental Toxicity: A significant decrease in the incidence of bilobed thoracic centra was seen at 100 ppm; however, study essentially becomes a single dose study and not amenable to dose-response modeling due to the high maternal toxicity at 300 ppm (10/16 maternal rats died at 300 ppm). Therefore, this study is not acceptable for POD derivation.
1,2-Dichloroethane, prenatal developmental toxicity	Reproductive/ Developmental Liver NOAEL = 16,000 Maternal Toxicity: LOAEL = 16,000	Schwetz et al. (1974), Vapor, SR Medium 7 hours/day 10 days Exposed on GD 6–15: 0, 3,800, 6,000 ppm; 0, 16,000, 24,300 mg/m^3	At 6,000 ppm: Increased relative liver weight (SGPT/ALT activity was not determined); an increased incidence of delayed ossification of sternabrae. At 3,800 ppm: decrease in maternal body weight gains observed LOAEL: 15,372 mg/m^3 (3,798 ppm). Study precluded for POD derivation because of several methodological and control issues.
1,2-Dichloroethane, liver	NOAEL = 2,527 LOAEL = 3,475	Brondeau et al. (1983), whole body inhalation chamber, SR Medium SD Rats – Male: 0, 618, 850, 1,056, 1,304 ppm; 0, 2,527, 3,475, 4,318, 5,332 mg/m^3	Significant increases in serum GLDH and SDH levels were seen at ≥850 ppm (3,475 mg/m^3); serum ALT and AST were significantly increased at 850 ppm (3,475 mg/m^3) but not at higher concentrations. Dose-response analysis inadequate. Histopathology and organ weight not assessed.
1,2-Dichloroethane, liver, metabolic, kidney, respiratory	Liver, Metabolic & Kidney (organ weight) Overall study NOAEL/LOAEL: Metabolic (body weight): NOAEL = 809 LOAEL = 2,428	Dow Chemical (2006), Vapor, SR High F344 Rats – Both sexes: 4 or 8 hours: 0, 50, 100, 150, 200, 600, or 2,000 ppm; 202, 405, 607, 809, 2,428 or 8,095 mg/m^3	Organ weight changes (liver, adrenal, kidney); histological changes (liver, kidney, olfactory mucosa); multiple FOB changes, bw changes were observed although most effects were inconsistent or transient but supportive of liver and kidney effects; the olfactory effect (degeneration of the olfactory neuroepithelial mucosa) from this study was used as the recommended POD (see first entry above).
1,2-Dichloroethane, liver/kidney relative organ weights	Liver (relative organ weight): NOAEC = 5,111 LOAEC = 6,134 Kidney (relative organ weight): NOAEC = N/A LOAEC = 4,089	Francovitch et al. (1986), Vapor, SR Medium CD-1 Mice – Male: 4 hours: 0, 1,000, 1,250, 1,500 ppm; 0, 4,089, 5,111 or 6,134 mg/m^3	Organ weight changes and histology (liver and kidney); however, exposure group where these changes occurred, and negative control data were not reported. Although study is supportive of liver and kidney effects, it is not suitable for dose-response analysis. Observed effects are occurring at higher concentrations than the recommended POD.

Chemical/ Endpoint	POD (mg/m^3)	Study Parameters	Comments
1,2-Dichloroethane, immunological/ streptococcal infection challenge	CD-1 (Female): NOAEC = 9.21 LOAEC = 21.6 SD Rats (Male): NOAEC = 801.2	Sherwood et al. (1987), Vapor, SR High CD-1 Mice – Female: 3-hour single exposure: 0, 2.3, 5.4, 10.8 ppm; 0, 9.21, 21.6, 43.3 mg/m^3 SD Rats – Male: 3-or 5-hour single exposure: 0, 10, 20, 50, 100, 200 ppm; 0, 40.1, 80.1, 200.3, 400.6 and 801.2 mg/m^3	Mice: Increased mortality from streptococcal challenge; decreased bactericidal activity; no effects in cell counts or phagocytic activity of alveolar macrophages; increased leucine aminopeptidase (LAP) activity. Rats: No effects observed
1,2-Dichloroethane, neurological	For 12 hours/day for 1 day: NOAEC: 2,500 LOAEC: 5,000 2, 4, or 6 hours/day for 1 day: LOAEC: 5,000	Zhang et al. (2010), Vapor, SR Medium SD Rats – Both sexes: 12 hours/day for 1 day: 0, 2,500, 5,000, 10,000 mg/m^3 2, 4, or 6 hours/day for 1 day: 0 or 5,000 mg/m^3	12 hours/day for 1 day: No mortality observed; signs of abnormal behavior; effects on brain histology (edema corresponding with water content in the cortex, no details on severity or dose-response). 2, 4, or 6 hours/day for 1 day: Effects on brain histology less severe than at 12 hours (edema corresponding with water content of cortex, perineural and perivascular spaces).
1,2-Dichloroethane, neurological	For 1.5 or 4 hours: NOAEC: 4,000	Zhou et al. (2016), Vapor, SR Medium SD Rats – Males: 1.5 or 4 hours; 0, 4,000, or 12,000 mg/m^3	Effects on the brain lesions with edema, and a significant decrease in the number of fiber tracts were observed compared to control. Study not suitable for dose- response analysis.
1,2-Dichloroethane, liver/kidney clinical chemistry	Liver Clinical Chemistry: NOAEC = 640 LOAEC = 2,020 Kidney weight/BUN: NOAEC = 640 LOAEC = 2,020 Mortality: NOAEC = 2,020 LOAEC = 4,339	Storer et al. (1984), Gas, SR High B6C3F1 Mice – Males: 4 hours: 0, 58, 499, 1,072, and 1,946 ppm; 0, 640, 2,020, 4,339, and 7,876 mg/m^3	Increased serum levels of IDH, ALT, and BUN; increased liver and kidney weights; evidence of DNA damage; and increased mortality (4/5 and 5/5 at $\geq$ 499 ppm) essentially reducing this study to a single dose study and unsuitable for dose-response analysis.

5.2.6.1.3 Non-Cancer PODs for Intermediate Exposures

Oral

There were 4 short-term (>1 to 30 days) and sub-chronic (>30 to 91 days)-duration animal toxicology studies from the 1,1-dichloroethane database rated as acceptable based on data quality evaluation using systematic review approaches (Table_Apx N-8). Three other studies met this exposure duration; however, they lacked concurrently run controls, had limited methodological details, and deficient data reporting. Studies, however, where applicable, were still integrated into hazard identification and weight of scientific evidence. Overall, the 1,1-dichloroethane database did not have enough information to

identify NOAELs and LOAELs by target organ/system. Identifying only overall non-cancer NOAELs and LOAELs yielded one study, Muralidhara et al. (2001) adequate for dose-response analysis and POD selection for the intermediate exposure duration. In this 13-week study following 1,1-dichloroethane exposure (Muralidhara et al., 2001), and further described above in Section 5.2.3, a NOAEL of 1,000 mg/kg-day and a LOAEL of 2,000 mg/kg-bw/day were identified for mortality (1/15 rats), CNS depression, and decreased body weight. At the high dose in this study (4,000 mg/kg-bw/day), the rats exhibited protracted narcosis, and 8/15 rats died between weeks 1 and 11, when the surviving rats in this group were also sacrificed. While this study was initially considered for intermediate exposure duration POD selection, lethality was observed at doses below the LD50 of 8,200 mg/kg-day (with 95% CLs of 4,800–14,100 mg/kg-day). Taken together with narcosis lacking sensitivity as a critical endpoint, Muralidhara et al. (2001) from the 1,1-dichloroethane database was not useable as a sub-chronic oral POD.

Thus, read-across from 1,2-dichloroethane was used for 1,1- dichloroethane to identify non-cancer intermediate oral and dermal PODs. For 1,2- dichloroethane, a total of 4 animal toxicity studies were available, and 3 of these studies had acceptable data quality for dose-response analysis and identification of the intermediate oral duration POD. There were no dermal data for the intermediate exposure duration.

As indicated prior, the database for 1,1-dichloroethane contained data gaps and the use of the 1,2-dichloroethane database was used to fill those gaps, a thorough evaluation for both ATSDR (2022) and (U.S. EPA, 2010), that identified the 13-week study by (NTP, 1991), where male and female F344/N, Sprague Dawley, and Osborne-Mendel rats as well as B6C3F1 mice exposed to 1,2-dichloroethane in drinking water was used to derive their respective values was evaluated.

A significant dose-related increase in kidney weight and the kidney-body-ratio of female F344 rats was identified at 58 mg/kg/day among the three rat strains. This study was considered as a potential candidate for POD derivation; however, the daily intake doses were estimated on a mg/kg body weight basis and not measured throughout the duration of exposure. The means by which the dosage estimates were calculated was by dividing the mean water consumption over the 13-week study by the initial and final body weights of ten animals. Additionally, weight gain depression was seen in males and females in the two higher dose groups throughout the study and was likely caused by dehydration. The study indicated that water consumption was substantially decreased with increasing dose with a decrease of as much as 60 percent in water intake was also seen in both male and female Osborne-Mendel rats at the highest concentration of 8,000 ppm (a range of 500–725 mg/kg/day) that indicates that the dose received by all exposed animals was less than the target dose. The authors indicated that as water intake was reduced at most exposure levels, equivalent exposure did not occur at different dose levels within a strain. Uncertainties to the actual dose delivered thus precluded the use of this study for dose-response. Additionally, increases in erythrocyte counts, mild decreases in mean cell volume, and the mild increases in blood urea nitrogen in the high dose male rats were also considered indicative of animal dehydration. The authors also suggest that the decrease in mean cell volume (hematocrit /erythrocytes) may be related to dehydration resulting in an increase in serum osmolarity, with a subsequent loss of water from the shrinkage of the erythrocytes.

Additionally, changes in kidney organ weight from the (NTP, 1991) drinking water studies for 1,2-dichloroethane possessed uncertainties precluding their consideration as potential candidate PODs. This includes uncertainty in the calculation for the estimated consumed concentrations by the authors due to a lack of reporting to the number of animals/cage and no correction to spillage of the test chemical

through the duration of the study. This lowered the confidence in the utility of the drinking water portion for dose-response, particularly to renal effects.

Within the study, stability of 1,2-dichloroethane in water, using gas chromatographic analysis, indicated that the solution maintained in the simulated animal-room conditions in clear glass drinking bottles under normal light resulted in losses of 1,2-dichloroethane of 13, 22, and 27 percent after 1, 2, and 3 days, respectively. As a result, the drinking water formulations were stored in sealed bottles for no longer than 3 weeks and changed at the end of each day. During the 13-week study, three complete sets of drinking water formulations were analyzed with 4 of 16 formulations out of specifications, with values ranging from 12 to 33 percent of the target formulation. An analysis of the dosage formulations that remained in the bottles after 24 hours indicated decreased concentrations that averaged approximately 29 percent, ranging from 13 to 53 percent of the target concentrations. Thus, fresh formulations were placed in the animal cages; however, the animal exposures were not consistent but ranged between the initial concentration and the concentration that was determined at the end of a 24-hour exposure period.

As renal tubular regeneration was observed in 9/10 rats in each group of dosed and control male rats with minimal to mild severity with no difference in severity between groups, EPA has low confidence that his histopathological finding is a critical effect of biological significance. The incidence of renal tubular regeneration in females; however, appeared to be dose related and was observed in 9/10 at 8,000 ppm, 3/10 at 4,000 ppm, 2/10 at 2,000 ppm, 1/10 at 1,000 ppm, 0/10 at 500 ppm, and in 0/10 controls. This lesion was also of minimal severity in all affected rats. The final ATSDR *Toxicological Profile for 1,2-Dichloroethane* identified renal tubular regeneration as the critical effect to calculate the intermediate duration oral MRL (ATSDR, 2024). EPA has low confidence in the use of these data for dose-response due to the apparent lack of histological specificity within the rat population. Characterization for this effect is also limited due to the lack of details presented in the study report regarding the histopathological methodology applied, the number of fields /slides analyzed or whether the pathologist was “blinded” to the samples.

Due to the uncertainty regarding the delivered dose, dehydration, lack of histological specificity and the inherit volatility associated with 1,2-dichloroethane, it was not recommended using this drinking water study for this dose-response assessment. The subchronic oral gavage study by NTP (1991) in F344 rats was thus identified for the dose-response assessment as greater confidence in the delivered dose could be attained without confounding due to the issues concerning volatility and spillage as was seen in the NTP (1991) drinking water study.

1,2-Dichloroethane and corn oil were mixed (w/v) to give the desired concentrations for the gavage studies. Stability studies of 1,2-dichloroethane in corn oil (approximately 10 mg/ml), using gas chromatography, established that the solutions maintained under simulated animal-room conditions (open to air and light for 3 hours) had a chemical loss of approximately 4 percent. During the studies, dose formulations were stored for no longer than 3 weeks at approximately 4°C in serum vials. Three complete sets of corn oil formulations were analyzed over the course of the 13-week studies, and all were within specifications (±10% of the target concentration). The analysis of the formulations remaining after dosing was completed gave results that were in reasonable agreement with those from samples taken immediately after mixing, indicating no loss of chemical during dose administration.

In the subchronic study by NTP (1991), oral gavage of 1,2-dichloroethane at the dosages of 0, 30, 60, 120, 240 or 480 mg/kg-bw/day for 13 weeks in male F344 rats, resulted in significant increases in absolute kidney weights at 30, 60, and 120 mg/kg/day (9, 21, and 25%, respectively) and significant

increases in relative kidney weights at 60 and 120 mg/kg-bw/day doses (15 and 26%, respectively). Female F344 rats dosed at 0, 18, 37, 75, 150, or 300 mg/kg/day at 5 days/week via oral gavage for 13 weeks caused significant increases in absolute kidney weights (12 and 23%) and relative kidney weights (10 and 21%) at 75 and 150 mg/kg-bw/day, respectively.

Based on the evidence integration within and across health effects, the selection of the NTP (1991) gavage study for dose-response was identified for the oral intermediate exposure duration based on kidney weight. As the NTP (1991) oral gavage study identified increases in absolute kidney weight at the dose of 30 mg/kg-day, increased relative kidney weight was also identified at 60 and 75 mg/kg-day in male and female F344 rats, respectively. The selection of the study is supported by other studies that identified relative renal toxicity as a critical health effect due to 1,2-dichloroethane at similar dosages.

In the subchronic 90 day (7 day/week for 13 weeks) oral gavage study by Daniel et al. (1994), male and female Sprague-Dawley rats treated with 0, 37.5, 75, or 150 mg/kg-bw/day of 1,2-dichloroethane resulted in increased relative kidney weights in both males and females (18 and 15 percent higher than controls, respectively) at the 75 and 150 mg/kg-bw/day.

Additionally, the subchronic 90-day oral gavage study in Wistar rats by van Esch et al. (1977) dosed at 0, 10, 30, or 90 mg/kg-bw/day resulted in a significantly increase in relative kidney weight of 17 and 16 percent higher than controls in males and females in the 90 mg/kg-bw/day, respectively.

As the NTP (1991) oral gavage study identified increases to both absolute and relative kidney weights, this study was selected as it was considered to detect the sensitive renal effects in F344 rats at 60 mg/kg-day dose group with concordance among the other oral gavage studies in the range of 60 to 90 mg/kg-day with regard to relative kidney weight. The BMD modeling of this study, when adjusted to a continuous study duration of 24 hours/day for 7 days, resulted in a $BMDL_{10}$ of 27 mg/kg-day (unadjusted value of 37.8 mg/kg-day) based on the increased relative kidney weight in the male F344 rats. This value was converted to HED values using a DAF of 0.24 (based on the body weight ¾ for rats) for occupational and continuous exposure of 9.1 and 6.5 mg/kg-day, respectively, with a benchmark MOE of 30 and used for risk assessment of intermediate oral/dermal exposure duration. This study was also used for risk assessment of chronic oral/dermal exposure duration with an applied MOE of 300.

Although the study by (Suguro et al., 2017) was a dermal cancer bioassay and not considered suitable for dose-response, renal toxicity due to 1,2-dichloroethane exposure in this study supports the selection of renal toxicity as a critical health effect based on histopathology as indicated by distal tubular mild karyomegaly and tubular degeneration. EPA has greater confidence as the histopathological methodology was provided in greater detail within the study report.

Several other studies were considered from across the 1,1- and 1,2-dichloroethane databases including sedation which was insensitive as a selected POD from 1,1-dichloroethane (Muralidhara et al., 2001), as discussed; changes in kidney organ weight from a drinking water study from 1,2-dichloroethane (NTP, 1991), as discussed; reproductive/developmental outcomes following exposure to 1,2-dichloroethane, including fetal resorptions and decreases in maternal body weight (Payan et al., 1995) and likely confounded results for fertility and implantation success for 1,2-dichloroethane (Lane et al., 1982).

The Munson (1982) 14-day short-term study in CD1 mice of both sexes were dosed with 1,2-dichloroethane via oral gavage at doses of 0, 4.9, or 49 mg/kg. Endpoints evaluated included body weight, hematology, gross necropsy, organ weights (liver, spleen, lungs, thymus, kidney, and brain), humoral immunity, and cell-mediated immunity. The treatment-related effect observed in this study was

immunosuppression based on observed suppression of a cell-mediated immune response at doses 4.9 and 49 mg/kg/day. Co-critical endpoints identified in this same Munson et al. (1982) study included an observed 30 percent decrease in leukocytes at 49 mg/kg/day, and a dose-dependent trend of antibody forming cells/spleen towards immune suppression with 25 and 40 percent suppression at 4.9 and 49 mg/kg/day, respectively. NTP (1991) provided additional support for immunotoxicity. It was a 13-week oral gavage study of F344/N rats dosed with 30, 60, 120, 240, or 480 mg/kg for males or 18, 37, 75, 150, or 300 for females of 1,2-dichloroethane that observed possible dose-related incidences of thymus necrosis. Female rat absolute thymus weight was decreased. The utilization of this endpoint was limited by the changes in thymus co-occuring with mortality.

The acute oral study by Zabrodskii et al. (2004) that identified immunotoxicity, at 930mg/kg was compared to the much lower POD of 4.9 mg/kg/day in the 1,2-dichloroethane Munson et al. (1982) multi-dose study and compared to other identified critical effects. This immunotoxicity finding as suggested by the metabolites presented in the study to be indicative of 1,2-dichloroethane further supports immunosuppression as a potential endpoint of consideration for 1,2-dichloroethane exposure. Important to underscore, immunotoxicity found in both the 1,1- and 1,2-dichloroethane databases, is recognized as a cancer mechanism (Hanahan and Weinberg, 2011). Specifically, inflammatory cell recruitment that can actively promote tumor formation and was observed in both the Munson et al. (1982) and Zabrodskii et al. (2004), through cell-mediated immune responses. However, due to limitation regarding immunotoxicity, this health effect was not identified for derivation of the intermediate POD.

The POD for the intermediate oral exposure route was thus based on renal toxicity, specifically increased relative kidney weight at a $BMDL_{10}$ of 27 based on NTP (1991) with a benchmark MOE of 30. The benchmark response of 10 percent for increased kidney weight is considered as an effect to be biologically significant in humans and protective of human health. The HED for the occupational (worker) scenario was then calculated as 9.1 mg/kg while the HED for the general population (continuous) scenario was calculated as 6.5 mg/kg.

Table 5-39, Figure 5-9, and Figure 5-10 show the recommended intermediate oral study and POD (in consideration of both 1,1-dichloroethane and 1,2-dichloroethane toxicity data) followed by co-critical endpoints (PODs within the range of the recommended study) and other studies considered in support of the recommended POD presented within exposure response arrays.

Inhalation

No other short/intermediate-term inhalation studies with a rating of acceptable were located for 1,1-dichloroethane except for Schwetz et al. (1974). Among the effects reported by Schwetz et al. (1974), only the decreased maternal body weight (LOAEL of 3,798 ppm) was considered to be a suitable endpoint for POD derivation. Uncertainties of the data from Schwetz et al. (1974) were (1) the evaluations of maternal endpoints did not include histopathology or effects in organs other than the liver, (2) the disparate findings on delayed ossification in the two control groups mean that a conclusion regarding this endpoint cannot be made with confidence, and (3) there are no supporting studies that evaluated comprehensive endpoints. A 4-week short-term study in male mice exposed to 1,2-dichloroethane by Zhang et al. (2017) was thus used based on read-across to 1,1-dichloroethane. A $BMCL_5$ and BMC_5 of 6.6 ppm (26.7 mg/m^3) and 5.24 ppm (21.2 mg/m^3), were identified based on decreased sperm concentration. The intermediate inhalation HEC for occupational and continuous exposure of 22 ppm (89 mg/m^3) and 5.2 ppm (21.2 mg/m^3), respectively, with a benchmark MOE of 100, was used for risk assessment of intermediate inhalation exposure (see Table 5-44).

The POD for the intermediate inhalation exposure route was thus based on male reproductive effects, specifically decreased sperm concentration at a $BMCL_5$ of 21.2 mg/m^3 based on Zhang (2017) with a benchmark MOE of 30. The HEC for the occupational (worker) scenario was then calculated as 22 ppm while the HEC for the general population (continuous) scenario was calculated as 5.2 ppm. EPA determined that the Zhang (2017) study was appropriate for dose-response based on the weight of scientific evidence that indicated the testes as a target organ for 1,2-dichloroethane. In this study, significant pathological changes in the testes including vacuolar degeneration of germ cells, decreased sperm concentration, motility, and progressive motility, and increased abnormalities of the sperm (head, body, and tail). Previous studies were evaluated but due to their limitations in reporting information on study design and results and a lack of evaluation of sperm parameters, Zhang (2017) was identified for POD selection based on sperm concentration and is considered by EPA as the use of the best available science as this health effect is considered both sensitive and supported by both histological and mechanistic data.

Zhang (2017), in addition to the measurement of sperm parameters and histopathology, also evaluated potential mechanisms of reproductive toxicity associated with inhalation to 1,2-dichloroethane. The study identified the induction of apoptosis in the germ cells of the mice exposed to 1,2-dichloroethane via inhalation as a potential mechanism for the sperm effects observed. Due to the limited mechanistic information from prior studies on the reproductive effects associated with 1,2-dichloroethane, the Agency is confident in the selection of this study for the POD based on decreased sperm concentration from this study for the intermediate duration at a benchmark response 5 percent is considered an effects to be biologically significant in humans and thus considered to be human health protective.

Table 5-40 and Figure 5-11 show the recommended intermediate inhalation study and POD (in consideration of both 1,1-dichloroethane and 1,2-dichloroethane toxicity data) followed by co-critical endpoints (PODs within the range of the recommended study) and other studies considered in support of the recommended POD presented within exposure response arrays.

Dermal

No short-term/subchronic exposure studies on 1,1-dichloroethane via the dermal route were located. Therefore, the intermediate oral HED for occupational and continuous exposures of 9.1 and 6.5 mg/kg-bw/day, respectively, was extrapolated for the dermal route, with a benchmark MOE of 30, and was used for risk assessment of intermediate dermal exposure (see Table 5-44).

Table 5-39. Intermediate Oral Non-Cancer POD-Endpoint Selection Table

Chemical/Endpoint	POD (mg/kg/day)	Study Parameters	Comments
POD selected for non-cancer risk evaluation for short-term/subchronic oral exposures			
1,2-Dichloroethane, kidney weight	NOAEL = 30 (males) LOAEL = 75 (females)	NTP (1991), Gavage, SR High F344 Rats – Both sexes: 13 weeks: 0, 30, 60, 120, 240, 480 mg/kg-day (males); 0, 18, 37, 75, 150, 300 mg/kg/day (females)	
Co-critical endpoints			
1,2-Dichloroethane, decreased leukocytes	LOAEL = 4.9	Munson et al. (1982), Gavage, SR High CD1 Mice – Both sexes: 14 days: 0, 4.9, 49 mg/kg-day	Supports cell-based immunosuppression endpoint.
Other studies/endpoints considered			
1,2-Dichloroethane, immunotoxicity • Humoral immune response to T-dependent and T-independent antigens • Antibody-dependent cell cytotoxicity • Delayed hypersensitivity (DTH) reaction	LOAEL = 930	Zabrodskii et al. (2004), Gavage, SR Medium Random-Bred Albino Rat – Both sexes: Single dose: 0, 930 mg/kg-bw	Qualitatively supports immunosuppression. A multi-day exposure produces more sensitive PODs for immune suppression than a single exposure study. However, dose is close to LD_{50}. Single acute exposure to one dose and monitored; various immune reactions and indices were evaluated 48 hours and 5 days after exposure.
1,2-Dichloroethane, sedation	$NOAEL_{adj}$ = 714	Muralidhara et al. (2001), Gavage, SR Medium SD Rats – Male: 13 weeks: 0, 500, 1,000, 2,000, 4,000 mg/kg-bw/day	1,2-Dichloroethane acute oral LD50 is 725 mg/kg (PubChem), the POD is near lethal doses, narcosis is well-known to occur at high doses and is not considered a sensitive endpoint in the database. This is the only study that passed SR with a useable subchronic oral POD.
1,2-Dichloroethane, immune (thymus)	NOAEL = 240 mg/kg-day (males); 150 mg/kg-day (females) LOAEL = 480 mg/kg-day for thymus necrosis in males; 300 mg/kg-day for thymus necrosis in females	NTP (1991), Gavage, SR High F344 Rats – Both sexes: 13 weeks: 0, 30, 60, 120, 240, 480 mg/kg-day (males); 0, 18, 37, 75, 150, 300 mg/kg/day (females)	Qualitatively supports immunosuppression. However, thymus necrosis occurs at dosages where mortality was also occurring therefore cannot be used as a POD.

Chemical/Endpoint	POD (mg/kg/day)	Study Parameters	Comments
1,2-Dichloroethane, decreased cell based immune response	LOAEL = 4.9	Munson et al. (1982), Gavage, SR High CD1 Mice – Both sexes: 14 days: 0, 4.9, 49 mg/kg-day	
1,2-Dichloroethane, fetal resorptions	NOAEL = 160 LOAEL = 200 (Data were not amenable for BMD modeling)	Payan et al. (1995), Gavage, Prenatal Developmental, SR High SD Rats – Female: Dosing GD6–20: 0, 120, 160, 200, or 240 mg/kg	The increases in non-implants and resorptions are difficult to interpret given the significant maternal toxicity at corresponding doses (30 and 49% at 200 and 240 mg/kg/day, respectively) consisting of decreases in maternal bw gain, and the fact that there was no effect on the number of live fetuses per litter despite the changes in non-surviving implants/litter and resorption sites/litter. Therefore, cannot be used as POD. NOAEL exceeds the maximum tolerated dose.
1,2-Dichloroethane, decreases in maternal body weight gain	NOAEL = 160 LOAEL = 200 (BMD = 99.1; BMDL = 41.8)	Payan et al. (1995), Gavage Pre-Natal Developmental, SR High SD Rats – Female: Dosing GD6–20: 0, 120, 160, 200, or 240 mg/kg	A dose-related reduction in adjusted (for gravid uterine weight) maternal bodyweight gain during treatment occurred, with statistical significance achieved at the two highest doses (30 and 49% reduction compared with controls, $p < 0.05$). NOAEL exceeds the maximum tolerated dose.
1,2-Dichloroethane, chronic 26-week dermal study decreased body weight in females; increased distal tubular mild karyomegaly (both sexes); renal karyomegaly and tubular degeneration (females)	LOAEL = 6,300	Suguro et al. (2017), Dermal, SR High CB6F1-Tg rasH2@Jcl (rasH2) mice – Both sexes: 3 days/week 26 weeks: 0, 126 mg; 0, 6,300 mg/kg-day	Not considered acceptable for dose response assessment as the study used a single dose using transgenic mice.

Table 5-40. Intermediate Inhalation Non-Cancer POD-Endpoint Selection Table

Chemical/Endpoint	POD (mg/m^3)	Study Parameters	Comments
POD selected for non-cancer risk evaluation for short-term/subchronic inhalation exposures			
1,2-Dichloroethane	$BMCL_5$ = 21.2 mg/m^3 NOAEC = 350 LOAEC = 700	Zhang et al. (2017), 4 week morphological analysis of sperm parameters, SR High Swiss Mice – Male: 6 hours/day, 7 days/week, 4 weeks: 0, 100, 350, 700 mg/m^3	Decreases in sperm concentration. Testicular (germinal epithelium) = $BMCL_{1SD}$ = 8.6 mg/m^3
Co-critical endpoints			
1,2-Dichloroethane, fetal development	Reproductive/ Developmental $BMCL_5$ = 25 Pup BW decreased at 613 $BMCL_{10}$ = 50 mg/m^3 NOAEC = 305 LOAEC = 613	Rao et al. (1980), Vapor, SR Medium SD Rats – Both sexes: Inhalation. Prior to mating, during gestation, and post-natally for two F1 generations: 0, 25, 75, 150 ppm; 0, 102, 305, or 613 mg/m^3	Decreased body weight of selected F1B male weanlings at 150 ppm. Study used for co-critical endpoints with $BMCL_5$ very close to that from the recommended endpoint. Considering NOAECs/LOAECs, using the recommended endpoint will be protective of the decreases in pup body weight. Also, portal of entry effects can be considered more sensitive than systemic effects.
Other studies/endpoints considered			
1,1-Dichloroethane, prenatal developmental toxicity	Reproductive/ Developmental Liver NOAEC = 16,000 Maternal Toxicity: LOAEC = 16,000	Schwetz et al. (1974), Vapor, SR Medium 7 hours/day 10 days Exposed on GD 6–15: 0, 3,800, 6,000 ppm; 0, 16,000, 24,300 mg/m^3	At 6,000 ppm: Increased relative liver weight (SGPT/ALT activity was not determined); an increased incidence of delayed ossification of sternabrae. At 3,800 ppm: decrease in maternal body weight gains observed LOAEC: 15,372 mg/m^3 (3,798 ppm). Study precluded for POD derivation because of several methodological and control issues.
1,2-Dichloroethane, liver	LOAEC = 3,424	Brondeau et al. (1983), Vapor, SR Medium SD Rats – Male: 6 hours/day for 2 or 4 days: 0 or 3,424 mg/m^3	6 hours/day for 2 days: Significant increases in serum ALT, GLDH and SDH levels; liver histopathology and organ weight were not assessed. 6 hours/day for 4 days: Serum SDH levels were significantly increased. Liver histopathology and organ weight were not assessed.
1,2-Dichloroethane, liver	LOAEC = 619	Igwe et al. (1986b), Vapor, SR High SD Rats – Male: 7 hours/day, 5 days/week, 4 weeks: 0, 153, 304, 455 ppm; 619, 1,230, and 1,842 mg/m^3	Increased relative liver weight and 5'-NT. Absolute liver weight was not reported. No changes in hepatic GST activity, hepatic DNA content, or serum enzymes ALT or SDH were observed at any concentration.

Chemical/Endpoint	POD (mg/m^3)	Study Parameters	Comments
1,2-Dichloroethane, liver/ reproductive/metabolic/mortality	Immune: NOAEC = 1,842 Reproductive: NOAEC = 1,842 Liver: LOAEC = 619 Mortality, Metabolic: NOAEC = 619 LOAEC = 1,230	Igwe et al. (1986b), Vapor, SR High SD Rats – Male: 7 hours/day, 5 days/week, 30 days: 0, 153, 304, 455 ppm; 619, 1,230, and 1,842 mg/m^3	Immune, Reproductive/Developmental: No effects on organ weight or histopathology. Liver: Increased relative liver weight, absolute liver weight was not reported. Mortality: Occurred in 1/12 and 2/12 animals in 1,230 and 1,842 mg/m^3, respectively Metabolic: Decreased body weight. NOAEC/LOAEC higher than recommended POD. Not amenable to BMD modeling
1,2-Dichloroethane, reproductive/ developmental/ maternal toxicity	Reproductive/ Developmental: NOAEC = 1,200 Maternal Toxicity: NOAEC = 1,000 LOAEC = 1,200	Payan et al. (1995), Vapor, SR High SD Rats – Both sexes: Inhalation exposure for 2 weeks. GD 6–20. 6 hours/day 7 days/week: 0, 150, 200, 250, 300 ppm; 0, 610, 820, 1,000, and 1,200 mg/m^3	Repro/Dev Toxicity: Pregnancy rate among females at 250 ppm was significantly lower, but not at 300 ppm; no other significant effects reported. Maternal Toxicity: 2/26 dams died at 300 ppm (highest dose). Maternal body weight gain at GD 6-21 was significantly decreased at 300 ppm. No mention of food consumption. NOAEC/LOAEC higher than recommended POD. Not amenable to BMD modeling.
1,2-Dichloroethane, reproductive/ developmental; maternal toxicity	Reproductive/ Developmental: LOAEC = 405 Maternal Toxicity: NOAEC: 405 LOAEC: 1214	Rao et al. (1980), Vapor, SR Medium SD Rats – Female: Inhalation exposure for 10 days; GD 6–15; 7 hours/day: 0, 100, 300 ppm (0, 405, and 1,214 mg/m^3)	Developmental Toxicity: A significant decrease in the incidence of bilobed thoracic centra was seen at 100 ppm; however, the study essentially becomes a single dose study and not amenable to dose-response modeling due to the high maternal toxicity at 300 ppm (10/16 maternal rats died at 300 ppm). Therefore, this study is not acceptable for POD derivation.
1,2-Dichloroethane, immunological/ streptococcal infection challenge	CD-1 Mice: NOAEC = 9.21 SD Rats: NOAEC = 400.6	Sherwood et al. (1987), Vapor, SR High CD-1 Mice – Female: 3 hour/day, 5 days/week, 5 days: 0, 2.3; 0, 9.21 mg/m^3 SD Rats – Male: 5 hour/day, 5 days/week, 12 days: 0, 10, 20, 50, 100; 0, 40.1, 80.1, 200.3, 400.6 mg/m^3	CD-1 mice and SD rats showed no effects.

Chemical/Endpoint	POD (mg/m^3)	Study Parameters	Comments
1,2-Dichloroethane, liver/metabolic	Liver: NOAEC = 350 Metabolic: NOAEC = 350 LOAEC = 700	Zeng et al. (2018), Aerosol, SR High Swiss Mice – Male: 6 hours/day, 7 days/week, 28 days: 0, 350, 700 mg/m^3	Liver: Increased absolute and relative liver weight, increased liver concentrations of glycogen, triglycerides, and free fatty acids at all concentrations; increased ALT (1.9-fold) at 700 mg/m^3; increased serum AST (1.3- to 1.7-fold) , triglycerides, and free fatty acids; decreased serum glucose at both exposure concentrations. Metabolic: Body weight significantly reduced at 700 mg/m^3.
1,2-Dichloroethane	Neurological, Reproductive, Immune/Hematological, Liver, Mortality, Metabolic, Kidney (Rat): Respiratory: NOAEC = 809 Liver, Metabolic, and Kidney (Guinea Pig): NOAEC = 405	Spencer et al. (1951), Vapor, SR Medium Wistar Rats – Both sexes 7 hours/day 5 days/week 212 days*: 0, 100, 200, 400 ppm; 0, 405, 809, 1,619 mg/m^3 *Although all exposure groups were intended for chronic duration exposures, animals at the high exposure level died within 14 days (females) and 56 days (males). Guinea Pigs – Both sexes: 7 hours/day 5 days/week 248 days: 0, 100, 200, 400 ppm; 0, 405, 809, 1,19 mg/m^3	Rats: High mortality at 400 ppm starting at 2 weeks; no other effects reported. Guinea Pigs: High mortality at 400 ppm starting at 2 weeks; reductions in body weight starting at 100 ppm; increases in liver weight; possible liver histopathology and changes in kidney weight, but incidence not reported.

5.2.6.1.4 Non-Cancer PODs for Chronic Exposures

Oral

In evaluating studies that could be considered suitable in identifying a chronic POD, no studies of chronic oral exposure in laboratory animals were considered suitable for POD determination (see Appendix N.2.5 for 1,1-dichloroethane and Appendix N.2.8 for 1,2-dichloroethane).

The NTP (1978), a chronic cancer bioassay in male and female Osborne-Mendel rats administered 1,1-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 382, or 764 mg/kg-day for males and 0, 475, or 950 mg/kg-day for females was also evaluated for non-cancer endpoints. The study evaluated, body weight gain, survival, clinical observations, gross and microscopic examination of all major tissues. Findings, including the overall poor survival, was confounded by high incidence of pneumonia observed in control and all treated groups. Additionally, this study did not indicate specific adverse body weight changes, histopathological changes or clinical observations in treated rats as compared to controls in both male and female rats. These observations, however, due to confounding by the observed murine pneumonia, also attributed to the preclusion in selection of findings of this study for dose-response.

Additionally, the NTP (1978) chronic cancer bioassay in male and female B6C3F1 mice administered 1,1-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 1,442, or 2,885 mg/kg-day for males and 0, 1,665, or 3,331 mg/kg-day for females also evaluated, body weight gain, survival, clinical observations, food consumption, gross and microscopic examination of all major tissues. Male mice exhibited poor survival in all groups while female mice showed better survival overall (survival within the untreated, vehicle low dose and high dose groups were at 35, 55, 62, and 32 percent in male mice and 80, 80, 80 and 50 percent in female mice). Mice exhibited incidences of murine pneumonia, though at lower occurrences to those identified in the parallel study in Osborne-Mendel rats and only in the treatment groups for both males and females. Additionally, this study did not indicate specific adverse body weight changes or clinical observations in treated mice as compared to controls in both male and female mice. The only specific histopathological changes were of endometrial polyps in the uterus in the 3331 mg/kg-day treated female mice with no other occurrences in the other dosage groups. Overall, for male mice, the NOAEL was identified as 1442 mg/kg-d based on reduced survival in the high dose group. For female mice, the NOAEL of 1665 mg/kg-d was identified based on reduced survival and increased incidence of endometrial stromal polyps in high dose females. The study wide NOAEL of 1665 mg/kg/day was thus proposed.

In Klaunig (1986), a chronic duration study evaluating tumorgenicity and tumor promotion potential of 1,1- and 1,2-dichloroethane, B6C3F1 hybrid male mice were administered 0, 0.835, or 2.5 mg/ml (equivalent to a calculated dose of 155 or 465 mg/kg/day for 1,1-dichloroethane and 159 or 475 mg/kg/day for 1,2-dichloroethane, respectively) of the chemicals continuously via drinking water *ad libitum* for 52 weeks. This study also evaluated the non-cancer endpoints of mortality, histological changes, and body weight that did not indicate any difference among treated animals as compared to controls for either 1,1- or 1,2-dichloroethane. Due to limitations in data reporting, specifically the reporting of growth and water intake data from controls on separate graphs from the experimental groups made independent evaluations and determinations of significance difficult. Additionally, statistical comparisons were only made against other treatment groups, rather than to controls.

In the study by Alumot (1976), male rats administered 1,2-dichloroethane in the diet at doses of 0, 250 or 500 ppm (equivalent to 21 and 42 mg/kg/day, using average body weight equal to 0.152 kg and mean food consumption rate of 0.0161 kg/day, chronic) for 3 hours/day (made available for 1 hour in the day and 2 hours at night) for 7 days/week during the 104 week treatment period. An evaluation of the health endpoints evaluated which included survival, body weight, and clinical chemistry did not identify differences between treated animals and controls and thus a NOAEL of 500 ppm (equivalent to 42 mg/kg/day) was determined.

The study by Storer (1995), that treated male and female ppG64 transgenic mice to 1,2-dichloroethane via oral gavage (0, 100 or 200 mg/kg-day for males or 0, 150 or 300 mg/kg in females) for 40 weeks necessitated a reduction in the high dose group due to mortality and decreased body weight. The final high dose groups were ultimately reduced from 200 to 100 and 300 to 150 in males and females, respectively, essentially down to the level of the low dose of the study. Endpoints included mortality, clinical signs, body weight, hematology, clinical chemistry, gross necropsy, and histopathology of multiple tissues from mice found dead or killed prior to terminal necropsy; however, no results were reported for most gross necropsy or clinical chemistry endpoints. Decreased survival and body weight gain occurred in females of both dose groups and light anemia was reported at terminal necropsy in females in the low-dose group. For males, animals that survived until terminal necropsy, histological examination was limited to the thymus, tumors and other gross ophthalmic changes with no results reported for most gross necropsy or clinical chemistry endpoints. After 40 weeks of exposure, decreased body weight gain occurred in high-dose males compared with controls and slight anemia with a

regenerative response (increased mean corpuscular volume) was reported at terminal necropsy of low-dose group males. No effects on survival or tumor incidence were observed in the male mice. As this study was intended to identify tumorgenicity due to exposure to 1,2-dichloroethane via the oral route, limitations in the data reporting for the non-cancer endpoints resulted in this study not identified as suitable for non-cancer dose-response.

The NTP (1978), a chronic cancer bioassay in male and female Osborne-Mendel rats administered 1,2-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 97, or 195 mg/kg-day for males and 0, 149, or 299 mg/kg-day for females was also evaluated for non-cancer endpoints. The study evaluated, body weight gain, survival, clinical observations, gross and microscopic examination of all major tissues. Findings, including the overall poor survival, was confounded by high incidence of pneumonia observed in control and all treated groups. This study indicated decreased body weight in females but no specific histopathological changes or clinical observations in treated rats as compared to controls in both male and female rats. These observations, however, due to confounding by the observed murine pneumonia, also attributed to the preclusion in selection of findings of this study for dose-response.

Additionally, the NTP (1978) chronic cancer bioassay in male and female B6C3F1 mice administered 1,1-dichloroethane via oral gavage in corn oil for 5 days/week for 78 weeks at time-weighted adjusted doses of 0, 1442, or 2885 mg/kg-day for males and 0, 1665, or 3331 mg/kg-day for females also evaluated, body weight gain, survival, clinical observations, food consumption, gross and microscopic examination of all major tissues. A LOAEL of 299 mg/kg-day was determined for this review, based on decreased body weights and increased mortality in female mice.

Although chronic studies were identified and evaluated for the chronic duration from both 1,1- and 1,2-dichloroethane due to limitations and uncertainties in those studies, as described in Section 0 of the risk evaluation, the identified intermediate POD in Section 5.2.6.1.2 was selected for derivation of the chronic POD. The intermediate continuous HED was 6.5 mg/kg-bw/day and the worker HED was 9.1 mg/kg-bw/day (see Appendix N.2.7). The benchmark MOE for this POD is 300 based on 3 for interspecies extrapolation when a dosimetric adjustment is used, 10 for human variability, and 10 for extrapolating from a subchronic study duration to a chronic study duration for chronic exposures (

Table 5-45).

Review of Existing PBPK Model

Under an Enforceable Consent Agreement for 1,2-dichloroethane, referred to as ethylene dichloride, (U.S. EPA, 2003, OPPT-2003-0010; FRL-7300-6), EPA agreed to consider the use of physiologically based pharmacokinetic (PBPK) modeling for route-to-route extrapolation of toxicity studies conducted by the oral route for quantitative evaluation of potential hazards posed by inhalation of 1,2-dichloroethane (Sweeney and Gargas, 2016; Sweeney et al., 2008; Sweeny and Gargas, 2006). The data for oral-to-inhalation extrapolation was performed with the intent of this PBPK model for extrapolation within the rat species and not to humans due to a lack of human toxicokinetic data. EPA has identified studies for consideration via the inhalation route for 1,2-dichloroethane for candidate PODs for this risk evaluation, so the use of this PBPK model was not identified as the preferred approach as parameters for this model have not been updated.

The POD for the chronic oral exposure route is based on the respective intermediate oral POD (an additional subchronic-to-chronic duration extrapolation uncertainty factor of 10× to account for the duration adjustment) with a total MOE of 300.

Table 5-41 show the recommended intermediate oral study used for the chronic duration and POD (in consideration of both 1,1-dichloroethane and 1,2-dichloroethane toxicity data) followed by co-critical endpoints (PODs within the range of the recommended study) and other studies considered in support of the recommended POD presented within exposure response arrays. Chronic studies from 1,1- and 1,2-dichloroethane that were evaluated but not identified and selected as the chronic POD are also presented in exposure response arrays (Figure 5-12).

Inhalation

In evaluating studies that could be considered suitable in identifying a chronic POD, no studies of chronic inhalation exposures were considered suitable from the 1,1- or 1,2-dichloroethane database for POD determination.

In the Hofmann (1971) inhalation study for 1,1-dichloroethane, Sprague Dawley rats, Pirbright-White guinea pigs, and rabbits were treated for at 0 or 500 ppm for 13 weeks (6 hours/day for 5 days/week) and then for an additional 13 weeks at 1,000 ppm. An evaluation of survival, body weights, clinical chemistry and histology did not indicate chemical-related effects on these parameters and thus a NOAEL of 750 ppm (3040 mg/m^3) was identified as a time-weighted average exposure concentration over the total 26-week study duration, though no effects were seen at the 1000 ppm level. Additionally, Hofmann (1971) also evaluated 1,1-dichloroethane vapor exposure in cats at 0 or 500/1000 ppm for 26 weeks. No treatment-related effects were reported on mortality, hematology, or liver parameters but reduced body weight gain, and increased BUN and serum creatinine were reported once the concentration was increased to 1000 ppm. Histological examination of the kidneys after 26 weeks exposure showed renal tubular dilation and degeneration in 3 of 4 cats tested; however, the study noted that weight loss during the 11th week of the study was due to an intercurrent catarrhal infection that confounded the interpretation of the study results to distinguish between effects caused by the 1,1-dichloroethane exposure and those caused by infection. In addition, effects on clinical pathology related to kidney function were largely attributed to one cat (sex not specified) that was removed from the study prematurely owing to poor general condition after 23 weeks. The incidence of kidney histology effects in controls was not explicitly specified. Overall, the analytical concentration for the 1000 ppm exposure concentration was identified as 1,150 ppm; however, there is uncertainty for the analytical concentration of the initial 500 ppm concentration as this was not measured in the study.

In Cheever (1990), male and female Sprague-Dawley rats treated with filtered air or 50 ppm (204 mg/m^3) of 1,2-dichloroethane vapor for 7 hours/day 5 days/week for 104 days and examined twice a day for signs of toxicity. Findings from this study included survival rate in exposed rats was similar to controls in both males and females. Additionally, no clinical signs of toxicity were noted during the study and terminal body weights were not significantly different from controls. No significant difference in food or water consumption was seen between exposed and control rats either. Absolute and relative liver weights were not different from controls with other organ weights not reported in the study. Gross testicular lesions were indicated to have been found in higher frequency in exposed males (24%) compared to control (10%) though the data were not presented, and gross pathologic observations were not evaluated statistically. Female exposed rats were noted to have showed a slight increase in the incidence of basophilic focal cellular changes in the pancreas, which were not apparent in the male rat yet the data were not provided within the study report.

In a chronic study by Spencer (1951), multiple species were evaluated based on a vapor inhalation exposure to 1,2-dichloroethane. Rabbits and monkeys were exposed at 0, 100 or 400 ppm for 7 hours/day 5 days/week for 248 or 212 days, respectively. Wistar rats and guinea pigs were exposed at 0, 100, 200 or 400 ppm for 7 hours/day 5 days/week for 212 or 248 days, respectively. Due to the small number of animals used for the rabbit study (2 males and 1 female/group) and monkey study (2 males/group), limited study details and insufficient data reporting, these data were not identified as suitable for dose-response. With regard to the rat study by Spencer (1951), at 400 ppm all female rats died within 14 days and all male rats died within 56 days. No further details on deaths or other endpoints were reported for this group. At 100 and 200 ppm, there were no evidence of adverse effects based on general appearance, behavior, growth, mortality, final body weight, organ weights, periodic hematological examinations, limited serum chemistry, or gross or microscopic examinations as compared with controls. Additionally, the specific control group of air-only or unexposed, used for comparison the treatments groups was not specified for some endpoints. Total lipid, phospholipid, neutral fat, and free and esterified cholesterol of the liver were also reported to not differ from controls. Due to lack of data reporting, the 100-ppm data could not be independently reviewed.

A growth curve and reporting of body weights and relative organ weights were presented as means only in the absence of variance for 200 ppm group only (Spencer et al., 1951). Negative findings at 100 ppm were only described in the text. Although the study indicated that statistical analysis was done on data to compare means, statistical results were not reported with the 200-ppm data. Thus, a NOAEL of 200 ppm for lack of adverse effects at this exposure level was determined. For the guinea pigs, exposure to 400 ppm resulted in severe intoxication and death with 100 percent of the males dying within 14 days and 100 percent of females dead within 32 days. Mortality in the controls was not reported. At 200 ppm, statistically significant reductions in final body weights were observed in males (16%) and females (9%), compared with air-only controls. Relative liver weights were also significantly increased in male rats (10.6%). Relative male kidney weights were also slightly elevated (6%), but did not reach statistical significance. Liver-lipid analyses in the 200-ppm group indicated a slight increase as compared the controls; however, the specific control group (air-only or unexposed) was not specified. It was also reported that microscopic examinations showed that about half of the guinea pigs examined, from both sexes, exhibited slight parenchymatous degeneration of the liver with a few fat vacuoles diffusely distributed.

Incidence values and indication as to whether any control animals exhibited these changes were not reported. At 100 pm, it was reported that there were no observed adverse effects in any endpoints; however, the data show female final body weights were statistically significantly decreased compared to

the unexposed controls. Compared to air-only controls, exposed female final body weights were not significantly decreased. Male final body weights were decreased relative to unexposed controls, but the magnitude of change was small (3%). Liver weights were significantly increased in both sexes compared to untreated controls, and female liver weights were increased by 11% compared with the air-only controls. Based on these findings, a NOAEC of 100 ppm was proposed based on lack of adverse effects at this dose. This study was not, however, considered suitable for dose-response as several limitations to the study were identified. These include data for an air-only control for males only at 100 ppm is not included in. An explanation was not provided in the text and statistical analysis of final body weight and organ weights for this exposure group were subsequently done using the unexposed groups as the controls (including in females despite the availability of an air-only control). Ambiguity as to the exact duration of exposure in the 100-ppm group and lack of data reporting for several endpoints also reduced the quality of this study.

In a study by IRFMN (1987a), Sprague Dawley rats were exposed to 1,2-dichloroethane for 7 hours/day, 5 days/week, at concentrations of 0, 5, 10, 50, or 150/250 ppm for 24 months. The dose of 250 ppm was lowered to 150 ppm after a few weeks of treatment due to severe acute toxicity. In males, there was a significant decrease in segmented neutrophils in the high exposure group. No other hematological changes were observed, and the study authors questioned the relevance of the finding. Serum chemistry changes either did not reach statistical significance, show clear relation to exposure concentration, and/or were not biologically significant as a tendency towards decreased serum LDH and ALP was observed rather than an increase). No urinary changes were observed. Due to the limited number of endpoints evaluated along with exposure conditions and exposure concentrations insufficiently reported, the data were not further considered for dose-response due to the inherent uncertainty.

A duration extrapolation from the 10-day inhalation study by Schwetz et al. (1974) was not conducted due to the inherent uncertainties when extrapolating from a 10-day study to a chronic duration. Likewise, a route-to-route extrapolation from the 13-week subchronic oral study Muralidhara et al. (2001) was not conducted given the differences in absorption rates across routes, method of dosing effects on blood levels and hazards (*i.e.*, gavage bolus dose vs. slower inhalation dosing), the lack of a PBPK model, and the inherent uncertainties when performing oral-to-inhalation route extrapolations for a volatile solvent (*i.e.*, most of it is eliminated in expired air). Therefore, there is inadequate data to identify an inhalation POD for the chronic duration scenario using 1,1-dichloroethane (see

Table 5-45). A 4-week short-term study in male mice exposed to 1,2-dichloroethane by Zhang et al. (2017) was thus used based on read-across to 1,1-dichloroethane. A duration extrapolation from the 4-week short-term/subchronic to a chronic duration was conducted in order to account for uncertainty. A subchronic to chronic UF of 10 was thus applied for extrapolating from a subchronic to chronic study duration. A $BMCL_5$ and BMC_5 of 6.6 ppm (26.7 mg/m^3) and 5.24 ppm (21.2 mg/m^3), were identified based on decreased sperm concentration. The intermediate inhalation HEC for occupational and continuous exposure of 22 ppm (89 mg/m^3) and 5.2 ppm (21.2 mg/m^3), respectively, with a benchmark MOE of 300, was used for risk assessment of chronic inhalation exposure.

Although an uncertainty regarding study duration may have been reduced while performing read-across by use of the chronic 104-week (Nagano et al., 2006) study that evaluated 1,2-dichloroethane, the study did not adequately evaluate non-cancer effects, preventing the determination of a non-cancer chronic POD. The study was directed to identify cancer endpoints at low doses and did not measure many non-cancer endpoints of concern. In mice, neither growth rate nor food consumption was suppressed in any 1,2-dichloroethane exposure group of either sex as compared with the respective control. The body weights of the 0, 10, 30 and 90 ppm 1,2-dichloroethane exposure groups at the end of the 2-year exposure period were 50.8 ± 6.5, 51.7 ± 6.1, 48.1 ± 8.2 and 50.7 ± 6.6 g for males and 36.6 ± 5.2, 35.8 ± 4.1, 37.4 ± 4.9 and 34.1± 4.0 g for females, respectively. No exposure related change in any hematological, blood biochemical, or urinary parameter was found in any 1,2-dichloroethane-exposed group of either sex.

Although chronic studies were identified and evaluated for the chronic duration from both 1,1- and 1,2-dichloroethane due to limitations and uncertainties in those studies, as described in Section 0 of the risk evaluation, the identified intermediate POD in Section 5.2.6.1.2 was selected for derivation of the chronic POD. The POD for the chronic inhalation exposure route is based on a duration adjusted from the respective intermediate inhalation POD (an additional subchronic-to-chronic duration extrapolation uncertainty factor of 10× to account for the duration adjustment) with a total MOE of 300.

Table 5-42 shows the recommended intermediate inhalation study used for the chronic duration and POD (in consideration of both 1,1-dichloroethane and 1,2-dichloroethane toxicity data) followed by co-critical endpoints (PODs within the range of the recommended study) and other studies considered in support of the recommended POD presented within exposure response arrays. Chronic studies from 1,1- and 1,2-dichloroethane that were evaluated but not identified and selected as the chronic POD are also presented in exposure response arrays (Figure 5-13 and Figure 5-14).

Dermal
In evaluating studies that could be considered suitable in identifying a chronic POD, no studies of chronic dermal exposures were identified from 1,1-dichloroethane database nor considered suitable from the 1,2-dichloroethane database for POD determination.

In Van Duuren (1979), a chronic cancer bioassay in which 0, 42 or 126 mg of 1,2-dichloroethane was applied to the dorsal skin of female Ha:ICR Swiss mice for 3 days/week for 581 days did not report non-cancer endpoints such as survival nor histological findings other than tumor incidences.

In Suguro (2017), a chronic cancer bioassay, male and female CB6F1-Tg rasH2@Jcl mice were dermally exposed to 1,2-dichloroethane on the shaved dorsal skin at 0 or 126 mg (equivalent to 6300 mg/kg-bw/day) in acetone, 3 days/week, for 26 weeks. This study evaluated a number of non-cancer endpoints that included clinical signs, body weights (measured weekly for the first 14 weeks and every other week thereafter), food consumption and water intake (over a 2-day period before each weighing), gross necropsy, organ weights, and histopathology. Five treated female mice were euthanized in a moribund condition, showing irregular respiration and/or emaciation during weeks 17 to 25. Treated females had significantly decreased body weight changes from week 18 to the end of the experiment compared to controls and no clinical signs or body weight effects were observed in males. At gross necropsy, discolored areas or nodules were found in the lungs of test substance-exposed animals with large-sized, discolored nodules were more prominent in females than males, with 100% lung neoplasm tumor incidence in females. The absolute and relative lung weights in treated females were significantly increased compared to those of controls. In the kidney, distal tubular mild karyomegaly was increased in test substance-exposed animals of both sexes. In females, karyomegaly was accompanied by tubular degeneration. No other chemical-related changes were observed in other organs examined at necropsy. As the study only tested one dose, a NOAEL could not be determined and made it not suitable for cancer slope factor derivation. Additionally, euthanasia prior to the scheduled termination of the study for the treated females also limited the use of this study for derivation of a chronic dermal POD.

Therefore, the chronic oral HED based on and duration adjusted from the intermediate oral POD (an additional subchronic-to-chronic duration extrapolation uncertainty factor of 10× to account for the duration adjustment) for occupational and continuous exposures of 9.1 and 6.5 mg/kg-bw/day, respectively, was extrapolated for the dermal route, with a benchmark MOE of 300, and was used for risk assessment of chronic dermal exposure (see

Table 5-45).

Table 5-41. Chronic, Oral, Non-Cancer POD-Endpoint Selection Table

Chemical/Endpoint	POD (mg/kg/day)	Study Parameters	Comments
POD selected for non-cancer risk evaluation for chronic oral exposures			
1,2-Dichloroethane, kidney weight	NOAEL = 30 (males) LOAEL = 75 (females)	NTP (1991), Gavage, SR High F344 Rats – Both sexes: 13 weeks: 0, 30, 60, 120, 240, 480 mg/kg-day (males); 0, 18, 37, 75, 150, 300 mg/kg/day (females)	
Co-critical endpoints			
1,2-Dichloroethane, decreased leukocytes	LOAEL = 4.9	Munson et al. (1982), Gavage SR High CD1 Mice – Both sexes: 14 days: 0, 4.9, 49 mg/kg-day	Supports cell-based immunosuppression endpoint.
Other studies considered			
1,1-Dichloroethane, immunotoxicity • Humoral immune response to T-dependent and T-independent antigens • Antibody-dependent cell cytotoxicity • Delayed hypersensitivity (DTH) reaction	LOAEL = 930	Zabrodskii et al. (2004), Gavage, SR Medium Random-Bred Albino Rat – Both sexes: Single dose: 0, 930 mg/kg-bw	Qualitatively supports immunosuppression. A multi-day exposure produces more sensitive PODs for immune suppression than a single exposure study. However, dose is close to LD50. Single acute exposure to one dose and monitored – various immune reactions and indices were evaluated 48 h and 5 days after exposure.
1,1-Dichloroethane, sedation	$NOAEL_{adj}$ = 714	Muralidhara et al. (2001), Gavage, SR Medium SD Rats – Male: 13 weeks: 0, 500, 1,000, 2,000, 4,000 mg/kg-bw/day	1,1-Dichloroethane Acute Oral LD50 is 725 mg/kg (PubChem), the POD is near lethal doses, Narcosis is well-known to occur at high doses and is not considered a sensitive endpoint in the database. This is the only study that passed SR with a useable subchronic oral POD. Would require a UF_S of 10 for duration extrapolation from sub-chronic to chronic and a database uncertainty factor.
1,2-Dichloroethane, immune (thymus)	NOAEL = 240 mg/kg-day (males); 150 mg/kg-day (females) LOAEL = 480 mg/kg-day for thymus necrosis in males; 300 mg/kg-	NTP (1991), Gavage, SR High (NTP 1991) F344 Rats – Both sexes: 13 weeks: 0, 30, 60, 120, 240, 480 mg/kg-day (males); 0, 18, 37, 75, 150, 300 mg/kg/day (females)	Qualitatively supports immunosuppression. However, thymus necrosis occurs at dosages where mortality was also occurring therefore cannot be used as a POD.

Chemical/Endpoint	POD (mg/kg/day)	Study Parameters	Comments
	day for thymus necrosis in females		
1,2-Dichloroethane, decreased cell based immune response	LOAEL = 4.9	Munson et al. (1982), Gavage, SR High CD1 Mice – Both sexes: 14 days: 0, 4.9, 49 mg/kg-day	
1,2-Dichloroethane, fetal resorptions	NOAEL = 160 LOAEL = 200 (data not amenable to modeling)	Payan et al. (1995), Gavage Pre-Natal Developmental, SR High SD Rats – Female: Dosing GD6–20: 0, 120, 160, 200, or 240 mg/kg	The increases in non-implants and resorptions are difficult to interpret given the significant maternal toxicity at corresponding doses (30 and 49% at 200 and 240 mg/kg/day, respectively) consisting of decreases in maternal bw gain, and the fact that there was no effect on the number of live fetuses per litter despite the changes in non-surviving implants/litter and resorption sites/litter. Therefore, cannot be used as POD.
1,2-Dichloroethane, decreases in maternal body weight gain	NOAEL = 160 LOAEL = 200 (BMD = 99.1; BMDL = 41.8)	Payan et al. (1995), Gavage Pre-Natal Developmental, SR High SD Rats – Female: Dosing GD6–20: 0, 120, 160, 200, or 240 mg/kg	A dose-related reduction in adjusted (for gravid uterine weight) maternal bodyweight gain during treatment occurred, with statistical significance achieved at the two highest doses above the maximum tolerated dose (30 and 49% reduction compared with controls, $p < 0.05$).
1,2-Dichloroethane, multigenerational/ reproductive pup weight	LOAEL = 50	Lane et al. (1982), Drinking Water, SR High ICR Mice – Both sexes: Reproductive Toxicity: 0, 5, 15 or 50 mg/kg-day	Drinking water not measured to confirm actual dosage. Also, not as sensitive (LOAE = 50) as the Immunotoxicity Endpoint (LOAEL = 4.9) Pup weight was biologically significantly (≥5%) decreased at ≥0.09 mg/ml (50 mg/kg/day) in F1/B mice.
1,2-Dichloroethane 40-week chronic study Body weight/lymphoma	LOAEL = 150 (females)	Storer et al. (1995), Gavage, SR Medium ppG64 Mice – Both sexes: 7 days/week for 40 weeks: 0, 150, 300 mg/kg-day (female); 0, 100, 200 mg/kg/day (males)	Minimal endpoints evaluated, only non-cancer endpoints were body weight and lymphoma at 150. Doses adjusted due to substantial mortality females at 300 mg/kg/day. Clear dose-response could not be assessed.
1,2-Dichloroethane, chronic 26-week dermal study	LOAEL = 6,300 Decreased body weight in females; increased distal tubular mild karyomegaly (both	Suguro et al. (2017), Dermal, SR High CB6F1- Tg rasH2@Jcl (rasH2) mice – Both sexes:	Cancer study, single dose using transgenic mice.

Chemical/Endpoint	POD (mg/kg/day)	Study Parameters	Comments
	sexes); renal karyomegaly & tubular degeneration (females)	3 days/week 26 weeks: 0, 126 mg; 0, 6,300 mg/kg-day	

Table 5-42. Chronic Inhalation Non-Cancer POD-Endpoint Selection Table

Chemical/ Endpoint	POD (mg/m^3)	Study Parameters	Comments
POD selected for non-cancer risk evaluation for chronic inhalation exposures			
1,2-Dichloroethane, male reproductive	BMCL5 = 21.2 mg/m^3 NOAEC = 350 LOAEC = 700	Zhang et al. (2017), 4 week morphological analysis of sperm parameters, SR High Swiss Mice – Male: 6 hours/day 7 days/week 4 weeks: 0, 100, 350, 700 mg/m^3	Decreases in sperm concentration.
Co-critical endpoints			
1,2-Dichloroethane, fetal development	Reproductive/ Developmental $BMCL_5$ = 25 Pup BW decreased at 613 $BMCL_{10}$ = 50 mg/m^3 NOAEC = 305 LOAEC = 613	Rao et al. (1980), Vapor, SR Medium SD Rats – Both sexes Inhalation. Prior to mating, rats were exposed for 60 days (6 hours/day, 5 days/week). The rest of the time, exposed to 6 hours/day, 7 days/week, except from gestational day 21-post natal day 4 maternal exposure stopped to allow for delivery and rearing of the young). Two F1 generations were evaluated: 0,25,75,150 ppm; 0, 102, 305 or 613 mg/m^3	Decreased body weight of selected F1B male weanlings at 150 ppm. Study used for co-critical endpoints with $BMCL_{10}$ very close to that from the recommended POD. Considering NOAECs/LOAECs, using the recommended endpoint will be protective of the decreases in pup body weight. Also, portal of entry effects can be considered more sensitive than systemic effects.
Other studies considered			
1,2-Dichloroethane	Reproductive/ Developmental NOAEC = 1,200 Maternal Toxicity: NOAEC = 1,000 LOAEC = 1,200	Payan et al. (1995), Vapor, SR High SD Rats – Both sexes: Inhalation exposure for 2 weeks. GD 6–20. 6 hours/day 7 days/week: 0, 150, 200, 250, 300 ppm; 0, 610, 820, 1,000, 1,200 mg/m^3	Repro/Dev Toxicity: Pregnancy rate among females at 250 ppm was significantly lower; not observed at the highest concentration of 300 ppm; no other significant effects reported. Maternal Toxicity: 2/26 dams died at 300 ppm (highest dose). Maternal body weight gain at GD 6–21 was significantly decreased at 300 ppm. No mention of food consumption. NOAEC/LOAEC higher than recommended POD. Not amenable to BMD modeling.
1,2-Dichloroethane	Reproductive/ Developmental LOAEL = 405	Rao et al. (1980), Vapor, SR Medium SD Rats – Female:	Developmental Toxicity: A significant decrease in the incidence of bilobed thoracic centra was seen at 100 ppm however study essentially becomes a

Chemical/ Endpoint	POD (mg/m^3)	Study Parameters	Comments
	Maternal Toxicity: NOAEC = 405 LOAEC = 1,214	Inhalation exposure for 10 days. GD 6–15. 7 hours/day: 0, 100, 300 ppm (0, 405, 1,214 mg/m^3)	single dose study and not amenable to dose-response modeling due to the high maternal toxicity at 300 ppm (10/16 maternal rats died at 300 ppm). Therefore, this study is not acceptable for POD derivation.
1,2-Dichloroethane	Hematological: NOAEC = 202 LOAEC = 607 Liver: LOAEC = 20 Kidney: NOAEC = 202 LOAEC = 607	IRFMN (1978), Vapor, SR Medium SD Rats – Both sexes: 7 hours/day, 5 days/week for 12 months: 0, 5, 10, 50, 150 ppm; 0, 20, 40, 202, 607 mg/m^3	Hemoglobin levels were significantly decreased in both sexes at 150 ppm; changes in hematocrit (increases rather than decreases) were of questionable biological significance and did not show a dose-response; decreases in cholesterol and calcium levels at ≥10 ppm; clinical chemistry signs of liver toxicity but did not show a dose-response, kidney BUN increases at 150 ppm; other kidney changes were male rat-specific and not relevant to humans.
1,2-Dichloroethane	Reproductive/ Developmental, Mortality & Metabolic: NOAEC = 204 Liver: LOAEC = 204	Cheever et al. (1990), Vapor, SR High SD Rats – Both sexes: 7 hours/day 5 days/week 104 weeks: 0, 50 ppm; 0, 204 mg/m^3	Gross testicular lesions were found in higher frequency in exposed males (24%) compared to control (10%) (data not shown and gross pathologic observations were not evaluated statistically); mortality similar in both treatment and control groups, survival rate in exposed rats (60 and 64%) was similar to control (58 and 54%) in males and females, respectively; absolute and relative liver weights were not different from controls.
1,2-Dichloroethane	Immunological/ Hematological, Liver, & Kidney: NOAEC = 809	IRFMN (1976), Vapor, SR Medium SD Rats – Both sexes: 7 hours/day 5 days/week 24 weeks: 0, 5, 10, 50, 150, 250 ppm; 0, 20, 40, 202, 607, 1,012 mg/m^3* *Animals in the highest exposure group were exposed to 250 ppm for "a few weeks" and then the exposure concentration was reduced to 150 ppm due to acute toxicity. A reliable TWA concentration cannot be determined based on the information available in this report, IRFMN (1978) suggested that the change occurred after 12 weeks of exposure. If this is accurate, then the TWA exposure concentration for the high exposure group was 200 ppm.	All observed hematological, serum chemistry, and urinalysis changes observed either did not reach statistical significance, showed no clear relation to exposure concentration, and/or were not biologically significant.
1,2-Dichloroethane	Immunological/ Hematological, Liver, and Kidney:	IRFMN (1987b), Vapor, SR Medium SD Rats – Both sexes:	Significant decrease in segmented neutrophils in the high exposure group in males; no other hematological changes were observed; serum liver

Chemical/ Endpoint	POD (mg/m^3)	Study Parameters	Comments
	NOAEC = 607	7 hours/day 5 days/week 78 weeks: 0, 5, 10, 50, 150, 250 ppm; 0, 20, 40, 202, 607, 1,012 mg/m^3* *Animals in the highest exposure group exposed to 250 ppm for "a few weeks" and then the exposure concentration was reduced to 150 ppm due to acute toxicity. A reliable TWA concentration cannot be determined based on the information available in this report, IRFMN (1978) suggested that the change occurred after 12 weeks of exposure. If this is accurate, then the TWA exposure concentration for the high exposure group was 200 ppm.	and kidney chemistry changes either did not reach statistical significance, showed no clear relation to exposure, concentration, and/or were not biologically significant; no urinary changes were observed.
1,2-Dichloroethane	Mortality (Rats): NOAEC = 654 Mortality (Mice): NOAEC = 368	Nagano et al. (2006) F344 Rats – Both sexes: 6 hours/day 5 days/week 104 weeks total: 0, 10, 40, 160 ppm; 0, 41, 164, or 654 mg/m^3 Crj:BDF1 Mice – Both sexes: 6 hours/day 5 days/week 104 weeks total: 0, 10, 30, 90 ppm; 0, 41, 123, or 368 mg/m^3	Endpoints evaluated included mortality, clinical signs of toxicity, body weight, food consumption, hematology, blood biochemistry, urinalysis, organ weight, gross necropsy of organs & histopathology. No significant effects reported.
1,2-Dichloroethane	Immune/ Hematological, Nutritional/ Metabolic, Liver, Mortality, and Kidney (Rats/Rabbits/ Guinea Pigs/ Cats): NOAEC = 405	Hofmann et al. (1971), Vapor, SR Medium SD Rats – Both sexes; Bunte Rabbits – Both sexes; Pirbright-White Guinea; Pigs – Both sexes; Cats – Both sexes: 6 hours/day 5 days/week 17 weeks: 0, 100 ppm; 0, 405 mg/m^3	The endpoints evaluated included mortality, body weights, hematological effects (blood counts, not further specified), liver effects (serum AST and ALT, liver weight, and liver histology), and renal effects (BUN and serum creatinine, urinary status – not further specified, kidney weight, and kidney histology); bromsulphthalein test in rabbits and cats does not indicate liver effects. Rats, cats, and guinea pigs: No significant effects reported. One of 4 rabbits showed increased BUN and kidney histology (not further specified); the observation of these effects in 1 rabbit was not considered adverse (or of questionable adversity).
1,2-Dichloroethane	Neurological, Liver, & Mortality (Rabbits): Not determined	Spencer et al. (1951), Vapor, SR Medium Rabbit – Both sexes: 7 hours/day 5 days/week 248 days*: 0, 100, 400 ppm; 0, 405, 1,619 mg/m^3	No significant effects reported in rabbits; histopathological changes reported in the liver and kidney in monkeys; mortality observed in rats and guinea pigs; uncertain signs of body weight changes, and possible

Chemical/ Endpoint	POD (mg/m^3)	Study Parameters	Comments
1,2-Dichloroethane	Hematological, Kidney, Liver, & Mortality (Monkeys): NOAEC: 405	*The exact duration of exposure is unclear. At 400 ppm rabbits "tolerated" exposure for 232 days" and at 100 ppm, rabbits "tolerated" exposure for 248 days without signs of adverse effects; the time of termination is not specified. Monkeys – Males: 7 hours/day 5 days/week 212 days*: 0, 100, 400 ppm; 0, 405, 1,619 mg/m^3 *At 400 ppm both Monkeys were killed in a moribund state after 8 and 12 exposures, respectively. The duration noted above applies only to the 100 ppm group. Wistar Rats – Both sexes: 7 hours/day 5 days/week 212 days*: 0, 100, 400 ppm; 0, 405, 1,619 mg/m^3 *Although all exposure groups were intended for chronic duration exposures, animals at the high exposure level died within 14 days (females) and 56 days (males). Guinea Pigs – Both sexes 7 hours/day 5 days/week 248 days: 0, 100, 200, 400 ppm; 0, 405, 809, 1,619 mg/m^3	signs of liver and kidney toxicity in guinea pigs but the data either did not show dose-response, or quantal data for these endpoints or incidence values and a statement whether any control animals exhibited these changes were not included.

5.2.6.1.5 Exposure-Response Arrays

The following exposure-response array provide a visualization of the studies identified for 1,1- and 1,2dichlororethane that were evaluated for consideration as candidate non-cancer PODs for dose-response. The studies are organized by study duration (acute, intermediate, or chronic), and route of exposure (oral or inhalation; no studies were identified by the dermal route for non-cancer PODs) and categorized by health outcome category and effect. The values for the data points and corresponding study reference key legend are presented in these exposure-response arrays are provided in supplemental file *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Human Health Hazard Exposure Response Array Data and Figures* (U.S. EPA, 2025g).

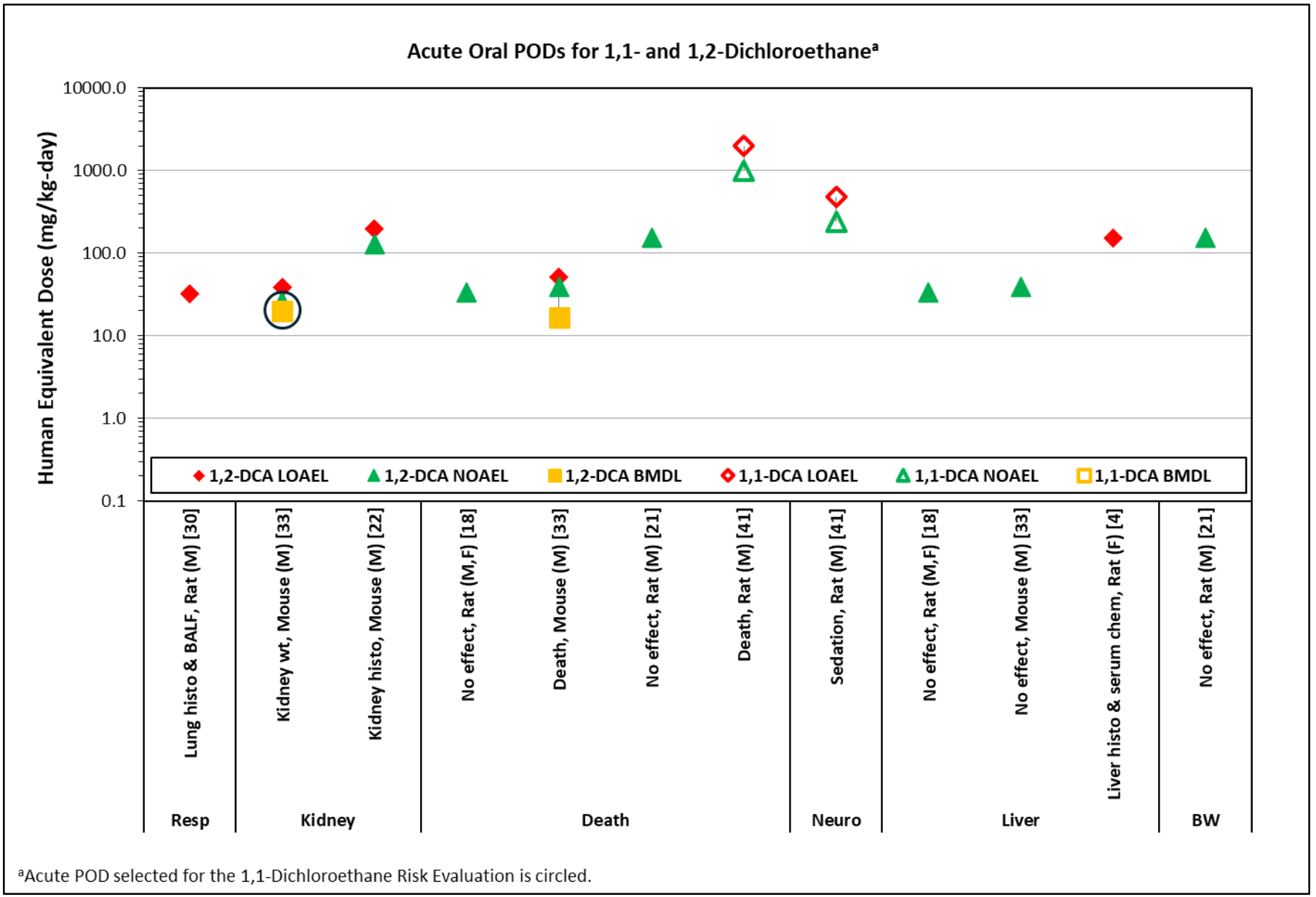


Figure 5-7. Acute Oral Exposure Response Array for 1,1- Dichloroethane and 1,2-Dichloroethane

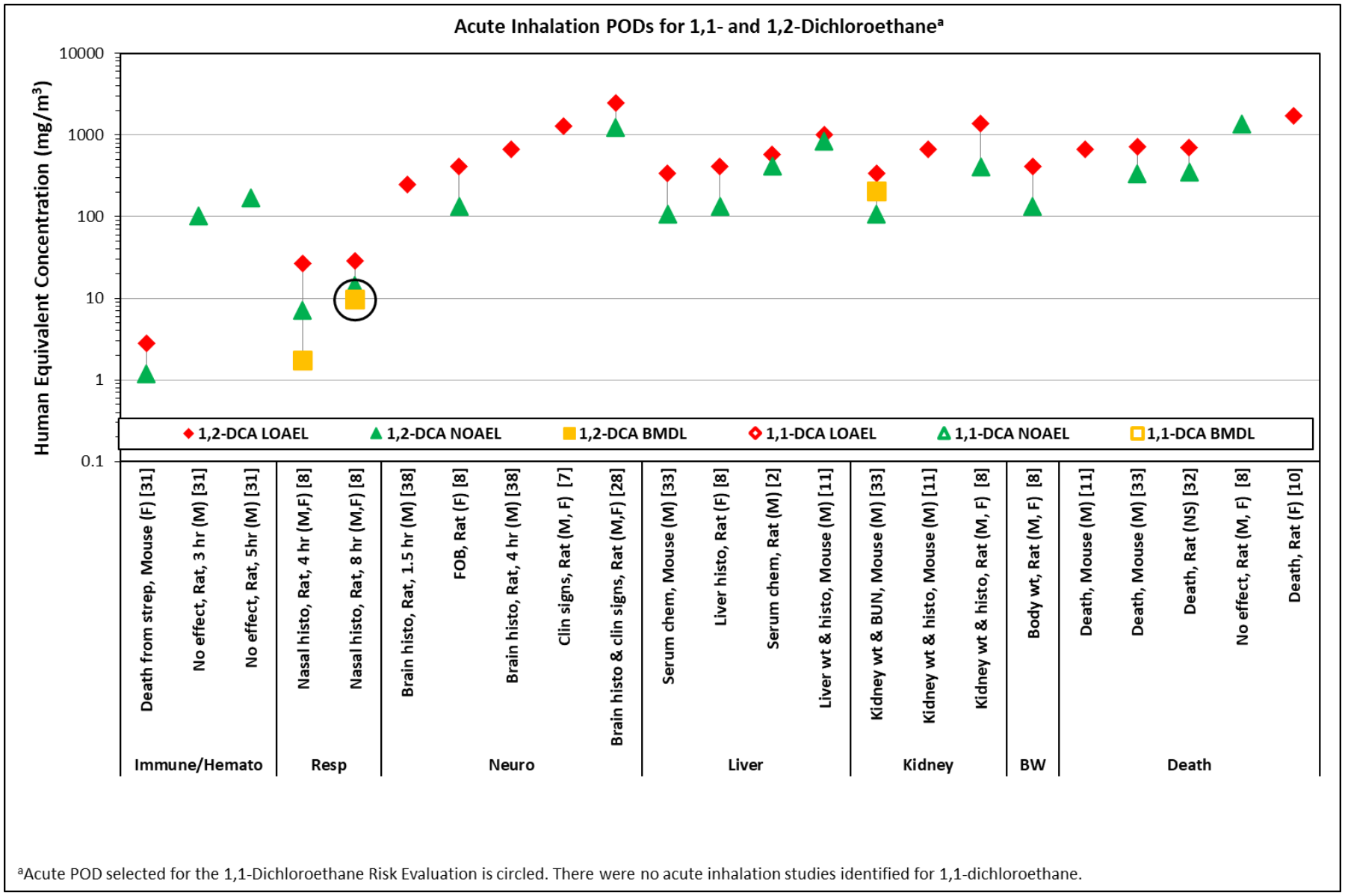
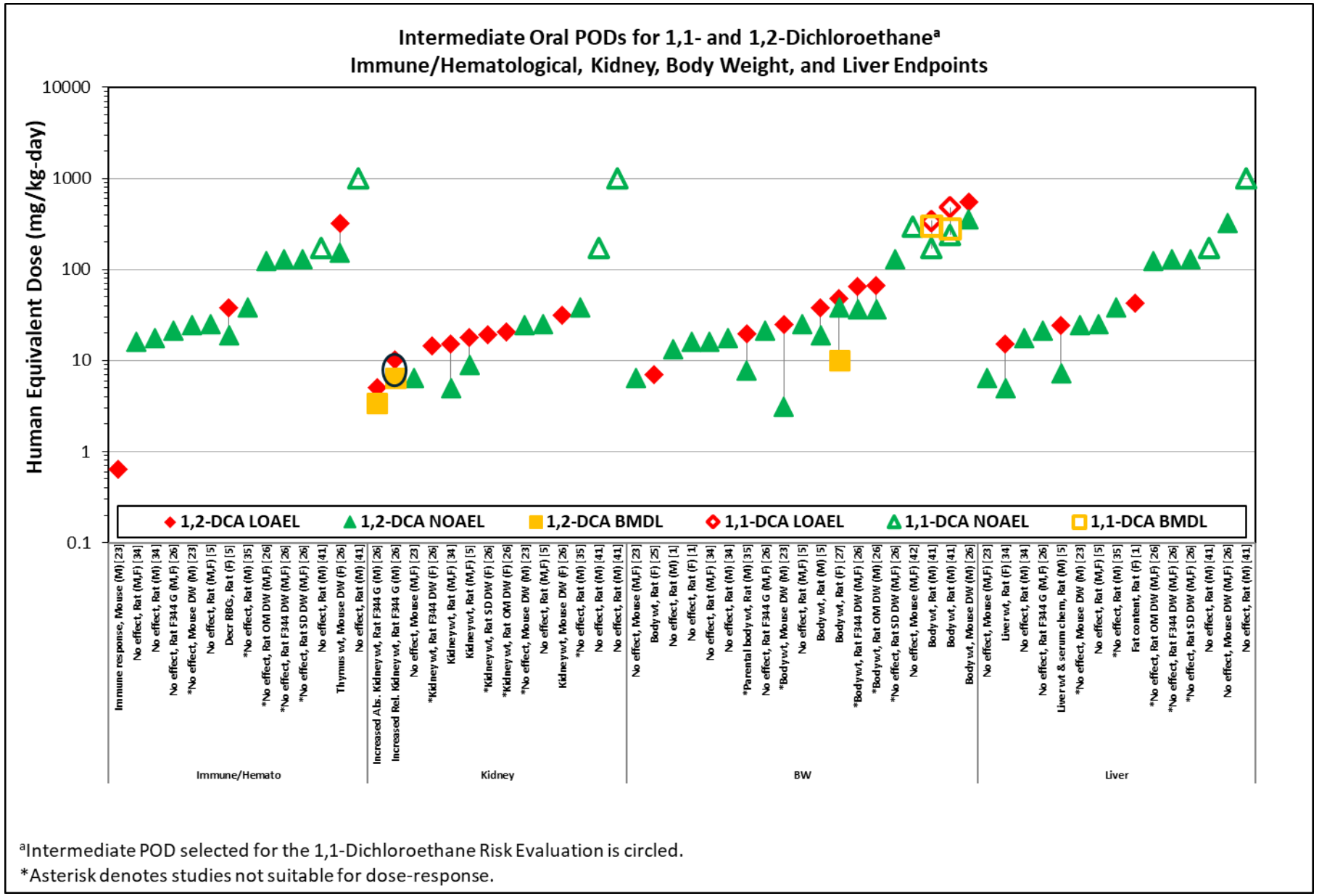


Figure 5-8. Acute Inhalation Exposure Response Array for 1,1-Dichloroethane and 1,2-Dichloroethane



[a]Intermediate POD selected for the 1,1-Dichloroethane Risk Evaluation is circled.
*Asterisk denotes studies not suitable for dose-response.

Figure 5-9. Intermediate Oral Exposure Response Array for 1,1- Dichloroethane and 1,2-Dichloroethane for Immune/ Hematological, Kidney, Body Weight, and Liver Endpoints

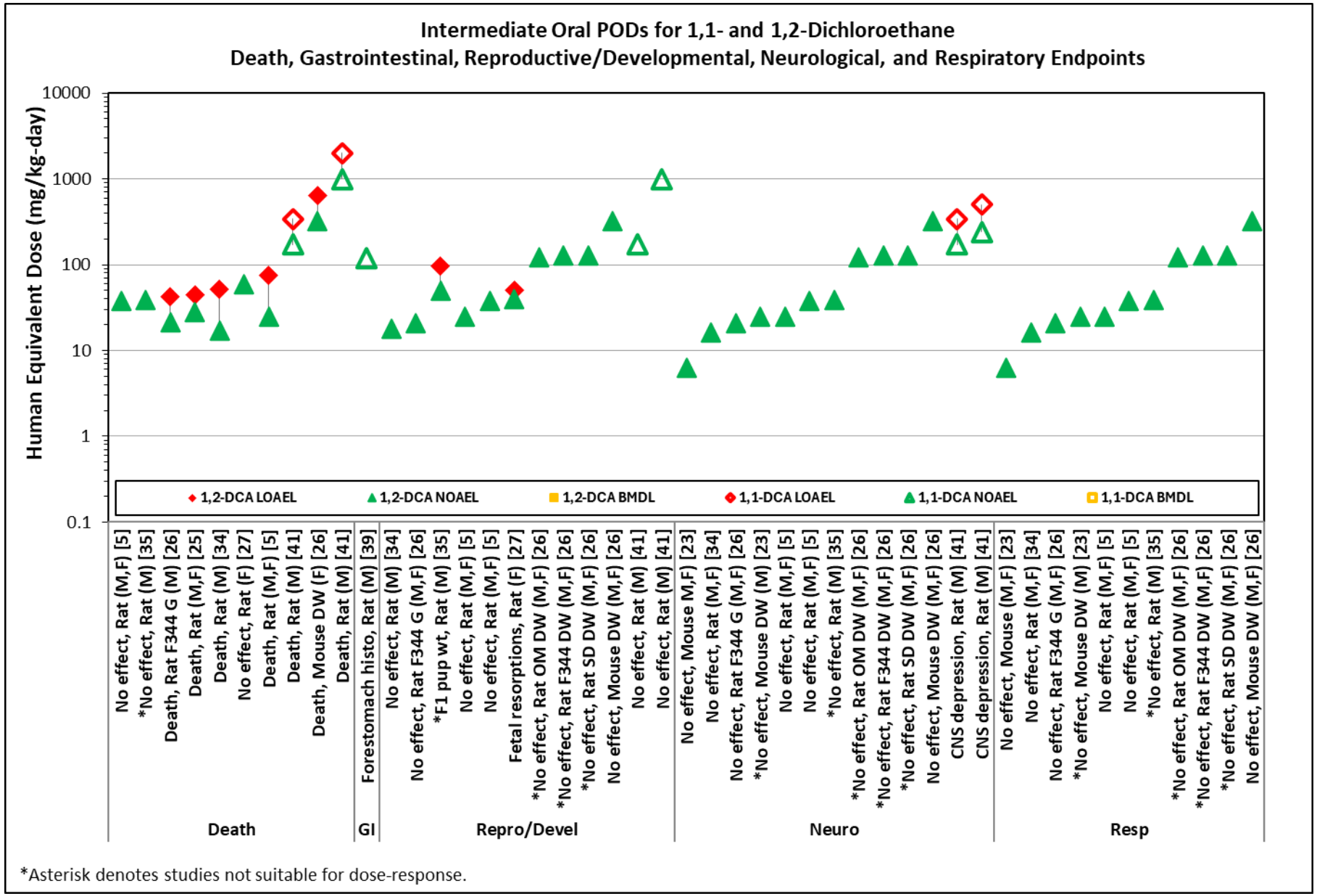


Figure 5-10. Intermediate Oral Exposure Response Array for 1,1-Dichloroethane and 1,2-Dichloroethane for Death, Gastrointestinal, Reproductive/Developmental, Neurological, and Respiratory Endpoints

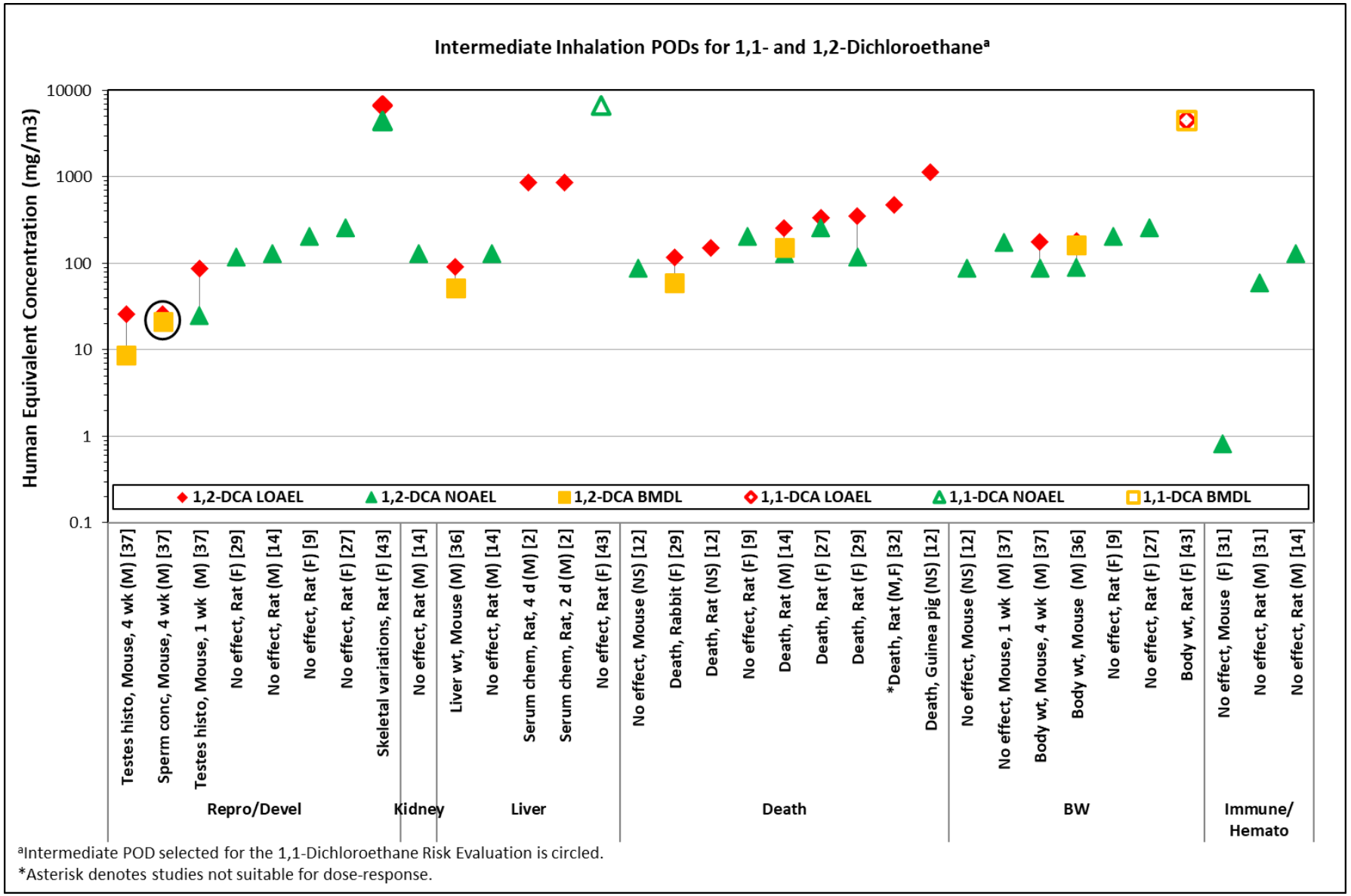


Figure 5-11. Intermediate Inhalation Exposure Response Array for 1,1-Dichloroethane and 1,2-Dichloroethane

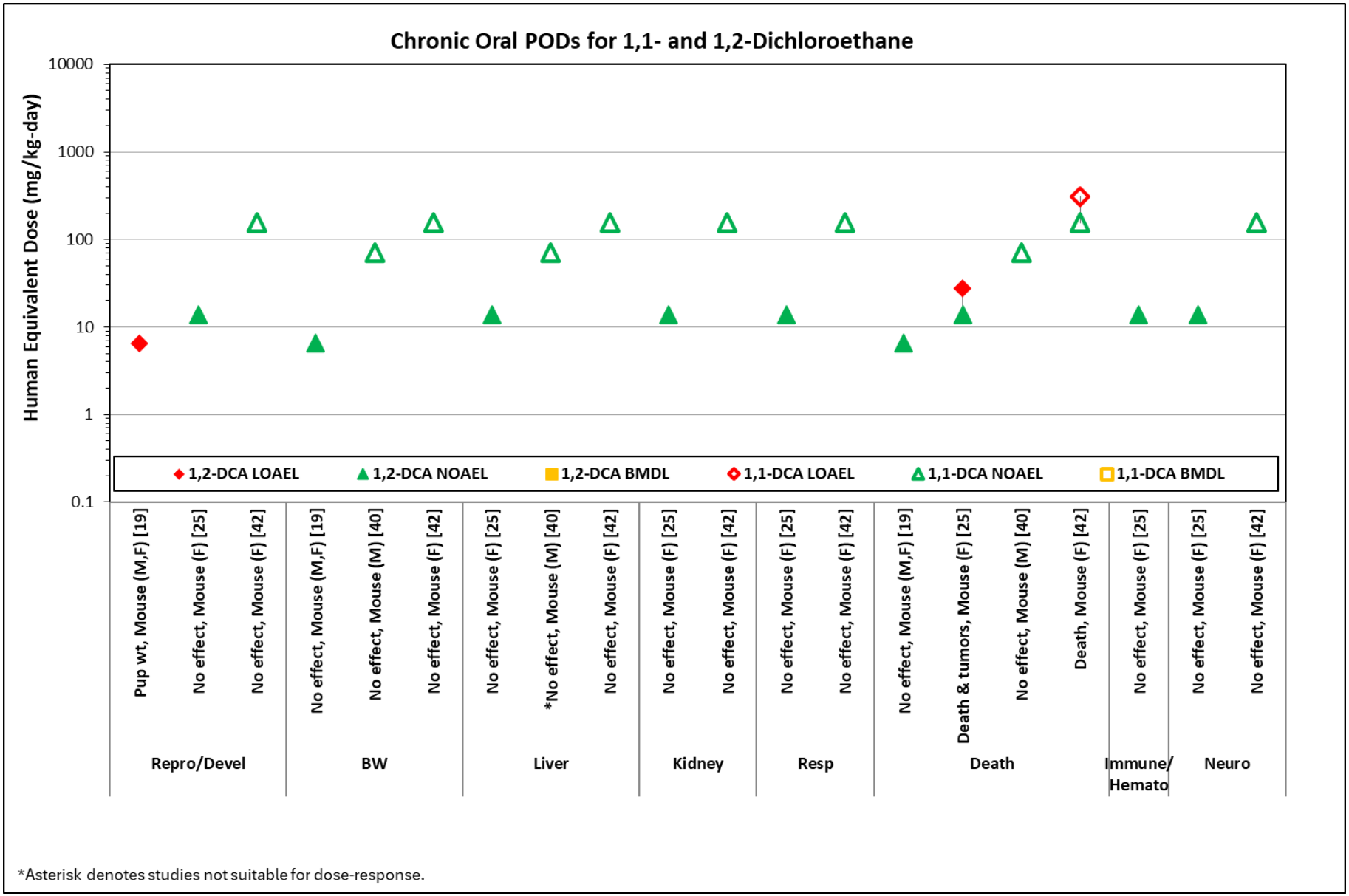


Figure 5-12. Chronic Oral Exposure Response Array for 1,1-Dichloroethane and 1,2-Dichloroethane

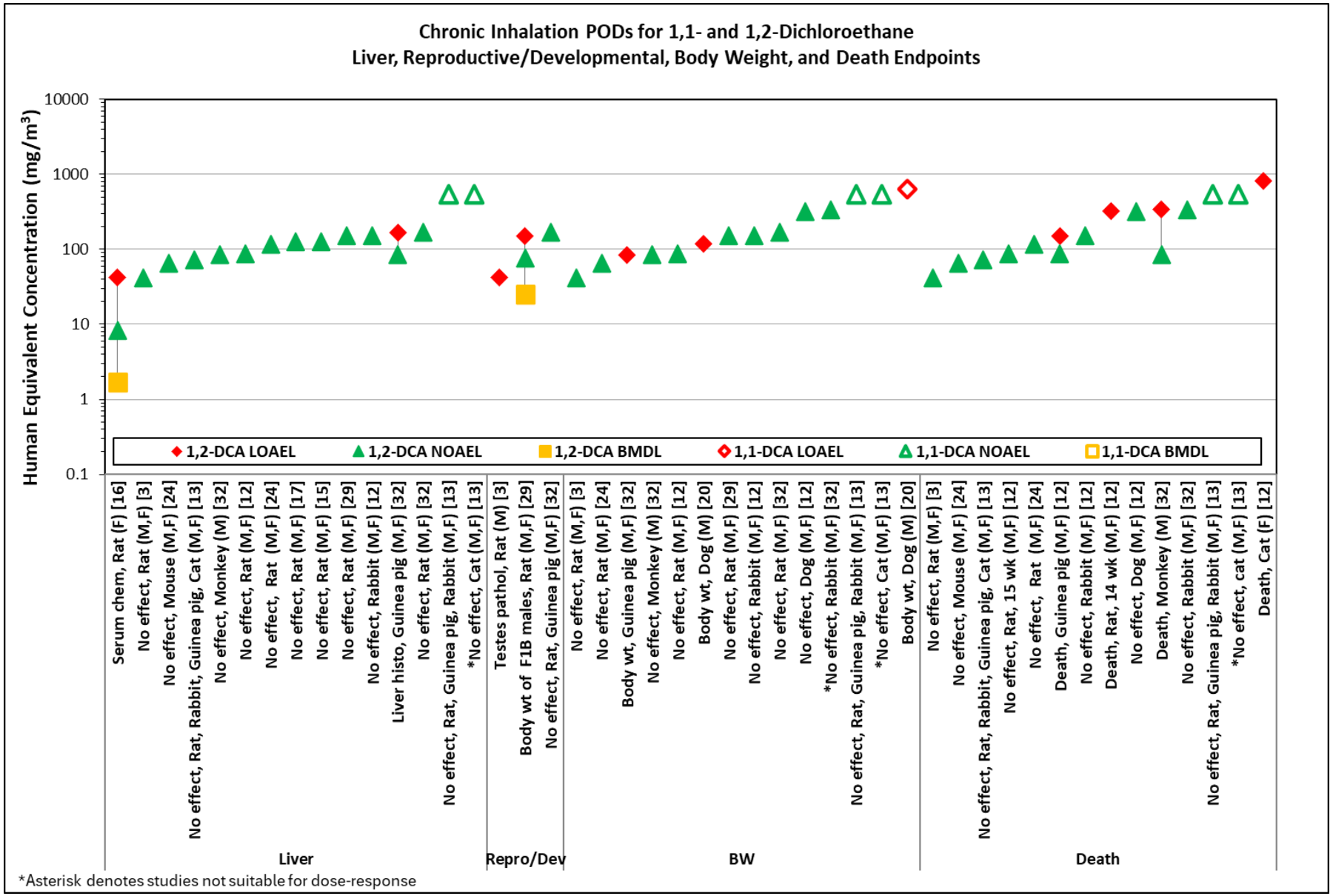


Figure 5-13. Chronic Inhalation Exposure Response Array for 1,1-Dichloroethane and 1,2-Dichloroethane for Liver, Reproductive, Developmental, Body Weight, and Death Endpoints

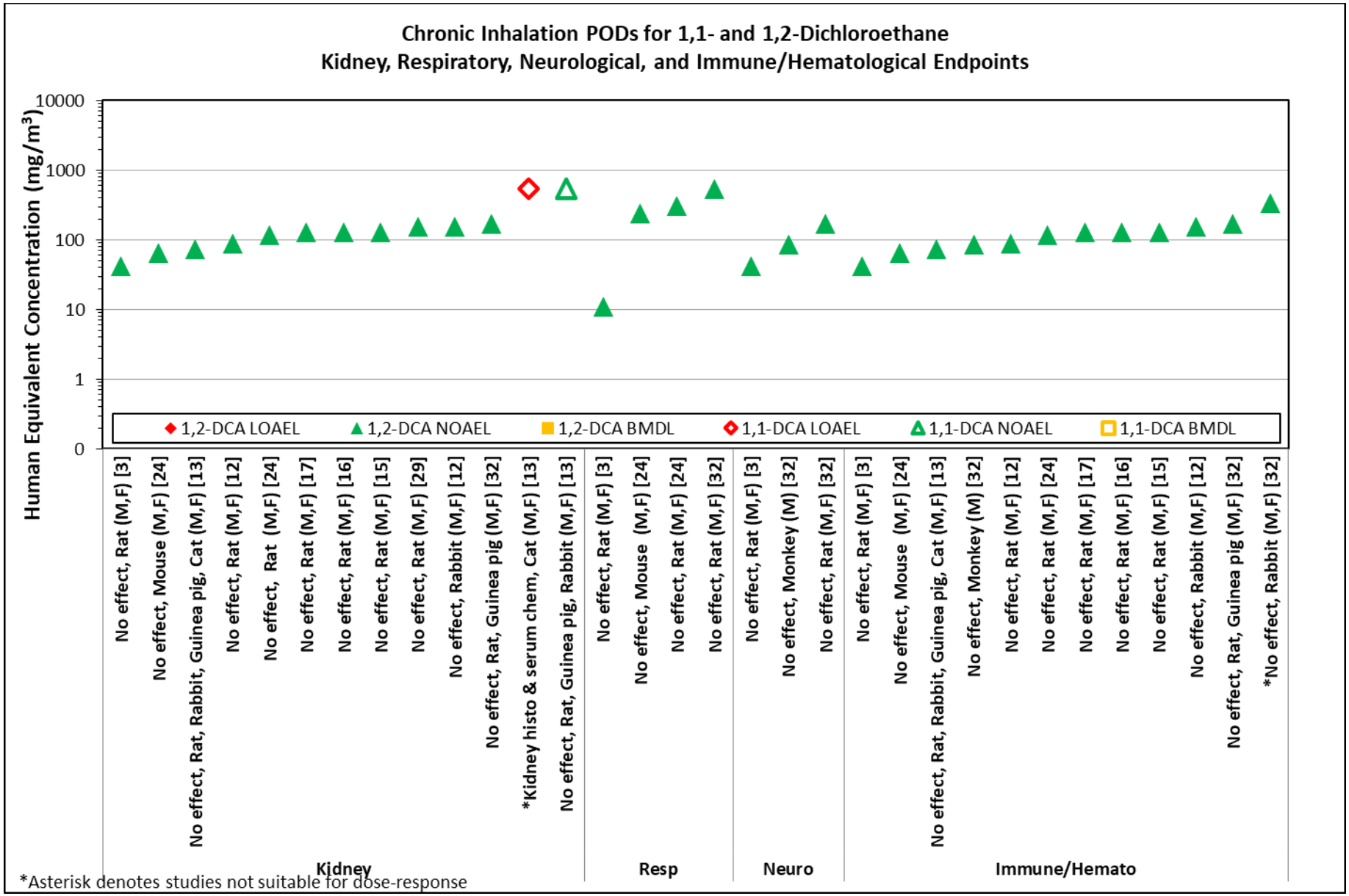


Figure 5-14. Chronic Inhalation Exposure Response Array for 1,1-Dichloroethane and 1,2-Dichloroethane for Kidney, Respiratory, Neurological, and Immune/Hematological Endpoints

5.2.6.2 Endpoint Derivation for Carcinogenic Dose-Response Assessment

1,2-Dichloroethane IUR for Inhalation Exposures (Read-Across to 1,1-Dichloroethane)

In 1987, the IRIS program derived an IUR of 2.6×10^{-5} (per µg/m^3) based on route-to-route extrapolation from the oral CSF derived within the report. The inhalation cancer bioassay by Nagano et al. (2006) was not available at the time of the IRIS assessment, thus allowing for the 1,1-dichloroethane risk evaluation to update and derive an IUR based on the inhalation route thus minimizing uncertainties associated with the route-to-route extrapolation. The (Nagano et al., 2006) inhalation study for 1,2-dichloroethane was used to derive the IUR value for read-across to 1,1-dichloroethane. Additionally, qualitative assessment of the oral gavage study by NTP (1978) for 1,2-dichloroethane identified similar tumor types to those identified in (Nagano et al., 2006).

A BMR of 10 percent extra risk was selected for all datasets. HECs were calculating using the ratio of blood:gas partition coefficients, as shown in Appendix N.1.2. Gargas and Andersen (1989) estimated blood:air partition coefficients for 1,2-dichloroethane of 19.5 and 30.4 in humans and rats, respectively. Because the rat partition coefficient is greater than the human partition coefficient, the default ratio of 1 is used in the calculation in accordance with U.S. EPA (1994) guidance. A blood:air partition coefficient for mice was not available from the literature reviewed; thus, the default ratio of 1 was used to calculate HECs for data in mice.

Details of the BMD modeling are provided in *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Benchmark Dose Modeling* (U.S. EPA, 2025e) and the BMCL, HEC, and IUR estimate for each dataset is shown in Appendix N.

The highest estimated IUR was calculated to be 6.2×10^{-6} (per µg/m^3) for combined mammary gland adenomas, fibroadenomas, and adenocarcinomas and subcutaneous fibromas in female rats in the inhalation study by Nagano et al. (2006) the incidences of the mammary tumors and subcutaneous fibromas showed a significant positive trend with increased concentration and were significantly different from the control group at 160 ppm (combined mammary tumors were also significantly different from controls at 40 ppm). The incidences of mammary tumors in the control group were at incidence rates that did not exceed the maximum tumor incidences when compared to historical controls and thus retained in the analysis.

CSF for Oral Exposures

The IRIS program derived an oral CSF of 9.1×10^{-2} (per mg/kg-bw/day) for 1,2-dichloroethane in 1987 based on the incidence of hemangiosarcomas in male rats in the chronic bioassay by NTP (1978); however, this study did not pass EPA systematic review. The oral CSF for male mice based on hepatocarcinomas was 6.2×10^{-3} (per mg/kg-bw/day) in NTP (1978). No oral cancer bioassays of 1,2-dichloroethane have been published since the IRIS 1987 assessment. The IRIS CSF was derived using time-to-tumor modeling to account for intercurrent mortality of the rats in the NTP (1978) study. No updates to the time-to-tumor modeling approach have been made since the IRIS 1987 assessment. Hemangiosarcomas in male rats were determined to be the most sensitive species, strain, and site; however, this study was not utilized in this assessment due to confounding associated with increased mortality in all dose groups. Due to limitations associated with both the rats and mice in the NTP (1978) study for 1,2-dichloroethane, EPA is not pursuing a quantitative assessment of cancer risks associated with a CSF derived from 1,2-dichloroethane hazard and exposures to 1,1-dichloroethane.

CSF for Dermal Exposures

There were no identified dermal cancer studies for either 1,1- or 1,2-dichloroethane for quantitative dose-response. The 1,2-dichloroethane dermal study by Suguro et al. (2017) did identify bronchioalveolar adenomas and adenocarcinomas; however, its single dose did not allow calculation of an accurate dermal linear low-dose cancer slope factor (Suguro et al., 2017). A dermal CSF was not derived from 1,1- or 1,2-dichloroethane via route-to-route extrapolation using oral data. Additionally, there are uncertainties associated with extrapolation using 1,2-dichloroethane data from both oral and inhalation dosing for 1,1-dichloroethane dermal route. Use of an oral POD for dermal extrapolation may not be preferred for chemicals known to undergo extensive liver metabolism because the "first-pass effect" that directs intestinally absorbed chemicals to the liver applies to oral ingestion. However, PBPK research also indicates extra-hepatic metabolism for 1,2-dichloroethane. The accuracy of extrapolation of inhalation toxicity data for dermal PODs is dependent on assumptions about inhalation exposure factors such as breathing rate and any associated dosimetric adjustments. However, whole-body inhalation studies may also already be incorporating some level of dermal exposure. Given these uncertainties, in the absence of 1,1-dichloroethane data to support derivation of a dermal CSF from an oral CSF or an inhalation IUR, a dermal CSF was not derived.

5.2.6.3 PODs for Non-Cancer and Cancer Human Health Hazard Endpoints

Table 5-43, Table 5-44, and Table 5-45 list the non-cancer PODs and corresponding HECs, HEDs, and UFs that EPA used in the 1,1-dichloroethane risk evaluation to estimate risks following acute, intermediate, and chronic exposure, respectively. Table 5-46 provides the cancer PODs for evaluating lifetime exposure.

Table 5-43. PODs and Toxicity Values Used to Estimate Non-Cancer Risks for Acute Exposure Scenarios[a]

Target Organ/ System [a]	Species/ Gender	Duration/ Route	Study POD/Type	Effect	Worker HEC [b] (ppm) [mg/m^3]	Continuous HEC [b] (ppm) [mg/m^3]	Worker HED [c] (mg/kg-bw/day)	Continuous HED [c] (mg/kg-bw/day)	Uncertainty Factors [g]	Total Uncertainty Factors	Reference	Data Quality
Renal	Mice (male)	**Oral 1,2-dichloroethane data** 1-day oral gavage	$BMDL_{10}$ = 153 mg/kg BMD = 270 mg/kg	Increased kidney weight	N/A	N/A	19.9	19.9	UF_A = 3 UF_H = 10	30 [d]	Storer et al. (1984)	High
Respiratory	Rats (males and females combined)	**Inhalation 1,2-dichloroethane data** 8-hour inhalation	$BMCL_{10}$ = 48.9 mg/m^3 or 12.1 ppm BMC_{10} = 81.4 mg/m^3 or 20.1 ppm	Degeneration with necrosis of the olfactory mucosa	10.14 ppm [41.1 mg/m^3]	2.42 ppm [9.78 mg/m^3]	N/A	N/A	UF_A = 3 UF_H = 10	30 [e]	Dow Chemical (2006)	High
Renal	Mice (male)	**Dermal (extrapolated from oral) 1,2-dichloroethane data** 1-day oral gavage	$BMDL_{10}$ = 153 mg/kg BMD = 270 mg/kg	Increased kidney weight	N/A	N/A	19.9	19.9	UF_A = 3 UF_H = 10	30 [f]	Storer et al. (1984)	High

[a] See Section 5.2.1.1 for details.
[b] BMCL10 of 48.9 mg/m^3 continuous adjusted × RGDR value (0.2) = 9.78 mg/m^3 for the HEC for continuous (adjusted for 24 hours). The HEC for the worker is the HEC_{cont} × 4.2 (hours in a week divided by the # of working hours in a week; 168/40) = 41.1 mg/m^3. Both HEC worker and continuous were converted to ppm by dividing by a factor of 4.05 (based 24.45/MW).
[c] $BMDL_{10}$ of 153 × DAF (0.13 $BW^{3/4}$ for mice) = 20.3 mg/kg. All oral PODs were first adjusted to 7 days/week and inhalation PODs adjusted to 24 hours/day, 7 days/week (continuous exposure). All continuous oral PODs were then converted to HEDs using DAFs. Dermal PODs were set equal to the oral HED. It is often necessary to convert between ppm and mg/m^3 due to variation in concentration reporting in studies and the default units for different OPPT models. Therefore, EPA presents all inhalation PODs in equivalents of both units to avoid confusion and errors. PODs converted for use in worker exposure scenarios were adjusted to 8 hours/day, 5 days/week and converted to HECs.
[d] No PODs were identified from acute exposure by the **oral route** to 1,1-dichloroethane; therefore, read-across from 1,2-dichloroethane was used to identify a POD. An acute-duration oral HED for both worker and continuous exposure of 19.9 mg/kg-bw/day was used for risk assessment of acute oral exposure, with a total uncertainty factor of 30, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used and 10 for human variability.
[e] No PODs were identified from acute exposure by the **inhalation route** to 1,1-dichloroethane; therefore, read-across from 1,2-dichloroethane was used to identify a POD. An acute-duration inhalation HEC of 10.14 ppm for worker and 2.42 ppm for continuous exposures was used for risk assessment of acute inhalation exposure, with a total uncertainty factor of 30, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used and 10 for human variability.
[f] No PODs were identified from acute exposure by the **dermal route** to 1,1-dichloroethane; therefore, route-to-route extrapolation from the oral route was used to identify a POD. An acute-duration dermal HED for both worker and continuous exposure of 19.9 mg/kg-bw/day was used for risk assessment of acute dermal exposure, with a total uncertainty factor of 30, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used and 10 for human variability.

Target Organ/ System [a]	Species/ Gender	Duration/ Route	Study POD/Type	Effect	Worker HEC [b] (ppm) [mg/m^3]	Continuous HEC [b] (ppm) [mg/m^3]	Worker HED [c] (mg/kg-bw/day)	Continuous HED [c] (mg/kg-bw/day)	Uncertainty Factors [g]	Total Uncertainty Factors	Reference	Data Quality

[g] UF = uncertainty factor; UF_A = extrapolation from animal to human (interspecies); UF_H = potential variation in sensitivity among members of the human population (intraspecies); UF_S = use of a short-term study for long-term risk assessment

Table 5-44. PODs and Toxicity Values Used to Estimate Non-Cancer Risks for Intermediate Exposure Scenarios[a]

Target Organ System	Species	Duration/ Route	Study POD/ Type	Effect	Worker HEC [b] (ppm) [mg/m^3]	Continuous HEC [b] (ppm) [mg/m^3]	Worker HED [c] (mg/kg-bw/day)	Continuous HED [c] (mg/kg-bw/day)	Uncertainty Factors [g]	Total Uncertainty Factors	Reference	Data Quality
Renal	Rats (male)	**Oral** **1,2-dichloroethane data** 13-weeks oral gavage	$BMDL_{10}$ = 27 mg/kg BMD_{10} = 33 mg/kg	Increased relative kidney weight	N/A	N/A	9.1	6.5	UF_A = 3 UF_H = 10	30 [d]	NTP (1991)	High
Reproductive	Mice (male)	**Inhalation** **1,2-dichloroethane data** 4-week morphological analysis of sperm parameters/ inhalation	$BMCL_5$ = 21.2 mg/m^3 or 5.2 ppm BMC_5 = 26.7 mg/m^3 or 6.6 ppm	Decreases in sperm concentration	22.0 ppm [89.0 mg/m^3]	5.2 ppm [21.2 mg/m^3]	N/A	N/A	UF_A = 3 UF_H = 10	30 [e]	Zhang et al. (2017)	High
Renal	Rats (male)	**Dermal (extrapolated from oral)** **1,2-dichloroethane data** 13-weeks oral gavage	$BMDL_{10}$ = 27 mg/kg BMD_{10} = 33 mg/kg	Increased relative kidney weight	N/A	N/A	9.1	6.5	UF_A = 3 UF_H = 10	30 [d]	NTP (1991)	High

Target Organ System	Species	Duration/ Route	Study POD/ Type	Effect	Worker HEC [b] (ppm) [mg/m^3]	Continuous HEC [b] (ppm) [mg/m^3]	Worker HED [c] (mg/kg-bw/day)	Continuous HED [c] (mg/kg-bw/day)	Uncertainty Factors [g]	Total Uncertainty Factors	Reference	Data Quality

[a] See Section 5.2.1.2.1 for details.

[b] BMCL5 = 21.2 mg/m^3 was adjusted to continuous adjusted (with no respiratory effects, there is no RGD; the blood:air ratio = 1, based on Equation_Apx N-9; therefore, the HEC_{cont} is the same as the adjusted POD of 21.2 mg/m^3. The HEC worker is the HEC_{cont} × 4.2 (hours in a week divided by the # of working hours in a week; 168/40) = 89.0 mg/m^3. Both HEC worker and continuous converted to ppm divided by a factor of 4.05 (based 24.45/MW).

[c] All oral PODs were first adjusted to 7 days/week. All continuous oral PODs were then converted to HEDs using DAFs. Dermal PODs were set equal to the oral HED. It is often necessary to convert between ppm and mg/m^3 due to variation in concentration reporting in studies and the default units for different OPPT models. Therefore, EPA presents all PODs in equivalents of both units to avoid confusion and errors. PODs converted for use in worker exposure scenarios were adjusted to 8 hours/day, 5 days/week and converted to HECs.

[d] No PODs were identified from short-term/subchronic exposure by the **oral route** to 1,1-dichloroethane; therefore, read-across from 1,2-dichloroethane was used to identify a POD. An intermediate oral HED for worker of 9.1 mg/kg-bw/day and a HED for continuous exposure of 6.5 mg/kg-bw/day was used for risk assessment of intermediate oral exposure, with a total uncertainty factor of 30, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used and 10 for human variability.

[e] No PODs were identified from short-term/subchronic exposure by the **inhalation route** to 1,1-dichloroethane. Therefore, read-across from 1,2-dichloroethane was used to identify a POD. A short-term/subchronic-duration inhalation HEC for worker exposure of 89.0 mg/m^3, and a HEC for continuous exposure of 21.2 mg/m^3, was used for risk assessment of short-term/subchronic inhalation exposure, with a total uncertainty factor of 30, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used and 10 for human variability.

[f] No PODs were identified from short-term/subchronic exposure by the **dermal route** to 1,1-dichloroethane; therefore, route-to-route extrapolation from the oral route was used to identify a POD. An intermediate dermal HED for worker of 9.1 mg/kg-bw/day and a HED for continuous exposure of 6.5 mg/kg-bw/day was used for risk assessment of intermediate dermal exposure, with a total uncertainty factor of 30, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used and 10 for human variability.

Table 5-45. PODs and Toxicity Values Used to Estimate Non-Cancer Risks for Chronic Exposure Scenarios[a]

Target Organ System	Species	Duration/ Route	Study POD/ Type	Effect	Worker HEC [b] (ppm) [mg/m^3]	Continuous HEC [b] (ppm) [mg/m^3]	Worker HED [c] (mg/kg-bw/day)	Continuous HED [c] (mg/kg-bw/day)	Uncertainty Factors [g]	Total Uncertainty Factors	Reference	Data Quality
Renal	Rats (male)	**Oral 1,2-dichloroethane data** 13-weeks oral gavage	$BMDL_{10}$ = 27 mg/kg BMD_{10} = 33 mg/kg	Increased relative kidney weight)	N/A	N/A	9.1	6.5	UF_A = 3 UF_H = 10 UF_S = 10	300[d]	NTP (1991)	High
Reproductive	Mice (male)	**Inhalation 1,2-dichloroethane data** 4-week morphological analysis of sperm parameters/ inhalation	$BMCL_5$ = 21.2 mg/m^3 or 5.2 ppm BMC_5 = 26.7 mg/m^3 or 6.6 ppm	Decreases in sperm concentration	22.0 ppm (89.0 mg/m^3)	5.2 ppm (21.2 g/m^3)	N/A	N/A	UF_A = 3 UF_H = 10 UF_S = 10	300[e]	Zhang et al. (2017)	High
Renal	Rats (male)	**Dermal (extrapolated from oral)** **1,2-dichloroethane data**	$BMDL_{10}$ = 27 mg/kg BMD_{10} = 33 mg/kg	Increased relative kidney weight)	N/A	N/A	9.1	6.5	UF_A = 3 UF_H = 10 UF_S = 10	300[d]	NTP (1991)	High

[a] See Section 5.2.1.2.1 for details.
[b] BMCL5 = 21.2 mg/m^3 was adjusted to continuous adjusted (with no respiratory effects, there is no RGD; the blood:air ratio = 1, based on Equation_Apx N-9; therefore, the HEC_{cont} is the same as the adjusted POD of 21.2 mg/m^3. The HEC worker is the HEC_{cont} × 4.2 (hours in a week divided by the # of working hours in a week; 168/40) = 89.0 mg/m^3. Both HEC worker and continuous converted to ppm divided by a factor of 4.05 (based 24.45/MW).
[c] All oral PODs were first adjusted to 7 days/week. All continuous oral PODs were then converted to HEDs using DAFs. Dermal PODs were set equal to the oral HED. It is often necessary to convert between ppm and mg/m^3 due to variation in concentration reporting in studies and the default units for different OPPT models. Therefore, EPA presents all PODs in equivalents of both units to avoid confusion and errors. PODs converted for use in worker exposure scenarios were adjusted to 8 hours/day, 5 days/week and converted to HECs.
[d] No PODs were identified from chronic exposure by the **oral route** to 1,1-dichloroethane; therefore, read-across from 1,2-dichloroethane was used to identify a POD. A chronic-duration oral HED for worker of 9.1 mg/kg-bw/day and a HED for continuous exposure of 6.5 mg/kg-bw/day was used for risk assessment of chronic oral exposure, with a total uncertainty factor of 30, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used and 10 for human variability.
[e] No PODs were identified from chronic exposure by the **inhalation route** to 1,1-dichloroethane. Therefore, read-across from 1,2-dichloroethane was used to identify a POD. The chronic-duration inhalation HEC for worker exposure of 89.0 mg/m^3, and a HEC for continuous exposure of 21.2 mg/m^3, was used for risk assessment of chronic inhalation exposure, with a total uncertainty factor of 300, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used, 10 for human variability, and 10 for extrapolating from a subchronic study duration to a chronic study duration.

Target Organ System	Species	Duration/ Route	Study POD/ Type	Effect	Worker HEC [b] (ppm) [mg/m^3]	Continuous HEC [b] (ppm) [mg/m^3]	Worker HED [c] (mg/kg-bw/day)	Continuous HED [c] (mg/kg-bw/day)	Uncertainty Factors [g]	Total Uncertainty Factors	Reference	Data Quality

[f] No PODs were identified from chronic exposure by the **dermal route** to 1,1-dichloroethane; therefore, route-to-route extrapolation from the oral route was used to identify a POD. A chronic-duration dermal HED for worker of 9.1 mg/kg-bw/day and a HED for continuous exposure of 6.5 mg/kg-bw/day was used for risk assessment of chronic dermal exposure, with a total uncertainty factor of 300, based on a combination of uncertainty factors: 3 for interspecies extrapolation when a dosimetric adjustment is used, 10 for human variability, and 10 for extrapolating from a subchronic study duration to a chronic study duration.

[g] UF = uncertainty factor; UF_A = extrapolation from animal to human (interspecies); UF_H = potential variation in sensitivity among members of the human population (intraspecies); UF_S = use of a short-term study for long-term risk assessment

Table 5-46. Cancer PODs for 1,1-Dichloroethane Lifetime Exposure Scenarios – Read-Across from 1,2-Dichloroethane Data

Exposure Assumption [a]	Oral Slope Factor [b]	Dermal Slope Factor [b]	Inhalation Unit Risk [c]	Drinking Water Unit Risk [d]	Extra Cancer Risk Benchmark
Continuous Exposure	Not Available	Not Available	7.1E−06 (per µg/m^3) 2.9E−2 (per ppm)	Not Available	1E−06 (general population)
Worker	Not Available	Not Available	2.4E−06 (per µg/m^3) 9.5E−3 (per ppm)	Not Available	1E−04 (occupational)

[a] Cancer slope factor and unit risk will be derived based on continuous exposure scenarios. Due to the exposure averaging time adjustments incorporated into lifetime exposure estimates, separate cancer hazard values for occupational scenarios are not required.
[b] Due to uncertainty and limitations regarding 1,1- and 1,2-dichloroethane studies, derivation of an oral slope factor was not performed. Additionally due to scarcity of data and inability for route-to-route extrapolation from an oral slope factor, a dermal slope factor was also not derived.
[c] Read-across using cancer inhalation PODs from 1,2-dichloroethane based on based on combined mammary gland adenomas, fibroadenomas, and adenocarcinomas and subcutaneous fibromas in female rats (Nagano et al., 2006).
[d] Due to uncertainty and limitations regarding 1,1- and 1,2-dichloroethane studies, derivation of a drinking water unit risk was not performed.

5.2.6.4 Human Health Hazard Values Used by Other Agencies

Historically, offices across EPA and other agencies (ATSDR), have developed their own assessments for 1,1- and 1,2-dichloroethane. A comparison of these assessments is outlined below and summarized in Table 5-47 for non-cancer hazard values based on exposure duration and route.

EPA first reviewed existing assessments of 1,1-and 1,2-dichloroethane conducted by regulatory and authoritative agencies such as ATSDR (2015) and ATSDR (2022), as well as several systematic reviews of studies of 1,1- and 1,2-dichloroethane published by U.S. EPA Integrated Risk Information System (IRIS) program (U.S. EPA, 1990, 1987) and U.S. EPA Provisional Peer-Reviewed Toxicity Values (U.S. EPA, 2010, 2006).

With regard to the U.S. EPA Integrated Risk Information System (IRIS) program (U.S. EPA, 1990, 1987) assessments for 1,1- and 1,2-dichloroethane, non-cancer exposure durations/routes were not assessed. Upon evaluation of the (ATSDR, 2015) *Toxicological Profile for 1,1-Dichloroethane* and U.S. EPA *Provisional Peer-Reviewed Toxicity Values for 1,1-Dichloroethane* ATSDR (2022) *Toxicological Profile for 1,2-Dichloroethane* and U.S. EPA *Provisional Peer-Reviewed Toxicity Values for 1,1-Dichloroethane* (U.S. EPA, 2006) and U.S. EPA *Provisional Peer-Reviewed Toxicity Values for 1,1-Dichloroethane* (U.S. EPA, 2010), the studies identified for minimal risk level (MRL) and provisional values, respectively, by these assessment were evaluated by the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). While there are many areas of agreement with these assessments, these assessments either did not derive values for exposure durations and/or routes, used studies that were not considered as "sensitive endpoints," or used studies that were identified as "uninformative for dose-response" based on systematic review for the subchronic duration scenarios.

For 1,1-dichloroethane, no provisional value was derived in U.S. EPA (2006) for the acute duration for any exposure route and the study by Muralidhara et al. (2001), based on sedation in male rats, was identified for the oral subchronic and chronic duration. This study was not used as the POD based on a NOAEL of 714 mg/kg/day in male rats with limited assessment of neurotoxicity.
Furthermore, as the database for 1,1-dichloroethane contained data gaps and the use of the 1,2-dichloroethane database was used to fill those gaps, a thorough evaluation for both ATSDR (2022) and

(U.S. EPA, 2010), that identified the 13-week study by NTP (1991), where male and female F344/N, Sprague Dawley, and Osborne-Mendel rats as well as B6C3F1 mice exposed to 1,2-dichloroethane in drinking water was used to derive their respective values. A significant dose-related increase in kidney weight and the kidney-body-ratio of female F344 rats was identified at 58 mg/kg-day among the three rat strains. This study was considered as a potential candidate for POD derivation; however, the daily intake doses were estimated on a mg/kg body weight basis and not measured throughout the duration of exposure. The means by which the dosage estimates were calculated was by dividing the mean water consumption over the 13-week study by the initial and final body weights of ten animals. Additionally, weight gain depression was seen in males and females in the two higher dose groups throughout the study and was likely caused by dehydration due to poor palatability of the formulated drinking water. The study also indicated that water consumption was substantially decreased with increasing dose. According to the study, a decrease of as much as 60 percent in water intake was also seen in both male and female Osborne-Mendel rats at the highest concentration of 8,000 ppm (a range of 500–725 mg/kg/day) that indicates that the dose received by all exposed animals was less than the target dose. The authors indicate that as water intake was reduced at most exposure levels, equivalent exposure did not; however, occur at different dose levels within a strain. Due to the uncertainty regarding the delivered dose and the inherit volatility associated with 1,2-dichloroethane, it was not recommended using this drinking water study for this dose-response assessment.

The final ATSDR *Toxicological Profile for 1,2-Dichloroethane* was also evaluated which has included updates to previously identified MRL values. Degeneration with necrosis of the olfactory epithelium, the basis of the acute inhalation MRL, was refined to 0.1 ppm (0.4 mg/m^3) from 0.3 ppm (1 mg/m^3). Additionally, renal tubular regeneration as the critical effect to calculate the intermediate oral MRL of 0.7 mg/kg-day replaced the original 0.2 mg/kg-day value based on increased kidney weight and the additional of an intermediate inhalation MRL of 0.1 ppm (0.4 mg/m^3) based on neurobehavioral changes not previously derived was incorporated (ATSDR, 2024).

NTP (1991), however, also included a 13-week gavage study that was rated “high” by systematic review and considered for a POD for intermediate exposures based on relative kidney weight and selected for the intermediate/chronic PODs for this assessment.

With regard to identification of a subchronic provisional reference concentration (p-RfC) in (U.S. EPA, 2010) for 1,-2-dichloroethane, the occupational Kozik (1957) study used identified in this assessment was rated “uninformative” by systematic review based on a number of limitations (poor data and test method reporting, lack of description of the analytical methodology, limited quantitative data and statistical analyses, unstated criteria for diagnosis of disease, limited number of study participants and no matched control group, lack of control for potential confounding, lack of exposure duration information). Furthermore, Kozik (1957) did not report any data that could be used for BMD modeling. Additionally, PPRTV also commented on the confidence of the study as well as confidence in the calculated p-RfC as being very low. This study was also used for the chronic p-RfC irrespective of this low confidence with additional uncertainty factor of 10 for the duration adjustment.

Table 5-47. Non-Cancer Human Health Hazard Values Used by Other Agencies and EPA Offices

Exposure	Chemical	Oral	Inhalation	Dermal	Comments
1,1-Dichloroethane risk evaluation					
Acute	1,1-Dichloroethane	Data inadequate – read-across from 1,2-dichloroethane.	Data inadequate – read-across from 1,2-dichloroethane.	Data inadequate – read-across from 1,2-dichloroethane.	
	1,2-Dichloroethane	$BMDL_{10}$ = 153 mg/kg-day BMD_{10} = 270 mg/kg-day Total UF = 30	$BMCL_{10}$ = 48.9 mg/m^3 (12.1 ppm) BMC_{10} = 81.4 mg/m^3 (20.1 ppm) Total UF = 30	$BMDL_{10}$ = 153 mg/kg-day BMD_{10} = 270 mg/kg-day Total UF = 30	
Intermediate	1,1-Dichloroethane	Data inadequate – read-across from 1,2-dichloroethane.	Data inadequate – read-across from 1,2-dichloroethane.	Data inadequate – read-across from 1,2-dichloroethane.	
	1,2-Dichloroethane	$BMDL_{10}$ = 27 mg/kg-day BMD_{10} = 33 mg/kg-day Total UF = 30	$BMCL_5$ = 21.2 mg/m^3 (5.2 ppm) BMC_5 = 26.7 mg/m^3 (6.6 ppm) Total UF = 30	$BMDL_{10}$ = 27 mg/kg-day BMD_{10} = 33 mg/kg-day Total UF = 30	
Chronic	1,1-Dichloroethane	Data inadequate –read-across from 1,2-dichloroethane.	Data inadequate – read-across from 1,2-dichloroethane.	Data inadequate – read-across from 1,2-dichloroethane.	
	1,2-Dichloroethane	$BMDL_{10}$ = 27 mg/kg-day BMD_{10} = 33 mg/kg-day Total UF = 300	$BMCL_5$ = 21.2 mg/m^3 (5.2 ppm) BMC_5 = 26.7 mg/m^3 (6.6 ppm) Total UF = 300	$BMDL_{10}$ = 27 mg/kg-day BMD_{10} = 33 mg/kg-day Total UF = 300	A UF_s of 10 added to intermediate study due chronic duration study not being identified.
IRIS (U.S. EPA, 1990, 1987)					
Acute	1,1- Dichloroethane	Not assessed under IRIS.			
	1,2- Dichloroethane				
Intermediate	1,1- Dichloroethane				
	1,2- Dichloroethane				
Chronic	1,1- Dichloroethane				
	1,2- Dichloroethane				
PPRTV (U.S. EPA, 2010, 2006)					
Acute	1,1- Dichloroethane	Did not derive a provisional value	Did not derive a provisional value	Did not derive a provisional value	Database considered inadequate.
	1,2- Dichloroethane	Did not derive a provisional value	Did not derive a provisional value	Did not derive a provisional value	Database considered inadequate.

Exposure	Chemical	Oral	Inhalation	Dermal	Comments
Intermediate	1,1-Dichloroethane	RfD = 2 mg/kg-day (Dividing $NOAEL_{adj}$ of 714 mg/kg-day by the total UF of 300) based on sedation (Muralidhara et al., 2001) for 13 weeks.	Available inhalation data in animals and humans considered inadequate for derivation of a RfC provisional.	Did not derive a provisional value	PPRTV commented confidence in the study is medium (and a UF_D of 3 was used in their total UF calculation), and overall confidence in the calculation of the provisional RfD is low.
	1,2-Dichloroethane	RfD = 0.02 mg/kg-day (Dividing LOAEL of 58 mg/kg-day by the total UF of 3000) based on increased kidney weights (NTP, 1991; Morgan et al., 1990), 90-day drinking water (DW)	1,2-Dichlorothane animal data was not used – human data was selected as the only feasible study for subchronic durations. RfC = 0.07 mg/m^3 (Dividing the $LOAEL_{HEC}$ of 22 mg/m^3 by the total UF of 300) based on neurobehavioral impairment (Kozik, 1957)	Did not derive a provisional value	For the oral route: PPRTV used a UF_D of 3 to account for database inadequacies.
Chronic	1,1-Dichloroethane	RfD = 2 mg/kg-day (Dividing $NOAEL_{adj}$ of 714 mg/kg-day by the total UF of 3,000) based on sedation (Muralidhara et al., 2001) for 13 weeks.	Available inhalation data in animals and humans considered inadequate for derivation of a RfC provisional value.	Did not derive a provisional value	Same study and conclusions as for the subchronic duration only added an additional UF of 10 for use of subchronic study for chronic duration to yield a total UF = 3,000.
	1,2-Dichloroethane	Did not derive a provisional value.	RfC = 0.007 mg/m^3 (Dividing the $LOAEL_{HEC}$ of 22 mg/m^3 by the total UF of 3000) based on neurobehavioral impairment (Kozik, 1957)	Did not derive a provisional value	For the RfC: Same study and conclusions as for the subchronic duration only added an additional UF of 10 for use of subchronic study for chronic duration to yield a total UF = 3,000.

Exposure	Chemical	Oral	Inhalation	Dermal	Comments
ATSDR (ATSDR, 2024, 2022, 2015)					
Acute	1,1-Dichloroethane	Did not derive an MRL	Did not derive an MRL	Did not derive an MRL	Database considered inadequate.
	1,2-Dichloroethane	Did not derive an MRL	Draft (ATSDR, 2022) Profile: 0.3 ppm (1 mg/m^3) Dividing $BMCL_{HEC}$ of 9.2 by total UF of 30 based on degeneration, with necrosis, olfactory epithelium in rats (Hotchkiss et al., 2010; Dow Chemical, 2006); ($BMCL_{10}$ = 57)	Did not derive an MRL	
			Final (ATSDR, 2024) Profile: 0.1 ppm (0.4 mg/m^3) Dividing $BMCL_{HEC}$ of 3.84 by total UF of 30 based on degeneration, with necrosis, olfactory epithelium in rats (Hotchkiss et al., 2010; Dow Chemical, 2006); ($BMCL_{10}$ = 57.62)		
Intermediate	1,1-Dichloroethane	Did not derive an MRL	Did not derive an MRL	Did not derive an MRL	Database considered inadequate
	1,2-Dichloroethane	Draft (ATSDR, 2022) Profile: 0.2 mg/kg/day (Dividing LOAEL of 58 mg/kg-day by the total UF of 3000) based on increased kidney weights (NTP, 1991; Morgan et al., 1990), 90-day drinking water (DW)	Draft (ATSDR, 2022) Profile: Did not derive an MRL	Did not derive an MRL	$BBMCL_{1SD-HEC}$ = Bayesian benchmark response of 1 standard deviation dosimetrically adjusted to a human equivalent concentration (HEC)
		Final (ATSDR, 2024) Profile: 0.7 mg/kg/day	Final (ATSDR, 2024) Profile: 0.1 ppm (0.4 mg/m^3)		

Exposure	Chemical	Oral	Inhalation	Dermal	Comments
Intermediate	1,2-Dichloroethane	(Dividing $BMDL_{10}$ of 70.1 mg/kg-day by the total UF of 100) based on kidney tubule regeneration, increased kidney weights (NTP, 1991; Morgan et al., 1990), 90-day drinking water (DW)	Dividing $BBMCL_{1SD\text{-}HEC}$ of 3.7 by total UF of 30 based neurobehavioral changes in mice (Zhong et al., 2022);		
Chronic	1,1- Dichloroethane	Did not derive an MRL	Did not derive an MRL	Did not derive an MRL	Database considered inadequate
	1,2- Dichloroethane	Did not derive an MRL	Did not derive an MRL	Did not derive an MRL	

5.2.7 Weight of Scientific Evidence Conclusions for Human Health Hazard

The weight of scientific evidence supporting the human health hazard assessment is based on the strengths, limitations, and uncertainties associated with the hazard studies identified. The weight of scientific evidence is summarized using confidence descriptors: robust, moderate, slight, or indeterminate. This approach is consistent with the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a). When weighing and integrating evidence to estimate the potential that 1,1-dichloroethane may cause a given non-cancer or cancer health hazard endpoint (*e.g.*, renal, olfactory, reproductive, mammary/subcutaneous tumors), EPA uses several factors adapted from Sir Bradford Hill (1965). These elements include consistency, dose-response relationship, strength of the association, temporal relationship, biological plausibility, and coherence among other considerations.

EPA considered evidence integration conclusions from Sections 5.2.3, 5.2.4, and 5.2.5 and additional factors when choosing studies for dose-response modeling and for each exposure scenario (acute, intermediate, and chronic), as described in Section 5.2.6. Additional considerations pertinent to the overall hazard confidence levels include evidence integration conclusions from Appendix N, selection of the critical endpoint and study, relevance to the exposure scenario, dose-response considerations and PESS sensitivity. Appendix N-12 presents a summary table of confidence for each hazard endpoint and exposure duration.

Several limitations exist for the 1,1-dichloroethane database. First, the database for studies in humans and animals consisted of a small number of studies, with limited evaluations performed in many of these studies, thereby precluding the identification of target organs for 1,1-dichloroethane. Second, no acceptable toxicological data were available by the dermal or drinking water route, and PBPK/PD models that would facilitate route-to-route extrapolation to the dermal route have not been identified for 1,1-dichloroethane. However, in oral dosing, the dose is rapidly absorbed and over 80% is exhaled through the lungs unchanged. Dermal exposures have similar elimination through the lungs. Therefore, oral PODs were used for extrapolation via the dermal route. Third, no adequate data were available to identify non-cancer PODs for the inhalation route for either acute or intermediate exposure durations. Data for the identified analog for 1,1-dichloroethane, 1,2-dichloroethane was used to read-across and fill identified data gaps (Section 5.2.1.1).

5.2.7.1 Overall Confidence – Strengths, Limitations, Assumptions, and Key Sources of Uncertainty in the Human Health Hazard Assessment

As discussed in Section 5.2.1.1, EPA identified data gaps for 1,1-dichloroethane for non-cancer PODs by the acute, intermediate, and chronic oral, dermal, and inhalation routes; and cancer PODs by the oral, inhalation, and dermal routes. A read-across approach was used to identify the best chemical analog to fill those data gaps. The analyses resulted in the identification of 1,2-dichloroethane (an isomer of 1,1-dichloroethane) as the most appropriate analog to fill the identified data gaps for 1,1-dichloroethane (see Section 5.2.1.3). Based on the identification and selection of the PODs, although not using chemical specific data for 1,1-dichloroethane for the derivation of these values, the Agency is confident that the use of 1,2-dichloroethane data is supported by the weight of scientific evidence and is also human health protective.

In addition, 1,1-dichloroethane and 1,2-dichloroethane both lacked adequate data by the dermal route for any exposure duration based on evaluation of epidemiological and animal studies that could be used for dose-response. Specifically, no studies for 1,1- or 1,2-dichloroethane were identified as suitable for dose-response. Therefore, EPA used a route-to-route extrapolation approach from the available 1,2-dichloroethane oral data to fill in the dermal data gap. EPA also has high confidence in this approach

and in assessing dermal exposure was able to incorporate test order data from 1,1-dichloroethane for risk estimates that was further corroborated by the *in silico* tool IH SkinPerm. Since both oral and dermal routes are similar metabolically and by-pass first pass metabolism through the liver, and since oral ADME studies showed that most of the 1,1-dichloroethane oral dose was eliminated unchanged in expired air, oral PODs were used for extrapolation via the dermal route.

EPA has high confidence in the human health hazard database for 1,2-dichloroethane and in the selection of the critical PODs. This is based on several reasons. First, all studies used to assess the hazards for 1,2-dichloroethane were rated high to medium in SR. Second, critical non-cancer effects that were ultimately selected as PODs for quantitative risk estimates (kidney, olfactory, and reproductive (sperm) toxicity), were considered the most sensitive and biologically relevant effects, supported by multiple lines of evidence that spanned across species, routes, and durations of exposure.

5.2.7.1.1 Acute Non-Cancer

Hazard ID Conclusions and Evidence Integration Judgements

The identified health effect of renal toxicity, specifically on increased relative kidney weight for the oral acute non-cancer POD from the 1,2-dichloroethane study by Storer et al. (1984), was supported by the weight of scientific evidence and considered appropriate for dose-response analysis.

Selection of Most Critical Endpoint and Study

EPA has the strongest confidence in the selection of study and endpoints representing renal and respiratory toxicity for oral and inhalation exposures, respectively. Although there are limitations in the number of studies that evaluated renal toxicity due to 1,1-dichloroethane exposure via the oral route, studies were identified that found associations between 1,2-dichloroethane exposure and increased kidney weights, BUN, and/or renal tubular histopathology in rats (both sexes) and mice.

Only the single-dose experiment by Muralidhara et al. (2001) was considered as a potential study adequate for evaluation of 1,1-dichloroethane toxicity and POD derivation following acute oral exposures. A NOAEL of 1,000 mg/kg-bw and a LOAEL of 2,000 mg/kg-bw were identified based on clinical signs of neurotoxicity characterized by the authors as “excitation followed by progressive motor impairment and sedation.” Although the acute-duration oral data are limited, the observation of central nervous system or CNS effects is consistent with the past use of 1,1-dichloroethane as a human anesthetic (ATSDR, 2015). This study, however, was not selected for the acute POD as this dose resulted in sedation/CNS depression but the methods that evaluated this endpoint were not provided. The data available for 1,1-dichloroethane in Muralidhara et al. (2001) were also near the LD_{50} value of 8,200 mg/kg-day and were not considered appropriate for use for POD identification. This effect was thus not considered a sensitive endpoint as the magnitude of this effect was also not quantitatively described in the study thus necessitating the integration of studies within the 1,2-dichloroethane database to identify a more sensitive endpoint.

Due to the lack of acute studies for 1,1-dichloroethane via the inhalation route, studies assessing the toxicological effects of 1,2-dichloroethane were identified as potential study candidates to derive PODs as read-across to 1,1-dichloroethane. As indicated previously, the 10-day inhalation study by Schwetz et al. (1974) was not used because the effects on developing fetuses and/or offspring were limited and inconclusive and were considered inadequate for derivation of an acute inhalation POD, and because the only effect reported were decreases in maternal body weight which occurred following 10-days of exposure. The identified health effect of respiratory (olfactory effects), specifically on degeneration with necrosis of the olfactory mucosa inhalation acute non-cancer POD from the 1,2-dichloroethane study by

Dow Chemical (2006), was supported by the weight of scientific evidence and considered appropriate for dose-response analysis.

Relevance to Exposure Scenarios
EPA has the strongest confidence in the selection of renal toxicity as the critical endpoint for acute oral exposure as supported by multiple studies in rats of various strains indicating this effect due to 1,2-dichloroethane exposure.

Dose-Response Considerations
EPA has strong confidence in dose-response considerations for renal effects, especially that associated with increased relative kidney weight. LOAEL/NOAEL values and BMD modeling from various studies were transformed to a HED/HEC value which resulted in PODs that were all within a few fold of each other.

PESS Sensitivity
Laboratory inbred animal strains were used for examination of all key endpoints and limited human evidence was available for non-cancer endpoints. Therefore, EPA was unable to quantify considerations from unique sensitivities to 1,1-dichlorethnae exposure due to limited data. An evaluation of the limited database in animals for 1,1-dichloroethane identified one study (Schwetz et al., 1974) with information on lifestages following exposure to 1,1-dichloroethane. The only effect reported was a decrease in maternal body weight (LOAEL of 3,798 ppm), with no observed effects on the fetuses or pups thus does not suggest greater biological susceptibility. Additionally, the reported delays in fetal ossification from this study were difficult to interpret as this effect also occurred in the two control groups. The only other effect considered for 1,1-dichloroethane was from a 13-week repeated-dose toxicity study by Muralidhara et al. (2001), with a $NOAEL_{continuous}$ and $LOAEL_{continuous}$ for CNS depression of 714 and 1,429 mg/kg-bw/day, respectively. This endpoint, however, was near lethal doses and was therefore not considered a sensitive endpoint for assessing potential biological susceptibility.

Although information on other considerations potentially impacting greater biological susceptibility (such as pre-existing disease, lifestyle activities, nutritional status, genetic predispositions, or other chemical co-exposures), was sparse, there is some information on 1,1-dichloroethane as impacting greater biological susceptibility. For example, ATSDR (2015) summarized the occupational health guidelines from CDC (1978) that indicated additional factors that could impact greater susceptibility in the general population. Individuals with skin disease may be of greater susceptibility; however, due to the identified exposure levels to 1,1-dichloroethane these individuals do not seem to be of greater susceptibility as a result of dermal irritation. Individuals with pre-existing diseases, particularly those that impact biotransformation and detoxification of 1,1-dichloroethane, such as those individuals with chronic kidney disease and impaired renal function, may be of greater susceptibility to 1,1-dichloroethane as data have indicated nephrotoxicity in animals exposed to 1,1-dichloroethane. Individuals with chronic respiratory disease may be of greater susceptibility; however, data were not identified on respiratory irritant effects induced by 1,1-dichloroethane. Additional potential populations that may be unusually susceptible to 1,1-dichloroethane include children and the elderly because of immature or compromised metabolic capabilities if metabolic pathways are impeded based on genetic polymorphisms and lifestyles activities such as drug/medication use or due to alcohol consumption.

Although information on other considerations potentially impacting greater biological susceptibility (such as pre-existing disease, lifestyle activities, nutritional status, genetic predispositions, or other chemical co-exposures), was sparse, there is some information on 1,2-dichloroethane as impacting greater biological susceptibility. For example, individuals with impaired renal function based on

evidence that 1,2-dichloroethane is nephrotoxic in animals, individuals with chronic respiratory disease because of the effects on the olfactory mucosa induced by 1,2-dichloroethane, and males with reproductive/fertility issues based on evidence that 1,2-dichloroethane causes decreases in sperm concentration in animals may be impacted due to greater biological susceptibility.

For PESS, specifically susceptibility, across both chemical databases for 1,1- and 1,2-dichloroethane, uncertainty exists based on limited number of studies, and the differences in results and comprehensiveness of endpoints assessed towards specific health outcomes across studies.

Overall Confidence
Based on the above factors, EPA has robust overall confidence for the evidence integration, study/endpoint selection, exposure scenario applicability, dose-response, PESS sensitivity of the conclusions, PODs for renal toxicity, and the most sensitive endpoint of respiratory (olfactory effects) for the oral and inhalation exposures, respectively**.** EPA has moderate overall confidence for the other critical hazard outcomes with PODs at very similar levels that further support the POD to be used for risk estimation.

5.2.7.1.2 Intermediate/Chronic Non-Cancer

Hazard ID Conclusions and Evidence Integration Judgements
The critical health effect domains associated with the renal and male reproductive systems were supported by the weight of scientific evidence and considered appropriate for dose-response analysis. Renal effects following sub-chronic exposure were observed across multiple studies in rats as well as observed in mice during the subchronic exposure duration. Male reproductive effects were observed in a dose-responsive manner; however, only in mice in the absence of any relevant epidemiological studies.

Due to the limited intermediate studies for 1,1-dichloroethane via the inhalation route, studies assessing the toxicological effects of 1,2-dichloroethane were identified as potential study candidates to derive PODs as read-across to 1,1-dichloroethane. As indicated previously, the 10-day inhalation study by Schwetz et al. (1974) was not used because the effects on developing fetuses and/or offspring were limited and inconclusive and were considered inadequate for derivation of an acute inhalation POD, and because the only effect reported were decreases in maternal body weight which occurred following 10-days of exposure.

In the study by Hofmann et al. (1971), a repeated 6-hour inhalation 13-week exposure to 500 ppm 1,1-dichloroethane or 1,2-dichloroethane in rats, guinea pigs, and rabbits indicated toxicity only in animals exposed to 1,2-dichloroethane. Although this study cannot be utilized quantitatively, qualitative evaluation based on this comparison of equivalent concentrations for 1,1-dichloroethane and 1,2-dichloroethane identifies 1,2-dichloroethane to possess greater toxicity among rats, guinea pigs and rabbits. Rats, as the most sensitive species, displayed an onset of dyspnea and death within the first five exposure sessions in contrast to the lack of any clinical or pathological changes in 1,1-dichloroethane exposed animals through the duration of the study. Taking this in account, Hofmann et al. (1971), suggest that 1,2-dichloroethane is approximately 5 times more toxic than 1,1-dichloroethane via the inhalation route based on this exposure scenario.

In the reproduction study by (Rao et al., 1980), male and female Sprague-Dawley rats were exposed to 0, 25, 75, or 150 ppm of 1,2-dichloroethane via whole body inhalation for 60 days, 6 hours/day and 5 days/week. After 60 days of exposure F_0 male and females of each respective treatment group were bred one-to-one to generate F_{1A} generation. Seven days after F_{1A} litter was sacrificed, F_0 rats were bred again to produce a F_{1B} generation. No exposure related effect in body weight, organ weights (liver and

kidney), or histology (liver, kidneys, ovaries, and testes) were seen in the F_0 rats. No significant differences in fertility index, gestation days, sex ratio, neonatal body weight or growth of pups were observed. Additionally, no exposure related change in liver or kidney weights or histology were seen in the F_1 generations. The apparent body weight decrease in selected male F_{1B} weanlings at 150 ppm was based on only five male weanlings per group, which was not a statistically significant difference from controls.

The study by (Payan et al., 1995), a 15-day study in female Sprague-Dawley rats exposed to 1,2-dichloroethane for 6 hours/day identified no significant effects in the body weight of dams nor pups in exposure groups up to 250 ppm. In addition, the pregnancy rate among females at 250 ppm was significantly lower than controls; however, the effect was not seen in the 300-ppm group, so it was assumed not to be related to exposure. At the highest concentration of 300 ppm, a decrease of maternal body weight was the only effect observed, similarly to Schwetz et al. (1974), but no significant morphological effects in pups were identified as compared to controls. In the 10-day teratogenicity study by (Rao et al., 1980), mated Sprague-Dawley rats (16–30/group) were exposed to 0, 100, 300 ppm of 1,2-dichloroethane for 7 hours/day on gestational day 6 to 15 via whole body inhalation. Dams were sacrificed on gestational day 21 and implantation resorption was evaluated for each exposure group; however, one litter was identified for the 300-ppm exposure group, as only one surviving female was pregnant at sacrifice in the 300 ppm exposure group. The embryotoxicity considered was thus considered secondary to the maternal toxicity.

The 4-week study by Zhang et al. (2017) was chosen for read-across from 1,2-dichloroethane to 1,1-dichloroethane to derive a POD for intermediate exposure via inhalation as the other studies indicated above using 1,2-dichloroethane were deemed inadequate for this determination due to study limitations.

Selection of Most Critical Endpoint and Study

EPA has the strongest confidence in the selection of renal toxicity as the critical endpoint for intermediate oral exposure as supported by multiple studies in rats of various strains indicating this effect due to 1,2-dichloroethane exposure.

Relevance to Exposure Scenarios

The renal toxicity endpoint based on increased relative kidney weight is relevant to the assigned exposure scenario as this study was based on a 13-week study duration (NTP, 1991). Although, several studies via the chronic exposure duration for both oral and inhalation exposures were identified these studies were not selected for the chronic POD due to study limitations and inherent uncertainties (see Section 0). Data based on the intermediate exposure duration was based on an overall weight of scientific evidence that identified an endpoint that was identified as appropriate and supported by other studies. The application of the UF_S of 10× was applied to account for the use of these intermediate studies for the long-term (chronic) duration. As a result, intermediate data were used for the chronic POD and an uncertainty factor (UF_s) of 10× was applied to account for the use of an intermediate study for long-term (chronic) assessment.

Dose-Response Considerations

EPA has strong confidence in dose-response considerations for renal effects, especially that associated with increased kidney weight. LOAEL/NOAEL values and BMD modeling from various studies were transformed to a HED/HEC value which resulted in PODs that were all within a few fold of each other.

PESS Sensitivity

Laboratory inbred animal strains were used for examination of all key endpoints and limited human evidence was available for non-cancer endpoints. Therefore, EPA was unable to quantify considerations from unique sensitivities to 1,1-dichlorethnae exposure due to limited data. An evaluation of the limited database in animals for 1,1-dichloroethane identified one study Schwetz et al. (1974) with information on lifestages following exposure to 1,1-dichloroethane. The only effect reported was a decrease in maternal body weight (LOAEL of 3,798 ppm), with no observed effects on the fetuses or pups thus does not suggest greater biological susceptibility. Additionally, the reported delays in fetal ossification from this study were difficult to interpret as this effect also occurred in the two control groups. The only other effect considered for 1,1-dichloroethane was from a 13-week repeated-dose toxicity study by Muralidhara et al. (2001), with a $NOAEL_{continuous}$ and $LOAEL_{continuous}$ for CNS depression of 714 and 1,429 mg/kg-bw/day, respectively. This endpoint, however, was near lethal doses and was therefore not considered a sensitive endpoint for assessing potential biological susceptibility.

Although information on other considerations potentially impacting greater biological susceptibility (such as pre-existing disease, lifestyle activities, nutritional status, genetic predispositions, or other chemical co-exposures), was sparse, there is some information on 1,1-dichloroethane as impacting greater biological susceptibility. For example, ATSDR (2015) summarized the occupational health guidelines from (CDC, 1978) that indicated additional factors that could impact greater susceptibility in the general population. Individuals with skin disease may be of greater susceptibility; however, due to the identified exposure levels to 1,1-dichloroethane these individuals do not seem to be of greater susceptibility as a result of dermal irritation. Individuals with pre-existing diseases, particularly those that impact biotransformation and detoxification of 1,1-dichloroethane, such as those individuals with chronic kidney disease and impaired renal function, may be of greater susceptibility to 1,1-dichloroethane as data have indicated nephrotoxicity in animals exposed to 1,1-dichloroethane. Individuals with chronic respiratory disease may be of greater susceptibility; however, data were not identified on respiratory irritant effects induced by 1,1-dichloroethane. Additional potential populations that may be unusually susceptible to 1,1-dichloroethane include children and the elderly because of immature or compromised metabolic capabilities if metabolic pathways are impeded based on genetic polymorphisms and lifestyles activities such as drug/medication use or due to alcohol consumption.

Although information on other considerations potentially impacting greater biological susceptibility (such as pre-existing disease, lifestyle activities, nutritional status, genetic predispositions, or other chemical co-exposures), was sparse, there is some information on 1,2-dichloroethane as impacting greater biological susceptibility. For example, individuals with impaired renal function based on evidence that 1,2-dichloroethane is nephrotoxic in animals, individuals with chronic respiratory disease because of the effects on the olfactory mucosa induced by 1,2-dichloroethane, and males with reproductive/fertility issues based on evidence that 1,2-dichloroethane causes decreases in sperm concentration in animals may be impacted due to greater biological susceptibility.

For PESS, specifically susceptibility, across both chemical databases for 1,1- and 1,2-dichloroethane, uncertainty exists based on limited number of studies, and the differences in results and comprehensiveness of endpoints assessed toward specific health outcomes across studies.

Overall Confidence

Based on the above factors, EPA has robust overall confidence for the evidence integration, study/endpoint selection, exposure scenario applicability, dose-response, PESS sensitivity of the conclusions, PODs for renal toxicity, and the most sensitive endpoint of reproductive (sperm effects) for the oral and inhalation exposures, respectively**.** EPA has moderate overall confidence for the other

critical hazard outcomes with PODs at very similar levels that further support the POD to be used for risk estimation.

For complete details on weight of scientific evidence conclusions for both within and across evidence streams, see the evidence profile tables for each organ domain in Appendix N.6 and N.7. For a more detailed description of the hazard database and weight of scientific evidence evaluation see 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) for details on the process of evidence evaluation and integration.

5.2.7.1.3 Cancer

EPA determined that evidence is inadequate to assess whether 1,1-dichloroethane exposure may cause cancer in humans under relevant exposure circumstances due to the limited number of studies in human or animals that were identified to determine an association. As 1,2-dichloroethane was identified as the analog to fulfill the cancer data gap, EPA determined that evidence indicates that 1,2-dichloroethane likely causes cancer under relevant exposure circumstances based on animals and mechanistic data.

Selection of Most Critical Endpoint and Study

The NTP (1978) chronic cancer bioassay in male and female Osborne-Mendel rats administered 1,1-dichloroethane via oral gavage was evaluated and identified overall poor survival confounded by high incidence of pneumonia observed in control and all treated groups. Additionally, the NTP (1978) chronic cancer bioassay in male and female B6C3F1 mice administered 1,1-dichloroethane via oral gavage also indicated that male mice exhibited poor survival in all groups while female mice showed better survival overall (survival within the untreated, vehicle low dose and high dose groups were at 35, 55, 62, and 32 percent in male mice and 80, 80, 80 and 50 percent in female mice). Mice also exhibited incidences of murine pneumonia, though at lower occurrences to those identified in the parallel study in Osborne-Mendel rats and only in the treatment groups for both males and females. The only specific histopathological changes were of endometrial polyps in the uterus in the 3331 mg/kg-day treated female mice with no other occurrences in the other dosage groups.

The IRIS program derived an oral CSF of 9.1×10^{-2} (per mg/kg-bw/day) for 1,2-dichloroethane in 1987 based on the incidence of hemangiosarcomas in male rats in the chronic bioassay by NTP (1978); however, this study did not pass EPA systematic review. The oral CSF for male mice based on hepatocarcinomas was determined to be 6.2×10^{-3} (per mg/kg-bw/day) also based on the same study NTP (1978). No oral cancer bioassays of 1,2-dichloroethane have been published since the IRIS 1987 assessment. The IRIS CSF was derived using time-to-tumor modeling to account for intercurrent mortality of the rats in the NTP (1978) study. No updates to the time-to-tumor modeling approach have been made since the IRIS 1987 assessment. Hemangiosarcomas in male rats were determined to be the most sensitive species, strain, and site; however, this study was not utilized in this assessment due to confounding associated with increased mortality in all dose groups. Due to limitations associated with both the rats and mice in the NTP (1978) study for 1,2-dichloroethane, EPA is not pursuing a quantitative assessment of cancer risks associated with a CSF derived from 1,2-dichloroethane hazard and exposures to 1,1-dichloroethane.

There were no identified dermal cancer studies for either 1,1- or 1,2-dichloroethane for quantitative dose-response. The 1,2-dichloroethane dermal study by Suguro et al. (2017) did identify bronchioalveolar adenomas and adenocarcinomas; however, its single dose did not allow calculation of an accurate dermal linear low-dose cancer slope factor (Suguro et al., 2017). A dermal CSF was not derived from 1,1- or 1,2-dichloroethane via route-to-route extrapolation using oral data. Additionally, there are uncertainties associated with extrapolation using 1,2-dichloroethane data from both oral and

inhalation dosing for 1,1-dichloroethane dermal route. Use of an oral POD for dermal extrapolation may not be preferred for chemicals known to undergo extensive liver metabolism because the "first-pass effect" that directs intestinally absorbed chemicals to the liver applies to oral ingestion. However, PBPK research also indicates extra-hepatic metabolism for 1,2-dichloroethane. The accuracy of extrapolation of inhalation toxicity data for dermal PODs is dependent on assumptions about inhalation exposure factors such as breathing rate and any associated dosimetric adjustments. However, whole-body inhalation studies may also already be incorporating some level of dermal exposure. Given these uncertainties, in the absence of 1,1-dichloroethane data to support derivation of a dermal CSF from an oral CSF or an inhalation IUR, a dermal CSF was not derived.

Animal data based on Nagano et al. (2006) was selected for dose-response analysis and derivation of the IUR due to female F344 rats that developed increased incidences of subcutaneous fibromas along with the occurrence of mammary gland adenomas, fibroadenomas, and adenocarcinomas due to 1,2-dichloroethane treatment.

Relevance to Exposure Scenarios
EPA utilized the Nagano et al. (2006) animal study performed for 104 weeks which is a standard duration for a cancer bioassay and relevant to the lifetime exposure scenario.

Dose-Response Considerations
BMD modeling of the combined tumor incidences in female rats was performed as the incidences of the mammary tumors and subcutaneous fibromas showed a significant positive trend with increased concentration and were significantly different from the control group at 160 ppm (combined mammary tumors were also significantly different from controls at 40 ppm). The incidences of mammary tumors in the control group were at incidence rates that did not exceed the maximum tumor incidences when compared to historical controls and thus retained in the modeling.

PESS Sensitivity
EPA was unable to quantitatively incorporate other considerations such as considerations lifestyle activities (*e.g.*, smoking), sociodemographic status, or nutrition.

Overall Confidence
As evidence was considered inadequate to assess whether 1,1-dichloroethane may cause cancer under relevant exposure scenarios, analog identification was used to fill this data gap. For the oral route, data from 1,1- and 1,2-dichloroethane was both confounded; however, EPA is confident that the data from 1,2-dichloroethane via the inhalation route would be human health protective for 1,1-dichloroethane. There is robust animal evidence of the association between incidences of mammary gland tumors and 1,2-dichloroethane exposure based on observations seen in male and female rats and in female mice exposed to 1,2-dichloroethane via inhalation in high-quality studies. There is additionally evidence of an association between the increased incidences of subcutaneous tumors and 1,2-dichloroethane exposure based on observations seen in male and female rats exposed to 1,2-dichloroethane via inhalation.

EPA combined the cancer risks from mammary gland tumors and subcutaneous tumors using the Multistage/Multi-tumor model as all the tumors were considered adverse and independent of each other. The purpose of Multistage/Multi-tumor Combo model in BMDS is to allow the user to calculate BMDs and BMDLs for a combination of tumors (corresponding to a defined risk of getting one or more of those tumors) when the individual tumor dose-responses have been modeled using a Multistage-Cancer model. Thus, the output of the run will present the results of fitting each individual tumor (including the BMD and BMDL for that tumor) plus the combined log - likelihood, BMD, and BMDL for the

combination of specified tumor responses. BMD modeling of the combined tumor incidences in female rats was performed as the incidences of the mammary tumors and subcutaneous fibromas showed a significant positive trend with increased concentration and were significantly different from the control group at 160 ppm (combined mammary tumors were also significantly different from controls at 40 ppm). The incidences of mammary tumors in the control group were at incidence rates that did not exceed the maximum tumor incidences when compared to historical controls and thus retained in the modeling. Nagano (2006) also concluded that the highest tested dose did not exceed the maximum tolerated dose thus the top dose is relevant for the analysis. EPA used the linear low dose of the curve to calculate the slope factor. EPA did not identify sufficient data to determine if 1,2-dichloroethane acts through a mutagenic MOA for carcinogenicity.

For complete details on weight of scientific evidence conclusions for both within and across evidence streams, see the evidence profile tables cancer in Appendix N.8 and N.9. For a more detailed description of the hazard database and weight of scientific evidence evaluation see 2021 Draft Systematic Review Protocol (U.S. EPA, 2021a) for details on the process of evidence evaluation and integration.

5.2.7.2 Hazard Considerations for Aggregate Exposure

EPA has defined aggregate exposure as "the combined exposures from a chemical substance across multiple routes and across multiple pathways" (89 FR 37028, May 3, 2024, to be codified at 40 CFR 702.33). For use in this risk evaluation and assessing risks from other exposure routes, the Agency conducted route-to-route extrapolation of the toxicity values from the oral studies for use in the dermal exposure routes and scenarios. Because the health outcomes are different for oral and inhalation studies, EPA did not consider it possible to aggregate risks across exposure routes for all exposure durations and endpoints for the selected PODs.

5.3 Human Health Risk Characterization

5.3.1 Risk Characterization Approach

The exposure scenarios, populations of interest, and toxicological endpoints used for evaluating risks from acute, intermediate, and chronic/lifetime exposures are summarized in Table 5-48.

Table 5-48. Exposure Scenarios, Populations of Interest, and Hazard Values

Populations of Interest and Exposure Scenarios	**Workers** Male and female adolescents and adults (≥16 years old) directly working with 1,1-dichloroethane under light activity (breathing rate of 1.25 m^3/hour) Exposure Durations • *Acute* – 8 hours for a single work day (most OESs) • *Intermediate* – 8 hours per work day for up to 22 working days • *Chronic* – 8 hours per work day for up to 250 days per year for 31 or 40 working years Exposure Routes – Inhalation and dermal
	Occupational Non-Users Male and female adolescents and adults (≥16 years old) indirectly exposed to 1,1-dichloroethane within the same work area as workers (breathing rate of 1.25 m^3/hour) Exposure Durations • *Acute, Intermediate, and Chronic* – Same as workers Exposure Route – Inhalation
	General Population Male and female infants, children, and adults exposed to 1,1-dichloroethane through drinking water, ambient water, ambient air, soil, and fish ingestion Exposure Durations • *Acute* – Exposed to 1,1-dichloroethane continuously for a 24-hour period • *Chronic* – Exposed to 1,1-dichloroethane continuously up to 78 years Exposure Routes – Inhalation, dermal, and oral (depending on exposure scenario)
Health Effects, Hazard Values, and Benchmarks	**Non-Cancer**[a] The **acute oral/dermal**[b] endpoint is increased relative kidney weight by 13 percent via a single oral gavage in male mice. • HED (occupational) = 19.9 mg/kg; HED (continuous) = 19.9 mg/kg • Acute uncertainty factors (Benchmark MOE) = 30 for oral and dermal ($UF_A = 3$; $UF_H = 10$) [c] The **intermediate oral/dermal**[b] endpoint is increased relative kidney weight by 18 percent in male rats via daily oral gavage for 90 days. • HED (occupational) = 9.1 mg/kg; HED (continuous) = 6.5 mg/kg • Short-term/subchronic uncertainty factors (benchmark MOE) = 30 for oral and dermal ($UF_A = 3$; $UF_H = 10$) [c] The **chronic oral/dermal**[b] endpoint is based on and duration adjusted from the identified intermediate POD of increased relative kidney weight seen in male rats treated with 1,2-dichloroethane via daily oral gavage for 90 days. • HED (occupational) = 9.1 mg/kg; HED (continuous) = 6.5 mg/kg • Chronic uncertainty factors (benchmark MOE) = 300 for oral and dermal ($UF_A = 3$; $UF_H = 10$; $UF_S = 10$) [c] The **acute inhalation endpoint** is olfactory effects– degeneration with necrosis of the olfactory mucosa. • HEC (occupational) = 41 mg/cm^3 or 10.14 ppm; HEC (continuous) = 9.78 mg/cm^3 or 2.42 ppm • Acute uncertainty factors (benchmark MOE) = 30 for inhalation ($UF_A = 3$; $UF_H = 10$) [c] The **intermediate inhalation endpoint** is decrease in sperm concentration.

Health Effects, Hazard Values, and Benchmarks	• HEC (occupational) = 89 mg/cm^3 or 22 ppm; HEC (continuous) = 21.2 mg/cm^3 or 5.2 ppm • Short-term/subchronic uncertainty factors (benchmark MOE) = 100 (UF_A = 3; UF_H = 10;) [c] The **chronic inhalation endpoint** is decrease in sperm concentration. • HEC (occupational) = 89 mg/cm^3 or 22 ppm; HEC (continuous) = 21.2 mg/cm^3 or 5.2 ppm • Chronic uncertainty factors (benchmark MOE) = 300 (UF_A = 3; UF_H = 10; UF_S = 10) [c] **Cancer**[a] The cancer endpoint was not quantified for the oral or dermal exposure • Oral/dermal cancer slope factor (continuous/worker) = not derived • Inhalation Unit Risk (IUR) (continuous) = 6E−06 per µg/m^3, IUR (worker) = 2E−06 per µg/m^3 • Drinking water (DW) unit risk (continuous) = not derived
[a] All non-cancer and cancer hazard values are based on data for 1,2-dicholorethane read directly across to 1,1-dichloroethane as an analog. [b] The dermal HED are extrapolated from the oral HED and are assumed to be equal. [c] Uncertainty factors in the benchmark MOE (margin of exposure): UF_A = interspecies (animal to human); UF_H = intraspecies (human variability); UF_S = subchronic to chronic	

5.3.1.1 Estimation of Non-Cancer Risks

EPA used a margin of exposure (MOE) approach to estimate non-cancer risks. The MOE is the ratio of the non-cancer hazard value divided by a human exposure dose. Acute, intermediate, and chronic MOEs for non-cancer inhalation and dermal risks were calculated using Equation 5-8:

Equation 5-8.

$$MOE = (Noncancer\ Hazard\ Value\ (POD))/(Human\ Exposure)$$

Where:

MOE	=	Margin of exposure for acute, intermediate, or chronic risk comparison (unitless)
$Noncancer\ Hazard\ Value\ (POD)$	=	HEC (mg/m^3) or HED (mg/kg-day)
$Human\ Exposure$	=	Exposure estimate (mg/m^3 or mg/kg-day)

MOE risk estimates may be interpreted in relation to benchmark MOEs. Benchmark MOEs are typically the total UF for each non-cancer hazard value. The MOE estimate is interpreted as a human health risk of concern if the MOE estimate is less than the benchmark MOE (*i.e.*, the total UF). On the other hand, if the MOE estimate is equal to or exceeds the benchmark MOE, risk is not considered to be of concern and mitigation is not needed. Typically, the larger the MOE, the more unlikely it is that a non-cancer adverse effect occurs relative to the benchmark. When determining if 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 in addition to risks identified in risk characterization.

5.3.1.2 Estimation of Cancer Risks

Extra cancer risks for repeated exposures to a chemical were estimated using Equation 5-9 or Equation 5-10:

Equation 5-9.

$$Inhalation\ Cancer\ Risk = Human\ Exposure \times IUR$$

Or

Equation 5-10.

$$\boldsymbol{Dermal\ or\ Oral\ Cancer\ Risk = Human\ Exposure \times CSF}$$

Where:

Risk	=	Extra cancer risk (unitless)
Human Exposure	=	Exposure estimate (LADC in ppm)
IUR	=	Inhalation unit risk (risk per mg/m^3)
CSF	=	Cancer slope factor (risk per mg/kg-day)

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

5.3.2 Risk Characterization for Potentially Exposed or Susceptible Subpopulations

EPA considered PESS throughout the exposure assessment and throughout the hazard identification and dose-response analysis. In general, the Agency evaluates several factors that may contribute to a group having increased exposure or biological susceptibility. Examples of these factors include lifestage, preexisting disease, occupational and certain consumer exposures, nutrition, and lifestyle activities.

For the 1,1-dichloroethane risk evaluation, EPA accounted for the following PESS groups: workers, infants exposed to drinking water during formula bottle feeding, subsistence and Tribal fishers, individuals with pre-existing conditions such as chronic kidney disease, people with the aldehyde dehydrogenase-2 polymorphism, lifestyle factors such as smoking cigarettes or secondhand smoke, and communities who live near facilities that emit 1,1-dichloroethane.

Table 5-49 summarizes how PESS were incorporated into the risk evaluation and the remaining sources of uncertainty related to consideration of PESS.

Additional information on other factors that could possibly impact greater biological susceptibility following exposure to 1,1-dichloroethane—such as more comprehensive information on pre-existing diseases in humans, lifestyle activities, nutritional status, or other chemical co-exposures and non-chemical stressors—was not reasonably available.

Table 5-49. Summary of PESS Categories in the Risk Evaluation and Remaining Sources of Uncertainty

PESS Categories	Potential Increased Exposures Incorporated into Exposure Assessment	Potential Sources of Biological Susceptibility Incorporated into Hazard Assessment
Lifestage	Lifestage-specific exposure scenarios included infants exposed to drinking water during formula bottle feeding. Exposure factors by age group were applied to calculate exposure. Other scenarios of children swimming or playing in soil may be considered for dermal and oral exposure. It is unclear how relevant dermal and ingestion estimates from soil exposure are as 1,1-dichloroethane is expected to either volatilize or migrate from surface soils to groundwater. Other factors by age may be relevant.	Direct evidence of a sperm effect was the basis for the chronic inhalation POD used for risk estimation. The inhalation POD selected is considered to be protective and data were incorporated in the weight of scientific evidence. The analog 1,2-dichloroethane partitions in the milk of women exposed dermally (ATSDR, 2024; Urusova, 1953) in toxicokinetic considerations. Children in households that smoke cigarettes, receiving secondhand smoke, may be exposed to higher levels of 1,1-dichloroethane (ATSDR, 2024; Wang et al., 2012). The increase in susceptibility due to secondhand smoke is not known and is a source of uncertainty in part reliant on proximity to the smoker, space ventilation, and frequency of smoking/number of cigarettes smoked. Evidence also from mice showed changes in sperm parameters in decreases in sperm count following short-term exposures to the analog 1,2-dichloroethane. Potential susceptibility of older adults due to toxicokinetic differences was addressed through a 10× UF for human variability.
Pre-Existing Disease	Not applicable	Application of a 10× UF_H to account for human variability. Especially susceptible individuals, such as those with chronic kidney disease, may not be accounted for by standard approaches. The increase in susceptibility due to pre-existing disease is not known and is a source of uncertainty.
Lifestyle Activities	EPA evaluated exposures resulting for subsistence and Tribal fishers and considered increased intake of fish in these populations. People that smoke cigarettes may be exposed to higher levels of 1,1-dichloroethane. Emissions from smoking cigarettes can contain between 51 and 110 µg 1,1-dichloroethane/cigarette (ATSDR, 2024; Wang et al., 2012).	EPA considered alcohol consumption and smoking as factors included in the human variability.
Occupational Exposures	EPA considered increased exposure specific to worker activities.	Not applicable.

PESS Categories	Potential Increased Exposures Incorporated into Exposure Assessment	Potential Sources of Biological Susceptibility Incorporated into Hazard Assessment
Sociodemographic	EPA evaluated exposure differences between groups, including women of reproductive age based on location of exposures to 1,1-dichloroethane in ambient air.	EPA utilized the most sensitive sex from rodent assays cancer modeling. EPA quantified sociodemographic differences based on sex alone.
Geography and Site-Specific	Potential for increased exposures included children under 5 and 18 years old because childcare centers and public schools were observed near several of the AERMOD TRI release sites. See Section 5.3.4. There is some uncertainty associated with the modeled distances from each release point and the associated exposure concentrations to which residential communities proximal to releasing facilities may be exposed.	Not applicable.
Nutrition	Not applicable.	EPA did not identify nutritional factors that influence susceptibility.
Genetics/ Epigenetics	Not applicable.	Genetic variants may increase susceptibility of the target organ was addressed through a 10× UF_H for human variability. A known metabolite of 1,1-dichloroethane is the reactive dichloroacetaldehyde supporting that a PESS group are people with the aldehyde dehydrogenase-2 polymorphism which may have a higher risk for several diseases affecting multiple organ systems including cancer, heart disease and osteoporosis. Hazard values are based on wild-type rodents and a broad occupational population and may underestimate risks for populations with sensitizing mutations.
Other Unique Activities	EPA did not identify unique activities that influence exposure.	EPA did not identify unique activities that influence susceptibility.
Aggregate Exposures	EPA assessed aggregate exposures to the general populations to the combined ambient air concentrations from several adjacent facility air releases. EPA did not aggregate routes of exposure as the endpoints are different and dependent on the corresponding route of exposure.	Not applicable.

PESS Categories	Potential Increased Exposures Incorporated into Exposure Assessment	Potential Sources of Biological Susceptibility Incorporated into Hazard Assessment
Other Chemical and Nonchemical Stressors	EPA did not identify other chemical and non-chemical factors influencing exposure.	EPA did not identify other chemical and nonchemical stressors that influence susceptibility.

5.3.3 Human Health Risk Characterization

5.3.3.1 Risk Estimates for Workers

For each condition of use, EPA assessed 1,1-dichloroethane inhalation exposures to workers and ONUs in occupational settings, presented as 8-hour (*i.e.*, full-shift) TWA described in Section 5.1.1. These estimated exposures were then used to calculate acute, intermediate, and chronic (non-cancer and cancer) inhalation exposures and dermal doses. These calculations require additional parameter inputs such as years of exposure, exposure duration and frequency, and lifetime years. EPA used combinations of point estimates of each parameter to estimate a central tendency and high-end for each final exposure metric result. EPA documented the method and rationale for selecting parametric combinations to be representative of central tendency and high-end.

EPA also assessed 1,1-dichloroethane dermal exposures to workers in occupational settings, presented as a dermal APDR. The APDRs are then used to calculate acute retained doses (ARD), intermediate retained dose (IRD), and chronic retained dose (CRD) for chronic non-cancer risks.

The input parameter values in Table 5-50 are used to calculate each of the above acute, subchronic, and chronic exposure estimates. For additional details on the parameters, refer to *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment* (U.S. EPA, 2025b).

Table 5-50. Parameter Values for Calculating Exposure Estimates

Parameter Name	Symbol	Value	Unit
Exposure Duration	ED	8	h/day
Breathing Rate Ratio	BR	2.04[a]	unitless
Exposure Frequency	EF	125–250[b]	days/year
Exposure Frequency, Subchronic	EF_{sc}	22	days
Days for Subchronic Duration	SCD	30	days
Working Years	WY	31 (50th percentile) 40 (95th percentile)	years
Lifetime Years, Cancer	LT	78	years
Averaging Time, Subchronic	AT_{sc}	720	hours
Averaging Time, Non-Cancer	AT	271,560 (central tendency) [c] 350,400 (high-end) [d]	hours
Averaging Time, Cancer	AT_c	683,280	hours
Body Weight	BW	80 (average adult worker) 72.4 (female of reproductive age)	kg

[a] EPA uses a breathing rate ratio, which is the ratio between the worker breathing rate and resting breathing rate, to account for the amount of air a worker breathes during exposure. The typical worker breathes about 10 m^3 of air in 8 hours, or 1.25 m^3/h (CEB, 1991) while the resting breathing rate is 0.6125 m^3/h (CEB, 1991). The ratio of these two values is equivalent to 2.04.
[b] Depending on OES; maximum number of exposure days was assumed to be 250 days per year.
[c] Calculated using the 95th percentile value for working years (WY).
[b] Calculated using the 50th percentile value for WY.

5.3.3.1.1 Acute Risk

Acute non-cancer (AC) is used to estimate workplace inhalation exposures for acute risks (*i.e.*, risks occurring as a result of exposure for less than one day), per Equation 5-11:

Equation 5-11.

$$AC = (C \times ED \times BR)/(AT_{acute})$$

Where:

AC	=	Acute exposure concentration
C	=	Contaminant concentration in air (TWA)
ED	=	Exposure duration (h/day)
BR	=	Breathing rate ratio (unitless)
AT_{acute}	=	Acute averaging time (h)

A sample calculation for the high-end acute inhalation exposure concentration (AC_{HE}) for the Manufacturing OES is demonstrated in Equation 5-12 below:

Equation 5-12.

$$AC_{HE} = (C_{HE} \times ED \times BR)/(A_{acute})$$

$$AC_{HE} = (1.1\ ppm \times 8\ hr/day \times 2.04)/(24\ hr/day) = \ 0.72 \text{ ppm}$$

Acute Retained Dose (*ARD*) is used to estimate workplace dermal exposures for acute risks and are calculated using Equation 5-13:

Equation 5-13.

$$ARD = APDR/BW$$

Where:

ARD	=	Acute retained dose (mg/kg-day)
$APDR$	=	Acute potential dose rate (mg/day)
BW	=	Body weight (kg)

A sample calculation for the high-end acute retained dose for the Manufacturing OES is demonstrated in Equation 5-14 below:

Equation 5-14.

$$ARD_{HE} = APDR_{HE}/BW$$

$$ARD_{HE} = (6.7\ mg/day)/(80\ kg) = 0.08\ mg\ /\ (kg - day)$$

5.3.3.1.2 Intermediate Risk

Intermediate non-cancer ($ADC_{intermediate}$) is used to estimate workplace inhalation exposures for intermediate risks and is estimated in Equation 5-15 and Equation 5-16, as follows:

Equation 5-15.

$$ADC_{intermediate} = (C \times ED \times EF_{intermediate} \times BR)/AT_{intermediate}$$

Equation 5-16.

$$AT_{intermediate} = ID \times 24\ hr/day$$

Where:

$ADC_{intermediate}$	=	Intermediate average daily concentration
$EF_{intermediate}$	=	Intermediate exposure frequency
$AT_{intermediate}$	=	Averaging time (hour) for intermediate exposure
ID	=	Days for intermediate duration (day)

A sample calculation for the intermediate exposure concentration ($ADC_{intermediate, HE}$) for the Manufacturing OES is demonstrated in Equation 5-17 below:

Equation 5-17.

$$ADC_{intermediate} = (C_{HE} \times ED \times EF_{intermediate} \times BR)/AT_{intermediate}$$

$$ADC_{intermediate,HE} = (1.1\ ppm \times 8\ "hr"/day \times 22\ "days"/year \times 2.04)/(24\ "hr"/day \times 30\ "days"/year) = 0.53\ ppm$$

Intermediate retained dose (IRD) is used to estimate workplace dermal exposures for intermediate risks, and is estimated using Equation 5-18:

Equation 5-18.

$$IRD = (AD \times EF_{intermediate} \times WY)/AT_{intermediate}$$

Where:

IRD	=	Intermediate retained dose (mg/kg-day)

A sample calculation for the high-end intermediate retained dose for the Manufacturing OES is demonstrated in Equation 5-19 below:

Equation 5-19.

$$IRD_{HE} = (ARD_{HE} \times EF_{intermediate} \times WY_{HE})/AT_{intermediate}$$

$$IRD_{HE} = (0.08\ mg/(kg-day) \times 22\ "day"/yr \times 40\ "yr")/(30\ "day") = 0.06\ mg\ /\ (kg-)\ day$$

5.3.3.1.3 Chronic Non-Cancer Risk

The Average daily concentration (ADC) is used to estimate workplace inhalation exposures for non-cancer risk. This exposure is estimated as follows in Equation 5-20 and Equation 5-21:

Equation 5-20.

$$ADC = (C \times ED \times EF \times WY \times BR)/AT$$

Equation 5-21.

$$AT = WY \times 365\ "day"\ /"yr"\ \times 24\ "hr"\ /"day"$$

Where:

ADC	=	Average daily concentration used for chronic non-cancer risk calculations
ED	=	Exposure duration (hr/day)
EF	=	Exposure frequency (day/year)
WY	=	Working years per lifetime (yr)
AT	=	Averaging time (hour) for chronic, non-cancer risk

A sample calculation for the high-end chronic non-cancer exposure concentration (ADC_{HE}) for the Manufacturing OES is demonstrated in Equation 5-22 below:

Equation 5-22.

$$ADC_{HE} = (C_{HE} \times ED \times EF \times WY \times BR)/AT$$

$$ADC_{HE} = (1.1\ ppm \times 8\ hr/day \times 250\ days/year \times 40\ years \times 2.04)/(40\ years \times 365\ days/yr \times 24\ hr/day) = 0.49\ ppm$$

The chronic retained dose (*CRD*) is used to estimate workplace dermal exposures for non-cancer risk and is calculated using Equation 5-23:

Equation 5-23.

$$CRD = (ARD \times EF \times WY)/(AT_{chronic}\ )$$

A sample calculation for the high-end chronic retained dose for the Manufacturing OES is demonstrated in Equation 5-24 below:

Equation 5-24.

$$CRD_{HE} = (ARD_{HE} \times EF \times WY)/(AT_{chronic}\ )$$

$$CRD_{HE} = (0.08\ mg/(kg - day) \times 250\ day/yr \times 40\ yr)/(14{,}600\ day) = 0.06\ (\ mg)\ /\ (kg-)\ day$$

5.3.3.1.4 Cancer Risk

Lifetime average daily concentration (LADC) is used to estimate workplace inhalation exposures for cancer risk. This exposure is estimated as follows in Equation 5-25 and Equation 5-26:

Equation 5-25.

$$LADC = (C \times ED \times EF \times WY \times BR)/AT_C$$

Equation 5-26.

$$AT_C\ = LT \times 365\ "day"\ /"yr"\ \times 24\ "hr"\ /"day"$$

Where:

$LADC$	=	Lifetime average daily concentration used for chronic cancer risk calculations
ED	=	Exposure duration (hr/day)
EF	=	Exposure frequency (day/year)
WY	=	Working years per lifetime (yr)
AT_C	=	Averaging time (hour) for cancer risk
LT	=	Lifetime years (yr) for cancer risk

A sample calculation for the high-end chronic cancer exposure concentration ($LADC_{HE}$) for the Manufacturing OES is demonstrated in Equation 5-27 below:

Equation 5-27.

$$LADC_{HE} = (C_{HE} \times ED \times EF \times WY \times BR)/(AT_C\ )$$

$$LADC_{HE} = (1.1\ ppm \times 8\ hr/day \times 250\ days/year \times 40\ years \times 2.04)/(78\ years \times 365\ days/year \times 24\ hr/day) = 0.25\ ppm$$

Lifetime chronic retained dose (LCRD) for cancer risk was not estimated as dermal cancer numbers for 1,1-dichloroethane were not derived.

5.3.3.1.5 Occupational Exposure and Risk Summary by OES

The occupational inhalation exposure metrics described in 5.3.3.1.1 through 5.3.3.1.4 are presented in Table 5-51, and the occupational dermal exposure metrics are presented in Table 5-53. EPA used the exposure metrics presented in Table 5-51 and Table 5-53 and the approach described in Sections 5.3.1.1 and 5.3.1.2 to develop risk estimates for each 1,1-dichloroethane exposure scenario. The risk estimates are presented below in Table 5-55. For additional details on the risk estimates, refer to *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Risk Calculator for Occupational Exposure.*

Under section 4(a) of TSCA, EPA issued a test order requiring manufactures and processors of 1,1-dichloroethane to develop and submit certain information for 1,1-dichloroethane. In response, the Vinyl Institute formed a testing consortium and provided data on occupational exposure (Stantec ChemRisk, 2023). The Vinyl Institute prepared a study plan for inhalation monitoring to collect inhalation monitoring data, which included identification of representative sites for sampling. The testing consortium provided information on 12 total sites from their members that manufacture 1,1-dichloroethane as an isolated intermediate and/or non-isolated byproduct and selected 4 representative sites for sampling following EPA's review and approval of the study plan. During the inhalation sampling study, operators wore half-face, air-purifying respirators (APF 10) during sample collection tasks (open or closed loop) and full-face respirators of varying types (APF 50 – 1,000) during other tasks with exposure potential such as process leak response activities. Maintenance technicians wore respiratory protection during major maintenance tasks (*e.g.,* line breaks and other equipment openings). It was noted that maintenance technicians may wear full-face airline respirators during line breaks and equipment opening tasks at certain facilities; however, the use of airline respirators was not observed during the inhalation sampling study. Logistics technicians wore half-face or full-face respirators during loading or offloading tasks, which required connecting and disconnecting process lines to railcars, tanks, and trucks. Certain lab personnel wore half-face air-purifying respirators during sample handling and processing, such as, during the preparation of dry standards on a benchtop surface. ONUs were not reported to wear respiratory protection during any routine daily tasks.

The information in the test order study report on the use of protective gloves and respirators when performing tasks with a potential for exposure, is consistent with observations made by EPA during a site visit to OxyChem's Geismar, Louisiana, facility in June 2023. This facility manufactures 1,2-dichloroethane and the site visit included a demonstration of a line break procedure to complete a maintenance task. Permits must be obtained prior to such a procedure and specific steps must be followed during the procedure. This included the use of protective gloves, protective clothing with a hood and use of a supplied-air respirator. After the repair was completed, a different set of checks and procedures needed to be followed before this part of the process could be placed back on-line.

Using the test order information, EPA prepared tables that included protection factors resulting from the use of PPE.

Table 5-52 provides occupational inhalation MOEs with and without PPE for the OES and worker categories assessed for 1,1-dichloroethane.

More generally, the Vinyl Institute test order provided data on the use of PPE (Stantec ChemRisk, 2023). The data were for the four facilities that were monitored for the test order and these facilities manufacture 1,1-dichloroethane as either an isolated intermediate or as a byproduct. Each representative facility utilized similar standard process area PPE, task-specific PPE, and emergency use PPE (*e.g.*, during an accidental release, spill, or leak). The type of PPE used depended on the process area and task performed. As such, individuals in each SEG required different types of PPE dependent on the process area in which they worked and the types of tasks they performed. For example, maintenance technicians wore standard process area PPE while conducting maintenance tasks in production process areas but donned additional PPE as necessary for specific maintenance tasks. Similarly, at one of the facilities, a laboratory technician cross-trained as an operator and was required to collect process samples, during which task they wore standard process area PPE and task-specific PPE required for sample collection. When the laboratory technician returned to the laboratory for sample processing, they donned appropriate laboratory PPE. When conducting process walkthroughs or other tasks that required them to enter process areas, ONUs donned standard process area PPE. Routine tasks conducted by ONUs (*e.g.*, office work) did not require access to process areas with exposure potential, and thus no PPE was required for these workers.

Table 5-54 provides occupational dermal MOEs with and without PPE for the OES and worker categories assessed for 1,1-dichloroethane. Where the MOE is less than the benchmark MOE, the Dermal Protection Factor (PF) needed to raise the MOE above the benchmark value is also provided.

Table 5-51. Summary of Occupational Inhalation Exposure Metrics

Occupational Exposure Scenario (OES)	Category	8-Hour TWA Exposures		Acute, Non-Cancer Exposures		Intermediate, Non-Cancer		Chronic, Non-Cancer Exposures		Chronic, Cancer Exposures	
		8-Hour TWA (ppm)		$AC_{8\text{-hr TWA}}$ (ppm)		$ADC_{8\text{-hr TWA}}$ (ppm)		$ADC_{8\text{-hr TWA}}$ (ppm)		$LADC_{8\text{-hr TWA}}$ (ppm)	
		Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End
Manufacturing as an isolated intermediate	Operator/Process Technician	7.8E−03	0.73	5.3E−03	0.50	3.89E−03	0.36	3.6E−03	0.34	1.4E−03	0.17
Manufacturing as an isolated intermediate	Operator/Process Technician (responding to line leaks)	1.9	1.9	1.3	1.3	4.4E−02	4.2E−02	–	–	–	–
Manufacturing as an isolated intermediate	Logistics Technician	2.8E−03	5.3E−03	1.9E−03	3.6E−03	1.41E−03	2.62E−03	1.3E−03	2.4E−03	5.2E−04	1.3E−03
Manufacturing as an isolated intermediate	Maintenance Technician	7.9E−02	0.41	5.4E−02	0.28	4.0E−02	0.21	3.7E−02	0.19	1.5E−02	9.9E−02
Manufacturing as an isolated intermediate	Laboratory technician	1.1E−03	2.4E−02	7.7E−04	1.6E−02	5.7E−04	1.2E−02	5.3E−04	1.1E−02	2.1E−04	5.6E−03
Manufacturing as an isolated intermediate	ONU	3.2E−03	2.0E−02	2.2E−03	1.4E−02	1.6E−03	1.0E−02	1.5E−03	9.4E−03	5.9E−04	4.8E−03
Processing as a reactive intermediate	Operator/Process Technician	7.8E−03	0.73	5.3E−03	0.50	3.89E−03	0.36	3.6E−03	0.34	1.4E−03	0.17
Processing as a reactive intermediate	Logistics Technician	2.8E−03	5.3E−03	1.9E−03	3.6E−03	1.41E−03	2.62E−03	1.3E−03	2.4E−03	5.2E−04	1.3E−03
Processing as a reactive intermediate	Maintenance technician	7.9E−02	0.41	5.4E−02	0.28	4.0E−02	0.21	3.7E−02	0.19	1.5E−02	9.9E−02
Processing as a reactive intermediate	Laboratory technician	1.1E−03	2.4E−02	7.7E−04	1.6E−02	5.7E−04	1.2E−02	5.3E−04	1.1E−02	2.1E−04	5.6E−03
Processing as a reactive intermediate	ONU	3.2E−03	2.0E−02	2.2E−03	1.4E−02	1.6E−03	1.0E−02	1.5E−03	9.4E−03	5.9E−04	4.8E−03
Processing – repackaging (loading, unloading, and cleaning)	Worker	3.5	13	2.4	8.8	1.8	6.4	0.17	3.1	6.8E−02	1.6

Occupational Exposure Scenario (OES)	Category	8-Hour TWA Exposures		Acute, Non-Cancer Exposures		Intermediate, Non-Cancer		Chronic, Non-Cancer Exposures		Chronic, Cancer Exposures	
		8-Hour TWA (ppm)		$AC_{8\text{-hr TWA}}$ (ppm)		$ADC_{8\text{-hr TWA}}$ (ppm)		$ADC_{8\text{-hr TWA}}$ (ppm)		$LADC_{8\text{-hr TWA}}$ (ppm)	
		Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End
Processing – repackaging (unloading and cleaning)	Worker	1.7	6.6	1.2	4.5	0.86	3.3	8.4E−02	1.6	3.3E−02	0.80
Processing – repackaging (loading)	Worker	1.7	6.6	1.2	4.5	0.86	3.3	8.4E−02	1.6	3.3E−02	0.81
Processing – repackaging	ONU	3.5	3.5	2.4	2.4	1.8	1.8	0.17	0.84	6.8E−02	0.43
Commercial use as a laboratory chemical	Laboratory Technician	1.1E−03	2.4E−02	7.7E−04	1.6E−02	5.7E−04	1.2E−02	3.7E−04	1.1E−02	1.5E−04	5.6E−03
	ONU	1.1E−03	1.1E−03	1.1E093	1.1E−03	7.7E−04	7.7E−04	3.7E−04	5.3E−04	1.5E−04	2.7E−04
General waste handling, treatment, and disposal	Worker	0.30	10	0.20	7.1	0.15	5.2	0.14	4.9	5.5E−02	2.5
	ONU	0.30	0.30	0.20	0.20	0.15	0.15	0.14	0.14	5.5E−02	7.1E−02
Waste handling, treatment, and disposal (POTW)	Worker	0.25	0.68	0.17	0.46	0.13	0.34	0.12	0.32	4.7E−02	0.16
	ONU	0.25	0.25	0.17	0.17	0.13	0.13	0.12	0.12	4.7E−02	6.1E−02

Table 5-52. Occupational Inhalation MOEs Without PPE and PPE Level Needed to Exceed Benchmark MOE

Occupational Exposure Scenario (OES)	Category	Exposure Level	Acute Non-Cancer (Benchmark = 30)		Intermediate Non-Cancer (Benchmark MOE = 30)		Chronic Non-Cancer (Benchmark MOE = 300)		Chronic Cancer (Benchmark MOE of 1E−04 Selected)	
			MOE – No PPE	MOE – APF[a]	MOE – No PPE	MOE – APF*a*	MOE – No PPE	MOE – APF[a]	MOE – No PPE	MOE – APF[a]
Manufacture of 1,1-dichloroethane as an isolated intermediate	Operator/Process Technician	Central Tendency	1,911	–	5,652	–	6,052	–	1.37E−05	–
		High-End	**20**	203 (APF 10)	60	–	**64**	643 (APF 10)	**1.67E−03**	6.67E−05 (APF 25)
	Operator/Process Technician (responding to line leaks)	Central Tendency	**8.0**	80 (APF 10)	520	–	–	–	–	–
		High-End	**7.7**	77 (APF 10)	504	–	–	–	–	–
	Maintenance Technician	Central Tendency	188	–	555	–	595	–	**1.40E−04**	1.40E−05 (APF 10)
		High-End	36	–	107	–	**114**	1,145 (APF 10)	**9.36E−04**	9.36E−05 (APF 10)
	Logistics/Distribution Technician	Central Tendency	5,284	–	1.6E04	–	1.7E04	–	4.96E−06	–
		High-End	2,837	–	8,394	–	8,987	–	1.19E−05	–
	Laboratory Technician	Central Tendency	1.3E04	–	3.9E04	–	4.2E04	–	1.99E−06	–
		High-End	631	–	1,866	–	1,998	–	5.36E−05	–
	ONUs	Central Tendency	8,327	–	2.5E04	–	2.6E04	–	3.15E−06	–
		High-End	811	–	2,398	–	2,568	–	4.17E−05	
Processing – repackaging	Worker: Scenario 1 – All Activities	Central Tendency	**4.2**	42 (APF 10)	**13**	126 (APF 10)	**129**	1,294 (APF 10)	**6.42E−04**	6.42E−05 (APF 10)
		High-End	**1.2**	58 (APF 50)	**3.4**	34 (APF 10)	**7.1**	357 (APF 50)	**1.50E−02**	1.5E−05 (APF 1,000)
	Worker: Scenario 2 – Unloading and Cleaning Activities Only	Central Tendency	**8.6**	86 (APF 10)	**25**	255 (APF 10)	**262**	2,622 (APF 10)	**3.17E−04**	3.17E−05 (APF 10)
		High-End	**2.3**	57 (APF 25)	**6.7**	67 (APF 10)	**14**	352 (APF 25)	**7.62E−03**	7.62E−06 (APF 1,000)
	Worker: Scenario 3 – Loading Activity Only	Central Tendency	**8.6**	86 (APF 10)	**26**	255 (APF 10)	**263**	2,625 (APF 10)	**3.16E−04**	3.16E−05 (APF 10)
		High-End	**2.3**	57 (APF 25)	**6.7**	67 (APF 10)	**14**	350 (APF 25)	**7.65E−03**	7.65E−06 (APF 1,000)

Occupational Exposure Scenario (OES)	Category	Exposure Level	Acute Non-Cancer (Benchmark = 30)		Intermediate Non-Cancer (Benchmark MOE = 30)		Chronic Non-Cancer (Benchmark MOE = 300)		Chronic Cancer (Benchmark MOE of 1E−04 Selected)	
			MOE – No PPE	MOE – APF[a]	MOE – No PPE	MOE – APF*a*	MOE – No PPE	MOE – APF[a]	MOE – No PPE	MOE – APF[a]
	ONUs	Central Tendency	**4.2**	42 (APF 10)	**13**	126 (APF 10)	**129**	1,294 (APF 10)	**6.42E−04**	6.42E−05 (APF 10)
		High-End	**4.2**	42 (APF 10)	**13**	126 (APF 10)	**129**	1,294 (APF 10)	**6.42E−04**	6.42E−05 (APF 10)
Processing as a reactive intermediate	Operator/Process Technician	Central Tendency	1,911	–	5,652	–	6,052		1.37E−05	
		High-End	**20**	203 (APF 10)	60	–	**64**	643 (APF 10)	**1.67E−03**	6.67E−05 (APF 25)
	Maintenance Technician	Central Tendency	188	–	555	–	595	–	**1.40E−04**	1.40E−05 (APF 10)
		High-End	36	–	107	–	**114**	1,145 (APF 10)	**9.36E−04**	9.36E−05 (APF 10)
	Logistics/ Distribution Technician	Central Tendency	5,284	–	1.6E04	–	1.7E04	–	4.96E−06	–
		High-End	2,837	–	8,394	–	8,987	–	1.19E−05	–
	Laboratory Technician	Central Tendency	1.3E04	–	3.9E04	–	4.2E04	–	1.99E−06	–
		High-End	631	–	1,866	–	1,998	–	5.36E−05	–
	ONUs	Central Tendency	8,327	–	2.5E04	–	2.6E=04	–	3.15E−06	–
		High-End	811	–	2,398	–	2,568	–	4.17E−05	–
Commercial use as a laboratory chemical	Laboratory Technician	Central Tendency	1.3E04	–	3.9E04	–	6.0E04	–	1.39E−06	–
		High-End	631	–	1,866	–	1,998	–	5.36E−05	–
	ONUs	Central Tendency	1.3E04	–	3.9E04	–	6.0E04	–	2.57E−06	–
		High-End	1.3E04	–	3.9E04	–	4.2E04	–	1.39E−06	–
Waste handling, treatment, and disposal – general	Worker	Central Tendency	50	–	149	–	**159**	1,592 (APF 10)	**5.22E−04**	5.22E−05 (APF 10)
		High-End	**1.4**	36 (APF 25)	**4.2**	42 (APF 10)	**4.5**	4,521 (APF 1,000)	**2.37E−02**	2.37E−5 (APF 10)
	ONUs	Central Tendency	50	–	149	–	**159**	1,592 (APF 10)	**5.22E−04**	5.22E−05 (APF 10)
		High-End	50	–	149	–	**159**	1,592 (APF 10)	**6.73E−04**	6.73E−05 (APF 10)

Occupational Exposure Scenario (OES)	Category	Exposure Level	Acute Non-Cancer (Benchmark = 30)		Intermediate Non-Cancer (Benchmark MOE = 30)		Chronic Non-Cancer (Benchmark MOE = 300)		Chronic Cancer (Benchmark MOE of 1E−04 Selected)	
			MOE – No PPE	MOE – APF[a]	MOE – No PPE	MOE – APF*a*	MOE – No PPE	MOE – APF[a]	MOE – No PPE	MOE – APF[a]
Waste handling, treatment, and disposal - POTW	Worker	Central Tendency	58	–	173	–	**185**	1,852 (APF 10)	**4.48E−04**	4.48E−05 (APF 10)
		High-End	**22**	219 (APF 10)	65	–	**69**	695 (APF 10)	**1.54E−03**	6.17E−05 (APF 25)
	ONU	Central Tendency	58	–	173	–	**185**	1,852 (APF 10)	**4.48E−04**	4.48E−05 (APF 10)
		High-End	58	–	173	–	**185**	1,852 (APF 10)	**4.48E−04**	4.48E−05 (APF 10)

APF = assigned protection factor; MOE = margin of exposure; POTW = publicly owned treatment works; PPE = personal protective equipment
[a] APF listed in parentheses is the level of protection needed for estimated inhalation MOEs to be above benchmark.
"–" = Inhalation APF not needed
"—" = chronic effects not evaluated for this SEG due to lower exposure frequency
Risk estimates that exceed the benchmark (*i.e.*, non-cancer risks less than the risk benchmark and cancer risks greater than the cancer risk benchmark) are bolded and shaded.

Table 5-53. Summary of Occupational Dermal Exposure Metrics

OES	Category	Estimation Method	Acute Potential Dose Rate		Acute Retained Dose		Intermediate Retained Dose, Non-Cancer		Chronic Retained Dose, Non-Cancer	
			APDR (mg/day)		ARD (mg/kg-day)		IRD (mg/kg-day)		CRD (mg/kg-day)	
			Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End
Manufacturing as an isolated intermediate	Operator/Process Technician	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.9E−02	5.8E−02
Manufacturing as an isolated intermediate	Operator/Process Technician	Probabilistic	3.2	5.5	4.0E−02	6.9E−02	3.0E−02	5.1E−02	2.8E−02	4.7E−02
Manufacturing as an isolated intermediate	Maintenance Technician	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.9E−02	5.8E−02
Manufacturing as an isolated intermediate	Maintenance Technician	Probabilistic	3.2	5.5	4.0E−02	6.9E−02	3.0E−02	5.1E−02	2.8E−02	4.7E−02
Manufacturing as an isolated intermediate	Laboratory Technician	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.9E−02	5.8E−02
Manufacturing as an isolated intermediate	Laboratory Technician	Probabilistic	3.2	5.5	4.0E−02	6.9E−02	3.0E−02	5.1E−02	2.8E−02	4.7E−02
Processing as a reactive intermediate	Operator/Process Technician	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.9E−02	5.8E−02
Processing as a reactive intermediate	Operator/Process Technician	Probabilistic	3.2	5.5	4.0E−02	6.9E−02	3.0E−02	5.1E−02	2.8E−02	4.7E−02
Processing as a reactive intermediate	Maintenance Technician	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.9E−02	5.8E−02
Processing as a reactive intermediate	Maintenance Technician	Probabilistic	3.2	5.5	4.0E−02	6.9E−02	3.0E−02	5.1E−02	2.8E−02	4.7E−02
Processing as a reactive intermediate	Laboratory Technician	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.9E−02	5.8E−02
Processing as a reactive intermediate	Laboratory Technician	Probabilistic	3.2	5.5	4.0E−02	6.9E−02	3.0E−02	5.1E−02	2.8E−02	4.7E−02
Processing – repackaging	Worker	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	2.0E−03	3.0E−02
Processing – repackaging	Worker	Probabilistic	3.2	5.5	4.1E−02	6.9E−02	3.0E−02	5.1E−02	8.0E−03	1.8E−02
Commercial use as a laboratory chemical	Worker	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.3E−02	5.8E−02

OES	Category	Estimation Method	Acute Potential Dose Rate		Acute Retained Dose		Intermediate Retained Dose, Non-Cancer		Chronic Retained Dose, Non-Cancer	
			APDR (mg/day)		ARD (mg/kg-day)		IRD (mg/kg-day)		CRD (mg/kg-day)	
			Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End	Central Tendency	High-End
Commercial use as a laboratory chemical	Worker	Probabilistic	3.2	5.5	4.0E−02	6.8E−02	2.9E−02	5.0E−02	2.4E−02	4.3E−02
General waste handling, treatment, and disposal	Worker	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.9E−02	5.8E−02
General waste handling, treatment, and disposal	Worker	Probabilistic	3.2	5.5	4.0E−02	6.9E−02	3.0E−02	5.1E−02	2.8E−02	4.7E−02
General waste handling, treatment, and disposal (dilute)	Worker	Deterministic	4.5E−02	0.13	5.6E−04	1.7E−03	4.1E−04	1.2E−03	3.8E−04	1.2E−03
General waste handling, treatment, and disposal (dilute)	Worker	Probabilistic	6.5E−02	0.11	8.1E−04	1.4E−03	6.0E−04	1.0E−03	5.6E−04	9.5E−04
Waste handling, treatment, and disposal (POTW)	Worker	Deterministic	2.2	6.7	2.8E−02	8.4E−02	2.1E−02	6.2E−02	1.9E−02	5.8E−02
Waste handling, treatment, and disposal (POTW)	Worker	Probabilistic	3.2	5.5	4.0E−02	6.9E−02	3.0E−02	5.1E−02	2.8E−02	4.7E−02
Waste handling, treatment, and disposal (POTW) (dilute)	Worker	Deterministic	4.5E−02	0.13	5.6E−04	1.7E−03	4.1E−04	1.2E−03	3.8E−04	1.2E−03
Waste handling, treatment, and disposal (POTW) (dilute)	Worker	Probabilistic	6.5E−02	0.11	8.1E−04	1.4E−03	6.0E−04	1.0E−03	5.6E−04	9.5E−04

Table 5-54. Occupational Dermal MOEs Without PPE and PPE Level Needed to Exceed Benchmark MOE

OES	Category	Exposure Level	Acute Non-Cancer (Benchmark = 30)		Intermediate Non-Cancer (Benchmark MOE = 30)		Chronic Non-Cancer (Benchmark MOE = 300)		Chronic Cancer – Not Evaluated for 1,1-Dichloroethane	
			MOE – No Gloves	MOE – Dermal PF	MOE – No Gloves	MOE – Dermal PF	MOE – No Gloves	MOE – Dermal PF	MOE – No Gloves	MOE – Dermal PF
Manufacture of 1,1-dichloroethane as an isolated intermediate	Operator/Process Technician	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
	Operator/Process Technician (responding to line leaks)	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
	Maintenance Technician	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
	Logistics/Distribution Technician	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
	Laboratory Technician	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
Processing – repackaging scenario 1	All Activities	Central Tendency	709		442		4,548		–	–
		High-End	236		147		308		–	–
Processing – repackaging scenario 2	Unloading and Cleaning	Central Tendency	709		442		4,548		–	–
		High-End	236		147		308		–	–
Processing – repackaging scenario 3	Loading	Central Tendency	709		442		4,548		–	–
		High-End	236		147		308		–	–
Processing as a reactive intermediate	Operator/Process Technician	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
	Maintenance Technician	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
	Logistics/Distribution Technician	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
	Laboratory Technician	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–

OES	Category	Exposure Level	Acute Non-Cancer (Benchmark = 30)		Intermediate Non-Cancer (Benchmark MOE = 30)		Chronic Non-Cancer (Benchmark MOE = 300)		Chronic Cancer – Not Evaluated for 1,1-Dichloroethane	
			MOE – No Gloves	MOE – Dermal PF	MOE – No Gloves	MOE – Dermal PF	MOE – No Gloves	MOE – Dermal PF	MOE – No Gloves	MOE – Dermal PF
Commercial use as a laboratory chemical		Central Tendency	709		442		680		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
General waste handling, treatment, and disposal	Dilute Scenario	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
		Central Tendency	3.5E04		2.2E04		2.4E04		–	–
		High-End	1.2E04		7,363		7,884		–	–
Waste handling, treatment, and disposal	POTW	Central Tendency	709		442		473		–	–
		High-End	236		147		**158**	788 (PF 5)	–	–
	POTW – Dilute Scenario	Central Tendency	3.5E04		2.2E04		2.4E04		–	–
		High-End	1.2E04		7,363		7,884		–	–

OES = occupational exposure scenario; MOE = margin of exposure; PF = protection factor; POTW = publicly owned treatment works; "–" = not evaluated
Chronic cancer risks were not evaluated for 1,1-dichloroethane.
Risk estimates that exceed the benchmark (*i.e.*, non-cancer risks less than the risk benchmark) are bolded and shaded.

Table 5-55. Occupational Risk Summary Table

Life Cycle Stage/ Category	Subcategory	OES Scenario	Population	Exposure Route and Duration	Exposure Level	Risk Estimates for Each Exposure Scenario			
						Acute Non-Cancer (Benchmark MOE: Dermal = 30; Inhalation = 30)	**Intermediate, Non-Cancer (Benchmark MOE: Dermal = 30 Inhalation = 30)**	**Chronic, Non-Cancer (Benchmark MOE: Dermal = 300; Inhalation=300)**	**Cancer (Benchmark = 10E−4)**
Manufacture/ Domestic Manufacturing	Domestic manufacture	Manufacturing	Operator/ Process Technician	Inhalation	Central Tendency	1,911	5,652	6,052	1.4E−05
					High-End	**20**	60	**64**	**1.7E−03**
			Operator / Process Technician (responding to line leaks)	Inhalation	Central Tendency	**8.0**	520	Not estimated for this SEG	Not estimated for this SEG
					High-End	**7.7**	504	Not estimated for this SEG	Not estimated for this SEG
			Maintenance Technician	Inhalation	Central Tendency	188	555	595	**1.4E−04**
					High-End	36	107	**114**	**9.4E−04**
			Logistics/ Distribution Technician	Inhalation	Central Tendency	5,284	1.6E04	1.7E04	5.0E−06
					High-End	2,837	8,394	8,987	1.2E−05
			Laboratory Technician	Inhalation	Central Tendency	1.3E04	3.9E04	4.2E04	2.0E−06
					High-End	631	1,866	1,998	5.4E−05
			Worker – Deterministic	Dermal	Central Tendency	709	442	473	NE
					High-End	236	147	**158**	NE
Manufacture/ Domestic Manufacturing	Domestic manufacture	Manufacturing	Worker – Probabilistic	Dermal	Central Tendency	492	307	328	NE
					High-End	288	179	**192**	NE
			ONU	Inhalation	Central Tendency	8,327	2.5E04	2.6E04	3.1E−06
					High-End	811	2,398	2,568	4.2E−05

Life Cycle Stage/ Category	Subcategory	OES Scenario	Population	Exposure Route and Duration	Exposure Level	Risk Estimates for Each Exposure Scenario			
						Acute Non-Cancer (Benchmark MOE: Dermal = 30; Inhalation = 30)	Intermediate, Non-Cancer (Benchmark MOE: Dermal = 30 Inhalation = 30)	Chronic, Non-Cancer (Benchmark MOE: Dermal = 300; Inhalation=300)	Cancer (Benchmark = 10E−4)
Processing	Intermediate in all other basic organic chemical manufacturing	Processing as a reactive intermediate	Operator / Process Technician	Inhalation	Central Tendency	1,911	5,652	6,052	1.4E−05
					High-End	**20**	60	**64**	**1.7E−03**
			Maintenance Technician	Inhalation	Central Tendency	188	555	595	**1.4E−04**
					High-End	36	107	**114**	**9.4E−04**
	Intermediate in all other chemical product and preparation manufacturing		Laboratory Technician	Inhalation	Central Tendency	1.3E04	3.9E04	4.2E04	2.0E−06
					High-End	631	1,866	1,998	5.4E−05
	Recycling		Logistics Technician	Inhalation	Central Tendency	5,284	1.6E04	1.7E04	5.0E−06
					High-End	2,837	8,394	8,987	1.2E−05
			ONU	Inhalation	Central Tendency	8,327	2.5E04	2.6E04	3.15E−06
					High-End	811	2,398	2,568	4.17E−05
			Worker – Deterministic	Dermal	Central Tendency	709	442	473	NE
					High-End	236	147	**158**	NE
			Worker – Probabilistic	Dermal	Central Tendency	492	307	328	NE
					High-End	288	179	**192**	NE
	Processing – Repackaging	Processing – repackaging	Worker (All Activities)	Inhalation	Central Tendency	**4.2**	**13**	**129**	**6.4E−04**
					High-End	**1.2**	**3.4**	**7.1**	**1.5E−02**
			Worker (Unloading and Cleaning)	Inhalation	Central Tendency	**8.6**	**25**	**262**	**3.2E−04**
					High-End	**2.3**	**6.7**	**14**	**7.6E−03**
			Worker (Loading)	Inhalation	Central Tendency	**8.6**	**26**	**263**	**3.2E−04**
					High-End	**2.3**	**6.7**	**14**	**7.6E−03**
			Worker – Deterministic	Dermal	Central Tendency	709	442	4,548	NE
					High-End	236	147	308	NE

Life Cycle Stage/ Category	Subcategory	OES Scenario	Population	Exposure Route and Duration	Exposure Level	Risk Estimates for Each Exposure Scenario			
						Acute Non-Cancer (Benchmark MOE: Dermal = 30; Inhalation = 30)	Intermediate, Non-Cancer (Benchmark MOE: Dermal = 30 Inhalation = 30)	Chronic, Non-Cancer (Benchmark MOE: Dermal = 300; Inhalation=300)	Cancer (Benchmark = 10E−4)
Processing	Processing – Repackaging	Processing – repackaging	Worker – Probabilistic	Dermal	Central Tendency	491	306	1,140	NE
					High-End	287	179	494	NE
			ONU	Inhalation	Central Tendency	**4.2**	**13**	**129**	6.4E−04
					High-End	**4.2**	**13**	**26**	4.1E−03
Commercial Use/ Laboratory Chemicals	Laboratory Chemicals Reference Material	Commercial use as a laboratory chemical	Worker	Inhalation	Central Tendency	1.3E04	3.9E04	6.0E04	1.4E−06
					High-End	631	1,866	1,998	5.4E−05
			Worker – Deterministic	Dermal	Central Tendency	709	442	680	NE
					High-End	236	147	**158**	NE
			Worker – Probabilistic	Dermal	Central Tendency	499	311	373	NE
					High-End	292	182	**214**	NE
			ONU	Inhalation	Central Tendency	1.3E04	3.9E04	6.0E04	1.4E−06
					High-End	1.3E04	3.9E04	4.2E04	2.6E−06
Disposal/ Disposal	Disposal	General waste handling, treatment, and disposal	Worker	Inhalation	Central Tendency	50	149	**159**	**5.2E−04**
					High-End	**1.4**	**4.2**	**4.5**	**2.4E−02**
			Worker – Deterministic	Dermal	Central Tendency	709	442	473	NE
					High-End	236	147	**158**	NE
			Worker – Deterministic (Dilute)	Dermal	Central Tendency	3.5E04	2.2E04	2.4E04	NE
					High-End	1.2E04	7,363	7,884	NE
Disposal/ Disposal	Disposal	General waste handling, treatment, and disposal	Worker – Probabilistic	Dermal	Central Tendency	492	307	328	NE
					High-End	288	179	**192**	NE
			Worker – Probabilistic (Dilute)	Dermal	Central Tendency	2.5E04	1.5E4	1.6E04	NE
					High-End	1.4E04	8,953	9,585	NE
			ONU	Inhalation	Central Tendency	50	149	**159**	**5.2E−04**
					High-End	50	149	**159**	**6.7E−04**

Life Cycle Stage/ Category	Subcategory	OES Scenario	Population	Exposure Route and Duration	Exposure Level	Risk Estimates for Each Exposure Scenario			
						Acute Non-Cancer (Benchmark MOE: Dermal = 30; Inhalation = 30)	Intermediate, Non-Cancer (Benchmark MOE: Dermal = 30 Inhalation = 30)	Chronic, Non-Cancer (Benchmark MOE: Dermal = 300; Inhalation=300)	Cancer (Benchmark = 10E−4)
Disposal/ Disposal	Disposal	Waste handling, treatment, and disposal (POTW)	Worker	Inhalation	Central Tendency	58	173	**185**	**4.5E−04**
					High-End	**22**	65	**69**	**1.5E−03**
			Worker – Deterministic	Dermal	Central Tendency	709	442	473	NE
					High-End	236	147	**158**	NE
			Worker – Deterministic (Dilute)	Dermal	Central Tendency	3.5E04	2.2E04	2.4E04	NE
					High-End	1.2E04	7,363	7,884	NE
			Worker – Probabilistic	Dermal	Central Tendency	492	307	328	NE
					High-End	288	179	**192**	NE
Disposal/ Disposal	Disposal	Waste handling, treatment, and disposal (POTW)	Worker – Probabilistic (Dilute)	Dermal	Central Tendency	2.5E04	1.5E04	1.6E04	NE
					High-End	1.4E04	8,953	9,585	NE
			ONU	Inhalation	Central Tendency	58	173	**185**	**4.5E−04**
					High-End	58	173	**185**	**5.8E−04**

NE = Not estimated. Lifetime chronic retained dose (LCRD) for cancer risk was not estimated as dermal cancer numbers for 1,1-dichloroethane were not derived.
Risk estimates that exceed the benchmark (*i.e.*, non-cancer risks < benchmark and cancer risks > cancer risk benchmark) are bolded and shaded.

5.3.3.2 Risk Estimates for the General Population

The following sections summarize the risk estimates and conclusions for inhalation, dermal and oral exposures for all general population exposure scenarios. The general population exposure assessment is described in Section 5.1.2. For those facilities where risk estimates based on ambient air concentrations exceed the benchmark MOE (*i.e.*, MOEs less than the benchmark MOE or cancer risks greater than the cancer risk benchmark of 1×10^{-6}) (see Table 5-56, Table 5-57, Table 5-58, Table 5-59, Table 5-60), EPA conducted a land use analysis of residences in proximity to 1,1-dichloroethane facility releases in Section 5.3.3.2.1. Specifically, EPA evaluated land use patterns to determine residential or industrial/commercial businesses or other public spaces relative to facilities emitting 1,1-dichloroethane and whether general population community risks may be reasonably anticipated. As shown in Table 5-61, EPA's land use analysis did not identify any residential, industrial/commercial businesses, or other public spaces within those 1,000 m where risk would exceed the cancer risk benchmark of 1×10^{-6}. Based on this characterization of land use patterns and expected risk estimates, EPA does not expect exposure and therefore does not expect a risk to the general population resulting from 1,1-dichloroethane releases via the ambient air pathway. As stated in Appendix D.3, additional land use analysis was not warranted for aggregate analysis. EPA acknowledges that land use patterns may change over time, but reasonably available information does not indicate that changes to land use patterns in these particular areas are reasonably foreseen, as opposed to speculative and unsubstantiated.

Table 5-56. Inhalation Lifetime Cancer Risks[a] Within 1 km of TRI Air Releases Based on 95th Percentile Modeled Ambient Air Exposure Concentrations

OES	Corresponding COUs		# Facilities		Maximum 95th Percentile Cancer Risks Estimated Within 10–1,000 m of Facilities [b c]							Overall Confidence
	Life Cycle Stage/Category	Subcategory	Total	Risk >1E−06	10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	
Manufacturing	Manufacturing/ Domestic manufacturing	Domestic manufacturing	9	7	1.6E−03	6.4E−04	4.9E−04	2.6E−04	1.2E−04	1.7E−05	2.9E−06	High
Processing as a reactive intermediate	Processing/ As a reactant, recycling	Intermediate in all other basic organic chemical manufacturing; Intermediate in all other chemical product and preparation manufacturing; Recycling	6	2	1.1E−04	4.5E−05	3.1E−05	1.8E−05	8.4E−06	1.2E−06	1.9E−07	High
General waste handling, treatment, and disposal	Disposal/ Disposal	Disposal	8	1	1.4E−04	6.6E−05	4.3E−05	2.8E−05	1.4E−05	1.0E−06	3.4E−07	High

[a] Lifetime cancer risks based on 78 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.
[c] Cancer risk estimates that exceed the benchmark (*i.e.,* cancer risks greater than the cancer risk benchmark) are listed However, based on the land use analysis (Section 5.3.3.2.1), EPA does not expect exposure in residential areas and therefore does not expect a risk to the general population.

Table 5-57. Inhalation Lifetime Cancer Risks[a] Within 1 km of NEI Air Releases Based on 95th Percentile Modeled Ambient Air Exposure Concentrations

OES	Corresponding COUs		# Facilities		Maximum 95th Percentile Cancer Risks Estimated Within 1,000 m of Releases [b c]							Overall Confidence
	Life Cycle Stage/Category	Subcategory	Total	Risk >1E−06	10 m	30 m	30 to 60 m	60 m	100 m	100 to 1,000 m	1,000 m	
Commercial use as a laboratory chemical	Commercial Use/ Other use	Laboratory chemicals	2	0	2.6E−07	8.2E−08	5.1E−08	3.0E−08	1.3E−08	1.4E−09	2.7E−10	Moderate
Manufacturing	Manufacturing/ Domestic manufacturing	Domestic manufacturing	9	4	1.5E−04	4.3E−05	4.3E−05	4.3E−05	4.1E−05	7.2E−06	8.6E−07	High
Processing as a reactive intermediate	Processing/As a reactant; Recycling	Intermediate in all other basic organic chemical manufacturing; Intermediate in all other chemical product and preparation manufacturing; Recycling	50	14	2.3E−04	8.7E−05	5.9E−05	3.5E−05	1.6E−05	1.9E−06	3.4E−07	High
General waste handling, treatment, and disposal	Disposal/ Disposal	Disposal	102	48	8.9E−05	5.9E−05	4.6E−05	2.9E−05	1.5E−05	1.5E−06	3.7E−07	High
Facilities not mapped to an OES			59	12	6.5E−05	2.6E−05	2.0E−05	1.1E−05	5.2E−06	8.4E−07	1.2E−07	N/A

[a] Lifetime cancer risks based on 78 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.
[c] Cancer risk estimates that exceed the benchmark (*i.e.*, cancer risks greater than the cancer risk benchmark) are listed. However, based on the land use analysis (Section 5.3.3.2.1), EPA does not expect exposure in residential areas and therefore does not expect a risk to the general population.

Table 5-58. Inhalation Lifetime Cancer Risks[a] Within 1 km of TRI Air Releases Based on 50th Percentile Modeled Ambient Air Exposure Concentrations

OES	Corresponding COUs		# Facilities		Maximum 50th Percentile Cancer Risks Estimated Within 10–1,000 m of Facilities[b c]							Overall Confidence
	Life Cycle Stage/Category	Subcategory	Total	Risk >1E−06	10 m	30 m	30–60 m	60 m	100 m	100–1,000 m	1,000 m	
Manufacturing	Manufacturing/ Domestic manufacturing	Domestic manufacturing	9	7	1.2E−03	4.7E−04	2.5E−04	1.9E−04	8.6E−05	3.2E−06	1.7E−06	High
Processing as a reactive intermediate	Processing/ as a reactant, recycling	Intermediate in all other basic organic chemical manufacturing; Intermediate in all other chemical product and preparation manufacturing; Recycling	6	2	6.0E−05	2.4E−05	1.5E−05	9.8E−06	4.6E−06	2.1E−07	1.0E−07	High
General waste handling, treatment, and disposal	Disposal/ Disposal	Disposal	8	1	3.7E−05	1.2E−05	7.5E−06	4.3E−06	2.0E−06	1.1E−07	4.6E−08	High

[a] Lifetime cancer risks based on 78 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.
[c] Cancer risk estimates that exceed the benchmark (*i.e.*, cancer risks greater than the cancer risk benchmark) are listed. However, based on the land use analysis (Section 5.3.3.2.1), EPA does not expect exposure in residential areas and therefore does not expect a risk to the general population.

Table 5-59. Inhalation Lifetime Cancer Risks[a] Within 1 km of NEI Air Releases Based on 50th Percentile Modeled Ambient Air Exposure Concentrations

OES	Corresponding COUs		# Releases		Maximum 50th Percentile Cancer Risks Estimated Within 1,000 m of Releases [b c]							Overall Confidence
	Life Cycle Stage/Category	Subcategory	Total	Risk >1E−06	10 m	30 m	30 to 60 m	60 m	100 m	100 to 1,000 m	1,000 m	
Commercial use as a laboratory chemical	Commercial Use/ Other use	Laboratory chemicals	2	0	1.3E−07	3.6E−08	1.9E−08	1.3E−08	5.5E−09	2.0E−10	1.0E−10	High
Manufacturing	Manufacturing/ Domestic manufacturing	Domestic manufacturing	9	3	9.2E−05	4.2E−05	4.1E−05	4.0E−05	3.4E−05	8.9E−07	3.9E−07	High
Processing as a reactive intermediate	Processing/ As a reactant; Recycling	Intermediate in all other basic organic chemical manufacturing; Intermediate in all other chemical product and preparation manufacturing; Recycling	50	14	1.8E−04	5.7E−05	3.2E−05	2.2E−05	9.7E−06	3.7E−07	2.0E−07	High
General waste handling, treatment, and disposal	Disposal/ Disposal	Disposal	102	39	4.8E−05	2.4E−05	1.3E−05	8.3E−06	3.6E−06	1.9E−07	7.4E−08	High
Facilities not mapped to an OES			59	9	5.1E−05	2.1E−05	1.2E−05	8.4E−06	4.0E−06	1.6E−07	8.5E−08	N/A

[a] Lifetime cancer risks based on 78 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.
[c] Cancer risk estimates that exceed the benchmark (*i.e.,* cancer risks greater than the cancer risk benchmark) are listed. However, based on the land use analysis (Section 5.3.3.2.1), EPA does not expect exposure in residential areas and therefore does not expect a risk to the general population.

Table 5-60. Inhalation Lifetime Cancer Risks[a] Within 1 km of Air Releases Based on 95th Percentile Modeled Exposure Concentrations for the Commercial Use as a Laboratory Chemical, and Processing – Repackaging for Laboratory Chemicals OESs

OES	Meteorology	Source	Land	Maximum 95th Percentile Cancer Risks Estimated Within 1,000 m of Releases[b c]						
				10 m	30 m	30 to 60 m	60 m	100 m	100 to 1,000 m	1,000 m
Processing – repackaging	High	Stack and Fugitive	Urban	6.6E−06	1.9E−06	1.4E−06	8.7E−07	7.1E−07	2.4E−07	1.0E−07
	High	Stack and Fugitive	Rural	6.6E−06	1.9E−06	1.5E−06	1.1E−06	1.0E−06	2.7E−07	8.9E−08
Commercial use as a laboratory chemical	High	Stack and Fugitive	Urban	1.1E−05	3.1E−06	2.5E−06	1.8E−06	1.7E−06	6.4E−07	2.8E−07
	High	Stack and Fugitive	Rural	1.1E−05	3.1E−06	2.8E−06	2.2E−06	2.5E−06	7.2E−07	2.4E−07

[a] Lifetime cancer risks based on 78 years of continuous inhalation exposure averaged over a 78-year lifetime. Estimated cancer risks calculated using maximum concentration by distance from the release point.
[b] Cancer risk estimates that exceed the benchmark (*i.e.,* cancer risks greater than the cancer risk benchmark) are listed. However, based on the land use analysis (Section 5.3.3.2.1), EPA does not expect exposure in residential areas and therefore does not expect a risk to the general population.
[c] Risk estimates based on 95th percentile modeled ambient air exposure concentrations.

5.3.3.2.1 Inhalation Exposure

EPA estimated risks of general population exposures to 1,1-dichloroethane released to air, with a focus on exposures in general populations residing near 1,1-dichloroethane emitting facilities. Risks were evaluated for air releases from industrial and commercial COUs based on exposure estimates from reported ambient air releases in Section 5.1.2.2 and human health hazard values (selected PODs) for chronic inhalation exposures in Section 5.2.6.3. Cancer and non-cancer risk estimates for general population exposures to ambient air within 10,000 m of 1,1-dichloroethane TRI-/NEI-reported releases were calculated for the 10th, 50th, and 95th percentiles of modeled air concentrations estimated in Section 3.3.1.2. Risk estimates were highest within 1,000 m of the releasing facilities and lower at distances beyond 1,000 m. Risks above benchmark (*i.e.,* MOEs less than the benchmark MOE or cancer risks greater than the cancer risk benchmark of 1×10^{-6}) were not indicated for any OESs/COUs beyond 1,000 m from a facility.

EPA calculated and found inhalation cancer risks greater than 1×10^{-6} for the 95th percentile (high-end) exposure concentrations within 1,000 m of the facilities based on TRI and NEI modeled exposure data, respectively (Table 5-56 and Table 5-57). EPA identified inhalation cancer risks greater than 1×10^{-6} for the 50th percentile air concentrations for manufacturing, processing, and disposal OESs/COUs at distances as far as 1,000 m from the releasing facility. No inhalation cancer risks were found for commercial use as a laboratory chemical OES/COU. Acute non-cancer risk estimates indicate risk relative to benchmark MOE based on the 95th percentile air concentrations for manufacturing OES/COU at 10 m from the releasing facility for one TRI facility within the OES/COU. No inhalation acute and chronic non-cancer risks were found based on the 50th percentile air concentrations—except for one TRI facility within the manufacturing OES/COU that shows chronic non-cancer risk at 10 m from the releasing facility. Chronic non-cancer risk estimates indicate risk relative to benchmark MOE based on the 95th percentile air concentrations for manufacturing OES/COU at distances as far as 30 m from the releasing facility (for one TRI facility within the OES/COU). Complete cancer and non-cancer risk results are provided in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025o), *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD Generic Releases Exposure and Risk Analysis* (U.S. EPA, 2025m), *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis* (U.S. EPA, 2025p)and in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD NEI Exposure and Risk Analysis* (U.S. EPA, 2025n).

Within the ambient air pathway, EPA also evaluated cancer and non-cancer risks from aggregate exposures from multiple neighboring facilities using a conservative screening methodology. The methodology for this analysis is consistent with what was previously described in the *Draft Supplement to the Risk Evaluation for 14-Dioxane* (U.S. EPA, 2023a). EPA identified four groups of two to six facilities reporting 1,1-dichloroethane 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 D.3.

EPA initially used U.S. Census Bureau census block data to assess and characterize populations in proximity of releasing facilities (see Section 3.3.1.2.4). However, as a refinement, EPA conducted a review of land use patterns around TRI facilities where cancer risk would exceed 1×10^{-6} (10 of the 23

GIS-mapped TRI facilities). 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* (accessed June 16, 2025).[20] This review was limited to those facilities with real Global Information System (GIS) locations. The land use analysis does not include generic facilities where alternative release estimates were modeled to estimate exposures since there is no real location around which to conduct the land use analysis. The purpose of this review was to determine if EPA can reasonably expect exposures to the general population within the modeled distances where cancer risk would exceed 1×10^{-6}. 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 cancer risk would exceed 1×10^{-6} for each facility based on the 95th percentile modeled air concentrations.

For locations where residential or industrial/commercial businesses or other public spaces are present within those radial distances indicating risk, EPA reasonably expects exposures and therefore associated potential risks to the general population. 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 exposures to the general population and therefore does not expect associated risks. The Agency did not consider possible future residential use of areas. EPA acknowledges that land use patterns may change over time, but reasonably available information does not indicate that changes to land use patterns in these areas are reasonably foreseen, as opposed to speculative and unsubstantiated. Also, as stated in Appendix D.3, additional land use analysis was not warranted for aggregate analysis.

As show in Table 5-61, EPA's land use analysis did not identify any residential, industrial/ commercial businesses, or other public spaces within those 1,000 m where risk estimates would exceed 1×10^{-6}. Based on this characterization of land use patterns and identified risk estimates, EPA does not expect exposures to the general population for any of the TRI facilities and aggregate groups (Appendix D.3) where cancer risk would exceed 1×10^{-6} for the 95th percentile modeled air concentrations. Therefore, EPA does not expect a risk to the general population resulting from 1,1-dichloroethane releases via the ambient air pathway. Individual facility summaries are available in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025o).

Table 5-61. Summary of the General Population Exposures Expected near Facilities Where TRI Modeled Air Concentrations Indicated Risk for 1,1-Dichloroethane

OES	COU	Total Number of Facilities Evaluated	Number of Facilities with Risk Indicated[a]	Number of Facilities with Risk Indicated and General Population Exposures Expected[b]
Manufacturing	Manufacturing	9	7	0
Processing as a reactive intermediate	Processing as a reactant	6	2	0
General waste handling, treatment, and disposal	Waste handling, disposal, and treatment	8	1	0

[a] TRI facilities where cancer risk would exceed 1×10^{-6}.
[b] A land use analysis was conducted to identify any residential communities within 1,000 m.

[20] https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/tsca-screening-level-approach-assessing-ambient-air-and (accessed June 16, 2025).

5.3.3.2.2 Dermal Exposures

No acute or chronic dermal risks were identified from the various exposure scenarios outlined in Section 5.1.2.2.3. Detailed calculations and results are presented in the supplemental file, *Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming High-End Exposure Estimates* (U.S. EPA, 2025r). Dermal cancer risks were not quantitatively assessed.

5.3.3.2.3 Oral Exposures

EPA estimated the possibility of risks associated with oral exposures from drinking water consumption. Facilities were identified with releases of 1,1-dichloroethane resulting in either the median (central tendency) or maximum exposures (see Section 5.1.2.4.1). None of the drinking water general population oral exposures were estimated to result in either acute or chronic risks (Table 5-62). Oral exposures from fish ingestion did not result in acute or chronic risks (Table 5-62). Oral cancer risks from ingestion via drinking water or fish ingestion could not be assessed as an acceptable cancer study is not available for either 1,1-dichloroethane or 1,2-dichloroethane.

EPA assumed that subsistence fishing is a likely scenario in receiving waters associated with the above listed COUs/OES. That is, it is common to fish in the bayous of Louisiana where the manufacturing facility releases occur and likely in the Navajo Nation in Arizona where the POTW releases occur. The high-end surface water concentrations are estimated in Arizona because the receiving water body, the Chinle Wash, may be intermittent, so that the effluent would in essence be the dominant source of surface water. Additional areas of exposure resulting in fish ingestion risk include a small tributary to San Jacinto Bay in Texas (associated with Processing as a reactant COU), Spring Creek in Ohio (Unknown COU) and South Fork of Arroyo Conejo Creek in California (Waste handling/remediation COU).

As presented in Sections 5.1.2.4.3, 5.1.2.4.4 adn Appendix N, the estimated oral exposures of 1,1-dichloroethane from incidental ingestion of surface water during swimming, ingestion of soil from biosolids land application or ingestion of soil containing 1,1-dichloroethane from air deposition are low compared to oral hazard values. Non-cancer risks below the benchmark MOE from these acute/chronic oral exposures are not expected.

Table 5-62. Summary of General Population Risk Estimates

Life Cycle Stage/ Category	Subcategory	OES	Exposure Route and Duration	Exposure Level	Risk Estimates for Each Exposure Scenario[c]		
					Acute Non-Cancer (Benchmark MOE: Oral = 30; Inhalation = 30)	Chronic Non-Cancer (Benchmark MOE: Oral = 300; Inhalation = 300)	Cancer (Benchmark = 1.0E10−6) [a]
Manufacture/ Domestic Manufacturing	Domestic manufacture	Manufacturing of 1,1-dichloroethane as an isolated intermediate	Ambient Air Inhalation	Central Tendency	1.4E02	1.2E02 (estimate at 10 m)	5.3E−04 (estimate at 10–1,000 m)
				High-End	1.7E01 (estimate at 10 m)	9.1E1 (estimate at 30 m)	7.0E−04 (estimate at 10–1,000 m)
			Drinking Water Ingestion[b]	Central Tendency	N/A	N/A	N/A
				High-End	N/A	N/A	N/A
			Fish Ingestion	Central Tendency	5.3E05	8.8E07	Not quantitatively assessed
				High-End	1.9E04	3.1E06	Not quantitatively assessed
Processing/As a Reactant	Intermediate in all other basic organic chemical manufacturing / Intermediate in all other chemical product and preparation manufacturing / Recycling	Processing as a reactive intermediate	Ambient Air Inhalation	Central Tendency	2.2E03	2.5E03	2.5E−05 (estimate at 10–100 m)
				High-End	2.8E02	1.4E03	4.6E−05 (estimate at 10–100 m)
			Drinking Water Ingestion	Central Tendency	5.7E08	7.8E10	Not quantitatively assessed
				High-End	6.5E06	7.7E08	Not quantitatively assessed
			Fish Ingestion	Central Tendency	3.3E06	5.5E08	Not quantitatively assessed
				High-End	1.2E05	2.0E07	Not quantitatively assessed
Processing/ Processing – Repackaging	Processing – Repackaging	Processing – repackaging	Ambient Air Inhalation	Central Tendency	N/A	2.42E08	1.4E−06 (estimate at 10 m)
				High-End	3.43E06	1.60E08	2.8E−06 (estimate at 10 m)
			Drinking Water Ingestion	Central Tendency	3.7E09	3.7E11	Not quantitatively assessed
				High-End	2.6E07	2.3E09	Not quantitatively assessed
			Fish Ingestion	Central Tendency	6.4E07	1.1E10	Not quantitatively assessed
				High-End	2.3E06	3.8E08	Not quantitatively assessed

Life Cycle Stage/ Category	Subcategory	OES	Exposure Route and Duration	Exposure Level	Risk Estimates for Each Exposure Scenario[c]		
					Acute Non-Cancer (Benchmark MOE: Oral = 30; Inhalation = 30)	Chronic Non-Cancer (Benchmark MOE: Oral = 300; Inhalation = 300)	Cancer (Benchmark = 1.0E10−6) [a]
Commercial Use/Other use	Laboratory Chemicals	Commercial use as a laboratory Chemical	Ambient Air Inhalation	Central Tendency	2.79E14	8.68E07	2.6E−06 (estimate at 10–30 m)
				High-End	1.48E06	5.87E07	4.6E−06 (estimate at 10–100 m)
			Drinking Water Ingestion[b]	Central Tendency	N/A	N/A	N/A
				High-End	N/A	N/A	N/A
			Fish Ingestion	Central Tendency	7.1E07	1.2E10	Not quantitatively assessed
				High-End	2.5E06	4.2E08	Not quantitatively assessed
Disposal/ Disposal	Disposal	General waste handling, treatment, and disposal	Ambient Air Inhalation	Central Tendency	5.8E03	4.1E03	1.6E−05 (Estimate at 1–60 m)
				High-End	3.1E02	3.1E03	5.8E−05 (Estimate at 1–100 m)
			Drinking Water Ingestion	Central Tendency	1.1E08	1.0E10	Not quantitatively assessed
				High-End	2.0E06	8.4E07	Not quantitatively assessed
			Fish Ingestion	Central Tendency	2.5E06	4.2E08	Not quantitatively assessed
				High-End	8.8E04	1.5E07	Not quantitatively assessed
Disposal/ Disposal	Disposal	Waste handling, treatment, and disposal (POTW)	Drinking Water Ingestion	Central Tendency	2.5E09	1.6E11	Not quantitatively assessed
				High- End	4.1E06	1.7E08	Not quantitatively assessed
			Fish Ingestion	Central Tendency	5.5E06	9.2E08	Not quantitatively assessed
				High- End	2.0E05	3.3E07	Not quantitatively assessed
Disposal/ Disposal	Disposal	Waste handling, treatment, and disposal (remediation)	Drinking Water Ingestion	Central Tendency	1.9E09	1.7E11	Not quantitatively assessed
				High-End	4.0E07	3.7E09	Not quantitatively assessed
			Fish Ingestion	Central Tendency	4.0E05	6.7E07	Not quantitatively assessed
				High-End	1.4E04	2.4E06	Not quantitatively assessed

Life Cycle Stage/ Category	Subcategory	OES	Exposure Route and Duration	Exposure Level	Risk Estimates for Each Exposure Scenario[c]		
					Acute Non-Cancer (Benchmark MOE: Oral = 30; Inhalation = 30)	Chronic Non-Cancer (Benchmark MOE: Oral = 300; Inhalation = 300)	Cancer (Benchmark = 1.0E10−6) [a]
Facilities not mapped to an OES/Facilities not mapped to an OES	Facilities not mapped to an OES	Facilities not mapped to an OES	Ambient Air Inhalation	Central Tendency	7.5E09	7.7E07	2.1E−05 (Estimate at 10–100 m)
				High-End	5.6E06	5.2E07	2.8E−05 (Estimate at 10–100 m)
			Drinking Water Ingestion	Central Tendency	9.6E08	1.0E11	Not quantitatively assessed
				High-End	1.4E07	6.0E08	Not quantitatively assessed
			Fish Ingestion	Central Tendency	2.2E06	3.7E08	Not quantitatively assessed
				High-End	7.8E04	1.3E07	Not quantitatively assessed

[a] Ambient inhalation risk estimate shown is the maximum risk value estimated from TRI and NEI air releases at any distance between 10 and 10,000 m. Distance range shown corresponds to distances where risk is exceeding benchmark. Based on land use analysis, there are no residential areas at the distances in parentheses, therefore EPA does not predict general population risks from 1,1-dichloroethane ambient air exposures.

[b] Drinking water risks were not assessed for this COU. Drinking water intakes were not identified downstream of the largest releasing facility within the COU. For drinking water: N/A designates – not assessed.

5.3.4 Risk Characterization of Aggregate and Sentinel Exposures

As stated in Section 5.1.4, EPA considered sentinel exposures by considering risks to populations who may have upper bound exposures; for example, workers who perform activities with higher exposure potential, or certain physical factors like body weight or skin surface area exposed. EPA characterized high-end exposures in evaluating exposure using both monitoring data and modeling approaches. Where statistical data are reasonably available, EPA typically uses the 95th percentile value of the reasonably available dataset to characterize high-end exposure for a given condition of use. In cases where sentinel exposures result in MOEs greater than the benchmark or cancer risk lower than the benchmark (*i.e.*, risks were not identified), EPA did no further analysis because sentinel exposures represent the worst-case scenario.

EPA aggregated ambient air concentrations to estimate inhalation risks from co-located facilities (see Section 5.1.3). EPA aggregated oral and dermal risks for the swimming scenario (U.S. EPA, 2025r) since endpoints for the selected PODs are the same. However, EPA did not aggregate risks across exposure routes for all exposure durations as the health outcomes (endpoints for the selected PODs) were different for oral/dermal and inhalation studies. EPA did not consider aggregate inhalation exposures to people who both work at and live near facilities releasing 1,1-dichloroethane since EPA does not have data showing that this is a likely exposure scenario.

5.3.5 Overall Confidence and Remaining Uncertainties in Human Health Risk Characterization

EPA took fate, exposure (occupational, and general population), and human health hazard considerations into account when characterizing the human health risks of 1,1-dichloroethane. Human health risk characterization evaluated confidence from occupational and general population exposures and human health hazards. Hazard confidence and uncertainty is represented by health outcome and exposure duration as reported in Section 5.2.7, which presents the confidence, uncertainties, and limitations of the human health hazards for 1,1-dichloroethane using 1,2-dichloroethane toxicity data as an analog for read-across. Confidence in the exposure assessment has been synthesized in the respective weight of scientific evidence conclusion sections for occupational exposures (see Section 5.3.5.1) and general population exposures (see Section 5.3.5.2). Appendix Q provides a summary of confidence for exposures and hazards for non-cancer endpoints for the COUs that resulted in any non-cancer risks; Appendix Q provides a confidence summary for cancer for the COUs that resulted in cancer risks.

5.3.5.1 Occupational Risk Estimates

Section 5.3.5.1 illustrates the confidence in the assessment of the occupational exposure scenarios.

Manufacture as an Isolated Intermediate

Manufacturing processes only occur in closed systems. Empirical inhalation monitoring data for 1,1-dichloroethane were collected via a TSCA section 4 test order from four sites. One of the four sites manufactures 1,1-dichloroethane as an isolated intermediate, while the other three sites manufacture 1,1-dichloroethane as a byproduct. Data from the one site that manufactures 1,1-dichloroethane as an isolated intermediate was used in this risk evaluation, as this site best represents the Manufacture as an Isolated Intermediate OES. The other three sites that manufacture 1,1-dichloroethane as a byproduct, are not considered to be representative of the Manufacture as an Isolated Intermediate OES. These three sites will be used in the 1,2-dichloroethane risk evaluation. Exposure groups were identified and monitored, including an exposure group for occupational non-users (ONUs). Within an exposure group, workers perform similar tasks. More details on the exposure groups are provided in Table 5-2. A total of 67 samples were collected over 5 exposure groups and 50th and 95th percentile exposures were

calculated to represent central tendency and high-end exposures. The highest exposures for operators/process technicians were associated with repairs of line leaks and these data were separated out into a separate exposure group and exposure frequency expected to be less than daily. EPA's confidence in this assessment based on these data is moderate to robust as these data represent actual exposures at a representative facility. EPA's general practice in occupational exposure estimates for risk evaluations is to use the 95th percentile for the high-end estimate.

Processing as a Reactive Intermediate

EPA did not identify monitoring data for the processing as a reactive intermediate OES; however, EPA assumed the exposures to be similar to manufacturing due to similar worker activities and the use of closed systems during processing. Therefore, EPA incorporated the manufacturing data into the processing as a reactive intermediate exposure estimates. EPA has used this assessment approach in previous risk evaluations, including the *Risk Evaluation for Perchloroethylene (PCE)* (U.S. EPA, 2020g). EPA assumed that the exposure groups for this OES would be similar to Manufacture with workers with exposure groups performing similar tasks. EPA includes both the central tendency as an estimate of exposure of workers in the middle of the distribution (EPA used the 50th percentile from the data set) and the 95th percentile as a high-end estimate of individual exposure. EPA's confidence in this assessment based on these data is moderate as these data represent actual exposures at a representative facility but were not for this scenario (based on Manufacturing OES).

Processing – Repackaging

EPA did not find any inhalation monitoring data for 1,1-dichloroethane from the systematic review to estimate inhalation exposures for the repackaging OES. EPA does have a generic scenario for repackaging and the scenario recommends the use of mass balance models to estimate inhalation exposure. The generic scenario recommended approach was used to estimate inhalation exposures to 1,1-dichloroethane for this OES. A strength of the assessment was the use of probabilistic modeling using Monte Carlo for the mass balance models. This allowed modeling of the ranges in values of model parameters in estimating the 50th percentile and 95th percentile inhalation exposures.

A limitation in the modeling approach is that EPA did not find any specific information on 1,1-dichloroethane going through a repackaging step. Due to this lack of information on production volume and how 1,1-dichloroethane is handled and repackaged, EPA used default values for the model, thus potentially over- estimating exposures, especially for activities that handle a small portion of the manufactured volume. The OES for repackaging was based on an EPA assumption that a repackaging step would need to take place prior to the use of 1,1-dichloroethane as a laboratory chemical. There were two manufacturing sites identified that manufacture 1,1-dichloroethane, which EPA assumed also conduct repackaging activities. Repackaging, however, may also occur at the 12 sites that process 1,1-dichloroethane. For modeling purposes, EPA used two sites that equates to 25,000 lbs/site/yr. EPA then used a Monte Carlo modeling approach that included varying the parameters such as container size to generate a distribution of estimates for exposure days and exposure concentrations to 1,1-dichloroethane. The parameters of lbs/yr for repackaging, number of sites and the daily amount handled are foundational parameters in the modeling approach that impact the daily exposure levels. The lack of 1,1-dichloroethane supporting information for lbs/yr for these parameters is a major uncertainty in the assessment of this OES and EPA therefore assigns a slight confidence rating for this OES. The SACC panel agreed that this OES is uncertain, and estimates should be considered as bounding estimates. In addition, estimates potentially overestimate exposure as repackaging activities in the manufacturing site would handle lower volumes with less frequency than the manufacturing operations.

Commercial Use as a Laboratory Chemical

The TSCA section 4 test order for 1,1-dichloroethane included the monitoring of laboratory technicians working in a lab at the manufacturing facility. The worker activities and procedures in this laboratory setting can be applied to the OES of laboratory chemical as an industrial/commercial use. However, there is uncertainty in whether the setting, activities, controls and PPE are representative of use as a laboratory chemical more generally for commercial use. The use of chemical-specific empirical monitoring data from a laboratory at the manufacturing facility to assess inhalation exposure for an analogous laboratory chemical OES is a more robust approach than the use of modeling for estimating exposures. EPA's confidence in this assessment is moderate (see Table 5-18).

Waste Handling Treatment and Disposal

EPA did not find any inhalation monitoring data for 1,1-dichloroethane from the systematic review of the scientific literature to estimate inhalation exposures for the waste handling, treatment, and disposal OES. EPA does not currently have a generic scenario for this OES to provide a characterization of exposure. To estimate inhalation exposure, EPA used surrogate monitoring data for chemicals with similar physical-chemical properties to estimate inhalation exposures to 1,1-dichloroethane for these OESs. For POTWs, there is uncertainty regarding whether the surrogate chemical volume throughput and concentrations, worker exposures, and waste streams are sufficiently similar to those for 1,1-dichloroethane. EPA's method for using surrogate data includes the application of a vapor pressure correction method based on assumption that Raoult's law is valid, to account for vapor pressure differences between the chemicals that were monitored and 1,1-dichloroethane. The metadata for the surrogate monitoring data used were limited. For example, the data did not include detail on the exposure groups that were monitored. This limits EPA's ability to understand exposure potential according to different exposure groups and tasks at facilities that handle, treat and dispose of was. Additionally, most facilities generating or using 1,1-dichloroethane are using it in processes in which it is either consumed as part of a reaction or generated as an impurity, resulting in low concentrations of 1,1-dichloroethane expected to be present in the waste streams. There is also, as stated by the SACC, uncertainty in whether the general disposal OES may be covered in the operator/processor and maintenance technician OESs. Because 1,1-dichloroethane-specific information (*e.g.,* inhalation monitoring data) on this OES is not available and a generic modeling scenario was also not available, EPA's confidence in this assessment is slight (see Table 5-18).

Dermal Exposure

The dermal exposure assessment used the DEVL model to estimate dermal exposures. A key strength of the approach used was the use of data on fractional absorption that was developed from a TSCA section 4 test order for 1,1-dichloroethane. Since 1,1-dichloroethane is a highly volatile chemical, any estimate of dermal exposure must take volatility into account. The fraction absorbed value from the test order showed a small percentage absorbed to the skin (0.03%) with the majority (99.7%) evaporating. This enabled EPA to generate a more accurate estimate of dose. The high-end dose estimate from the model that includes the fraction absorbed is 6.7 mg/day. Using the same model without accounting for evaporation of this highly volatile chemical would result in an estimated high-end dose of 2,271 mg/day.

The dermal loading values (mg/cm^2) used for the DEVL model are based on experimental studies. The experimental values are very similar to the dermal loading values for pesticides from data in PHED (see Table 5-14). However. EPA does not know if the experimental values or data in PHED are the same for exposure scenarios that are encountered in the industrial and commercial settings for the OES identified for 1,1-dichloroethane. The modeling approach does include a weight fraction parameter that accounts for differences in the weight fraction of 1,1-dichloroethane between OES. However, it does not account for other differences that may exist among the OES that impact dermal exposure such as differences in

dermal loading, skin surface area exposed, and frequency of contact. Central-tendency and high-end exposures reflect differences in the magnitude of exposure each day. The OES of Manufacture and Processing as a reactive intermediate only occurs in a closed system. Although manufacturing and processing only occurs in closed systems, monitoring data supports that there is still the potential for dermal exposure during activities such as connecting/disconnecting transfer lines and sampling. These types of activities can be done on a daily basis although the fraction of the workers' shift associated with these activities is expected to be low. While high-end dermal exposures are possible during these types of activities (*i.e.*, acute and/or intermediate exposure), high-end dermal exposures are likely to be infrequent. EPA believes the central tendency from the closed system monitoring data is a more representative and appropriate exposure estimate for a frequent, repeated dermal exposure (*i.e.*, chronic) and is health protective for risk estimation for closed system processes, as high-end exposures from daily connecting/disconnecting of transfer lines and sampling is not realistic.

At this time, based on limitations in reasonably available data, EPA could not quantitatively assess the cancer from dermal exposure for all COUs to workers and ONUs. Without a quantitative dermal cancer assessment, EPA qualitatively characterized the dermal cancer risk, as described in Section 5.2.5. Further discussion can be found in Sections 5.2.6 and Appendix N. Due to this lack of quantitative data, dermal cancer risk is uncertain and not quantitatively assessed.

5.3.5.2 General Population Risk Estimates

Section 5.3.5.2 illustrates the confidence in the assessment of the general population exposure scenarios.

Air Pathway

For the air pathway EPA conducted tiered analyses for estimating ambient air exposures and associated risks. 1,1-Dichloroethane ambient air concentrations were modeled using facility release data reported in TRI and NEI corresponding to TSCA COU or alternative release estimates where facility specific data were not available. EPA performed a full analysis using AERMOD and used the reported air release estimates as direct inputs for the model to estimate exposure concentrations at various distances from a releasing facility.

Overall confidence in risk estimates is high for OESs/COUs that rely primarily on release data reported to TRI and NEI (based on high levels of confidence in underlying release information used to estimate exposures). Overall confidence in risk estimates is medium for OESs/COUs for which release estimates are based on modeled information.

As described in Section 3.3.5.1, EPA has high confidence in the air concentrations estimated from TRI and NEI release data using AERMOD. As described in Section 5.1.2.5.1 the overall confidence in exposure estimates varies due to variable levels of confidence in underlying release information used to support the analysis (high levels of confidence for release data reported to TRI and NEI and medium levels of confidence for modeled release estimates).

Inhalation cancer estimates relative to the benchmark were estimated for the high-end, 95th percentile air concentration for 10 of the 23 TRI facilities representing three of the five COUs. However, based on characterization of land use patterns, residential fenceline communities were not identified and fenceline inhalation exposures are anticipated to be lower than levels that would be a concern for cancer. Therefore, there is no inhalation cancer risk above the benchmark is expected for these communities.

Distance Where Risk Identified
IIOAC and AERMOD provided both 50th percentile and 95th percentile ambient air exposure concentrations at discrete distances from air releases representing high-end and central tendency potential exposures. EPA likewise calculated central tendency and high-end risk at corresponding modeled discrete distances. Therefore, there is uncertainty of risk between the two distances modeled. For example, if risk was found risk at 1,000 m and not at 2,500 m, EPA is uncertain if there is risk at 1,001 to 2,499 m. To not underestimate risk beyond the risk showing distance (*e.g.,* at 1,001 meters), or overestimate risk closer to the distance where risk was not found (*e.g.,* at 2,499 meters), remodeling may be required to determine exposure concentrations, and thus calculating risk between the two discrete distances previously modeled. Additionally, reported TRI facility's location data (latitude/longitude) may not represent the actual location of the releasing source (*e.g.,* a processes stack).

However, for 1,1-dichloroethane, residential fenceline community exposures are not at levels of 1,1-dichloroethane concentrations that present risk. That is, the fenceline community locations are beyond the location of non-cancer or cancer risk relative to the benchmark. EPA has high confidence in the estimate of general population exposures as a basis for confidence in the absence of risk to the general population.

Uncertainties associated with the general population exposures assessment included the uncertainty in the precision of site-specific information, particularly associated with TRI release location data, and the complexity of the assessed exposure scenarios. Even with these uncertainties, EPA's land use analysis has provided confidence that people were not residing in locations where high-end 1,1-dichloroethane ambient air concentrations from facility releases were estimated.

Other General Population Exposure Pathways
EPA quantitatively assessed general population central tendency and high-end exposures to 1,1-dichloroethane based on the reported facility-specific TSCA releases to surface waters and soil. High-end estimates were based on the 95th percentile distribution of drinking water intake, fish ingestion, incidental ingestion via swimming and pica of soil as based on EPA's *Exposure Factors Handbook* and represent exposures to people that consume more drinking water, fish or soil. Estimated risks based on both central tendency and high-end exposures did not result in risks below the benchmark, for all these pathways, namely, drinking water exposures, fish ingestion, incidental oral ingestion from swimming or soil and dermal exposures from swimming. EPA is confident in the conservative high-end and central tendency exposure estimates (see Section 5.1.2.4.5) as they are based on reported facility-specific release data and is confident in the risk conclusion of no unreasonable risks below the benchmark for the assessed pathways. EPA is also confident that there are no unreasonable risks resulting from the high-end exposures that are protective of various lifestages, PESS, and tribal nations with higher fish consumption.

In addition, EPA is also confident that there are no unreasonable risks to the general population from TSCA conditions of use that were not identified that could have resulted in risks below the benchmark (see Section 1.1.2.1 for pathways of exposure). EPA does acknowledge the uncertainties associated with the presence and exposure of 1,1-dichloroethane from releases from other sources such as the degradation of other chlorinated solvents to 1,1-dichloroethane but does not expect these exposures to significantly change risk estimates.

At this time, based on limitations in reasonably available data, EPA could not quantitatively assess cancer from dermal exposure exposures route for all COUs to the general population. Without a quantitative dermal cancer assessment, EPA qualitatively characterized the dermal cancer risk, as

described in Section 5.2.5. Additionally, due to limitation in reasonably available oral data, EPA could not quantitatively assess the oral cancer exposure for all COUs to the general population. Due to the lack of quantitative oral/dermal cancer estimates, oral/dermal cancer risks are uncertain.

5.3.5.3 Hazard Values

EPA identified data gaps for 1,1-dichloroethane for non-cancer PODs by the acute, intermediate, and chronic oral, dermal, and inhalation routes; and cancer PODs by the oral, inhalation, and dermal routes. A read-across approach was used to identify the best chemical analog to fill those data gaps. The analyses resulted in the identification of 1,2-dichloroethane (an isomer of 1,1-dichloroethane) as the most appropriate analog to fill the identified data gaps for 1,1-dichloroethane (see Section 5.2.1.3). Based on the identification and selection of the PODs, although not using chemical specific data for 1,1-dichloroethane for the derivation of these values, the Agency is confident that the use of 1,2-dichloroethane data is supported by the weight of scientific evidence and thus human health. EPA has high confidence that the 1,2-dichloroethane isomer data accurately reflects the human health hazards of 1,1-dichloroethane where there are data gaps. In addition, 1,1- and 1,2-dichloroethane lacked adequate data by the dermal route for any exposure duration based on evaluation of epidemiological and animal studies. Therefore, EPA used a route-to-route extrapolation approach from the available 1,2-dichloroethane oral data to fill in the dermal data gap. The Agency has high confidence in this approach and in assessing dermal exposure and also incorporated test order data from 1,1-dichloroethane for risk estimates that was further corroborated by the *in silico* tool IH SkinPerm. Furthermore, as both oral and dermal routes are similar metabolically and bypass first pass metabolism through the liver, and since oral ADME studies showed that most of the 1,1-dichloroethane oral dose was eliminated unchanged in expired air, oral PODs were used for extrapolation via the dermal route.

EPA has high confidence in the human health hazard database for 1,2-dichloroethane and in the selection of the PODs. This is based on several reasons. First, all studies used to assess the hazards for 1,2-dichloroethane were rated high to medium in systematic review. Second, non-cancer effects that were ultimately selected as PODs for quantitative risk estimates (kidney toxicity, olfactory effects, and reproductive [sperm] effects) were considered the most sensitive and biologically relevant effects, supported by multiple lines of evidence that spanned across species, routes, and durations of exposure (see tables and exposure-response arrays [figures] in Section 5.2.6.1.5).

Although several studies via the chronic exposure duration for both oral and inhalation exposures were identified, these studies were not selected for the chronic POD due to study limitations and inherit uncertainties (see Section 0). Data based on the intermediate exposure duration were based on an overall weight of scientific evidence that identified an endpoint that was identified as appropriate and supported by other studies. The application of the UF_S of 10× was applied to account for the duration adjustment in using the identified intermediate studies for the corresponding long-term (chronic) duration under the assumption that the POD value at the intermediate duration would be approximately 10-times higher than a POD identified from a chronic study.

6 UNREASONABLE RISK DETERMINATION

TSCA section 6(b)(4) requires EPA to conduct a risk evaluation to determine whether a chemical substance presents an unreasonable risk of injury to human health or the environment—without consideration of costs or other non-risk factors—including an unreasonable risk to a potentially exposed or susceptible subpopulation (PESS) identified by the Agency as relevant to the risk evaluation, under the conditions of use (COUs).

Environment

EPA did not identify unreasonable risk of injury to the environment due to exposures via soil, air, surface water, and sediment *(e.g.*, reproductive effects to aquatic invertebrates, growth and developmental effects to algae) under the COUs.

Workers and Occupational Non-Users (ONUs)

EPA evaluated eight COUs for 1,1-dichloroethane, listed in Table 1-1. The Agency determined that 1,1-dichloroethane presents an unreasonable risk of injury to human health due to non-cancer health effects (*e.g.*, olfactory, male reproductive effects) or cancer risk (tumors) to workers from inhalation exposures driven by three of the eight COUs. EPA did not identify unreasonable risk of injury to human health due to non-cancer health effects (renal effects) for workers from dermal exposure or non-cancer health effects (*i.e.*, olfactory, male reproductive effects) or cancer risk (tumors) to ONUs from inhalation exposures. The three COUs that significantly contribute to the unreasonable risk determination for 1,1-dichloroethane due to identified unreasonable risk to workers are listed below:

- Processing as a reactant – intermediate in all other basic organic chemical manufacturing;
- Processing as a reactant – intermediate in all other chemical product and preparation manufacturing; and
- Processing – recycling.

For these three processing COUs, (see Section 5.3.3.1.5), when respirators that achieve a minimum APF 10 to 25 (depending on the expected workplace activity, represented in the risk evaluation by the various Similar Exposure Groups (SEGs)) are worn or other exposure controls (*e.g.*, engineering controls) that may be equally or more effective in reducing worker exposures are used, the unreasonable risk identified would no longer be unreasonable.

The five of the eight COUs that EPA determined do not significantly contribute to the determination of unreasonable risk of injury to human health are listed below:

- Manufacture (domestic manufacture);
- Processing – repackaging;
- Commercial use in laboratory chemicals;
- Distribution in commerce; and
- Disposal.

The determination for the Manufacturing COU is informed by an EPA test order to a consortium of companies where the consortium characterized the facility control operations that are known to be representative of workplace controls at 12 facilities (see Section 6.2.1).

General Population

EPA did not identify unreasonable risk of injury to human health due to non-cancer health effects (*e.g.*, renal, olfactory, male reproductive effects) from inhalation and dermal exposures or cancer risk (tumors) from inhalation exposures to the general population under the COUs.

As noted in the Executive Summary, 1,1-dichloroethane is a highly volatile organic compound mainly used to manufacture 1,1,1-trichloroethane (CASRN 71-55-6) and other chlorinated solvents—including 1,2-dichloroethane (accessed June 16, 2025) (CASRN 107-06-2) that is currently undergoing risk evaluation. Exposure to 1,1-dichloroethane is generally isolated to a few regions in the southern United States, and CDR reports between 2012 and 2020 include facilities solely located in Louisiana and Texas. There are no commercial or consumer applications besides commercial use in laboratory chemicals. EPA has not received any information to indicate the import of 1,1-dichloroethane is intended, known, or reasonably foreseen; therefore, import was not evaluated as a COU of 1,1-dichloroethane.

This unreasonable risk determination is based on the information provided in previous sections of this risk evaluation, the appendices, and supplemental documents (see Appendix B), in accordance with TSCA section 6(b). This risk evaluation discusses important assumptions and key sources of uncertainty in the risk characterization; these are described in more detail in the respective weight of scientific evidence conclusions sections for fate and transport (Section 2.2.3), environmental release (Section 3.2.2), environmental exposures (Section 4.1.5), environmental hazards (Section 4.2.4), human health hazards (Section 5.2.7), human health risk characterization (Section 5.3.5), and Appendix Q. It also includes overall confidence and remaining uncertainties sections for human health and environmental risk characterizations. In general, EPA makes an unreasonable risk determination based on risk estimates that have an overall confidence rating of moderate or robust because those confidence ratings indicate the scientific evidence is adequate to characterize risk estimates despite uncertainties or is such that it is unlikely the uncertainties could have a significant effect on the risk estimates. This unreasonable risk determination and the underlying evaluation are consistent with the best available science (TSCA section 26(h)) and based on the weight of scientific evidence (TSCA section 26(i)).

EPA will initiate risk management for 1,1-dichloroethane by applying one or more of the requirements under TSCA section 6(a) to the extent necessary so that 1,1-dichloroethane no longer presents an unreasonable risk. The Agency expects risk management requirements to focus on those COUs that drive the determination of unreasonable risk under TSCA section 6(a). EPA may select from among a suite of risk management options related to manufacture (including import), processing, distribution in commerce, commercial use, and disposal to address the unreasonable risk. For instance, the Agency may seek to regulate upstream COUs (*e.g.*, processing, distribution in commerce) to address downstream COUs that significantly contribute to unreasonable risk (*e.g.*, use) if necessary to address such unreasonable risk. EPA expects this to be the exception, not the norm. The Agency could also consider whether such risk may be prevented or reduced to a sufficient extent by action taken under another federal law, such that referral to another agency under TSCA section 9(a) or use of another EPA-administered authority to protect against such risk pursuant to TSCA section 9(b) may be appropriate.

6.1 Environment

Calculated risk quotients (RQs) can provide a risk profile by presenting a range of estimates for different environmental hazard effects for different COUs. An RQ equal to 1 indicates that the exposures are the same as the concentration that causes effects. An RQ less than 1, when the exposure is less than the effect concentration, generally indicates that there is no risk of injury to the environment that would support a determination of unreasonable risk for the chemical substance. An RQ greater than 1, when the exposure is greater than the effect concentration, generally indicates that there is risk of injury to the environment that would support a determination of unreasonable risk for the chemical substance. Additionally, if an RQ is 1 or greater, EPA evaluates whether the RQ is 1 or greater for the days of exceedance before making a determination of unreasonable risk.

EPA evaluated aquatic RQs and days of exceedance across two days-of-release scenarios: (1) at a hazard based-release duration (15 or 21 consecutive days of release); or (2) at the total number of operating days assumed as the maximum release duration. These are 350 days per year for the Manufacturing of 1,1-dichloroethane as an isolated intermediate and Processing as a reactive intermediate occupational exposure scenarios (OESs); 260 days per year at the Processing – repackaging and Commercial use as a laboratory chemical OESs; 250 days per year for the General waste handling, treatment, and disposal OES; and 365 days per year for the Waste handling, treatment, and disposal (POTW) and Waste handling, treatment, and disposal (remediation) OESs.

Because EPA did not have substantial information to support the lower-end, hazard-based release scenario, or the assumption that annual loads are released in consecutive days, the Agency based its environmental risk determination on the operating days release scenario. Although the operating days are estimated, estimates are based on known or generic patterns of operation for each OES.

6.1.1 Basis for No Unreasonable Risk to the Environment

Based on the risk evaluation for 1,1-dichloroethane—including the populations and exposures assessed, the environmental effects, the derived risk estimates, and consideration of uncertainties—EPA did not identify unreasonable risk of injury to the environment for 1,1-dichloroethane.

Due to chemical and physical properties and the low amounts of 1,1-dichloroethane undergoing wastewater treatment, land application of biosolids from 1,1-dichloroethane wastewater treatment was not expected to be a significant exposure pathway; therefore, EPA did not expect exposure to 1,1-dichloroethane from wastewater treatment to present unreasonable risk to terrestrial organisms. Similarly, the Agency did not expect exposure to 1,1-dichloroethane via biosolids to present unreasonable risk to the environment. Additionally, although EPA expected larger releases of 1,1-dichloroethane to surface water during storm events, the Agency also expected greater flow rates; thus, the low amounts of 1,1-dichloroethane in the total released surface water was not expected to present unreasonable risk to aquatic organisms. EPA's overall environmental risk characterization confidence levels were varied and are summarized in Section 4.3.6.

6.2 Human Health

Calculated risk estimates (margin of exposures [MOEs[21]] or cancer risk estimates[22]) can provide a risk profile of 1,1-dichloroethane by presenting a range of estimates for different health effects for different COUs. When characterizing the risk to human health from occupational exposures during risk evaluation under TSCA, EPA conducts baseline assessments of risk and makes its determination of unreasonable risk in a manner that takes in consideration reasonably available information (*e.g.,* test order information, site visits). It should be noted that, in some cases, baseline conditions may reflect certain mitigation measures, such as engineering controls, respiratory protection or other PPE, in instances where exposure estimates are based on monitoring data at facilities that have such controls in place. In this risk evaluation, monitoring data submitted pursuant to a test order allowed EPA to make its unreasonable risk determination taking into consideration specific information regarding workers wearing PPE. In addition, the risk estimates are based on exposure scenarios with monitoring data that reflect existing requirements, such as those established by OSHA (*i.e.,* permissible exposure limits [PELs]) or industry or sector best practices.

[21] EPA derives non-cancer MOEs by dividing the non-cancer POD (HEC [mg/m^3] or HED [mg/kg-day]) by the exposure estimate (mg/m^3 or mg/kg-day). Section 5.3.1 has additional information on the risk assessment approach for human health.
[22] Section 5.3.1 explains how cancer risk estimates are calculated.

An MOE that is less than the benchmark MOE is a starting point for informing a determination of unreasonable risk of injury to health, based on non-cancer effects. Similarly, a calculated cancer risk estimate that is greater than the cancer benchmark is a starting point for informing a determination of unreasonable risk of injury to health from cancer. Inhalation cancer risk estimates represent the incremental increase in probability of an individual in an exposed population developing cancer over a lifetime (excess lifetime cancer risk [ELCR]) following exposure to the chemical. Standard cancer benchmarks used by EPA and other regulatory agencies are an increased cancer risk ranging from 1 in 1,000,000 to 1 in 10,000 (*i.e.*, 1×10^{-6} to 1×10^{-4}), depending on the subpopulation exposed. In this assessment the Agency considers 1×10^{-4} as the appropriate benchmark for increased cancer risk for workers, including ONUs.

It is important to emphasize that these calculated risk estimates alone are not "bright-line" indicators of unreasonable risk. In the risk determination, EPA considered risk-related factors beyond exceedance of benchmarks—including the Agency's confidence in the data, an evaluation of the strengths, limitations, uncertainties, and confidences associated with the information used to inform the risk estimate and risk characterization. Descriptions of risk estimates that are based on highly refined hazard and exposure information would be considered differently than risk estimates based on conservative assumptions on both hazard and exposure. The process of determining unreasonable risk is made on a case-by-case basis, given the inherently unique nature of chemical-specific risk evaluations.

6.2.1 Basis for the Unreasonable Risk to Workers

EPA analyzed dermal and inhalation exposure in the occupational scenarios using a time-weighted average (TWA) for a typical 8-hour shift (see Section 5.3). Separate estimates of central tendency and high-end inhalation and dermal exposures were made for male and female adolescent (16–21 years) and adult (21+ years) workers directly working with 1,1-dichloroethane, as well as separate estimates for inhalation exposures for ONUs not directly working with 1,1-dichloroethane, as appropriate. Because 1,1-dichloroethane is primarily used at manufacturing and processing facilities, EPA does not expect workers at such facilities to be younger than 18 years old. Non-cancer risk estimates were calculated from acute, intermediate, and chronic exposures. For most OESs, acute refers to an exposure timeframe of one 8-hour workday, intermediate refers to an exposure time frame of 22 workdays (8 hours per day), and chronic refers to an exposure time frame of 250 days per year for 31 to 40 years (8 hours per day).

EPA analyzed the individual COUs in this risk evaluation under both central tendency and high-end estimates for workers and ONUs, based on the parameters and assumptions used in the OESs used to evaluate each COU. For all COUs with sufficient confidence to support a risk determination, based on the reasonably available information and the Agency's confidence and uncertainties described earlier in this risk evaluation, EPA is basing its unreasonable risk determination on the high-end for inhalation exposures. For all COUs with sufficient confidence to support a risk determination, based on the reasonably available information, and the Agency's confidence and uncertainties described earlier in this risk evaluation, EPA is basing its unreasonable risk determination for acute and intermediate dermal exposure on the high-end, and for chronic dermal exposure on the central tendency. The central tendency risk estimates were identified as more appropriate than the high-end for chronic dermal exposures due to differences in the magnitude and frequency of expected workplace exposures (see also Section 5.3.5). Additional information on occupational risk estimates is provided in Section 5.3.5.

Workers

As noted previously, based on the risk evaluation for 1,1-dichloroethane—including the populations and exposures assessed, the human health effects, the derived risk estimates, and consideration of

uncertainties—EPA found unreasonable risk of injury to human health from inhalation non-cancer and cancer risk estimates from 1,1-dichloroethane for workers driven by three COUs:

- Processing as a reactant – intermediate in all other basic organic chemical manufacturing,
- Processing as a reactant – intermediate in all other chemical product and preparation manufacturing, and
- Processing – recycling.

The unreasonable risk for these three COUs would no longer be unreasonable when using respirators that achieve a minimum APF 10 to 25 (depending on the expected workplace activity, represented in the risk evaluation by the various SEGs) or implementing other exposure controls (*e.g.*, engineering controls) that may be equally or more effective in reducing worker exposures, as described in Section 5.3.3.1.

The following five COUs do not significantly contribute to the determination of unreasonable risk of injury to human health for workers or ONUs:

- Manufacture (domestic manufacture);
- Processing – repackaging;
- Commercial use in laboratory chemicals;
- Distribution in commerce; and
- Disposal.

EPA used the test order data to estimate and assess exposures under five COUs: Manufacturing – domestic manufacturing; Processing as a reactant – intermediate in all other basic organic chemical manufacturing; Processing as a reactant – intermediate in all other chemical product and preparation manufacturing; Processing – recycling; and Commercial use – Laboratory chemical. Based on the workplace exposure monitoring data, EPA has moderate to robust confidence that the inhalation risk estimates are sufficient for determining whether a COU significantly contributes to unreasonable risk. The Agency used the high-end exposure levels as the basis of the unreasonable risk determination for the inhalation exposure to 1,1-dichloroethane for the COUs evaluated with monitoring data.

For the manufacturing COU, though the risk estimates are below the benchmark MOE without PPE, based on reasonably available information received in response to an EPA test order, the Agency accounted for known workplace controls for the manufacturing COU. The test order characterized the facility control operations known and expected to be in place depending on the potential exposure during standard, task-specific, and emergency activities—including engineering controls, administrative controls, PPE (*e.g.*, respirators achieving a level of APF 10–1,000), and chemical safety plans (Stantec ChemRisk, 2023). EPA is taking the PPE information into account in the determination for the Manufacturing COU. Therefore, due to the known workplace controls and use of respirators with a minimum APF 10 to 25 (depending on the expected workplace activity, represented in the risk evaluation by the various SEGs) at the representative manufacturing facility for 1,1-dicholorethane, EPA did not determine the Manufacturing COU significantly contributed to unreasonable risk.

The Processing – repackaging COU has MOEs below the benchmark at the central tendency and high-end for both non-cancer and cancer exposures. However, as discussed in Sections 5.1.1.3 and 5.3.5, EPA has slight confidence in these MOEs for both the non-cancer and cancer exposures because of the uncertainties in the modeled exposure. These uncertainties include the potentially over-estimated exposures, the lack of inhalation monitoring data or facility information, and the assumptions of quantity of 1,1-dichloroethane repackaged daily. EPA did not find any specific information on repackaging sites, but evaluated exposures based on the assumption that a repackaging step would need to take place prior

to the use of 1,1-dichloroethane as a laboratory chemical. EPA assumed that repackaging could occur at two manufacturing sites. If repackaging were to occur at a higher number of processing sites, due to the low volumes of 1,1-dichloroethane going to laboratory uses, the highly uncertain exposures would be further reduced. Therefore, Processing – repackaging COU was found to not significantly contribute to the unreasonable risk of 1,1-dichloroethane because of the Agency's slight confidence in the risk estimates.

The Disposal COU MOEs are below the benchmark at the central tendency and high-end for both non-cancer and cancer exposures. However, as discussed in Sections 5.1.1.3 and 5.3.5 EPA has slight confidence in these MOEs for both the non-cancer and cancer exposures because of the uncertainties in the modeled exposure. This includes uncertainties regarding whether the surrogate chemical volume throughput and concentrations, worker exposures, and waste streams are sufficiently similar to those for 1,1-dichloroethane. Additionally, most facilities generating or using 1,1-dichloroethane are using it in processes in which it is either consumed as part of a reaction or generated as an impurity, resulting in low concentrations of 1,1-dichloroethane expected to be present in the waste streams. Therefore, these MOEs reflect highly conservative inputs in the modeled exposure. Therefore, Disposal COU was found to not significantly contribute to the unreasonable risk of 1,1-dichloroethane because of the Agency's slight confidence in the risk estimates.

Inhalation MOEs for the Commercial use in laboratory chemicals COU are above the benchmark at the central tendency or high-end for both non-cancer and cancer. For dermal non-cancer, the chronic MOEs are below the benchmark at the high-end but not the central tendency. Based on the differences in magnitude of exposure and frequency of workplace exposure in the dermal assessment, described in Section 5.3.5, EPA used the central tendency for the chronic dermal risk determination and did not identify unreasonable risk under the Commercial use in laboratory chemicals COU.

Additionally, the Agency characterized distribution in commerce qualitatively because EPA had limited data about exposures from this COU besides those exposures from other COUs already quantified with release estimates. Although the Agency cannot calculate risk estimates for distribution in commerce separately from the risk related to loading and unloading from transport vehicles already estimated for other relevant COUs, EPA has concluded that distribution in commerce does not drive the determination of unreasonable risk.

Risk estimates based on high-end exposure levels (*e.g.,* 95th percentile) are generally intended to cover individuals exposed at sentinel exposure levels, whereas risk estimates at the central tendency exposure are intended to cover average or typical exposure. To determine the unreasonable risk EPA may consider chemical-specific information and risk-related factors, including (1) how the central tendency and high-end risk estimates best represent each COU (*e.g.,* where EPA may rely on central tendency exposures when the high-end risk estimates may not represent sentinel exposure levels accurately); or (2) rely on the high-end exposure levels in the absence of chemical-specific data or information. EPA used the central tendency risk estimates for the dermal exposure and the high-end estimates for the inhalation exposure as the basis of the unreasonable risk determination for 1,1-dichloroethane to workers. The use of the central tendency for dermal exposures and the high-end for inhalation exposures to 1,1-dichloroethane to make a determination of unreasonable risk for workers is based on the risk estimates for the COUs, the reasonably available information, and best representativeness of the average or typical exposure to workers and process within that COU. For COUs where the Agency was not able to estimate ONU inhalation exposure from monitoring data or models, the ONU exposure was assumed to be equivalent to the central tendency exposure for workers for the corresponding COU, as described in Section 5.1.1.1.4.

Dermal Considerations: EPA derived dermal risk estimates for both a deterministic and probabilistic calculation. The deterministic model used a single set of representative parameters but did not address variability in exposure duration and frequency. The probabilistic model did use the full distribution for most of the modeled parameters (except for fraction absorbed and event frequency). EPA used the probabilistic model as the basis for the unreasonable risk determination for 1,1-dichloroethane because the Agency had increased confidence in the probabilistic model as further discussed in Section 5.1.1.1.5.

Based on the uncertainties described in Section 5.3.5 of the risk evaluation, EPA has moderate to robust confidence that the dermal risk estimates generated by the model are sufficient for determining whether a COU presents unreasonable risk. EPA used the high-end exposure estimates for acute and intermediate dermal risk determination, and the central tendency exposure estimates for chronic dermal risk determination. The chronic dermal risk determination used the central tendency estimates due to the differences in magnitude and frequency of expected workplace exposures. EPA did not receive any information about the concentration of 1,1-dichloroethane during disposal. Thus, to be health protective, for disposal EPA used the neat scenario rather than the dilute scenario as the basis of the unreasonable risk determination for 1,1-dichloroethane.

Other Considerations: EPA did not have enough data to calculate risk estimates for all COUs and characterized the risk for those COUs by integrating limited amounts of reasonably available information in a qualitative characterization. At this time, based on limitations in reasonably available data, EPA could not quantitatively assess the cancer risks from dermal exposure for all COUs to workers, ONUs, and the general population. Without a quantitative dermal cancer assessment, the Agency qualitatively characterized the dermal cancer risk, as described in Section 5.2.5.

Although EPA aggregated a few exposure routes, the Agency did not aggregate risks across exposure routes for all exposure durations as the health outcomes (endpoints for the selected PODs) were different for oral/dermal and inhalation studies. EPA aggregated ambient air exposures from multiple neighboring facilities and oral and dermal risks from swimming for risk to the general population. The Agency has not characterized aggregate risk to workers or inhalation risk to people who both work at and live near facilities releasing 1,1-dichloroethane since EPA does not have data showing that this is a likely exposure scenario. More information on how the Agency characterized sentinel and aggregate risks is provided in Section 5.3.4.

Hazards: The acute and intermediate benchmark MOE for 1,1-dichloroethane is 30; the chronic benchmark MOE is 300. Derived from the total UFs, these benchmark MOEs are conservative given the reasonably available information as described in Section 5.2.6.1. The non-cancer PODs are based on susceptible populations. The acute POD is based on renal effects from dermal exposure and olfactory effects from inhalation exposure, whereas the intermediate and chronic PODs are based on renal effects from dermal exposure and male reproductive effects from inhalation exposure.

Although there is likely to be variability in susceptibility across the human population, EPA did not identify specific human groups that are expected to be more susceptible to cancer or non-cancer effects following 1,1-dichloroethane exposure. As described in Section 5.2.1.3, because acceptable human health hazard data were not available for 1,1-dichloroethane,1,2-dichloroethane studies were utilized for read-across to 1,1-dichloroethane for all non-cancer PODs and the inhalation cancer slope factor. EPA described their similar chemical properties and noted the greater reactivity of 1,2-dichloroethane compared to 1,1-dichloroethane. The Agency was not able to quantify the toxicological differences between 1,1-dichloroethane and 1,2-dichloroethane due to the limited data available for 1,1-dichloroethane, EPA identified 1,2-dichloroethane the most appropriate analog for the risk evaluation,

while recognizing it was a conservative and therefore health protective read-across approach. A dermal cancer slope factor was not reasonably able to be derived, and dermal cancer risk could not be assessed quantitatively, as described in Section 5.2.5.

Exposures: EPA used accepted approaches to estimate inhalation exposures in occupational settings as explained in Section 5.1.1. These include using specific inhalation monitoring data from an EPA-issued test order and other inhalation modeling data, including surrogate monitoring data and statistical modeling data as explained in Sections 5.1.1.1.3 and 5.1.1.1.4. Lacking 1,1-dichloroethane chemical specific data, the Agency used collected surrogate monitoring data from methylene chloride (accessed June 16, 2025) (assessed in previous EPA risk evaluations) and 1,2-dichloroethane (accessed June 16, 2025) (assessed in an ongoing EPA risk evaluation) because there are similarities in chemical properties, nature of workplace environment, and worker activities associated with certain uses of 1,1-dichloroethane. When EPA did not identify reasonably available surrogate monitoring data, modeled data were used.

6.2.2 Basis for No Unreasonable Risk to the General Population

EPA used the high-end exposure levels to make a determination of unreasonable risk for the general population to capture vulnerable populations that are expected to have higher exposures (*e.g.,* communities who live near facilities that emit 1,1-dichloroethane). Based on the risk estimates, calculated using releases from manufacturing, processing, and commercial uses of 1,1-dichloroethane and related risk factors, EPA did not identify unreasonable risk of injury to the general population based on either cancer or non-cancer risks from 1,1-dichloroethane from any assessed routes of exposure (ambient air inhalation, indoor air inhalation, incidental dermal from swimming, drinking water exposure, fish ingestion, incidental oral ingestion from swimming, soil ingestion**)**. Oral and dermal cancer risks were not quantitatively assessed due to limitations in reasonably available data to derive cancer risk estimates. Additionally, due to limitation in reasonably available oral data, EPA could not quantitatively assess the oral cancer exposure route for all COUs to the general population. Further discussion can be found in Section 5.3.

6.3 Supporting Basis for the Risk Determination

Table 6-1 summarizes the basis for this unreasonable risk determination of injury to human health by identifying the type of effect (*e.g.,* non-cancer and cancer for human health) and the exposure route to the population or receptor that results in such significant contribution presented in this 1,1-dichloroethane risk evaluation. In Table 6-1, the bolded numbers indicate that the COU significantly contributes to the unreasonable risk. The identified PPE in parentheses indicates the minimum identified controls needed in the absence of other exposure controls (*e.g.,* engineering controls) so that the risk is no longer unreasonable. If EPA did not identify unreasonable risk under the COU when considering all reasonably available information and risk-related factors substantiating the use of the known to be used and identified PPE based on the test order or that the exposure route does not drive the unreasonable risk determination, the numbers are not bolded. For the manufacturing COU, unreasonable risk was not identified when using respirators in a manner that achieves a minimum APF 10 to 25, depending on the workplace expected activities and exposure potential, represented by the various SEGs, as described in the submitted test order for manufacturing. As explained in Section 6, for this unreasonable risk determination, EPA considered the effects of 1,1-dichloroethane to human health for workers, ONUs, and the general population, as well as effects of 1,1-dichloroethane to human health and the environment from the exposures associated with the TSCA COUs, risk estimates, and uncertainties in the analysis.

Table 6-1. Supporting Basis for the Unreasonable Risk Determination for Human Health

COU			Population	Exposure Route	Human Health Effects and Identified Controls [a]			
Life Cycle Stage	**Category**	**Subcategory**			**Acute Non-Cancer (Benchmark = 30)**	**Intermediate Non-Cancer (Benchmark = 30)**	**Chronic Non-Cancer (Benchmark = 300)**	**Lifetime Cancer**
Manufacturing	Domestic manufacture	Domestic manufacture [b]	Worker	Dermal Central Tendency	492	307	328	NE
				Dermal High-End	288	179	195 (788 with PF 5)	NE
			Worker – Operator/ Process Technician	Inhalation Central Tendency	1,911	5,652	6,052	1.4E−05
				Inhalation High-End	20 (203 with APF 10)	60	64 (643 with APF 10)	1.7E−03 (6.67 E−05 with APF 25)
			Worker – Operator/Process Technician (responding to line leaks)	Inhalation Central Tendency	8 (80 with APF 10)	520	LF	LF
				Inhalation High-End	7.7 (77 with APF 10)	504	LF	LF
			Worker – Maintenance Technician	Inhalation Central Tendency	188	555	595	1.4E−04 (1.40 E−05 with APF 10)
				Inhalation High-End	36	107	114 (1,145 with APF 10)	9.4E−04 (9.36 E−05 with APF 10)
			Worker – Logistics/ Distribution Technician	Inhalation Central Tendency	5,284	1.6E04	1.7E04	5.0E−06
				Inhalation High-End	2,837	8,394	8,987	1.2E−05
			Worker – Laboratory Technician	Inhalation Central Tendency	1.3E04	3.9E04	4.2E04	2.0E−06
				Inhalation High-End	631	1,866	1,998	5.4E−05
			ONU	Inhalation Central Tendency	8,327	2.5E04	2.6E04	3.1E−06
				Inhalation High-End	811	2,398	2,568	4.2E−05

COU			Population	Exposure Route	Human Health Effects and Identified Controls [a]			
Life Cycle Stage	**Category**	**Subcategory**			**Acute Non-Cancer (Benchmark = 30)**	**Intermediate Non-Cancer (Benchmark = 30)**	**Chronic Non-Cancer (Benchmark = 300)**	**Lifetime Cancer**
Processing	Processing as a reactant	Intermediate in all other basic organic chemical manufacturing	Worker	Dermal Central Tendency	492	307	328	NE
				Dermal High-End	288	179	192 (788 with PF 5)	NE
			Worker – Operator/ Process Technician	Inhalation Central Tendency	1,911	5,652	6,052	1.4E−05
				Inhalation High-End	**20 (203 with APF 10)**	60	**64 (643 with APF 10)**	**1.7E−03 (6.67 E−05 with APF 25)**
			Worker – Maintenance Technician	Inhalation Central Tendency	188	555	595	**1.4E−04 (1.40 E−05 with APF 10)**
				Inhalation High-End	36	107	**114 (1,145 with APF 10)**	**9.4E−04 (9.36 E−05 with APF 10)**
			Worker – Logistics/ Distribution Technician	Inhalation Central Tendency	5,284	1.6E04	1.7E04	5.0E−06
				Inhalation High-End	2,837	8,394	8,987	1.2E−05
			Worker – Laboratory Technician	Inhalation Central Tendency	1.3E04	3.9E04	4.2E04	2.0E−06
				Inhalation High-End	631	1,866	1,998	5.4E−05
			ONU	Inhalation Central Tendency	8,327	2.5E04	2.6E04	3.1E−06
				Inhalation High-End	811	2,398	2,568	4.2E−05

COU			Population	Exposure Route	Human Health Effects and Identified Controls [a]			
Life Cycle Stage	**Category**	**Subcategory**			**Acute Non-Cancer (Benchmark = 30)**	**Intermediate Non-Cancer (Benchmark = 30)**	**Chronic Non-Cancer (Benchmark = 300)**	**Lifetime Cancer**
Processing	Processing as a reactant	Intermediate in all other chemical product and preparation manufacturing	Worker	Dermal Central Tendency	492	307	328	NE
				Dermal High-End	288	179	192 (788 with PF 5)	NE
			Worker – Operator/ Process Technician	Inhalation Central Tendency	1,911	5,652	6,052	1.4E−05
				Inhalation High-End	**20 (203 with APF 10)**	60	**64 (643 with APF 10)**	**1.7E−03 (6.67 E−05 with APF 25)**
			Worker – Maintenance Technician	Inhalation Central Tendency	188	555	595	**1.4E−04 (1.40 E−05 with APF 10)**
				Inhalation High-End	36	107	**114 (1,145 with APF 10)**	**9.4E−04 (9.36 E−05 with APF 10)**
			Worker – Logistics/ Distribution Technician	Inhalation Central Tendency	5,284	1.6E04	1.7E04	5.0E−06
				Inhalation High-End	2,837	8,394	8,987	1.2E−05
			Worker – Laboratory Technician	Inhalation Central Tendency	1.3E04	3.9E04	4.2E04	2.0E−06
				Inhalation High-End	631	1,866	1,998	5.4E−05
			ONU	Inhalation Central Tendency	8,327	2.5E04	2.6E04	3.1E−06
				Inhalation High-End	811	2,398	2,568	4.2E−05

COU			Population	Exposure Route	Human Health Effects and Identified Controls [a]			
Life Cycle Stage	**Category**	**Subcategory**			**Acute Non-Cancer (Benchmark = 30)**	**Intermediate Non-Cancer (Benchmark = 30)**	**Chronic Non-Cancer (Benchmark = 300)**	**Lifetime Cancer**
Processing	Recycling	Recycling	Worker	Dermal Central Tendency	492	307	328	NE
				Dermal High-End	288	179	192 (without PF 5)	NE
			Worker – Operator/ Process Technician	Inhalation Central Tendency	1,911	5,652	6,052	1.4E−05
				Inhalation High-End	**20 (203 with APF 10)**	60	**64 (643 with APF 10)**	**1.7E−03 (6.67 E−05 with APF 25)**
			Worker – Maintenance Technician	Inhalation Central Tendency	188	555	595	**1.4E−04 (1.40 E−05 with APF 10)**
				Inhalation High-End	36	107	**114 (1,145 with APF 10)**	**9.4E−04 (9.36 E−05 with APF 10)**
			Worker – Logistics/ Distribution Technician	Inhalation Central Tendency	5,284	1.6E04	1.7E04	5.0E−06
				Inhalation High-End	2,837	8,394	8,987	1.2E−05
			Worker – Laboratory Technician	Inhalation Central Tendency	1.3E04	3.9E04	4.2E04	2.0E−06
				Inhalation High-End	631	1,866	1,998	5.4E−05
			ONU	Inhalation Central Tendency	8,327	2.5E04	2.6E04	3.1E−06
				Inhalation High-End	811	2,398	2,568	4.2E−05

COU			Population	Exposure Route	Human Health Effects and Identified Controls [a]			
Life Cycle Stage	**Category**	**Subcategory**			**Acute Non-Cancer (Benchmark = 30)**	**Intermediate Non-Cancer (Benchmark = 30)**	**Chronic Non-Cancer (Benchmark = 300)**	**Lifetime Cancer**
Processing	Repackaging	Repackaging	Worker	Dermal Central Tendency	*	*	*	NE
				Dermal High-End	*	*	*	
			Worker (All Activities)	Inhalation Central Tendency	*	*	*	*
				Inhalation High-End	*	*	*	*
			Worker (Unloading and Cleaning)	Inhalation Central Tendency	*	*	*	*
				Inhalation High-End	*	*	*	*
			Worker (Loading)	Inhalation Central Tendency	*	*	*	*
				Inhalation High-End	*	*	*	*
			ONU	Inhalation Central Tendency	*	*	*	*
				Inhalation High-End	*	*	*	*
Commercial Use	Other uses	Laboratory chemicals	Worker	Dermal Central Tendency	499	311	373	NE
				Dermal High-End	292	182	214 (788 with PF 5)	NE
			Worker	Inhalation Central Tendency	1.3E04	3.9E04	6.0E04	1.4E−06
				Inhalation High-End	631	1,866	1,998	5.4E−05
			ONU	Inhalation Central Tendency	1.3E04	3.9E04	6.0E04	1.4E−06
				Inhalation High-End	1.3E04	3.9E04	4.2E04	2.6E−06
Distribution in Commerce	Distribution in commerce	Distribution in Commerce	Assessed qualitatively					

COU			Population	Exposure Route	Human Health Effects and Identified Controls [a]			
Life Cycle Stage	Category	Subcategory			Acute Non-Cancer (Benchmark = 30)	Intermediate Non-Cancer (Benchmark = 30)	Chronic Non-Cancer (Benchmark = 300)	Lifetime Cancer
Disposal	Disposal	Disposal [c]	Worker	Dermal Central Tendency	*	*	*	NE
				Dermal High-End	*	*	*	NE
			Worker (Dilute)	Dermal Central Tendency	*	*	*	NE
				Dermal High-End	*	*	*	NE
			Worker	Inhalation Central Tendency	*	*	*	*
				Inhalation High-End	*	*	*	*
			ONU	Inhalation Central Tendency	*	*	*	*
				Inhalation High-End	*	*	*	*
		Disposal [d]	Worker	Dermal Central Tendency	*	*	*	NE
				Dermal High-End	*	*	*	NE
			Worker (Dilute)	Dermal Central Tendency	*	*	*	NE
				Dermal High-End	*	*	*	NE
			Worker	Inhalation Central Tendency	*	*	*	*
				Inhalation High-End	*	*	*	*
			ONU	Inhalation Central Tendency	*	*	*	*
				Inhalation High-End	*	*	*	*

COU			Population	Exposure Route	Human Health Effects and Identified Controls [a]			
Life Cycle Stage	**Category**	**Subcategory**			**Acute Non-Cancer (Benchmark = 30)**	**Intermediate Non-Cancer (Benchmark = 30)**	**Chronic Non-Cancer (Benchmark = 300)**	**Lifetime Cancer**
NE = not estimated; lifetime chronic retained dose (LCRD) for cancer risk was not estimated as dermal cancer numbers for 1,1-dichloroethane were not derived. LF = low frequency; the chronic non-cancer and cancer effects were not evaluated due to lower exposure frequency. [a] The identified PPE in parentheses indicates the minimum identified controls needed in the absence of other exposure controls (*e.g.*, engineering controls) so that the risk is no longer unreasonable. If EPA did not identify unreasonable risk under the COU when considering all reasonably available information and risk-related factors substantiating the use of the known to be used and identified PPE based on the test order, or that the exposure route does not drive the unreasonable risk determination, the numbers are not bolded. When PPE would mitigate the risk, but was not known to be used, the numbers are bolded to represent that unreasonable risk was identified when PPE was not used. [b] EPA did not identify unreasonable risk under the COU when considering all reasonably available information and risk-related factors substantiating the use of the identified PPE. [c] Occupational exposure scenario: General waste handling, treatment, and disposal [d] Occupational exposure scenario: Waste handling, treatment, and disposal (POTW) * Based on the lack of inhalation monitoring data, facility information, and the quantity of 1,1-dichloroethane repackaged daily, or disposed of, EPA does not find the quantified numbers to be reliable due to the uncertainties and low confidence and they are therefore not presented. EPA did not find that these COUs significantly contribute to the unreasonable risk of injury to human health.								

REFERENCES

3M Environmental Lab. (1984). [Redacted] Data summary report on the tests for acute & chronic toxicity of fluorochemicals to Daphnia magna (water flea). (Lab Request No. 81098). St. Paul, MN: 3M.

Adamson, DT; Mahendra, S; Walker, KL, Jr; Rauch, SR; Sengupta, S; Newell, CJ. (2014). A multisite survey to identify the scale of the 1,4-dioxane problem at contaminated groundwater sites. Environ Sci Technol Lett 1: 254-258. http://dx.doi.org/10.1021/ez500092u

Ahmad, N; Benoit, D; Brooke, L; Call, D; Carlson, A; DeFoe, D; Huot, J; Morairity, A; Richter, J. (1984). Aquatic toxicity tests to characterise the hazard of volatile organic chemicals in water: A toxicity data summary – parts I and II. (EPA 600/3-84-009). Duluth, MN: U.S. Environmental Protection Agency. https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=91005AR3.txt

Alexander, GR. (1977). Food of vertebrate predators on trout waters in north central lower Michigan. Mich Acad 10: 181-195.

Alumot, E; Nachtomi, E; Mandel, E; Holstein, P. (1976). Tolerance and acceptable daily intake of chlorinated fumigants in the rat diet. Food Cosmet Toxicol 14: 105-111. http://dx.doi.org/10.1016/S0015-6264(76)80252-0

Alvarez-Cohen, L; Speitel, GE. (2001). Kinetics of aerobic cometabolism of chlorinated solvents. Biodegradation 12: 105-126. http://dx.doi.org/10.1023/A:1012075322466

Ansari, GA; Singh, SV; Gan, JC; Awasthi, YC. (1987). Human erythrocyte glutathione S-transferase: A possible marker of chemical exposure. Toxicol Lett 37: 57-62. http://dx.doi.org/10.1016/0378-4274(87)90167-6

Arfellini, G; Bartoli, S; Colacci, A; Mazzullo, M; Galli, MC; Prodi, G; Grilli, S. (1984). In vivo and in vitro binding of 1,2-dibromoethane and 1,2-dichloroethane to macromolecules in rat and mouse organs. J Cancer Res Clin Oncol 108: 204-213. http://dx.doi.org/10.1007/BF00402468

Arthur D. Little Inc. (1983). Cell transformation assays of 11 chlorinated hydrocarbon analogs: Final report: ICAIR work assignment no. 10 [TSCA Submission]. (EPA/OTS Doc #40-8324457). Cambridge, MA. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0509392.xhtml

ASTM. (2014). Standard guide for conducting acute toxicity tests on test materials with fishes, macroinvertebrates, and amphibians. (E 29-96(2014)). West Conshohocken, PA. https://compass.astm.org/document/?contentCode=ASTM%7CE0729-96R14%7Cen-US

ATSDR. (2015). Toxicological profile for 1,1-dichloroethane. (TP133). Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry. https://www.atsdr.cdc.gov/toxprofiles/tp133.pdf

ATSDR. (2022). Toxicological profile for 1,2-dichloroethane: Draft for public comment [ATSDR Tox Profile]. Atlanta, GA. https://wwwn.cdc.gov/TSP/ToxProfiles/ToxProfiles.aspx?id=592&tid=110

ATSDR. (2024). Toxicological profile for 1,2-dichloroethane [ATSDR Tox Profile]. Atlanta, GA. https://wwwn.cdc.gov/TSP/ToxProfiles/ToxProfiles.aspx?id=592&tid=110

Austin, SG; Schnatter, AR. (1983). A case-control study of chemical exposures and brain tumors in petrochemical workers. J Occup Environ Med 25: 313-320.

Aziz, CE; Smith, AP; Newell, CJ; Gonzales, J. (2000). BIOCHLOR: Chlorinated solvent plume database report. San Antonio, TX: Air Force Center for Environmental Excellence (AFCEE).

Baertsch, A; Lutz, WK; Schlatter, C. (1991). Effect of inhalation exposure regimen on DNA binding potency of 1,2-dichloroethane in the rat. Arch Toxicol 65: 169-176. http://dx.doi.org/10.1007/BF02307305

Banerjee, S. (1988). DNA damage in rodent liver by 1,2-dichloroethane, a hepatocarcinogen. Cancer Biochem Biophys 10: 165-173.

Barnthouse, LW; DeAngelis, DL; Gardner, RH; O'Neill, RV; Suter, GW; Vaughan, DS. (1982). Methodology for Environmental Risk Analysis. (ORNL/TM-8167). Oak Ridge, TN: Oak Ridge National Laboratory.

BASF. (2005). Letter: Subject: Supplemental information regarding prior TSCA Section 8(e) submission - Preliminary results from a cancer incidence study of employees assigned to a BASF Corporation former chemical manufacturing unit in Geismar, LA that ceased operations in 1987 (EPA Control number: 8EHQ-02-15135) [TSCA Submission]. (8EHQ-02-15135B. 89050000455). BASF Corporation.

Beeman, RE; Suflita, JM. (1987). Microbial ecology of a shallow unconfined ground water aquifer polluted by municipal landfill leachate. Microb Ecol 14: 39-54. http://dx.doi.org/10.1007/BF02011569

Beeman, RE; Suflita, JM. (1990). Environmental factors influencing methanogenesis in a shallow anoxic aquifer: a field and laboratory study. 5: 45-57. http://dx.doi.org/10.1007/BF01569605

Belford, B. (2023). 6.2: Similarity coefficients [Encyclopedia]. In Cheminformatics. Davis, CA: LibreTexts. https://chem.libretexts.org/Courses/Intercollegiate_Courses/Cheminformatics/06%3A_Molecular_Similarity/6.02%3A_Similarity_Coefficients

Bell, J; Melcer, H; Monteith, H; Osinga, I; Steel, P. (1993). Stripping of volatile organic compounds at full-scale municipal wastewater treatment plants. Water Environ Res 65: 708-716. http://dx.doi.org/10.2175/WER.65.6.2

Benoit, DA; Puglisi, FA; Olson, DL. (1982). A fathead minnow Pimephales promelas early life stage toxicity test method evaluation and exposure to four organic chemicals. Environ Pollut Ser A 28: 189-197. http://dx.doi.org/10.1016/0143-1471(82)90075-7

Benson, LO; Teta, MJ. (1993). Mortality due to pancreatic and lymphopoietic cancers in chlorohydrin production workers. Br J Ind Med 50: 710-716. http://dx.doi.org/10.1136/oem.50.8.710

Bigsby, PR; Myers, NC. (1989). Hydrogeology and ground-water-quality conditions at the Geary County landfill, northeast Kansas, 1988. (Water-Resources Investigations Report 89-4114). Denver, CO: U.S. Geological Survey. http://dx.doi.org/10.3133/wri894114

Borzelleca, JF; Carchman, RA. (1982). Effects of selected organic drinking water contaminants on male reproduction. Richmond, Va: Medical College of Virginia. https://search.proquest.com/docview/13543677?accountid=171501

Bove, FJ. (1996). Public drinking water contamination and birthweight, prematurity, fetal deaths, and birth defects. Toxicol Ind Health 12: 255-266.

Bove, FJ; Fulcomer, MC; Klotz, JB; Esmart, J; Dufficy, EM; Savrin, JE. (1995). Public drinking water contamination and birth outcomes. Am J Epidemiol 141: 850-862. http://dx.doi.org/10.1093/oxfordjournals.aje.a117521

Brack, W; Rottler, H. (1994). Toxicity testing of highly volatile chemicals with green algae: A new assay. Environ Sci Pollut Res Int 1: 223-228. http://dx.doi.org/10.1007/BF02986534

Brender, JD; Shinde, MU; Zhan, FB; Gong, X; Langlois, PH. (2014). Maternal residential proximity to chlorinated solvent emissions and birth defects in offspring: A case-control study. Environ Health 13: 96. http://dx.doi.org/10.1186/1476-069X-13-96

Brondeau, MT; Bonnet, P; Guenier, JP; De, CJ. (1983). Short-term inhalation test for evaluating industrial hepatotoxicants in rats. Toxicol Lett 19: 139-146. http://dx.doi.org/10.1016/0378-4274(83)90274-6

Buszka, PM; Yeskis, DJ; Kolpin, DW; Furlong, ET; Zaugg, SD; Meyer, MT. (2009). Waste-indicator and pharmaceutical compounds in landfill-leachate-affected ground water near Elkhart, Indiana, 2000-2002. Bull Environ Contam Toxicol 82: 653-659. http://dx.doi.org/10.1007/s00128-009-9702-z

Casciola, LAF; Ivanetich, KM. (1984). Metabolism of chloroethanes by rat liver nuclear cytochrome P-450. Carcinogenesis 5: 543-548. http://dx.doi.org/10.1093/carcin/5.5.543
CDC. (1978). Occupational health guideline for 1,1-Dichloroethane. September 1978. Occupational health guidelines for chemical hazards. DHHS (NIOSH) publication number 81-123. https://www.cdc.gov/niosh/docs/81-123/default.html
CEB. (1991). Chemical Engineering Branch manual for the preparation of engineering assessments: Volume I. CEB engineering manual. Washington, DC: Office of Pollution Prevention and Toxics, US Environmental Protection Agency. https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P10000VS.txt
Cheever, KL; Cholakis, JM; El-Hawari, AM; Kovatch, RM; Weisburger, EK. (1990). Ethylene dichloride: The influence of disulfiram or ethanol on oncogenicity, metabolism, and DNA covalent binding in rats. Toxicol Sci 14: 243-261. http://dx.doi.org/10.1016/0272-0590(90)90205-X
Chen, CS; Zoltek, J, Jr. (1995). Organic priority pollutants in wetland-treated leachates at a landfill in central Florida. Chemosphere 31: 3455-3464. http://dx.doi.org/10.1016/0045-6535(95)00198-H
Cheng, TJ; Chou, PY; Huang, ML; Du, CL; Wong, RH; Chen, PC. (2000). Increased lymphocyte sister chromatid exchange frequency in workers with exposure to low level of ethylene dichloride. Mutat Res 470: 109-114. http://dx.doi.org/10.1016/S1383-5742(00)00045-4
Cheng, TJ; Huang, ML; You, NC; Du, CL; Chau, TT. (1999). Abnormal liver function in workers exposed to low levels of ethylene dichloride and vinyl chloride monomer. J Occup Environ Med 41: 1128-1133. http://dx.doi.org/10.1097/00043764-199912000-00018
Christensen, TH; Kjeldsen, P; Bjerg, PL; Jensen, DL; Christensen, JB; Baun, A; Albrechtsen, HJ; Heron, G. (2001). Biogeochemistry of landfill leachate plumes. Appl Geochem 16: 659-718. http://dx.doi.org/10.1016/S0883-2927(00)00082-2
Chroust, K; Jowett, T; Farid-Wajidi, MF; Huang, JY; Ryskova, M; Wolf, R; Holoubek, I. (2001). Activation or detoxification of mutagenic and carcinogenic compounds in transgenic Drosophila expressing human glutathione S-transferase. Mutat Res 498: 169-179. http://dx.doi.org/10.1016/S1383-5718(01)00280-7
CITI. (1996a). Acute immobilization test on Daphnia magna exposed to 1,2-dichloroethane (translation). (Test No. 91502). Tokyo, Japan: Ministry of the Environment.
CITI. (1996b). Algal growth inhibition test of Selenastrum capricornurum exposed to 1,2-dichloroethane (translation). (Test No. 91501). Tokyo, Japan: Ministry of the Environment.
CITI. (1996c). Prolonged toxicity test (21 days) on medaka (Oryzias latipes) exposed to 1,2-dichloroethane (translation). (Test No. 91505). Tokyo, Japan: Ministry of the Environment.
CITI. (1996d). Reproductive inhibition test on Daphnia magna exposed to 1,2-dichloroethane (translation). (Test No. 91503). Tokyo, Japan: Ministry of the Environment.
Colacci, A; Arfellini, G; Mazzullo, M; Prodi, G; Grilli, S. (1985). Genotoxicity of 1,1-dichloroethane. Res Commun Chem Pathol Pharmacol 49: 243-254.
Cottalasso, D; Barisione, G; Fontana, L; Domenicotti, C; Pronzato, MA; Nanni, G. (1994). Impairment of lipoglycoprotein metabolism in rat-liver cells induced by 1,2-dichloroethane. Occup Environ Med 51: 281-285. http://dx.doi.org/10.1136/oem.51.4.281
Cottalasso, D; Domenicotti, C; Traverso, N; Pronzato, M; Nanni, G. (2002). Influence of chronic ethanol consumption on toxic effects of 1,2-dichloroethane: glycolipoprotein retention and impairment of dolichol concentration in rat liver microsomes and Golgi apparatus. Toxicology 178: 229-240. http://dx.doi.org/10.1016/S0300-483X(02)00235-4
Cottalasso, D; Fontana, L; Gazzo, P; Dapino, D; Domenicotti, C; Pronzato, MA; Nanni, G. (1995). Effects of 1,2-dichloroethane intoxication on dolichol levels and glycosyltransferase activities in rat liver microsomes and Golgi apparatus. Toxicology 104: 63-71. http://dx.doi.org/10.1016/0300-483X(95)03130-8

COWI AS. (2018). Screening programme 2017: Suspected PBT compounds. Trondheim, Norway: The Norwegian Environment Agency. https://www.miljodirektoratet.no/globalassets/publikasjoner/m1063/m1063.pdf
Crebelli, R; Andreoli, C; Carere, A; Conti, L; Crochi, B; Cotta-Ramusino, M; Benigni, R. (1995). Toxicology of halogenated aliphatic hydrocarbons: Structural and molecular determinants for the disturbance of chromosome segregation and the induction of lipid peroxidation. Chem Biol Interact 98: 113-129. http://dx.doi.org/10.1016/0009-2797(95)03639-3
Crebelli, R; Benigni, R; Franekic, J; Conti, G; Conti, L; Carere, A. (1988). Induction of chromosome malsegregation by halogenated organic solvents in Aspergillus nidulans: Unspecific or specific mechanism? Mutat Res 201: 401-411. http://dx.doi.org/10.1016/0027-5107(88)90027-9
Crebelli, R; Carere, A; Leopardi, P; Conti, L; Fassio, F; Raiteri, F; Barone, D; Ciliutti, P; Cinelli, S; Vericat, JA. (1999). Evaluation of 10 aliphatic halogenated hydrocarbons in the mouse bone marrow micronucleus test. Mutagenesis 14: 207-215. http://dx.doi.org/10.1093/mutage/14.2.207
Crespi, CL; Seixas, GM; Turner, TR; Ryan, CG; Penman, BW. (1985). Mutagenicity of 1,2-dichloroethane and 1,2-dibromoethane in two human lymphoblastoid cell lines. Mutat Res 142: 133-140. http://dx.doi.org/10.1016/0165-7992(85)90053-3
D'Souza, RW; Francis, WR; Andersen, ME. (1988). Physiological model for tissue glutathione depletion and increased resynthesis after ethylene dichloride exposure. J Pharmacol Exp Ther 245: 563-568.
D'Souza, RW; Francis, WR; Bruce, RD; Andersen, ME. (1987). Physiologically based pharmacokinetic model for ethylene dichloride and its application in risk assessment. In Pharmacokinetics in risk assessment: V 8, drinking water and health. Washington, DC: National Academy Press.
Daigle, JHJ; Cole, DN; Carlson, J; Lee, WR; Wilson, VL. (2009). Ethylene Dichloride Disruption of Fertility in Male Mice. The Open Toxicology Journal 3: 39-46. http://dx.doi.org/10.2174/1874340400903010039
Daniel, FB; Robinson, M; Olson, GR; York, RG; Condie, LW. (1994). Ten and ninety-day toxicity studies of 1,2-dichloroethane in Sprague-Dawley rats. Drug Chem Toxicol 17: 463-477. http://dx.doi.org/10.3109/01480549409014312
Davis, B. (2012). Endometrial stromal polyps in rodents: Biology, etiology, and relevance to disease in women [Review]. Toxicol Pathol 40: 419-424. http://dx.doi.org/10.1177/0192623311431466
Dietz, AC; Schnoor, JL. (2001). Phytotoxicity of chlorinated aliphatics to hybrid poplar (Populus deltoides x nigra DN34). Environ Toxicol Chem 20: 389-393. http://dx.doi.org/10.1002/etc.5620200221
Doherty, AT; Ellard, S; Parry, EM; Parry, JM. (1996). An investigation into the activation and deactivation of chlorinated hydrocarbons to genotoxins in metabolically competent human cells. Mutagenesis 11: 247-274. http://dx.doi.org/10.1093/mutage/11.3.247
Dosemeci, M; Cocco, P; Chow, WH. (1999). Gender differences in risk of renal cell carcinoma and occupational exposures to chlorinated aliphatic hydrocarbons. Am J Ind Med 36: 54-59. http://dx.doi.org/10.1002/(sici)1097-0274(199907)36:1<54::aid-ajim8>3.0.co;2-0
Dow Chemical. (1947). Results of range-finding toxicological studies on Ethylidene Dichloride [TSCA Submission]. (OTS0515950. 86-870002160. TSCATS/309472).
Dow Chemical. (1956). Results of skin absorption studies on carbon tetrachloride, ethylene dichloride, tetrachloroethylene, trichloroethylene, and chlorothene [TSCA Submission]. (OTS0515981. 86-870002191. TSCATS/309536). https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0515981.xhtml
Dow Chemical. (1962). Topical application of various solvents and solutions to evaluate dermal irritation [TSCA Submission]. (OTS0515970. 86-870002180. TSCATS/309514). https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0515970.xhtml

Dow Chemical. (1988). Letter from Dow Chem Co to U.S. EPA regarding submission of final study reports for 1,2-dichloropropane with attachments. (EPA/OTS Doc #40-8867156). Midland, MI. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0527733.xhtml
Dow Chemical. (1989). Comparison of the acute lethality of selected hydrocarbons via intratracheal and oral routes (final report) with attachments, cover sheets and letter dated 061989 [TSCA Submission]. (Laboratory Project Study ID T2.02-194-000-002. OTS0520615. 86-890000576. TSCATS/404074).
Dow Chemical. (2005). Ethylene dichloride: Acute vapor inhalation toxicity study in Fischer 344 rats. (041089). Millwood, VA: HAP Task Force for Ethylene Dichloride.
Dow Chemical. (2006a). 1,2-Dichloroethane (EDC): Limited pharmacokinetics and metabolism study in Fischer 344 rats. (041093). Millwood, VA: HAP Task Force.
Dow Chemical. (2006b). Re: Testing consent order for ethylene dichloride; final report (docket no . OPPT-2003-0010) [TSCA Submission]. (Study ID No. 041115. 40060000065). HAP Task Force for Ethylene Dichloride.
Dow Chemical. (2010). [Redacted] Reanalysis of algal growth inhibition data from 1,2-dichloropropane report "1,2-Dichloropropane: The toxicity to Skeletonema costatum". (ES-2014). Midland, MI.
Dow Chemical. (2014). [Redacted] Investigation of the mode of action for 1,2-dichloroethane-induced mammary tumors in female F344/DuCrl rats. (121180).
Dow Chemical. (2017). [Redacted] 1,2-Dichloroethane: Acute vapor inhalation toxicity study in F344/DuCrl rats. (171002). Brussels, Belgium: ReachCentrum S.A.
Dreher, EL; Beutel, KK; Myers, JD; Lübbe, T; Krieger, S; Pottenger, LH. (2014). Chloroethanes and chloroethylenes. In B Elvers (Ed.), Ullmann's encyclopedia of industrial chemistry (6th ed., pp. 1-81). Hoboken, NJ: Wiley-VCH Verlag GmbH & Co. http://dx.doi.org/10.1002/14356007.o06_o01.pub2
EC/HC. (2011). Screening assessment report on hexabromocyclododecane. Ottawa, Canada. http://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=7882C148-1#a4
ECB. (2003). Technical guidance document on risk assessment: Part II. (EUR 20418 EN/2). Luxembourg: Office for Official Publications of the European Communities. http://ihcp.jrc.ec.europa.eu/our_activities/public-health/risk_assessment_of_Biocides/doc/tgd/tgdpart2_2ed.pdf
EFSA. (2023). Risk assessment for birds and mammals. EFSA J 21: 7790. http://dx.doi.org/10.2903/j.efsa.2023.7790
Ellis, PA; Rivett, MO. (2007). Assessing the impact of VOC-contaminated groundwater on surface water at the city scale. J Contam Hydrol 91: 107-127. http://dx.doi.org/10.1016/j.jconhyd.2006.08.015
Elsevier. (2019). Reaxys: physical-chemical property data for 1,1-dichloroethane. CAS Registry Number: 75-34-3. Available online
Enwright Associates. (1985). Groundwater & wastewater monitoring report with cover letter dated 120385 [TSCA Submission]. (EPA/OTS Doc #878216227). Morristown, NJ: Allied Corporation. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0206891.xhtml
Etterson, M. (2020a). Species Sensitivity Distribution (SSD) Toolbox. Duluth, MN: U.S. Environmental Protection Agency. Retrieved from https://www.epa.gov/sciencematters/species-sensitivity-distribution-toolbox-new-tool-identify-and-protect-vulnerable
Etterson, M. (2020b). Technical Manual: SSD Toolbox Version 1.0. (EPA/600/R-19/104). Duluth, MN: U.S. Environmental Protection Agency. https://www.epa.gov/sciencematters/species-sensitivity-distribution-toolbox-new-tool-identify-and-protect-vulnerable
Fan, C; Wang, GS; Chen, YC; Ko, CH. (2009). Risk assessment of exposure to volatile organic compounds in groundwater in Taiwan. Sci Total Environ 407: 2165-2174. http://dx.doi.org/10.1016/j.scitotenv.2008.12.015

Ferrario, JB; Lawler, GC; Deleon, IR; Laseter, JL. (1985). Volatile organic pollutants in biota and sediments of Lake Pontchartrain. Bull Environ Contam Toxicol 34: 246-255. http://dx.doi.org/10.1007/BF01609730

Foster, G; Tullis, RE. (1984). A quantitative structure-activity relationship between partition coefficients and the acute toxicity of naphthalene derivatives in Artemia salina nauplii. Aquat Toxicol AMST: 245-254. http://dx.doi.org/10.1016/0166-445X(84)90023-7

Foster, GD; Tullis, RE. (1985). Quantitative structure-toxicity relationships with osmotically stressed Artemia salina nauplii. Environ Pollut Ser A 38: 273-281. http://dx.doi.org/10.1016/0143-1471(85)90132-1

Francovitch, RJ; Schor, NA; George, WJ. (1986). Effects of SKF 525A, phenobarbital, and 3-methylcholanthrene on ethylene dichloride toxicity following inhalation exposure. Int J Toxicol 5: 117-126. http://dx.doi.org/10.3109/10915818609141016

Frasch, HF; Barbero, AM. (2009). A paired comparison between human skin and hairless guinea pig skin in vitro permeability and lag time measurements for 6 industrial chemicals. Cutan Ocul Toxicol 28: 107-113. http://dx.doi.org/10.1080/15569520902950474

Frasch, HF; Barbero, AM; Alachkar, H; Mcdougal, JN. (2007). Skin penetration and lag times of neat and aqueous diethyl phthalate, 1,2-dichloroethane and naphthalene. Cutan Ocul Toxicol 26: 147-160. http://dx.doi.org/10.1080/15569520701212274

Fuchsman, PC. (2003). Modification of the equilibrium partitioning approach for volatile organic compounds in sediment. Environ Toxicol Chem 22: 1532-1534.

Gajjar, RM; Kasting, GB. (2014). Absorption of ethanol, acetone, benzene and 1,2-dichloroethane through human skin in vitro: a test of diffusion model predictions. Toxicol Appl Pharmacol 281: 109-117. http://dx.doi.org/10.1016/j.taap.2014.09.013

Garcia, E; Hurley, S; Nelson, DO; Hertz, A; Reynolds, P. (2015). Hazardous air pollutants and breast cancer risk in California teachers: A cohort study. Environ Health 14: 1-175. http://dx.doi.org/10.1186/1476-069X-14-14

Gargas, ML; Andersen, ME. (1989). Determining kinetic constants of chlorinated ethane metabolism in the rat from rates of exhalation. Toxicol Appl Pharmacol 99: 344-353. http://dx.doi.org/10.1016/0041-008X(89)90016-1

Gargas, ML; Burgess, RJ; Voisard, DE; Cason, GH; Andersen, ME. (1989). Partition coefficients of low-molecular-weight volatile chemicals in various liquids and tissues. Toxicol Appl Pharmacol 98: 87-99. http://dx.doi.org/10.1016/0041-008x(89)90137-3

Gargas, ML; HJ, C; Andersen, ME. (1990). Gas uptake inhalation techniques and the rates of metabolism of chloromethanes, chloroethanes, and chloroethylenes in the rat. Inhal Toxicol 2: 295-319. http://dx.doi.org/10.3109/08958379009145260

Geiger, DL; Northcott, CE; Call, DJ; Brooke, LT. (1985). Acute toxicities of organic chemicals to fathead minnows (Pimephales promelas): Volume II. Superior, WI: Center for Lake Superior Environmental Studies, University of Wisconsin-Superior.

Ghanayem, BI; Maronpot, RR; Matthews, HB. (1986). Association of chemically induced forestomach cell proliferation and carcinogenesis. Cancer Lett 32: 271-278. http://dx.doi.org/10.1016/0304-3835(86)90179-5

Ghassemi, M; Quinlivan, S; Bachmaier, J. (1984). Characteristics of leachates from hazardous waste landfills. J Environ Sci Health A Environ Sci Eng 19: 579-620. http://dx.doi.org/10.1080/10934528409375180

Gigante, V; Pauletti, GM; Kopp, S; Xu, M; Gonzalez-Alvarez, I; Merino, V; McIntosh, MP; Wessels, A; Lee, BJ; Rezende, KR; Scriba, GK; Jadaun, GPS; Bermejo, M. (2021). Global testing of a consensus solubility assessment to enhance robustness of the WHO biopharmaceutical classification system. ADMET & DMPK 9. http://dx.doi.org/10.5599/admet.850

Giri, AK; Que Hee, SS. (1988). In vivo sister chromatid exchange induced by 1,2-dichloroethane on bone marrow cells of mice. Environ Mol Mutagen 12: 331-334. http://dx.doi.org/10.1002/em.2860120307

Gotoh, M; Sekitani, Y; Aramaki, T; Kobayashi, H; Ogino, K; Hobara, T. (1992). Pollution due to volatile halocarbon compounds in biota. Bull Environ Contam Toxicol 49: 186-191. http://dx.doi.org/10.1007/BF00191753

Government of Canada. (2021). Fact sheet: 1,1-dichloroethane. https://gost.tpsgc-pwgsc.gc.ca/Contfs.aspx?ID=4&lang=eng

Grostern, A; Edwards, EA. (2006). A 1,1,1-trichloroethane-degrading anaerobic mixed microbial culture enhances biotransformation of mixtures of chlorinated ethenes and ethanes. Appl Environ Microbiol 72: 7849. http://dx.doi.org/10.1128/AEM.01269-06

Guengerich, FP; Crawford, WM, Jr; Domoradzki, JY; McDonald, TL; Watanabe, PG. (1980). In vitro activation of 1,2-dichloroethane by microsomal and cytosolic enzymes. Toxicol Appl Pharmacol 55: 303-317. http://dx.doi.org/10.1016/0041-008X(80)90092-7

Guengerich, FP; Kim, DH; Iwasaki, M. (1991). Role of human cytochrome P-450 IIE1 in the oxidation of many low molecular weight cancer suspects. Chem Res Toxicol 4: 168-179. http://dx.doi.org/10.1021/tx00020a008

Guo, XL; Niu, Q. (2003). [The relationship between excitatory amino acids and acute intoxicated encephalopathy induced by 1,2-dichloroethane]. Zhonghua Laodong Weisheng Zhiyebing Zazhi 21: 83-85.

Gwinn, MR; Johns, DO; Bateson, TF; Guyton, KZ. (2011). A review of the genotoxicity of 1,2-dichloroethane (EDC) [Review]. Mutat Res 727: 42-53. http://dx.doi.org/10.1016/j.mrrev.2011.01.001

Hachiya, N; Motohashi, Y. (2000). Examination of lacZ mutant induction in the liver and testis of Muta(TM)Mouse following injection of halogenated aliphatic hydrocarbons classified as human carcinogens. Ind Health 38: 213-220. http://dx.doi.org/10.2486/indhealth.38.213

Hamonts, K; Kuhn, T; Maesen, M; Bronders, J; Lookman, R; Kalka, H; Diels, L; Meckenstock, RU; Springael, D; Dejonghe, W. (2009). Factors determining the attenuation of chlorinated aliphatic hydrocarbons in eutrohic river sediment impacted by discharging polluted groundwater. Environ Sci Technol 43: 5270-5275. http://dx.doi.org/10.1021/es8035994

Hanahan, D; Weinberg, RA. (2011). Hallmarks of cancer: The next generation [Review]. Cell 144: 646-674. http://dx.doi.org/10.1016/j.cell.2011.02.013

Hannah, SA; Austern, BM; Eralp, AE; Wise, RH. (1986). Comparative removal of toxic pollutants by six wastewater treatment processes. J Water Pollut Control Fed 58: 27-34.

Hatch, GG; Mamay, PD; Ayer, ML; Casto, BC; Nesnow, S. (1983). Chemical enhancement of viral transformation in Syrian hamster embryo cells by gaseous and volatile chlorinated methanes and ethanes. Cancer Res 43: 1945-1950.

Heck, BA; Myers, NC; Hargadine, DA. (1992). Hydrogeology and ground-water quality conditions at the Reno County Landfill, South-Central Kansas, 1990-91. (92-4169). Heck, BA; Myers, NC; Hargadine, DA. http://dx.doi.org/10.3133/wri924169

Hellman, B; Brandt, I. (1986). Effects of carcinogenic halogenated aliphatic hydrocarbons on [3H]thymidine incorporation into various organs of the mouse. A comparison between 1,2-dibromoethane and 1,2-dichloroethane. Mutat Res 163: 193-199. http://dx.doi.org/10.1016/0027-5107(86)90048-5

Heppel, LA; Neal, PA; Perrin, TL; Endicott, KM; Porterfield, VT. (1945). The toxicology of 1,2-dichloroethane (ethylene). III. Its acute toxicity and the effect of protective agents. J Pharmacol Exp Ther 84: 53-63.

Heppel, LA; Neal, PA; Perrin, TL; Endicott, KM; Porterfield, VT. (1946). The toxicology of 1,2-dichloroethane (ethylene dichloride): V. The effects of daily inhalations. J Ind Hyg Toxicol 28: 113-120.
Hill, AB. (1965). The environment and disease: Association or causation? Proc R Soc Med 58: 295-300. http://dx.doi.org/10.1177/003591576505800503
Hilton, GM; Adcock, C; Akerman, G; Baldassari, J; Battalora, M; Casey, W; Clippinger, AJ; Cope, R; Goetz, A; Hayes, AW; Papineni, S; Peffer, RC; Ramsingh, D; Williamson Riffle, B; Sanches da Rocha, M; Ryan, N; Scollon, E; Visconti, N; Wolf, DC; Yan, Z; Lowit, A. (2022). Rethinking chronic toxicity and carcinogenicity assessment for agrochemicals project (ReCAAP): A reporting framework to support a weight of evidence safety assessment without long-term rodent bioassays. Regul Toxicol Pharmacol 131: 105160. http://dx.doi.org/10.1016/j.yrtph.2022.105160
Hofmann, HT; Birnstiel, H; Jobst, P. (1971). On inhalation toxicity of 1,1- and 1,2-dichloroethane. Arch Toxikol 27: 248-265. http://dx.doi.org/10.1007/BF00315048
Hopple, JA; Delzer, GC; Kingsbury, JA. (2009). Anthropogenic organic compounds in source water of selected community water systems that use groundwater, 2002-05 (pp. 76). (SIR 2009-5200). Reston, VA: U.S. Geological Survey. http://dx.doi.org/10.3133/sir20095200
Horvath, RS. (1972). Microbial co-metabolism and the degradation of organic compounds in nature. Bacteriol Rev 36: 146-155. http://dx.doi.org/10.1128/br.36.2.146-155.1972
Hotchkiss, JA; Andrus, AK; Johnson, KA; Krieger, SM; Woolhiser, MR; Maurissen, JP. (2010). Acute toxicologic and neurotoxic effects of inhaled 1,2-dichloroethane in adult Fischer 344 rats. Food Chem Toxicol 48: 470-481. http://dx.doi.org/10.1016/j.fct.2009.10.039
HSDB. (2008). Hexabromocyclododecane (HBCD). Bethesda, MD: National Library of Medicine. http://toxnet.nlm.nih.gov/cgi-bin/sis/search/f?./temp/~XQzOHy:1
Huang, M; Zhong, Y; Lin, L; Liang, B; Liu, J; Jiang, J; Hu, M; Huang, Y; Lin, X; Lu, L; Bian, Z; Zhong, W; Wu, J; Zheng, J; Rong, W; Zhang, Y; Jiang, L; Wu, J; Zhang, X; Yang, X; Hu, Q; Huang, Z. (2020). 1,2-Dichloroethane induces cerebellum granular cell apoptosis via mitochondrial pathway in vitro and in vivo. Toxicol Lett 322: 87-97. http://dx.doi.org/10.1016/j.toxlet.2020.01.004
Huang, Y; Su, T; Wang, L; Wang, N; Xue, Y; Dai, W; Lee, SC; Cao, J; Ho, SSH. (2019). Evaluation and characterization of volatile air toxics indoors in a heavy polluted city of northwestern China in wintertime. Sci Total Environ 662: 470-480. http://dx.doi.org/10.1016/j.scitotenv.2019.01.250
Huff, GF; Braun, CL; Lee, RW. (2000). Assessment of potential for natural attenuation of chlorinated ethenes and ethanes in ground water at a petrochemical reclamation site, Harris County, Texas. Austin, TX: Geological Survey Water Resources Division. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/PB2001106225.xhtml
Hunt, J; Birch, G; Warne, MSJ. (2007). Deriving trigger values for, and assessing hazard posed by, volatile chlorinated hydrocarbons in a sydney estuary. Australasian Journal of Ecotoxicology 13: 33-42.
Igwe, OJ; Que Hee, SS; Wagner, WD. (1986a). Effect of disulfiram pretreatment on the tissue distribution, macromolecular binding, and excretion of [U-1,2-14C]dichloroethane in the rat. Drug Metab Dispos 14: 65-72.
Igwe, OJ; Que Hee, SS; Wagner, WD. (1986b). Interaction between 1,2-dichloroethane and disulfiram. I. Toxicologic effects. Fundam Appl Toxicol 6: 733-746. http://dx.doi.org/10.1016/0272-0590(86)90186-7
Igwe, OJ; Que Hee, SS; Wagner, WD. (1986c). Interaction between 1,2-dichloroethane and tetraethylthiuram disulfide (disulfiram). II. Hepatotoxic manifestations with possible mechanism of action. Toxicol Appl Pharmacol 86: 286-297. http://dx.doi.org/10.1016/0041-008X(86)90059-1

Inskeep, PB; Koga, N; Cmarik, JL; Guengerich, FP. (1986). Covalent binding of 1,2-dihaloalkanes to DNA and stability of the major DNA adduct, S-[2-(N7-guanyl)ethyl]glutathione. Cancer Res 46: 2839-2844.
IPCS. (1995). 1,2-Dichloroethane (Second edition) [WHO EHC]. Geneva, Switzerland: World Health Organization. http://www.inchem.org/documents/ehc/ehc/ehc176.htm
IRFMN. (1976). Clinical chemistry results after 6 months inhalatory exposure to ethylene dichloride [TSCA Submission]. (OTS0515738. 86-870001662. TSCATS/309048). Shell Oil Company. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0515738.xhtml
IRFMN. (1978). Clinical chemistry results in adult rats exposed to ethylene dichloride by inhalation for 12 months [TSCA Submission]. (OTS0515737. 86-870001661). Shell Oil Company. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0515737.xhtml
IRFMN. (1987a). Long-term effects on rats and mice of ethylene dichloride, administered by inhalation [TSCA Submission]. (OTS0515739. 86-870001663. TSCATS/309050). Shell Oil Company. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0515739.xhtml
IRFMN. (1987b). Report on the clinical chemistry results after 18 months inhalatory exposure - ethylene dichloride [TSCA Submission]. (OTS0517059. 86-870002269. TSCATS/309692). Dow Chemical. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0517059.xhtml
Jakobson, I; Wahlberg, JE; Holmberg, B. (1982). Uptake via the blood and elimination of 10 organic solvents following epicutaneous exposure of anesthetized guinea pigs. Toxicol Appl Pharmacol 63:181-187.
Jean, PA; Reed, DJ. (1992). Utilization of glutathione during 1,2-dihaloethane metabolism in rat hepatocytes. Chem Res Toxicol 5: 386-391. http://dx.doi.org/10.1021/tx00027a011
Jeffers, PM; Ward, LM; Woytowitch, LM; Wolfe, NL. (1989). Homogeneous hydrolysis rate constants for selected chlorinated methanes, ethanes, ethenes, and propanes. Environ Sci Technol 23: 965-969. http://dx.doi.org/10.1021/es00066a006
Jin, X; Liao, Y; Tan, X; Guo, J; Wang, G; Zhao, F; Jin, Y. (2018a). Involvement of the p38 MAPK signaling pathway in overexpression of matrix metalloproteinase-9 during the course of brain edema in 1,2-dichloroethane-intoxicated mice. Neurotoxicology 69: 296-306. http://dx.doi.org/10.1016/j.neuro.2018.07.022
Jin, X; Liao, Y; Tan, X; Wang, G; Zhao, F; Jin, Y. (2018b). Involvement of CYP2E1 in the course of brain edema induced by subacute poisoning with 1,2-dichloroethane in mice. Front Pharmacol 9: 1317. http://dx.doi.org/10.3389/fphar.2018.01317
Johnson, MK. (1967). Metabolism of chloroethanol in the rat. Biochem Pharmacol 16: 185-199. http://dx.doi.org/10.1016/0006-2952(67)90199-2
Kaiser K, LE; Mckinnon, MB; Stendahl, DH; Pett, WB. (1995). Response threshold levels of selected organic compounds for rainbow trout (Oncorhynchus mykiss). Environ Toxicol Chem 14: 2107-2113. http://dx.doi.org/10.1002/etc.5620141214
Kanada, M; Miyagawa, M; Sato, M; Hasegawa, H; Honma, T. (1994). Neurochemical profile of effects of 28 neurotoxic chemicals on the central nervous system in rats (1) Effects of oral administration on brain contents of biogenic amines and metabolites. Ind Health 32: 145-164. http://dx.doi.org/10.2486/indhealth.32.145
KemI. (2008). Risk assessment: Hexabromocyclododecane. Ispra, Italy: European Chemicals Bureau. https://echa.europa.eu/documents/10162/661bff17-dc0a-4475-9758-40bdd6198f82
Kerler, F; Schoenherr, J. (1988). Permeation of lipophilic chemicals across plant cuticles prediction from partition coefficients and molar volumes. Arch Environ Contam Toxicol 17: 7-12. http://dx.doi.org/10.1007/BF01055147
Kernan, GJ; Ji, BT; Dosemeci, M; Silverman, DT; Balbus, J; Zahm, SH. (1999). Occupational risk factors for pancreatic cancer: A case-control study based on death certificates from 24 U.S.

states. Am J Ind Med 36: 260-270. http://dx.doi.org/10.1002/(SICI)1097-0274(199908)36:2<260::AID-AJIM5>3.0.CO;2-P

Kettering Laboratory. (1943). The physiological effects upon rabbits of exposure to 1,2-dichloroethane and 1,2-dibromoethane [TSCA Submission]. (OTS0516127. 86-870001224. TSCATS/400132). University of Cincinnati. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0516127.xhtml

King County DNRP. (2004). 2003 Biosolids Quality Summary. Seattle, WA: King County Department of Natural Resources and Parks, Wastewater Treatment Division.

Kingsbury, JA; Delzer, GC; Hopple, JA. (2008). Anthropogenic organic compounds in source water of nine community water systems that withdraw from streams, 2002–05 (pp. 68). (Scientific Investigations Report 2008–5208). Reston, VA: U.S. Geological Survey. http://pubs.usgs.gov/sir/2008/5208/

Kitchin, KT; Brown, JL; Kulkarni, AP. (1993). Predicting rodent carcinogenicity of halogenated hydrocarbons by in vivo biochemical parameters. Birth Defects Res B Dev Reprod Toxicol 13: 167-184. http://dx.doi.org/10.1002/tcm.1770130403

Klaunig, JE; Ruch, RJ; Pereira, MA. (1986). Carcinogenicity of chlorinated methane and ethane compounds administered in drinking water to mice. Environ Health Perspect 69: 89-95. http://dx.doi.org/10.1289/ehp.866989

Kozik, IV. (1957). [Problems of occupational hygiene in the use of dichloroethane in the aviation industry]. Gig Tr Prof Zabol 1: 31-38.

Kronevi, T; Wahlberg, JE; Holmberg, B. (1981). Skin pathology following epicutaneous exposure to seven organic solvents. Int J Tissue React 3: 21-30.

Kuhn, TK; Hamonts, K; Dijk, JA; Kalka, H; Stichler, W; Springael, D; Dejonghe, W; Meckenstock, RU. (2009). Assessment of the intrinsic bioremediation capacity of an eutrophic river sediment polluted by discharging chlorinated aliphatic hydrocarbons: a compound-specific isotope approach. Environ Sci Technol 43: 5263-5269. http://dx.doi.org/10.1021/es803600s

L, L; Albrechtsen, HJ; DB, R; Ekelund, F; Christensen, TH. (1999). Distribution and composition of microbial populations in a landfill leachate contaminated aquifer (Grindsted, Denmark). Microb Ecol 37: 197-207. http://dx.doi.org/10.1007/s002489900143

Labcorp Early Development. (2024). 1,1-Dichloroethane - Test Order: Rates of penetration through human skin using a flow through in vitro system. (8479195). Washington, DC: Stantec ChemRisk, Vinyl Institute 1,1-Dichloroethane Test Order Consortium.

Landmeyer, JE; Campbell, BG. (2014). Assessment of ethylene dibromide, dibromochloropropane, other volatile organic compounds, radium isotopes, radon, and inorganic compounds in groundwater and spring water from the Crouch Branch and McQueen Branch aquifers near McBee, South Carolina, 2010-2012.

Lane, RW; Riddle, BL; Borzelleca, JF. (1982). Effects of 1,2-dichloroethane and 1,1,1-trichloroethane in drinking water on reproduction and development in mice. Toxicol Appl Pharmacol 63: 409-421. http://dx.doi.org/10.1016/0041-008X(82)90270-8

Lebaron, MJ; Hotchkiss, JA; Zhang, F; Koehler, MW; Boverhof, DR. (2021). Investigation of potential early key events and mode of action for 1,2-dichloroethane-induced mammary tumors in female rats. J Appl Toxicol 41: 362-374. http://dx.doi.org/10.1002/jat.4048

Leblanc, GA. (1980). Acute toxicity of priority pollutants to water flea (Daphnia magna). Bull Environ Contam Toxicol 24: 684-691. http://dx.doi.org/10.1007/BF01608174

Li, M; Mathieu, J; Yang, Y; Fiorenza, S; Deng, Y; He, Z; Zhou, J; Alvarez, PJ. (2013). Widespread distribution of soluble di-iron monooxygenase (SDIMO) genes in Arctic groundwater impacted by 1,4-dioxane. Environ Sci Technol 47: 9950-9958. http://dx.doi.org/10.1021/es402228x

Li, M; Van Orden, ET; Devries, DJ; Xiong, Z; Hinchee, R; Alvarez, PJ. (2015a). Bench-scale biodegradation tests to assess natural attenuation potential of 1,4-dioxane at three sites in California. Biodegradation 26: 39-50. http://dx.doi.org/10.1007/s10532-014-9714-1

Li, W; Chen, L; Su, Y; Yin, H, ua; Pang, Y; Zhuang, Z. (2015b). 1,2-Dichloroethane induced nephrotoxicity through ROS mediated apoptosis in vitro and in vivo. Toxicology Research 4: 1389-1399. http://dx.doi.org/10.1039/c5tx00056d

Li, Y; Cakmak, S; Zhu, J. (2019). Profiles and monthly variations of selected volatile organic compounds in indoor air in Canadian homes: Results of Canadian national indoor air survey 2012-2013. Environ Int 126: 134-144. http://dx.doi.org/10.1016/j.envint.2019.02.035

Liang, B; Zhong, Y; Wang, B; Lin, L; Liu, J; Lin, X; Huang, Y; Hu, M; Zhang, B; Meng, H; Jiang, L; Jiang, J; Wu, J; Zhang, Y; Rong, W; Yang, X; Huang, Z. (2021). 1,2-Dichloroethane induces apoptosis in the cerebral cortexes of NIH Swiss mice through microRNA-182-5p targeting phospholipase D1 via a mitochondria-dependent pathway. Toxicol Appl Pharmacol 430: 15728-15728. http://dx.doi.org/10.1016/j.taap.2021.115728

Lindroth, RL; Batzli, GO. (1984). Food habits of the meadow vole (Microtus pennsylvanicus) in bluegrass and prairie habitats. Journal of Mammalogy 65: 600-606. http://dx.doi.org/10.2307/1380843

Lindstrom, AB; Proffitt, D; Fortune, CR. (1995). Effects of modified residential construction on indoor air quality. Indoor Air 5: 258-269. http://dx.doi.org/10.1111/j.1600-0668.1995.00005.x

Liss, PS; Slater, PG. (1974). Flux of gases across the air-sea interface. Nature 247: 181-184. http://dx.doi.org/10.1038/247181a0

Livesey, JC. (1982) Studies on the metabolism and toxicity of 1,2-dihaloethanes. (Doctoral Dissertation). University of Minnesota, Minneapolis, MN. Retrieved from https://login.libpdb.d.umn.edu:2443/login?url=

Lizarraga, LE; Dean, JL; Kaiser, JP; Wesselkamper, SC; Lambert, JC; Zhao, QJ. (2019). A case study on the application of an expert-driven read-across approach in support of quantitative risk assessment of p,p'-dichlorodiphenyldichloroethane. Regul Toxicol Pharmacol 103: 301-313. http://dx.doi.org/10.1016/j.yrtph.2019.02.010

Lizarraga, LE; Suter, GW; Lambert, JC; Patlewicz, G; Zhao, JQ; Dean, JL; Kaiser, P. (2023). Advancing the science of a read-across framework for evaluation of data-poor chemicals incorporating systematic and new approach methods. Regul Toxicol Pharmacol 137: 105293. http://dx.doi.org/10.1016/j.yrtph.2022.105293

Logue, JM; Small, MJ; Stern, D; Maranche, J; Robinson, AL. (2010). Spatial variation in ambient air toxics concentrations and health risks between industrial-influenced, urban, and rural sites. J Air Waste Manag Assoc 60: 271-286. http://dx.doi.org/10.3155/1047-3289.60.3.271

Lone, MI; Nazam, N; Hussain, A; Singh, SK; Dar, AH; Najar, RA; Al-Qahtani, MH; Ahmad, W. (2016). Genotoxicity and immunotoxic effects of 1,2-dichloroethane in Wistar rats. J Environ Sci Health C Environ Carcinog Ecotoxicol Rev 34: 169-186. http://dx.doi.org/10.1080/10590501.2016.1193924

Mackay, D; Di Guardo, A; Paterson, S; Kicsi, G; Cowan, CE. (1996). Assessing the fate of new and existing chemicals: A five-stage process. Environ Toxicol Chem 15: 1618-1626. http://dx.doi.org/10.1002/etc.5620150929

Maltoni, C; Valgimigli, L; Scarnato, C. (1980). Long-term carcinogenic bioassays on ethylene dichloride administered by inhalation to rats and mice. In B Ames; P Infante; R Reitz (Eds.), Ethylene dichloride: A potential health risk? (pp. 3-29). Cold Spring Harbor, NY: Cold Spring Harbor Laboratory.

Manufacturing Chemists Association. (1979). Third report on distribution and metabolism of 1,2-dichloroethane (EDC) in experimental animals with attachments and cover letter dated 041179 [TSCA Submission] (pp. 21). (OTS0516163. 86-870001582. TSCATS/400204). Institute of

Pharmacology. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0516163.xhtml
Martí, V; Jubany, I; Pérez, C; Rubio, X; De Pablo, J; Giménez, J. (2014). Human health risk assessment of a landfill based on volatile organic compounds emission, immission and soil gas concentration measurements. Appl Geochem 49: 218-224. http://dx.doi.org/10.1016/j.apgeochem.2014.06.018
Matsuoka, A; Hayashi, M; Sofuni, T. (1998). In vitro clastogenicity of 19 organic chemicals found in contaminated water and 7 structurally related chemicals. Environmental Mutagen Research 20: 159-165.
Mayer, FL, Jr; Ellersieck, MR. (1986). Manual of acute toxicity: Interpretation and data base for 410 chemicals and 66 species of freshwater animals. In USDI Fish and Wildlife Service, Publication No160, Washington, DC. (PB86239878). Columbia, MO: U.S. Department of the Interior, Fish and Wildlife Service. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/PB86239878.xhtml
McCall, SN; Jurgens, P; Ivanetich, KM. (1983). Hepatic microsomal metabolism of the dichloroethanes. Biochem Pharmacol 32: 207-213. http://dx.doi.org/10.1016/0006-2952(83)90545-2
Mcdermott, C; Heffron, JJA. (2013). Toxicity of Industrially Relevant Chlorinated Organic Solvents In Vitro. Int J Toxicol 32: 136-145. http://dx.doi.org/10.1177/1091581813482006
Mellon Institute. (1947). Repeated exposure of rats and dogs to vapors of eight chlorinated hydrocarbons [TSCA Submission]. (OTS0515559. 86-870001397. TSCATS/308690). Carbide and Carbon Chemicals Corporation. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0515559.xhtml
Mellon Institute. (1948). The toxicity of ethylene dichloride [TSCA Submission]. (Report 11-40. OTS0515565. 86-870001403). Union Carbide Corporation.
Mellor, CL; Marchese Robinson, RL; Benigni, R; Ebbrell, D; Enoch, SJ; Firman, JW; Madden, JC; Pawar, G; Yang, C; Cronin, MTD. (2019). Molecular fingerprint-derived similarity measures for toxicological read-across: Recommendations for optimal use. Regul Toxicol Pharmacol 101: 121-134. http://dx.doi.org/10.1016/j.yrtph.2018.11.002
Milman, HA; Story, DL; Riccio, ES; Sivak, A; Tu, AS; Williams, GM; Tong, C; Tyson, CA. (1988). Rat liver foci and in vitro assays to detect initiating and promoting effects of chlorinated ethanes and ethylenes. Ann N Y Acad Sci 534: 521-530. http://dx.doi.org/10.1111/j.1749-6632.1988.tb30143.x
Mitoma, C; Steeger, T; Jackson, SE; Wheeler, KP; Rogers, JH; Milman, HA. (1985). Metabolic disposition study of chlorinated hydrocarbons in rats and mice. Drug Chem Toxicol 8: 183-194. http://dx.doi.org/10.3109/01480548508999169
Mitsubishi Chemical Medience Corporation. (2009a). Acute immobilization test on Daphnia magna exposed to 1,1-dichloroethane (translation). (A090026). Tokyo, Japan: Ministry of the Environment (Japan).
Mitsubishi Chemical Medience Corporation. (2009b). Acute toxicity test on killifish (Oryzias latipes) exposed to 1,1-dichloroethane (translation). (A090025). Tokyo, Japan: Ministry of the Environment (Japan).
Mitsubishi Chemical Medience Corporation. (2009c). Algal growth inhibition test of Pseudokirchneriella subcapitata exposed to 1,1-dichloroethane (translation). (A090027). Tokyo, Japan: Ministry of the Environment (Japan).
Mitsubishi Chemical Medience Corporation. (2009d). Reproduction test on Daphnia magna exposed to 1,1-dichloroethane (translation). (A090028). Tokyo, Japan: Ministry of the Environment (Japan).
Mohr, TKG; DiGuiseppi, WH. (2010). Case Studies of 1,4-Dioxane Releases, Treatment, and Drinking Water Contamination. In T Mohr; B Diguiseppi; JK Anderson; JW Hatton (Eds.), Environmental investigation and remediation (2nd ed., pp. 369-420). Boca Raton, FL: CRC Press. http://dx.doi.org/10.1201/EBK1566706629-c8

Moody, DE; James, JL; Clawson, GA; Smuckler, EA. (1981). Correlations among the changes in hepatic microsomal components after intoxication with alkyl halides and other hepatotoxins. Mol Pharmacol 20: 685-693.
Morel, G; Ban, M; Hettich, D; Huguet, N. (1999). Role of SAM-dependent thiol methylation in the renal toxicity of several solvents in mice. J Appl Toxicol 19: 47-54. http://dx.doi.org/10.1002/(SICI)1099-1263(199901/02)19:1<47::AID-JAT536>3.0.CO;2-L
Morgan, DL; Bucher, JR; Elwell, MR; Lilja, HS; Murthy, AS. (1990). Comparative toxicity of ethylene dichloride in F344/N, Sprague-Dawley and Osborne-Mendel rats. Food Chem Toxicol 28: 839-845. http://dx.doi.org/10.1016/0278-6915(90)90057-T
Morgan, DL; Cooper, SW; Carlock, DL; Sykora, JJ; Sutton, B; Mattie, DR; McDougal, JN. (1991). Dermal absorption of neat and aqueous volatile organic chemicals in the Fischer 344 rat. Environ Res 55: 51-63. http://dx.doi.org/10.1016/S0013-9351(05)80140-9
Munson, AE; Sanders, VM; Douglas, KA; Sain, LE; Kauffmann, BM; White Jr., KL. (1982). In vivo assessment of immunotoxicity. Environ Health Perspect 43: 41-52. http://dx.doi.org/10.1289/ehp.824341
Muralidhara, S; Ramanathan, R; Mehta, SM; Lash, LH; Acosta, D; Bruckner, JV. (2001). Acute, subacute, and subchronic oral toxicity studies of 1,1-dichloroethane in rats: Application to risk evaluation. Toxicol Sci 64: 135-145. http://dx.doi.org/10.1093/toxsci/64.1.135
Nagano, K; Umeda, Y; Senoh, H; Gotoh, K; Arito, H; Yamamoto, S; Matsushima, T. (2006). Carcinogenicity and chronic toxicity in rats and mice exposed by inhalation to 1,2-dichloroethane for two years. J Occup Health 48: 424-436. http://dx.doi.org/10.1539/joh.48.424
Natsyuk, MV; Chekman, IS. (1975). Content of nicotinamide coenzymes in liver and myocardium of rats poisoned with dichloroethane. Bull Exp Biol Med 79: 408-409. http://dx.doi.org/10.1007/BF00832711
Natsyuk, MV; Fedurov, VV. (1974). Effect of methyluracil on oxidative phosphorylation in the hepatic mitochondria of rats poisoned with dichloroethane. Bull Exp Biol Med 77: 391-393.
NCBI. (2020a). PubChem Compound Summary for CID 6365: 1,1-Dichloroethane.
NCBI. (2020b). PubChem database: compound summary: 1,1-dichloroethane. Available online at https://pubchem.ncbi.nlm.nih.gov/compound/1%2C1-dichloroethane
NCEA. (1987). 1,2-Dichloroethane: IRIS summary. U.S. Environmental Protection Agency.
NCI. (1978). Bioassay of 1,1-dichloroethane for possible carcinogenicity (CAS No. 75-34-3) (pp. 1-107). (ISSN 0163-7185/NCI-CG-TR-66). Bethesda, MD. https://ntp.niehs.nih.gov/ntp/htdocs/lt_rpts/tr066.pdf
Neuhauser, EF; Durkin, PR; Malecki, MR; Anatra, M. (1986). Comparative toxicity of ten organic chemicals to four earthworm species. Comp Biochem Physiol C Comp Pharmacol Toxicol 83: 197-200. http://dx.doi.org/10.1016/0742-8413(86)90036-8
Neuhauser, EF; Loehr, RC; Malecki, MR; Milligan, DL; Durkin, PR. (1985). The toxicity of selected organic chemicals to the earthworm Eisenia fetida. J Environ Qual 14: 383-388. http://dx.doi.org/10.2134/jeq1985.00472425001400030015x
Niehoff, NM; Gammon, MD; Keil, AP; Nichols, HB; Engel, LS; Sandler, DP; White, AJ. (2019). Airborne mammary carcinogens and breast cancer risk in the Sister Study. Environ Int 130: 104897. http://dx.doi.org/10.1016/j.envint.2019.06.007
NIOSH. (2007). NIOSH pocket guide to chemical hazards. (DHHS Publication No. (NIOSH) 2005-149; CBRNIAC-CB-112149). Cincinnati, OH. http://www.cdc.gov/niosh/docs/2005-149/
NIOSH. (2018). NIOSH pocket guide to chemical hazards: 1,1-dichloroethane. Atlanta, GA: United States Department of Health and Human Services, Centers for Disease Control and Prevention, National Institute for Occupational Safety and Health. http://www.cdc.gov/niosh/npg/npgd0194.html

NITE. (1995a). J-Check data: Acute immobilization test of 1,2-dichloropropane to Daphnia magna. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=28173&mno=2-0081&cno=78-87-5&request_locale=en
NITE. (1995b). J-Check Data: Reproduction inhibition test of 1,2-dichloropropane to Daphnia magna. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=28174&mno=2-0081&cno=78-87-5&request_locale=en
NITE. (2023a). 1-Chlorobutane (CAS RN 109-69-3), Test #28: Biodegradation in water, screening tests. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=3196&mno=2-0060&cno=109-69-3&request_locale=en
NITE. (2023b). 1,2-Dichloroethane (CAS RN 107-06-2), Test #28: Biodegradation in water, screening tests. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=753&mno=2-0081&cno=78-87-5&request_locale=en
NITE. (2023c). 1,2-Dichloropropane (CAS RN 78-87-5), Test #28: Biodegradation in water, screening tests. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=753&mno=2-0081&cno=78-87-5&request_locale=en
NITE. (2023d). 1,2,3-Trichloropropane (CAS RN 96-18-4), Test #28: Biodegradation in water, screening tests. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=1816&mno=2-0083&cno=96-18-4&request_locale=en
NITE. (2023e). 1,4-Dichlorobutane (CAS RN 110-56-5), Test #28: Biodegradation in water, screening tests. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=3297&mno=2-0061&cno=110-56-5&request_locale=en
NITE. (2023f). 2-Chloropropane (CAS RN 75-29-6), Test #28: Biodegradation in water, screening tests. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=493&mno=2-0080&cno=75-29-6&request_locale=en
NITE. (2023g). Chloroethane (CAS RN 75-00-3), Test #28: Biodegradation in water, screening tests. Available online at https://www.nite.go.jp/chem/jcheck/template.action?ano=426&mno=2-0053&cno=75-00-3&request_locale=en
NLM. (2018). PubChem: Hazardous Substance Data Bank: 1,1-Dichloroethane, 75-34-3. Available online at https://pubchem.ncbi.nlm.nih.gov/compound/6365#source=HSDB
Nouchi, T; Miura, H; Kanayama, M; Mizuguchi, O; Takano, T. (1984). Fatal intoxication by 1,2-dichloroethane - a case report. Int Arch Occup Environ Health 54: 111-113. http://dx.doi.org/10.1007/BF00378513
NRCe. (2009). Query/download NRC FOIA data [Database]. Retrieved from http://www.nrc.uscg.mil/foia.html
NTP. (1978). Bioassay of 1,2-dichloroethane for possible carcinogenicity [NTP]. In National Cancer Institute carcinogenesis technical report series, no 55. (TR 55). Bethesda, Maryland: U.S. Department of Health and Human Services, National Institutes of Health, National Cancer Institute. https://ntp.niehs.nih.gov/sites/default/files/ntp/htdocs/lt_rpts/tr055.pdf?vvv
NTP. (1991). Toxicity studies of 1,2-dichloroethane (ethylene bichloride) (CAS No. 107-06-2) in F344/N rats, Sprague Dawley rats, Osborne-Mendel rats, and B6C3F1 mice (drinking water and gavage studies). (NTP TOX 4; NIH Publication No. 91-3123). Research Triangle Park, NC. https://ntp.niehs.nih.gov/publications/reports/tox/000s/tox004
NTP. (2021). Report on carcinogens, fifteenth edition. (ISSN 1551-8280). Research Triangle Park, NC: U.S. Department of Health and Human Services, Public Health Service. http://dx.doi.org/10.22427/NTP-OTHER-1003
NWQMC. (2022). Water quality portal: 1,4-Dioxane [Database]. Washington, DC. Retrieved from https://acwi.gov/monitoring/waterqualitydata.html

O'Neil, MJ. (2013). Ethylidene chloride. 75-34-3. [1,1-Dichloroethane]. In MJ O'Neill; PE Heckelman; PH Dobbelaar; KJ Roman; CM Kenney; LS Karaffa (Eds.), The Merck index: An encyclopedia of chemicals, drugs, and biologicals (15th ed., pp. 705). Cambridge, UK: The Royal Society of Chemistry.
OECD. (2009). Emission scenario document on transport and storage of chemicals. Paris, France. http://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=env/jm/mono(2009)26&doclanguage=en
OECD. (2015). Fundamental and guiding principles for (Q)SAR analysis of chemical carcinogens with mechanistic considerations. (ENV/JM/MONO(2015)46). Paris, France. http://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=env/jm/mono(2015)46&doclanguage=en
OSHA. (1997). Occupational exposure to methylene chloride. Fed Reg 62: 1493-1619.
OSHA. (2019). Permissible exposure limits: OSHA annotated table Z-2. United States Department of Labor, Occupational Safety & Health Administration. https://www.osha.gov/dsg/annotated-pels/tablez-2.html
Ott, MG; Teta, J; Greenberg, HL. (1989). Lymphatic and hematopoietic tissue cancer in a chemical manufacturing environment. Am J Ind Med 16: 631-644. http://dx.doi.org/10.1002/ajim.4700160603
Pang, Y; Qi, G; Jiang, S; Zhou, Y; Li, W. (2018). 1,2-Dichloroethane induced hepatotoxicity and apoptosis by inhibition of ERK 1/2 pathways. Can J Physiol Pharmacol 96: 1119-1126. http://dx.doi.org/10.1139/cjpp-2017-0677
Paolini, M; Mesirca, R; Pozzetti, L; Sapone, A; Biagi, GL; Trieff, NM; Cantelli-Forti, G. (1994). Correlation between murine liver cytochrome P450 2B1 induction by halogenated hydrocarbons and toxicity. Toxicol Environ Chem 44: 55-64. http://dx.doi.org/10.1080/02772249409358043
Patlolla, BP; Patlolla, AK; Tchounwou, PB. (2005). Cytogenetic effects of 1,1-dichloroethane in mice bone marrow cells. Int J Environ Res Public Health 2: 101-106. http://dx.doi.org/10.3390/ijerph2005010101
Payan, JP; Beydon, D; Fabry, JP; Brondeau, MT; Ban, M; de Ceaurritz, J. (1993). Urinary thiodiglycolic acid and thioether excretion in male rats dosed with 1,2-dichloroethane. J Appl Toxicol 13: 417-422. http://dx.doi.org/10.1002/jat.2550130608
Payan, JP; Saillenfait, AM; Bonnet, P; Fabry, JP; Langonne, I; Sabate, JP. (1995). Assessment of the developmental toxicity and placental transfer of 1,2-dichloroethane in rats. Toxicol Sci 28: 187-198. http://dx.doi.org/10.1006/faat.1995.1159
Pestana, CB; Firman, JW; Cronin, MTD. (2021). Incorporating lines of evidence from New Approach Methodologies (NAMs) to reduce uncertainties in a category based read-across: A case study for repeated dose toxicity. Regul Toxicol Pharmacol 120: 104855. http://dx.doi.org/10.1016/j.yrtph.2020.104855
Plaa, GL; Larson, RE. (1965). Relative nephrotoxic properties of chlorinated methane, ethane, and ethylene derivatives in mice. Toxicol Appl Pharmacol 7: 37-44. http://dx.doi.org/10.1016/0041-008X(65)90072-4
Pontolilloand, J; Eganhouse, RP. (2001). Search for Reliable Aqueous Solubility (Sw) and Octanol-Water Partition Coefficient (Kow) Data for Hydrophobic Organic Compounds: DDT and DDE as a Case Study. (NTIS/02935761_a). Pontolilloand, J; Eganhouse, RP.
Poole, SK; Poole, CF. (1999). Chromatographic models for the sorption of neutral organic compounds by soil from water and air. J Chromatogr A 845: 381-400. http://dx.doi.org/10.1016/S0021-9673(98)01085-1
Poulsen, MM; Kueper, BH. (1992). A FIELD EXPERIMENT TO STUDY THE BEHAVIOR OF TETRACHLOROETHYLENE IN UNSATURATED POROUS-MEDIA. Environ Sci Technol 26: 889-895.

Prodi, G; Arfellini, G; Colacci, A; Grilli, S; Mazzullo, M. (1986). Interaction of halocompounds with nucleic acids. Toxicol Pathol 14: 438-444. http://dx.doi.org/10.1177/019262338601400409
Prodi, G; Colacci, A; Grilli, S; Lattanzi, G; Mazzullo, M; Turina, P. (1988). Comparison of the covalent binding of various chloroethanes with nucleic acids. In F Feo; P Pani; A Columbano; R Garcea (Eds.), Chemical carcinogenesis (pp. 93-102). Boston, MA: Springer. http://dx.doi.org/10.1007/978-1-4757-9640-7_10
Qin-li, Z; Qiao, N; Lai-yu, L; Li-jun, Y; Xiao-li, G; Jian-xun, H; Lin-ping, W; You-xin, L. (2010). Toxic encephalopathy induced by occupational exposure to 1,2-dichloroethane and toxicological effect on animal model. In Proceedings of the 5th International Academic Conference on Environmental and Occupational Medicine. Shanghai, China: Journal of Environmental & Occupational Medicine.
Que, SSH; Igwe, OJ; Boyle, JR. (1988). Elemental alterations during the exposure of 1,2-dichloroethane (EDC), disulfiram (DSF), and EDC-DSF to male Sprague-Dawley rats. Biol Trace Elem Res 18: 9-28. http://dx.doi.org/10.1007/BF02917485
R Core Team. (2022). R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.r-project.org/
Raimondo, S, ., D.,.N. Vivian, and M.G. Barron. (2010). Web-Based Interspecies Correlation Estimation (Web-ICE) for Acute Toxicity: User Manual Version 3.1. (600R10004). Raimondo, S., D.N. Vivian, and M.G. Barron. http://nepis.epa.gov/exe/ZyPURL.cgi?Dockey=P10068ND.txt
Raimondo, S; Lilavois, C; Nelson, SA. (2023). Uncertainty analysis and updated user guidance for interspecies correlation estimation models and low toxicity compounds. Integr Environ Assess Manag. http://dx.doi.org/10.1002/ieam.4884
Rao, KS; Murray, JS; Deacon, MM; John, JA; Calhoun, LL; Young, JT. (1980). Teratogenicity and reproduction studies in animals inhaling ethylene dichloride. In B Ames; P Infante; R Reitz (Eds.), Banbury report: Ethylene dichloride: A potential health risk (pp. P149-P166). Cold Spring Harbor, NY: Cold Spring Harbor Laboratory.
Ras-Mallorqui, MR; Marce-Recasens, RM; Borrull-Ballarin, F. (2007). Determination of volatile organic compounds in urban and industrial air from Tarragona by thermal desorption and gas chromatography-mass spectrometry. Talanta 72: 941-950. http://dx.doi.org/10.1016/j.talanta.2006.12.025
Reid, JB; Muianga, CV. (2012). Saturated Halogenated Aliphatic Hydrocarbons Two to Four Carbons. In Patty's Toxicology. http://dx.doi.org/10.1002/0471435139.tox063.pub2
Reitz, RH; Fox, TR; Domoradzki, JY; Quast, JF; Langvardt, P; Watanabe, PG. (1980). Pharmacokinetics and macromolecular interactions of ethylene dichloride: Comparison of oral and inhalation exposures. In B Ames; P Infante; R Reitz (Eds.), Ethylene dichloride: A potential health risk? (pp. 135-148). Cold Spring Harbor Laboratory: Cold Spring Harbor, NY.
Reitz, RH; Fox, TR; Ramsey, JC; Quast, JF; Langvardt, PW; Watanabe, PG. (1982). Pharmacokinetics and macromolecular interactions of ethylene dichloride in rats after inhalation or gavage. Toxicol Appl Pharmacol 62: 190-204. http://dx.doi.org/10.1016/0041-008X(82)90117-X
Richter, JE; Peterson, SF; Kleiner, CF. (1983). Acute and chronic toxicity of some chlorinated benzenes, chlorinated ethanes, and tetrachloroethylene to Daphnia magna. Arch Environ Contam Toxicol 12: 679-684. http://dx.doi.org/10.1007/BF01060751
Ritz, C; Baty, F; Streibig, JC; Gerhard, D. (2015). Dose-Response Analysis Using R. PLoS ONE 10: e0146021. http://dx.doi.org/10.1371/journal.pone.0146021
RIVM. (2007). Ecotoxicologically based environmental risk limits for several volatile aliphatic hydrocarbons (pp. 217). (601782002/2007). Bilthoven, Netherlands: National Institute for Public Health and the Environment (RIVM). https://www.rivm.nl/bibliotheek/rapporten/601782002.pdf

Romert, L; Magnusson, J; Ramel, C. (1990). The importance of glutathione and glutathione transferase for somatic mutations in Drosophila melanogaster induced in vivo by 1,2-dichloroethane. Carcinogenesis 11: 1399-1402. http://dx.doi.org/10.1093/carcin/11.8.1399
Roose, P; Brinkman, UA. (1998). Determination of volatile organic compounds in marine biota. J Chromatogr A 799: 233-248. http://dx.doi.org/10.1016/S0021-9673(97)01081-9
Rosenberg, R; Grahn, O; Johansson, L. (1975). Toxic effects of aliphatic chlorinated by-products from vinyl chloride production on marine animals. Water Res 9: 607-612. http://dx.doi.org/10.1016/0043-1354(75)90164-5
Rosner, B. (1983). Percentage points for a generalized esd many-outlier procedure. Technometrics 25: 165-172. http://dx.doi.org/10.1080/00401706.1983.10487848
Roy F. Weston Inc. (1986). Installation restoration program phase ii-confirmation/quantification, stage 1. Final report for Burlington Air national guard base, Burlington, Vermont. (F33615-80-D-4006). https://search.proquest.com/docview/19070510?accountid=171501
Rumble, JR. (2018a). 1,1-Dichloroethane. In CRC handbook of chemistry and physics (99 ed.). Boca Raton, FL: CRC Press. Taylor & Francis Group.
Rumble, JR. (2018b). Flammability of chemical substances. In CRC Handbook of Chemistry and Physics (99 ed.). Boca Raton, FL: CRC Press. Taylor & Francis Group.
Rumble, JR. (2018c). Viscosity of liquids. In CRC Handbook of Chemistry and Physics (99 ed.). Boca Raton, FL: CRC Press. Taylor & Francis Group. https://hbcp.chemnetbase.com/faces/documents/06_37/06_37_0001.xhtml
SAB. (2007). Science Advisory Board (SAB) review of the Estimation Programs Interface Suite (EPI Suite). (EPA-SAB-07-11). Washington, DC: U.S. Environmental Protection Agency.
Sabel, GV; Clark, TP. (1984). Volatile organic compounds as indicators of municipal solid waste leachate contamination. Waste Manag Res 2: 119-130. http://dx.doi.org/10.1016/0734-242X(84)90135-6
Salmon, AG; Jones, RB; Mackrodt, WC. (1981). Microsomal dechlorination of chloroethanes: Structure-reactivity relationships. Xenobiotica 11: 723-734. http://dx.doi.org/10.3109/00498258109045876
Salovsky, P; Shopova, V; Dancheva, V; Yordanov, Y; Marinov, E. (2002). Early pneumotoxic effects after oral administration of 1,2-dichloroethane. J Occup Environ Med 44: 475-480. http://dx.doi.org/10.1097/00043764-200205000-00016
Sasaki, YF; Saga, A; Akasaka, M; Ishibashi, S; Yoshida, K; Su, YQ; Matsusaka, N; Tsuda, S. (1998). Detection in vivo genotoxicity of haloalkanes and haloalkenes carcinogenic to rodents by the alkaline single cell gel electrophoresis (comet) assay in multiple mouse organs. Mutat Res 419: 13-20. http://dx.doi.org/10.1016/S1383-5718(98)00114-4
Sato, A; Nakajima, T; Koyama, Y. (1983). Interaction between ethanol and carbohydrate on the metabolism in rat liver of aromatic and chlorinated hydrocarbons. Toxicol Appl Pharmacol 68: 242-249. http://dx.doi.org/10.1016/0041-008X(83)90008-X
Schäfer, H; Hettler, H; Fritsche, U; Pitzen, G; Röderer, G; Wenzel, A. (1994). Biotests using unicellular algae and ciliates for predicting long-term effects of toxicants. Ecotoxicol Environ Saf 27: 64-81. http://dx.doi.org/10.1006/eesa.1994.1007
Schenk, L; Rauma, M; Fransson, MN; Johanson, G. (2018). Percutaneous absorption of thirty-eight organic solvents in vitro using pig skin. PLoS ONE 13: e0205458. http://dx.doi.org/10.1371/journal.pone.0205458
Scheutz, C; Durant, ND; Hansen, MH; Bjerg, PL. (2011). Natural and enhanced anaerobic degradation of 1,1,1-trichloroethane and its degradation products in the subsurface--a critical review [Review]. Water Res 45: 2701-2723. http://dx.doi.org/10.1016/j.watres.2011.02.027
Schrab, GE; Brown, KW; Donnelly, KC. (1993). Acute and genetic toxicity of municipal landfill leachate. Water Air Soil Pollut 69: 99-112. http://dx.doi.org/10.1007/BF00478351

Schwetz, BA; Leong, BKJ; Gehring, PJ. (1974). Embryo- and fetotoxicity of inhaled carbon tetrachloride, 1,1-dichloroethane and methyl ethyl ketone in rats. Toxicol Appl Pharmacol 28: 452-464. http://dx.doi.org/10.1016/0041-008X(74)90230-0

Sharpe, A; Carter, D. (1993). Substrate specificity of rat liver aldehyde dehydrogenase with chloroacetaldehydes. J Biochem Toxicol 8: 155-160. http://dx.doi.org/10.1002/jbt.2570080307

Sherwood, RL; O'Shea, W; Thomas, PT; Ratajczak, HV; Aranyi, C; Graham, JA. (1987). Effects of inhalation of ethylene dichloride on pulmonary defenses of mice and rats. Toxicol Appl Pharmacol 91: 491-496. http://dx.doi.org/10.1016/0041-008X(87)90071-8

Sigma-Aldrich. (2020). 1,1-Dichloroethane analytical standard. Sigma-Aldrich. https://www.sigmaaldrich.com/catalog/product/supelco/48512?lang=en&region=US

Simmon, VF; Kauhanen, K; Tardiff, RG. (1977). Mutagenic activity of chemicals identified in drinking water. In D Scott; B Bridges; F Sobel (Eds.), Progress in genetic toxicology: Proceedings of the Second International Conference on Environmental Mutagens (pp. 249-258). New York, NY: Elsevier/North Holland Press.

Şimşir, B; Yan, J; Im, J; Graves, D; Löffler, FE. (2017). Natural Attenuation in Streambed Sediment Receiving Chlorinated Solvents from Underlying Fracture Networks. Environ Sci Technol 51: 4821-4830. http://dx.doi.org/10.1021/acs.est.6b05554

Smith, AD; Bharath, A; Mallard, C; Orr, D; Smith, K; Sutton, JA; Vukmanich, J; Mccarty, LS; Ozburn, GW. (1991). The acute and chronic toxicity of ten chlorinated organic compounds to the American flagfish (Jordanella floridae). Arch Environ Contam Toxicol 20: 94-102. http://dx.doi.org/10.1007/BF01065334

Smithers. (2023). 1,1,2-Trichloroethane - Sediment-water chironomid (Chironomus riparius) life-cycle toxicity test using spiked sediment, following OECD Guideline 233. (Smithers Study No. 14331.6105). Pittsburgh, PA: Stantec ChemRisk.

Smithers. (2024a). [14C]1,2-Dichloropropane – Acute toxicity to midges (Chironomus riparius) under static conditions. (Smithers Study No. 14339.6107). Washington, DC: PDC Consortium.

Smithers. (2024b). Acute toxicity to midges (Chironomus riparius) under static-renewal conditions. (Smithers Study No. 14331.6102). Pittsburgh, PA: Stantec ChemRisk, The Vinyl Institute 1,1-Dichloroethane Consortia.

Sobel, W; Bond, GG; Skowronski, BJ; Brownson, PJ; Cook, RR. (1987). A soft tissue sarcoma case control study in a large multi-chemical manufacturing facility. Chemosphere 16: 2095-2099. http://dx.doi.org/10.1016/0045-6535(87)90214-1

Spencer, HC; Rowe, VK; Adams, EM; McCollister, DD; Irish, DD. (1951). Vapor toxicity of ethylene dichloride determined by experiments on laboratory animals. Arch Ind Hyg Occup Med 4: 482-493.

Spreafico, F; Zuccato, E; Marcucci, F; Sironi, M; Paglialunga, S; Madonna, M; Mussini, E. (1980). Pharmacokinetics of ethylene dichloride in rats treated by different routes and its long-term inhalatory toxicity. In B Ames; P Infante; R Reitz (Eds.), Ethylene dichloride: A potential health risk? (Banbury Report 5 ed., pp. 107-133). Cold Spring Harbor, NY: Cold Spring Harbor Laboratory.

Stantec ChemRisk. (2023). 1,1-Dichloroethane - Test Order - Final study report: Inhalation monitoring of 1,1-dichloroethane (CASRN 75-34-3). Washington, DC: Vinyl Institute Consortium.

Stauffer Chem Co. (1973). Acute oral toxicity and eye and skin irritation properties of ethylene dichloride [TSCA Submission]. In Toxicological studies of 1,2-dichloroethane with attachments and cover letter dated 072387. (Toxicology Labaoratory Report T-4408. OTS0515133. 86-870000606).

Storer, RD; Cartwright, ME; Cook, WO; Soper, KA; Nichols, WW. (1995). Short-term carcinogenesis bioassay of genotoxic procarcinogens in PIM transgenic mice. Carcinogenesis 16: 285-293. http://dx.doi.org/10.1093/carcin/16.2.285

Storer, RD; Conolly, RB. (1983). Comparative in vivo genotoxicity and acute hepatotoxicity of three 1,2-dihaloethanes. Carcinogenesis 4: 1491-1494. http://dx.doi.org/10.1093/carcin/4.11.1491
Storer, RD; Conolly, RB. (1985). An investigation of the role of microsomal oxidative metabolism in the in vivo genotoxicity of 1,2-dichloroethane. Toxicol Appl Pharmacol 77: 36-46. http://dx.doi.org/10.1016/0041-008X(85)90265-0
Storer, RD; Jackson, NM; Conolly, RB. (1984). In vivo genotoxicity and acute hepatotoxicity of 1,2-dichloroethane in mice: Comparison of oral, intraperitoneal, and inhalation routes of exposure. Cancer Res 44: 4267-4271.
Story, DL; Meierhenry, EF; Tyson, CA; Milman, HA. (1986). Differences in rat liver enzyme-altered foci produced by chlorinated aliphatics and phenobarbital [Review]. Toxicol Ind Health 2: 351-362. http://dx.doi.org/10.1177/074823378600200402
Suarez, MP; Rifai, HS. (1999). Biodegradation rates for fuel hydrocarbons and chlorinated solvents in groundwater. Bioremediat J 3: 337-362. http://dx.doi.org/10.1080/10889869991219433
Suguro, M; Numano, T; Kawabe, M; Doi, Y; Imai, N; Mera, Y; Tamano, S. (2017). Lung tumor induction by 26-week dermal application of 1,2-dichloroethane in CB6F1-Tg rasH2 mice. Toxicol Pathol 45: 427-434. http://dx.doi.org/10.1177/0192623317701003
Sun, BL; Griffin, BM; Ayala-Del-Rio, HL; Hashsham, SA; Tiedje, JM. (2002). Microbial dehalorespiration with 1,1,1-trichloroethane. Science 298: 1023-1025. http://dx.doi.org/10.1126/science.1074675
Sun, Q; Liao, Y; Wang, T; Tang, H; Wang, G; Zhao, F; Jin, Y. (2016a). 2-Chloroethanol Induced Upregulation of Matrix Metalloproteinase-2 in Primary Cultured Rat Astrocytes Via MAPK Signal Pathways. Frontiers in Neuroscience 10: 593. http://dx.doi.org/10.3389/fnins.2016.00593
Sun, Q; Liao, Y; Wang, T; Wang, G; Zhao, F; Jin, Y. (2016b). Alteration in mitochondrial function and glutamate metabolism affected by 2-chloroethanol in primary cultured astrocytes. Toxicol In Vitro 37: 50-60. http://dx.doi.org/10.1016/j.tiv.2016.09.005
Sun, Q; Wang, G; Gao, L; Shi, L; Qi, Y; Lv, X; Jin, Y. (2016c). Roles of CYP2e1 in 1,2-dichloroethane-induced liver damage in mice. Environ Toxicol 31: 1430-1438. http://dx.doi.org/10.1002/tox.22148
Suzuki, T; Nezu, K; Sasaki, H; Miyazawa, T; Isono, H. (1994). Cytotoxicity of chlorinated hydrocarbons and lipid peroxidation in isolated rat hepatocytes. Biol Pharm Bull 17: 82-86. http://dx.doi.org/10.1248/bpb.17.82
Sweeney, LM; Gargas, ML. (2016). Route-to-route extrapolation of 1,2-dichloroethane studies from the oral route to inhalation using physiologically based pharmacokinetic models. Regul Toxicol Pharmacol 81: 468-479. http://dx.doi.org/10.1016/j.yrtph.2016.10.005
Sweeney, LM; Saghir, SA; Gargas, ML. (2008). Physiologically based pharmacokinetic model development and simulations for ethylene dichloride (1,2-dichloroethane) in rats. Regul Toxicol Pharmacol 51: 311-323. http://dx.doi.org/10.1016/j.yrtph.2008.05.002
Sweeny, LM; Gargas, ML. (2006). Physiologically based pharmacokinetic model development and simulations for ethylene dichloride (1,2-Dichloroethane) in rats [with cover letter dated 07/07/2006]. Millwood, Virginia: HAP Task Force.
Tabak, HH; Quave, SA; Mashni, CI; Barth, EF. (1981). Biodegradability studies with organic priority pollutant compounds. J Water Pollut Control Fed 53: 1503-1518.
Tafazoli, M; Baeten, A; Geerlings, P; Kirsch-Volders, M. (1998). In vitro mutagenicity and genotoxicity study of a number of short-chain chlorinated hydrocarbons using the micronucleus test and the alkaline single cell gel electrophoresis technique (Comet assay) in human lymphocytes: a structure-activity relationship (QSAR) analysis of the genotoxic and cytotoxic potential. Mutagenesis 13: 115-126. http://dx.doi.org/10.1093/mutage/13.2.115
Take, M; Takanobu, K; Takeuchi, T; Haresaku, M; Matsumoto, M; Nagano, K; Yamamoto, S; Fukushima, S. (2013). Distribution of blood and tissue concentrations in rats by inhalation

exposure to 1,2-dichloroethane. J Environ Sci Health A Tox Hazard Subst Environ Eng 48: 1031-1036. http://dx.doi.org/10.1080/10934529.2013.773765

Tan, EL; Hsie, AW. (1981). Mutagenicity and cytotoxicity of haloethanes as studied in the CHO/HGPRT system. Mutat Res 90: 183-191. http://dx.doi.org/10.1016/0165-1218(81)90081-1

Taningher, M; Parodi, S; Grilli, S; Colacci, A; Mazzullo, M; Bordone, R; Santi, L. (1991). Lack of correlation between alkaline DNA fragmentation and DNA covalent binding induced by polychloroethanes after in vivo administration. Problems related to the assessment of a carcinogenic hazard. Cancer Detect Prev 15: 35-39.

Teta, MJ; Ott, MG; Schnatter, AR. (1991). An update of mortality due to brain neoplasms and other causes among employees of a petrochemical facility. J Occup Med 33: 45-51. http://dx.doi.org/10.1097/00043764-199101000-00013

Thomas, L; Defeo, B; Mariani, MF; van Rossum, GD. (1989). Comparison of metabolic effects of carbon tetrachloride and 1,2-dichloroethane added in vitro to slices of rat liver. Toxicol In Vitro 3: 59-68. http://dx.doi.org/10.1016/0887-2333(89)90025-8

Tomasi, A; Albano, E; Bini, A; Botti, B; Slater, TF; Vannini, V. (1984). Free radical intermediates under hypoxic conditions in the metabolism of halogenated carcinogens. Toxicol Pathol 12: 240-246. http://dx.doi.org/10.1177/019262338401200306

Tsuruta, H. (1975). Percutaneous absorption of organic solvents: 1) comparative study of the in vivo percutaneous absorption of chlorinated solvents in mice. Ind Health 13: 227-236. http://dx.doi.org/10.2486/indhealth.13.227

Tu, AS; Murray, TA; Hatch, KM; Sivak, A; Milman, HA. (1985). In vitro transformation of BALB/c-3T3 cells by chlorinated ethanes and ethylenes. Cancer Lett 28: 85-92. http://dx.doi.org/10.1016/0304-3835(85)90096-5

U.S. BLS. (2023). May 2023 Occupational Employment and Wage Estimates: National industry-specific estimates. Washington, DC. https://www.bls.gov/oes/tables.htm

U.S. Census Bureau. (2017). 2017 SUSB Annual Data Tables by Establishment Industry: U.S. & states, 6-digit NAICS. Suitland-Silver Hill, MD. https://www.census.gov/data/tables/2017/econ/susb/2017-susb-annual.html

U.S. EPA. (1982). Fate of priority pollutants in publicly owned treatment works, Volume i. (EPA 440/1-82/303). Washington, DC: Effluent Guidelines Division. http://nepis.epa.gov/exe/ZyPURL.cgi?Dockey=000012HL.txt

U.S. EPA. (1987). Integrated Risk Information System (IRIS) chemical assessment summary: 1,2-dichloroethane; CASRN: 107-06-2. Washington, DC: U.S. Environmental Protection Agency, National Center for Environmental Assessment. https://cfpub.epa.gov/ncea/iris/iris_documents/documents/subst/0149_summary.pdf

U.S. EPA. (1988). National survey of solid waste (municipal) landfill facilities [EPA Report]. (EPA/530-SW88-034). Washington, DC: Office of Solid Waste and Emergency Response, U.S. Environmental Protection Agency.

U.S. EPA. (1990). Integrated Risk Information System (IRIS) chemical assessment summary: 1,1-dichloroethane; CASRN 75-34-3. Washington, DC: U.S. Environmental Protection Agency, National Center for Environmental Assessment. https://cfpub.epa.gov/ncea/iris/iris_documents/documents/subst/0409_summary.pdf

U.S. EPA. (1992). A laboratory method to determine the retention of liquids on the surface of hands [EPA Report]. (EPA/747/R-92/003). Washington, DC.

U.S. EPA. (1993a). Reference Dose (RfD): description and use in health risk assessments background document 1A, March 15, 1993. Washington, DC: U.S. Environmental Protection Agency, Integrated Risk Information System. https://www.epa.gov/iris/reference-dose-rfd-description-and-use-health-risk-assessments

U.S. EPA. (1993b). Wildlife exposure factors handbook [EPA Report]. (EPA/600/R-93/187). Washington, DC: Office of Research and Development. http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=2799
U.S. EPA. (1994). Methods for derivation of inhalation reference concentrations and application of inhalation dosimetry [EPA Report]. (EPA600890066F). Research Triangle Park, NC. https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=71993&CFID=51174829&CFTOKEN=25006317
U.S. EPA. (1996). Guidelines for reproductive toxicity risk assessment [EPA Report]. (EPA/630/R-96/009). Washington, DC: U.S. Environmental Protection Agency, Risk Assessment Forum. https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=30004YQB.txt
U.S. EPA. (1998). Guidelines for ecological risk assessment [EPA Report]. (EPA/630/R-95/002F). Washington, DC: U.S. Environmental Protection Agency, Risk Assessment Forum. https://www.epa.gov/risk/guidelines-ecological-risk-assessment
U.S. EPA. (1999). Category for persistent, bioacculative, and toxic new chemical substances. In US Environmental Protection Agency (pp. 60194-60204). (ISSN 0097-6326/EISSN 2167-2520/213). Federal Register. https://www.gpo.gov/fdsys/pkg/FR-1999-11-04/pdf/99-28888.pdf
U.S. EPA. (2000a). Letter from vulcan chemicals to usepa submitting comments concerning 1,1-dichloroethane and 1,1,2,2-tetrachloroethane as well as the proposed 14-day subacute oral testing procotol. (EPA/OTS; Doc #40-90106032).
U.S. EPA. (2000b). Supplementary guidance for conducting health risk assessment of chemical mixtures (pp. 1-209). (EPA/630/R-00/002). Washington, DC: U.S. Environmental Protection Agency, Risk Assessment Forum. http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=20533
U.S. EPA. (2002a). Guidance on cumulative risk assessment of pesticide chemicals that have a common mechanism of toxicity [EPA Report]. Washington, D.C.
U.S. EPA. (2002b). A review of the reference dose and reference concentration processes [EPA Report]. (EPA630P02002F). Washington, DC. https://www.epa.gov/sites/production/files/2014-12/documents/rfd-final.pdf
U.S. EPA. (2003a). Attachment 1-3 Guidance for Developing Ecological Soil Screening Levels (Eco-SSLs): Evaluation of Dermal Contact and Inhalation Exposure Pathways for the Purpose of Setting Eco-SSLs. (OSWER9285755E). Washington, DC: .S. Environmental Protection Agency, Office of Solid Waste and Emergency Response. https://www.epa.gov/sites/production/files/2015-09/documents/ecossl_attachment_1-3.pdf
U.S. EPA. (2003b). Attachment 1-4. Guidance for developing ecological soil screening levels (Eco-SSLs): Review of background concentration for metals. (OSWER Directive 92857-55). Washington, DC. https://www.epa.gov/sites/default/files/2015-09/documents/ecossl_attachment_1-4.pdf
U.S. EPA. (2003c). Framework for cumulative risk assessment [EPA Report]. (EPA/630/P-02/001F). Washington, DC. https://www.epa.gov/sites/production/files/2014-11/documents/frmwrk_cum_risk_assmnt.pdf
U.S. EPA. (2003d). Methodology for deriving ambient water quality criteria for the protection of human health (2000), technical support document. Volume 2: Development of national bioaccumulation factors [EPA Report]. (EPA/822/R-03/030). Washington, DC. http://www.epa.gov/waterscience/criteria/humanhealth/method/tsdvol2.pdf
U.S. EPA. (2005a). Guidance for developing ecological soil screening levels [EPA Report]. (OSWER Directive 92857-55). Washington, DC: U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response. http://www.epa.gov/chemical-research/guidance-developing-ecological-soil-screening-levels

U.S. EPA. (2005b). Guidelines for carcinogen risk assessment [EPA Report]. (EPA630P03001F). Washington, DC. https://www.epa.gov/sites/production/files/2013-09/documents/cancer_guidelines_final_3-25-05.pdf
U.S. EPA. (2006a). 2006 community water system survey - Volume I: Overview [EPA Report]. (EPA 815-R-09-001). Washington, DC: U.S. Environmental Protection Agency, Office of Water. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1009JJI.txt
U.S. EPA. (2006b). Provisional peer-review toxicity values for 1,1-dichloroethane (CASRN 75-34-3). Cincinnati, OH: U.S. Environmental Protection Agency, National Center for Environmental Assessment, Superfund Health Risk Technical Support Center. https://hhpprtv.ornl.gov/issue_papers/Dichloroethane11.pdf
U.S. EPA. (2007). Attachment 4-3 Guidance for Developing Ecological Soil Screening Levels (Eco-SSLs) Eco-SSL Standard Operating Procedure (SOP) #4: Wildlife Toxicity Reference Value Literature Review, Data Extraction and Coding. (OSWER9285755F). http://nepis.epa.gov/exe/ZyPURL.cgi?Dockey=P100CDHC.txt
U.S. EPA. (2010). Provisional peer-reviewed toxicity values for dichloroethane, 1,2. (EPA/690/R-10/011F). Washington, DC. https://cfpub.epa.gov/ncea/pprtv/documents/Dichloroethane12.pdf
U.S. EPA. (2011a). Exposure factors handbook: 2011 edition [EPA Report]. (EPA/600/R-090/052F). Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment. https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P100F2OS.txt
U.S. EPA. (2011b). Recommended use of body weight 3/4 as the default method in derivation of the oral reference dose. (EPA100R110001). Washington, DC. https://www.epa.gov/sites/production/files/2013-09/documents/recommended-use-of-bw34.pdf
U.S. EPA. (2012a). Advances in inhalation gas dosimetry for derivation of a reference concentration (RfC) and use in risk assessment (pp. 1-140). (EPA/600/R-12/044). Washington, DC. https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=244650&CFID=50524762&CFTOKEN=17139189
U.S. EPA. (2012b). Benchmark dose technical guidance [EPA Report]. (EPA100R12001). Washington, DC: U.S. Environmental Protection Agency, Risk Assessment Forum. https://www.epa.gov/risk/benchmark-dose-technical-guidance
U.S. EPA. (2012c). Estimation Programs Interface Suite™ for Microsoft® Windows, v 4.11 [Computer Program]. Washington, DC. https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface
U.S. EPA. (2013a). Ground water issue paper: Synthesis report on state of understanding of chlorinated solvent transformation. (EPA/600/R-13/237). Washington, DC. https://nepis.epa.gov/Exe/ZyPDF.cgi/P100JDPP.PDF?Dockey=P100JDPP.PDF
U.S. EPA. (2013b). Updating CEB's method for screening-level estimates of dermal exposure. Chemical Engineering Branch.
U.S. EPA. (2014a). Exposure and Fate Assessment Screening Tool Version 2014 (E-FAST 2014). Washington, DC: Office of Pollution Prevention and Toxics. https://www.epa.gov/tsca-screening-tools/e-fast-exposure-and-fate-assessment-screening-tool-version-2014
U.S. EPA. (2014b). Flame retardant alternatives for hexabromocyclododecane (HBCD) [EPA Report]. (EPA/740/R-14/001). Washington, D.C. http://www2.epa.gov/saferchoice/partnership-evaluate-flame-retardant-alternatives-hbcd-publications
U.S. EPA. (2014c). Framework for human health risk assessment to inform decision making. Final [EPA Report]. (EPA/100/R-14/001). Washington, DC: U.S. Environmental Protection, Risk Assessment Forum. https://www.epa.gov/risk/framework-human-health-risk-assessment-inform-decision-making

U.S. EPA. (2015). ChemSTEER user guide - Chemical screening tool for exposures and environmental releases. Washington, D.C. https://www.epa.gov/sites/production/files/2015-05/documents/user_guide.pdf
U.S. EPA. (2016a). Ecological effects test guidelines: OCSPP 850.1075: Freshwater and saltwater fish acute toxicity test [EPA Report]. (EPA 712-C-16-007; EPA-HQ-OPPT-2009-0154-0035). Washington, DC: U.S. Environmental Protection Agency, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/document/EPA-HQ-OPPT-2009-0154-0035
U.S. EPA. (2016b). Public database 2016 chemical data reporting (May 2017 release). Washington, DC: US Environmental Protection Agency, Office of Pollution Prevention and Toxics. https://www.epa.gov/chemical-data-reporting
U.S. EPA. (2016c). Weight of evidence in ecological assessment [EPA Report]. (EPA/100/R-16/001). Washington, DC: Office of the Science Advisor. https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P100SFXR.txt
U.S. EPA. (2017a). Second Five Year Review report: Hudson River PCBs Superfund Site - Appendix 11: Human health and ecological risks.
U.S. EPA. (2017b). Toxics Release Inventory (TRI) basic plus data file, Hexabromocyclododecane (CAS # 25637-99-4), reporting year 2017. https://www.epa.gov/toxics-release-inventory-tri-program/tri-basic-plus-data-files-calendar-years-1987-2017
U.S. EPA. (2017c). Unregulated Contaminant Monitoring Rule (UCMR 3) (2013-2015) data: 1,4-dioxane. Washington, DC. https://www.epa.gov/dwucmr/occurrence-data-unregulated-contaminant-monitoring-rule#3
U.S. EPA. (2017d). Update for Chapter 5 of the Exposure Factors Handbook: Soil and dust ingestion [EPA Report]. (EPA/600R-17/384F). Washington, DC: National Center for Environmental Assessment, Office of Research and Development. https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P100TTX4.txt
U.S. EPA. (2018a). Application of systematic review in TSCA risk evaluations. (740-P1-8001). Washington, DC: U.S. Environmental Protection Agency, Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/sites/production/files/2018-06/documents/final_application_of_sr_in_tsca_05-31-18.pdf
U.S. EPA. (2018b). User's Guide for the AMS/EPA Regulatory Model (AERMOD). (EPA Document Number: EPA-454/B-18-001). U.S. EPA.
U.S. EPA. (2019a). Exposure factors handbook chapter 3 (update): Ingestion of water and other select liquids [EPA Report]. (EPA/600/R-18/259F). Washington, DC. https://cfpub.epa.gov/ncea/efp/recordisplay.cfm?deid=343661
U.S. EPA. (2019b). Guidelines for human exposure assessment [EPA Report]. (EPA/100/B-19/001). Washington, DC: Risk Assessment Forum. https://www.epa.gov/sites/production/files/2020-01/documents/guidelines_for_human_exposure_assessment_final2019.pdf
U.S. EPA. (2019c). Point Source Calculator: A Model for Estimating Chemical Concentration in Water Bodies. Washington, DC: U.S. Environmental Protection Agency, Office of Chemical Safety and Pollution Prevention.
U.S. EPA. (2019d). User's Guide: Integrated Indoor-Outdoor Air Calculator (IIOAC). Washington, DC: U.S. EPA.
U.S. EPA. (2020a). 2020 CDR data [Database]. Washington, DC: U.S. Environmental Protection Agency, Office of Pollution Prevention and Toxics. https://www.epa.gov/chemical-data-reporting/access-cdr-data
U.S. EPA. (2020b). Final Scope of the Risk Evaluation for 1,1-Dichloroethane; CASRN 75-34-3. (EPA 740-R-20-004). Washington, DC: Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/sites/default/files/2020-09/documents/casrn_75-34-3_11-dichloroethane_finalscope.pdf

U.S. EPA. (2020c). Final Scope of the Risk Evaluation for 1,1,2-Trichloroethane. (EPA-740-R-20-003). Washington, DC: U.S. Environmental Protection Agency, Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/sites/default/files/2020-09/documents/casrn_79-00-5_112-trichloroethane_finalscope.pdf
U.S. EPA. (2020d). Final scope of the risk evaluation for 1,1,2-trichloroethane; CASRN 79-00-5. (EPA 740-R-20-003). Washington, DC: Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/sites/default/files/2020-09/documents/casrn_79-00-5_112-trichloroethane_finalscope.pdf
U.S. EPA. (2020e). Final scope of the risk evaluation for 1,2-dichloroethane; CASRN 107-06-2. (EPA 740-R-20-005). Washington, DC: Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/sites/default/files/2020-09/documents/casrn_107-06-2_12-dichloroethane_final_scope.pdf
U.S. EPA. (2020f). Final Scope of the Risk Evaluation for 1,2-Dichloropropane. (EPA-740-R-20-006). Washington, DC: U.S. Environmental Protection Agency, Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/sites/default/files/2020-09/documents/casrn_78-87-5_12-dichloropropane_finalscope.pdf
U.S. EPA. (2020g). Risk evaluation for perchloroethylene (Ethene, 1,1,2,2-Tetrachloro-); CASRN 127-18-4 [EPA Report]. (740-R1-8011). Washington, DC: Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/document/EPA-HQ-OPPT-2019-0502-0058
U.S. EPA. (2020h). The Technical Support Document for the Hazardous Waste Delisting Risk Assessment Software (DRAS). Washington, DC: U.S. Environmental Protection Agency, Office of Solid Waste. https://www.epa.gov/hw/technical-support-document-hazardous-waste-delisting-risk-assessment-software-dras
U.S. EPA. (2021a). Announcement of final regulatory determinations for contaminants on the Fourth Drinking Water Contaminant Candidate List. https://www.govinfo.gov/content/pkg/FR-2021-03-03/pdf/2021-04184.pdf
U.S. EPA. (2021b). Contaminant Candidate List (CCL) and regulatory determination: Regulatory determination 4. Available online at https://www.epa.gov/ccl/regulatory-determination-4 (accessed April 27, 2021).
U.S. EPA. (2021c). Draft systematic review protocol supporting TSCA risk evaluations for chemical substances, Version 1.0: A generic TSCA systematic review protocol with chemical-specific methodologies. (EPA Document #EPA-D-20-031). Washington, DC: Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/document/EPA-HQ-OPPT-2021-0414-0005
U.S. EPA. (2021d). Final Regulatory Determination 4 Support Document [EPA Report]. (EPA 815R21001). U.S. Environmental Protection Agency (EPA). https://www.regulations.gov/document/EPA-HQ-OW-2019-0583-0284
U.S. EPA. (2022a). Chemical repackaging - Generic scenario for estimating occupational exposures and environmental releases (revised draft) [EPA Report]. Washington, DC.
U.S. EPA. (2022b). Cumulative impacts: Recommendations for EPA's Office of Research and Development. (EPA/600/R-22/014a). Washington, DC: Office of Research and Development, U.S. Environmental Protection Agency. https://www.epa.gov/system/files/documents/2022-09/Cumulative%20Impacts%20Research%20Final%20Report_FINAL-EPA%20600-R-22-014a.pdf
U.S. EPA. (2022c). Discharge Monitoring Report (DMR) data for 1,4-dioxane, 2013-2019. Washington, DC. https://echo.epa.gov/trends/loading-tool/water-pollution-search
U.S. EPA. (2022d). Ecological structure activity relationships (ECOSAR) predictive model, v2.2. Washington, DC. https://www.epa.gov/tsca-screening-tools/ecological-structure-activity-relationships-ecosar-predictive-model

U.S. EPA. (2022e). Safe Drinking Water Information System (SDWIS) Fed Data Warehouse: Sensitive drinking water-related information, 2022Q2. Washington, DC. https://www.epa.gov/ground-water-and-drinking-water/safe-drinking-water-information-system-sdwis-federal-reporting
U.S. EPA. (2022f). Toxics Release Inventory (TRI) data for 1,4-dioxane, 2013-2019. Washington, DC. https://www.epa.gov/toxics-release-inventory-tri-program/tri-data-and-tools
U.S. EPA. (2023a). Biosolids Tool (BST) User's Guide: Version 1, February 2023. (822D23002). Washington, DC: U.S. Environmental Protection Agency, Office of Water.
U.S. EPA. (2023b). Draft Supplement to the Risk Evaluation for 1,4-Dioxane. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/document/EPA-HQ-OPPT-2022-0905-0027
U.S. EPA. (2023c). Use of laboratory chemicals - Generic scenario for estimating occupational exposures and environmental releases (Revised draft generic scenario) [EPA Report]. Washington, DC: U.S. Environmental Protection Agency, Office of Pollution Prevention and Toxics, Existing Chemicals Risk Assessment Division.
U.S. EPA. (2024a). Environmental Hazard Assessment for Diisodecyl Phthalate (DIDP). Washington, DC: Office of Pollution Prevention and Toxics. https://www.regulations.gov/document/EPA-HQ-OPPT-2024-0073
U.S. EPA. (2024b). Web-based Interspecies Correlation Estimation (Web-ICE) for Acute Toxicity: User Manual, Version 4.0. (EPA/600/B-24/158). Gulf Breeze, FL: Office of Research and Development, Gulf Ecosystem Measurement and Modeling Division. https://www3.epa.gov/webice/iceManual.html
U.S. EPA. (2025a). Nontechnical Summary of the TSCA Risk Evaluation for 1,1-Dichloroethane. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114
U.S. EPA. (2025b). Risk Evaluation for 1,1-Dichloroethane - Supplemental File: Systematic Review Protocol. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114
U.S. EPA. (2025c). Risk Evaluation for 1,1-Dichloroethane - Supplemental Information File: Environmental Release and Occupational Exposure Assessment. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114
U.S. EPA. (2025d). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: AERMOD Input Specifications. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114
U.S. EPA. (2025e). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Ambient Monitoring Technology Information Center (AMTIC), 1,1-Dichloroethane Monitoring Data 2015 to 2020. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114
U.S. EPA. (2025f). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Benchmark Dose Modeling. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114
U.S. EPA. (2025g). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: EFAST Drinking Water Exposure Estimates. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114
U.S. EPA. (2025h). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Human Health Hazard Exposure Response Array Data and Figures. Washington, DC: Office of Pollution

Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025i). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: In Vitro Dermal Absorption Study Analysis. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025j). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: In Vitro Dermal Absorption Study Calculation Sheet. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025k). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Laboratory Chemical Occupational Exposure and Environmental Release Modeling Results. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025l). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Occupational Exposure Scenario Mapping Results. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025m). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Repackaging Environmental Release Modeling Results. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025n). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Repackaging Occupational Exposure Modeling Results. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025o). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Risk Calculator for Occupational Exposure. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025p). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD Generic Exposure and Risk Analysis. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025q). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD NEI Exposure and Risk Analysis. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025r). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025s). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on EPI Suite Modeling Results in the Fate Assessment. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025t). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis. Washington, DC: Office

of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025u). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming Central Tendency Exposure Estimates. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025v). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming High-End Exposure Estimates. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025w). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: TRV Calculator. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025x). Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Water Quality Portal Data 2015 to 2020. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025y). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Extraction Information for Environmental Hazard and Human Health Hazard Animal Toxicology and Epidemiology. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025z). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Extraction Information for General Population, Consumer, and Environmental Exposure. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025aa). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Dermal Absorption. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025ab). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025ac). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025ad). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025ae). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for Environmental Hazard. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025af). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for General Population, Consumer, and Environmental

Exposure. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025ag). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for Human Health Hazard Animal Toxicology. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA. (2025ah). Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for Human Health Hazard Epidemiology. Washington, DC: Office of Pollution Prevention and Toxics, Office of Chemical Safety and Pollution Prevention. https://www.regulations.gov/docket/EPA-HQ-OPPT-2024-0114

U.S. EPA; USGS. (2016). National hydrography dataset plus - NHDPlus edition 2.10. U.S. Environmental Protection Agency. https://www.epa.gov/waterdata/nhdplus-national-hydrography-dataset-plus

Umezu, T; Shibata, Y. (2014). Different behavioral effect dose-response profiles in mice exposed to two-carbon chlorinated hydrocarbons: influence of structural and physical properties. Toxicol Appl Pharmacol 279: 103-112. http://dx.doi.org/10.1016/j.taap.2014.05.012

Union Carbide. (1989). Lymphatic and hematopoietic tissue cancer in a chemical manufacturing environment with attached tables and cover letter dated 022189 [TSCA Submission]. (OTS0513414-2. 8EHQ-0289-0698. 89-890000005. TSCATS/311144). https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS05134172.xhtml

Urusova, TP. (1953). [Possibility of penetration of dichloroethane into milk in mothers exposed to preparation in industry]. Gig Sanit 60: 36-37.

Utsumi, H; Hakoda, M; Kiyoshige, K; Manabe, H; Mitade, C; Murayama, J; Han, SK; Hamada, A. (1992). Cytotoxicity and mutagenicity of micropollutants in drinking water. Water Sci Technol 25: 325-332. http://dx.doi.org/10.2166/wst.1992.0309

Van Duuren, BL; Goldschmidt, BM; Loewengart, G; Smith, AC; Melchionne, S; Seidman, I; Roth, D. (1979). Carcinogenicity of halogenated olefinic and aliphatic hydrocarbons in mice. J Natl Cancer Inst 63: 1433-1439.

Van Dyke, RA; Wineman, CG. (1971). Enzymatic dechlorination: Dechlorination of chloroethanes and propanes in vitro. Biochem Pharmacol 20: 463-470. http://dx.doi.org/10.1016/0006-2952(71)90082-7

Van Eekert, MH; Stams, AJ; Field, JA. (1999). Gratuitous dechlorination of chloroethanes by methanogenic granular sludge. Appl Microbiol Biotechnol 51: 46-52. http://dx.doi.org/10.1007/s002530051361

van Esch, GJ; Kroes, R; van Logten, MJ; den Tonkelaar, EM. (1977). Ninety-day toxicity study with 1,2-dichloroethane (DCE) in rats. (Report 195/77 Alg.Tox). Bilthoven, the Netherlands: National Institute of Public Health and Environmental Protection.

Versar. (2014). Exposure and Fate Assessment Screening Tool (E-FAST 2014) - Documentation manual. Washington, DC: U.S. Environmental Protection Agency. https://www.epa.gov/tsca-screening-tools/e-fast-exposure-and-fate-assessment-screening-tool-version-2014

Vogel, TM; McCarty, PL. (1987). Abiotic and biotic transformations of 1,1,1-trichloroethane under methanogenic conditions. Environ Sci Technol 21: 1208-1213. http://dx.doi.org/10.1021/es00165a008

Vozovaia, MA. (1977). [The effect of dichloroethane on the sexual cycle and embryogenesis of experimental animals]. Akush Ginekol 2: 57-59.

Walbridge, CT; Fiandt, JT; Phipps, GL; Holcombe, GW. (1983). Acute toxicity of ten chlorinated aliphatic hydrocarbons to the fathead minnow (Pimephales promelas). Arch Environ Contam Toxicol 12: 661-666. http://dx.doi.org/10.1007/BF01060748

Wang, B, ei; Ho, S; Ho, K; Huang, Y, u; Chan, C; Feng, N; Ip, S. (2012). An Environmental Chamber Study of the Characteristics of Air Pollutants Released from Environmental Tobacco Smoke. Aerosol Air Qual Res 12: 1269-1281. http://dx.doi.org/10.4209/aaqr.2011.11.0221

Wang, G; Qi, Y; Gao, L; Li, G; Lv, X; Jin, YP. (2013). Effects of subacute exposure to 1,2-dichloroethane on mouse behavior and the related mechanisms. Hum Exp Toxicol 32: 983-991. http://dx.doi.org/10.1177/0960327112470270

Wang, G; Yuan, Y; Gao, L; Tan, X; Yang, G; Zhao, F; Jin, Y. (2018a). Disruption of Intracellular ATP Generation and Tight Junction Protein Expression during the Course of Brain Edema Induced by Subacute Poisoning of 1,2-Dichloroethane. Frontiers in Neuroscience 12: 12. http://dx.doi.org/10.3389/fnins.2018.00012

Wang, G; Yuan, Y; Zhang, J; Gao, L; Tan, X; Yang, G; Lv, X; Jin, Y. (2014). Roles of aquaporins and matrix metalloproteinases in mouse brain edema formation induced by subacute exposure to 1,2-dichloroethane. Neurotoxicol Teratol 44: 105-112. http://dx.doi.org/10.1016/j.ntt.2014.06.005

Wang, T; Jin, X; Liao, Y; Sun, Q, i; Luo, C; Wang, G; Zhao, F; Jin, Y. (2018b). Association of NF-kappa B and AP-1 with MMP-9 overexpression in 2-Chloroethanol exposed rat astrocytes. 7. http://dx.doi.org/10.3390/cells7080096

Wang, T; Liao, Y; Sun, Q; Tang, H; Wang, G; Zhao, F; Jin, Y. (2017). Upregulation of matrix metalloproteinase-9 in primary cultured rat astrocytes induced by 2-chloroethanol via MAPK signal pathways. Front Cell Neurosci 11: 218. http://dx.doi.org/10.3389/fncel.2017.00218

Washington, JW; Cameron, BA. (2001). Evaluating degradation rates of chlorinated organics in groundwater using analytical models. Environ Toxicol Chem 20: 1909-1915. http://dx.doi.org/10.1002/etc.5620200908

Watanabe, K; Liberman, RG; Skipper, PL; Tannenbaum, SR; Guengerich, FP. (2007). Analysis of DNA adducts formed in vivo in rats and mice from 1,2-dibromoethane, 1,2-dichloroethane, dibromomethane, and dichloromethane using HPLC/accelerator mass spectrometry and relevance to risk estimates. Chem Res Toxicol 20: 1594-1600. http://dx.doi.org/10.1021/tx700125p

Webb, WW; Elfarra, AA; Webster, KD; Thom, RE; Anders, MW. (1987). Role for an episulfonium ion in S-(2-chloroethyl)-DL-cysteine-induced cytotoxicity and its reaction with glutathione. Biochemistry 26: 3017-3023. http://dx.doi.org/10.1021/bi00385a010

Wehmas, LC; Deangelo, AB; Hester, SD; Chorley, BN; Carswell, G; Olson, GR; George, MH; Carter, JH; Eldridge, S. R.; Fisher, A; Vallanat, B; Wood, CE. (2017). Metabolic Disruption Early in Life is Associated With Latent Carcinogenic Activity of Dichloroacetic Acid in Mice. Toxicol Sci 159: 354-365. http://dx.doi.org/10.1093/toxsci/kfx146

Weisburger, EK. (1977). Carcinogenicity studies on halogenated hydrocarbons. Environ Health Perspect 21: 7-16. http://dx.doi.org/10.1289/ehp.77217

Welke, B; Ettlinger, K; Riederer, M. (1998). Sorption of volatile organic chemicals in plant surfaces. Environ Sci Technol 32: 1099-1104.

Wesely, ML; Doskey, PV; Shannon, JD. (2002). Deposition parameterizations for the Industrial Source Complex (ISC3) model. (ANL/ER/TR-01/003). Argonne, IL: Argonne National Lab.

Westinghouse Savannah River Company. (1997). Sanitary landfill groundwater monitoring report. Fourth quarter 1996 and 1996 summary (pp. 506). International Programme on Chemical Safety (IPCS). https://search.proquest.com/docview/17587104?accountid=171501

Wiedemeier, TH; Rifai, HS; Newell, CJ; Wilson, JT. (1999). Natural attenuation of fuels and chlorinated solvents in the subsurface. New York, NY: John Wiley & Sons, Inc. http://dx.doi.org/10.1002/9780470172964

WIL Research. (2015). An extended one-generation drinking water reproductive toxicity study of ethylene dichloride in rats [TSCA Submission]. (Sec4-15-0042. WIL-417007). Millwood, VA: HAP Task Force.

Williams, GM; Mori, H; Mcqueen, CA. (1989). Structure-activity relationships in the rat hepatocyte DNA-repair test for 300 chemicals [Review]. Mutat Res 221: 263-286. http://dx.doi.org/10.1016/0165-1110(89)90039-0

Wilson, JT; Mcnsbb, JF; Wilson, RH. (1983). Biotransformation of selected organic pollutants in ground water. In Developments in Industrial Microbiology Volume 24 (pp. 225-233). (ISSN 0070-4563). Arlington, VA: Society for Industrial Microbiology. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/PB84101526.xhtml

Withey, JR; Collins, BT; Collins, PG. (1983). Effect of vehicle on the pharmacokinetics and uptake of four halogenated hydrocarbons from the gastrointestinal tract of the rat. J Appl Toxicol 3: 249-253. http://dx.doi.org/10.1002/jat.2550030506

Withey, JR; Karpinski, K. (1985). The fetal distribution of some aliphatic chlorinated hydrocarbons in the rat after vapor phase exposure. Biol Res Pregnancy Perinatol 6: 79-88.

Witt, KL; Knapton, A; Wehr, CM; Hook, GJ; Mirsalis, J; Shelby, MD; Macgregor, JT. (2000). Micronucleated erythrocyte frequency in peripheral blood of B6C3F(1) mice from short-term, prechronic, and chronic studies of the NTP carcinogenesis bioassay program. Environ Mol Mutagen 36: 163-194. http://dx.doi.org/10.1002/1098-2280(2000)36:3<163::AID-EM1>3.0.CO;2-P

Zabrodskii, PF; Troshkin, NM; Mandych, VG. (2004). Stimulation of immunotoxicity of chemicals metabolizing in vivo into highly toxic compounds by the monooxygenase system inductors. Bull Exp Biol Med 138: 369-371. http://dx.doi.org/10.1007/s10517-005-0044-5

Zamora, PO; Benson, JM; Li, AP; Brooks, AL. (1983). Evaluation of an exposure system using cells grown on collagen gels for detecting highly volatile mutagens in the CHO/HGPRT mutation assay. Environ Mutagen 5: 795-801. http://dx.doi.org/10.1002/em.2860050604

Zeiger, E; Anderson, B; Haworth, S; Lawlor, T; Mortelmans, K. (1992). Salmonella mutagenicity tests: V. Results from the testing of 311 chemicals. Environ Mol Mutagen 19: 2-141. http://dx.doi.org/10.1002/em.2850190603

Zeng, N; Jiang, H; Fan, Q; Wang, T; Rong, W; Li, G; Li, R; Xu, D; Guo, T; Wang, F; Zeng, L; Huang, M; Zheng, J; Lu, F; Chen, W; Hu, Q; Huang, Z; Wang, Q. (2018). Aberrant expression of miR-451a contributes to 1,2-dichloroethane-induced hepatic glycerol gluconeogenesis disorder by inhibiting glycerol kinase expression in NIH Swiss mice. J Appl Toxicol 38: 292-303. http://dx.doi.org/10.1002/jat.3526

Zhang, L; Jin, YP. (2019). Toxic effects of combined treatment of 1,2-dichloroethane and ethanol on mouse brain and the related mechanisms. J Biochem Mol Toxicol 33: 1. http://dx.doi.org/10.1002/jbt.22294

Zhang, Q; Niu, Q; Li, LY; Yang, L; Guo, XL; Huang, JX; Wang, LP; Liang, YX. (2011). Establishment of a poisoned animal model of toxic encephalopathy induced by 1,2-dichloroethane. Int J Immunopathol Pharmacol 24: 79S-83S.

Zhang, Y; Li, G; Zhong, Y; Huang, M; Wu, J; Zheng, J; Rong, W; Zeng, L; Yin, X; Lu, F; Xie, Z; Xu, D; Fan, Q; Jia, X; Wang, T; Hu, Q; Chen, W; Wang, Q; Huang, Z. (2017). 1,2-dichloroethane induces reproductive toxicity mediated by the CREM/CREB signaling pathway in male NIH Swiss mice. Toxicol Sci 160: 299-314. http://dx.doi.org/10.1093/toxsci/kfx182

Zhao, SF; Bao, YS; Zhang, XC. (1989). Studies on the effects of 1,2-dichloroethane on reproductive function. Zhonghua Yufang Yixue Zazhi 23: 199-202.

Zhao, SF; Zhang, XC; Zhang, LF; Zhou, SS; Zhang, F; Wang, QF; Wang, YL; Bao, YS. (1997). The evaluation of developmental toxicity of chemicals exposed occupationally using whole embryo culture. Int J Dev Biol 41: 275-282.

Zhong, Y; Liang, B; Hu, M; Liu, J; Lin, L; Jiang, J; Lin, X; Huang, Y; Lu, L; Jiang, L; Wu, J; Jia, X; Zhang, Y; Rong, W; Xie, Z; Zeng, L; Zhang, X; Yang, X; Huang, Z. (2020). MicroRNA-29b-3p

aggravates 1,2-dichloroethane-induced brain edema by targeting aquaporin 4 in Sprague-Dawley rats and CD-1 mice. Toxicol Lett 319: 160-167. http://dx.doi.org/10.1016/j.toxlet.2019.11.011

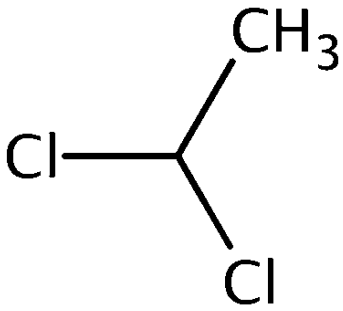
Zhong, Y; Liang, B; Meng, H; Ye, R; Li, Z; Du, J; Wang, B; Zhang, B; Huang, Y; Lin, X; Hu, M; Rong, W; Wu, Q; Yang, X; Huang, Z. (2022). 1,2-Dichloroethane induces cortex demyelination by depressing myelin basic protein via inhibiting aquaporin 4 in mice. Ecotoxicol Environ Saf 231: 113180. http://dx.doi.org/10.1016/j.ecoenv.2022.113180

Zhou, X; Cao, Y; Leuze, C; Nie, B; Shan, B; Zhou, W; Cipriano, P; Xiao, BO. (2016). Early non-invasive detection of acute 1,2-dichloroethane-induced toxic encephalopathy in rats. In Vivo 30: 787-793. http://dx.doi.org/10.21873/invivo.10995

[This page intentionally left blank]

Appendices: Risk Evaluation for 1,1-Dichloroethane

CASRN 75-34-3



June 2025

APPENDICES TABLE OF CONTENTS

LIST OF APPENDIX TABLES

LIST OF APPENDIX FIGURES

APPENDICES

Appendix A REGULATORY AND ASSESSMENT HISTORY

A.1 Federal Laws and Regulations

Table_Apx A-1. Federal Laws and Regulations

Statutes/Regulations	Description of Authority/Regulation	Description of Regulation[a]
EPA statutes/regulations		
Toxic Substances Control Act (TSCA) – section 6(b)	EPA is directed to identify high-priority chemical substances for risk evaluation; and conduct risk evaluations on at least 20 high priority substances no later than three and one-half years after the date of enactment of the Frank R. Lautenberg Chemical Safety for the 21st Century Act.	1,1-Dichloroethane is one of the 20 chemicals EPA designated as a High-Priority Substance for risk evaluation under TSCA (84 FR 71924, December 30, 2019). Designation of 1,1-dichloroethane as a high-priority substance constitutes the initiation of the risk evaluation on the chemical.
TSCA – section 8(a)	The TSCA section 8(a) CDR Rule requires manufacturers (including importers) to give EPA basic exposure-related information on the types, quantities, and uses of chemical substances produced domestically and imported into the United States.	1,1-Dichloroethane manufacturing (including importing), processing, and use information is reported under the CDR rule (85 FR 20122, January 23, 2025).
TSCA – section 8(b)	EPA must compile, keep current, and publish a list (the TSCA Inventory) of each chemical substance manufactured (including imported) or processed for commercial purposes in the United States.	Ethane, 1,1-dichloro (CASRN 75-34-3) was on the initial TSCA Inventory and therefore not subject to EPA's new chemicals review process under TSCA section 5 (60 FR 16309, March 29, 1995).
TSCA – section 8(d)	Provides EPA with authority to issue rules requiring manufacturers (including importers), processors, and distributors of a chemical substance or mixture to submit lists and/or copies of ongoing and completed, unpublished health and safety studies. EPA's Health and Safety Data Reporting Rule at 40 CFR part 716 generally requires such submissions for manufacturers (including importers) and (if specified) processors of substances covered by part 716.	One health and safety study received for 1,1-dichloroethane (2021) On June 29, 2021, EPA issued a final rule requiring manufacturers (including importers) of 1,1-dichloroethane to submit lists and copies of certain unpublished health and safety studies to EPA; the submission deadline in that rule was later extended to December 1, 2021 (86 FR 34147, June 29, 2021; 86 FR 54386, Oct. 1, 2021). (U.S. EPA, ChemView; accessed Jan. 29, 2025).
TSCA – section 8(e)	Manufacturers (including importers), processors, and distributors must immediately notify EPA if they obtain information that supports the conclusion that a chemical substance or mixture presents a substantial risk of injury to health or the environment.	Two substantial risk reports received for 1,1-dichloroethane (1993–1995: 2991004) (U.S. EPA, ChemView; accessed Jan. 3, 2025)

Statutes/Regulations	Description of Authority/Regulation	Description of Regulation[a]
TSCA – section 4	Provides EPA with authority to issue rules and consent agreements and orders requiring manufacturers (including importers) and processors to test chemical substances and mixtures.	Eight chemical data submissions from test rules and enforceable consent agreements were received for 1,1-dichloroethane: Environmental fate (3), Physical and chemical properties (5). (U.S. EPA, ChemView; accessed Jan. 27, 2025). The Frank R. Lautenberg Chemical Safety for the 21st Century Act revised TSCA section 4 to add authority to issue test orders. Under this authority, EPA issued test orders for information on environmental hazard and occupational exposure for 1,1-dichloroethane on January 19, 2021.
Emergency Planning and Community Right-to-Know Act (EPCRA) – section 313	EPCRA section 313—also known as the Toxics Release Inventory (TRI)—requires annual reporting from facilities in specific industry sectors that employ 10 or more full-time equivalent employees and that manufacture, process, or otherwise use a TRI-listed chemical in quantities above threshold levels. A facility that meets reporting requirements must submit a reporting form for each chemical for which it triggered reporting, providing data across a variety of categories, including activities and uses of the chemical, releases, and other waste management (*e.g.*, quantities recycled, treated, combusted) and pollution prevention activities (under section 6607 of the Pollution Prevention Act). These data include on- and off-site data as well as multimedia data (*i.e.*, air, land, and water).	1,1-Dichloroethane (ethylidene dichloride) is a listed substance subject to reporting requirements under 40 CFR 372.65 effective as of January 1, 1994.
Clean Air Act (CAA) – section 112(b)	Contains the original list of 189 hazardous air pollutants (HAPs) that Congress added in 1990. Under 112(c) of the CAA, EPA must identify and list source categories that emit HAP and then set emission standards for those listed source categories under CAA section 112(d). CAA section 112(b)(3)(A) specifies that any person may petition the Administrator to modify the list of HAP by adding or deleting a substance. Since 1990, EPA has both removed HAPs from and added HAPs to the original list.	1,1-Dichloroethane is listed as a HAP (42 U.S. Code Section 7412).

Statutes/Regulations	Description of Authority/Regulation	Description of Regulation[a]
CAA – section 112(d)	Directs EPA to establish, by rule, National Emission Standards for Hazardous Air Pollutants (NESHAPs) for each category or subcategory of listed major sources and area sources of HAPs (listed pursuant to section 112(c)). For major sources, the standards must require the maximum degree of emission reduction that EPA determines is achievable by each particular source category. This is generally referred to as maximum achievable control technology (MACT). For area sources, the standards must require generally achievable control technology (GACT) though may require MACT. Section 112(d)(6) requires EPA to review, and revise, as necessary, (taking into account developments in practices, processes and control technologies) the emission standards every 8 years.	EPA has established NESHAP for a number of source categories that emit 1,1-dichloroethane to air.
CAA – sections 112(d) and 112(f)	Section 112(f)(2) requires EPA to conduct risk assessments for each source category subject to section 112(d) NESHAP that require MACT and to determine if additional standards are needed to reduce remaining risks; this is required within 8 years of promulgating the NESHAP.	EPA has promulgated a number of RTR NESHAPs and will do so, as required, for the remaining source categories with NESHAPs.
Clean Water Act (CWA) – sections 301, 304, 306, 307 and 402	Clean Water Act Section 307(a) establishes a list of toxic pollutants or combination of pollutants under the CWA. The statute specifies a list of families of toxic pollutants also listed at 40 CFR 401.15. The "priority pollutants" specified by those families are listed in 40 CFR Part 423 Appendix A. These are pollutants (along with non-conventional pollutants) for which best available technology (BAT) effluent limitations must be established on either a national basis through rules (CWA sections 301(b), 304(b), 307(b), 306) or on a case-by-case best professional judgement basis in National Pollutant Discharge Elimination System (NPDES) permits, see section 402(a)(1)(B). EPA identifies BATs as economically achievable for that industry after considering statutorily prescribed factors and sets regulatory requirements based on the performance of that technology.	1,1-Dichloroethane is designated as a priority pollutant under section 307(a)(1) of the CWA and as such is subject to effluent limitations. Under CWA section 304, 1,1-dichloroethane is included in the list of total toxic organics (TTO) for at least one point source category (Coil Coating (40 CFR 465.02(j); Electroplating (40 CFR 413.02(i)); Metal Finishing (40 CFR 433.11(e)).

Statutes/Regulations	Description of Authority/Regulation	Description of Regulation[a]
Safe Drinking Water Act (SDWA) – section 1412(b)	Every 5 years, EPA must publish a list of contaminants that: (1) are not subject to any proposed or promulgated national primary drinking water regulations, (2) are known or anticipated to occur in public water systems (PWSs), and (3) may require regulation under SDWA. EPA must also determine whether to regulate at least 5 contaminants from the list every 5 years.	1,1-Dichloroethane was identified on CCL1 (1998), CCL2 (2005), CCL3 (2009), and CCL4 (2016). Contaminant Candidate List (CCL) 63 FR 10274, March 2, 1998; 70 FR 9071, February 24, 2005; 74 FR 51850, October 8, 2009; 81 FR 81099, November 17, 2016.
SDWA – section 1445(a)	Every 5 years, EPA must issue a new list of no more than 30 unregulated contaminants to be monitored by PWSs. The data obtained must be entered into the National Drinking Water Contaminant Occurrence Database.	1,1-Dichloroethane was identified in the third Unregulated Contaminant Monitoring Rule (UCMR3), issued in 2012 (77 FR 26071, May 2, 2012).
Resource Conservation and Recovery Act (RCRA) – section 3001	Directs EPA to develop and promulgate criteria for identifying the characteristics of hazardous waste, and for listing hazardous waste, taking into account toxicity, persistence, and degradability in nature, potential for accumulation in tissue, and other related factors such as flammability, corrosiveness, and other hazardous characteristics.	1,1-Dichloroethane is included on the list of hazardous wastes pursuant to RCRA 3001. RCRA Hazardous Waste Code: U076 (40 CFR 261.33).
Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) – sections 102(a) and 103	Authorizes EPA to promulgate regulations designating as hazardous substances, in addition to those referred to in section 101(14) of CERCLA, those elements, compounds, mixtures, solutions, and substances which, when released into the environment, may present substantial danger to the public health or welfare or the environment. EPA must also promulgate regulations establishing the quantity of any hazardous substance the release of which must be reported under section 103. Section 103 requires persons in charge of vessels or facilities to report to the National Response Center if they have knowledge of a release of a hazardous substance above the reportable quantity threshold. CERCLA Hazardous substances listed under 40 CFR Table 302.4 are subject to EPCRA section 304 notification requirements.	1,1-Dichloroethane is a hazardous substance under CERCLA. Releases of 1,1-dichloroethane in excess of 1,000 lb must be reported (40 CFR 302.4).

Statutes/Regulations	Description of Authority/Regulation	Description of Regulation[a]
Superfund Amendments and Reauthorization Act (SARA)	Requires EPA to revise the hazardous ranking system and update the National Priorities List of hazardous waste sites, increases state and citizen involvement in the superfund program and provides new enforcement authorities and settlement tools.	1,1-Dichloroethane is listed on SARA, an amendment to CERCLA, and the CERCLA Priority List of Hazardous Substances. This list includes substances most commonly found at facilities on the CERCLA National Priorities List (NPL) that have been deemed to pose the greatest threat to public health.
Other federal statutes/regulations		
Occupational Safety and Health Act (OSH Act)	Requires employers to provide their workers with a place of employment free from recognized hazards to safety and health, such as exposure to toxic chemicals, excessive noise levels, mechanical dangers, heat or cold stress, or unsanitary conditions (29 U.S.C section 651 et seq.). Under the Act, OSHA can issue occupational safety and health standards including such provisions as permissible exposure limits (PELs), exposure monitoring, engineering and administrative control measures, and respiratory protection.	In 1971, OSHA issued occupational safety and health standards for 1,1-dichloroethane that included a PEL of 100 ppm TWA. (29 CFR 1910.1000). See OSHA Annotated Table Z-1 (accessed Jan. 27, 2025).
Hazardous Materials Transportation Act (HMTA)	Section 5103 of the Act directs the Secretary of Transportation to: • Designate material (including an explosive, radioactive material, infectious substance, flammable or combustible liquid, solid or gas, toxic, oxidizing or corrosive material, and compressed gas) as hazardous when the Secretary determines that transporting the material in commerce may pose an unreasonable risk to health and safety or property. • Issue regulations for the safe transportation, including security, of hazardous material in intrastate, interstate, and foreign commerce.	1,1-Dichloroethane is listed as a hazardous material with regard to transportation and is subject to regulations prescribing requirements applicable to the shipment and transportation of listed hazardous materials (70 FR 34381, June 14, 2005).
Department of Energy	Protective Action Criteria (PAC)	PAC listed for 1,1-dichloroethane (accessed Jan. 29, 2025).

[a] Unless noted otherwise, all hyperlinks accessed June 11, 2025.

A.2 State Laws and Regulations

Table_ Apx A-2. State Laws and Regulations

State Actions[a]		Description of Action
State Air Regulations	Michigan Initial Threshold Screening Level (ITSL) (Michigan Administrative Code R.336.1229 List of Screening Levels)	State ITSL: 500 µg/m^3
	New Hampshire Allowable Ambient Levels (AAL) (Env-A 1400: Regulated Toxic Air Pollutants)	24-hour AAL: 2,037 µg/m^3 Annual AAL: 1,358 µg/m^3
	New York Annual Guidance Concentration (AGC) (6 NYCRR Part 212)	State AGC: 0.63 µg/m^3
	Rhode Island Annual Acceptable Ambient Level (Air Pollution Regulation No. 22)	State AAL: 0.6 µg/m^3
State Drinking Water Standards and Guidelines	California (Cal Code Regs. Title 26, § 22-64444)	Maximum Contaminant Level (MCL): 5 µg/L Detection Limit for Reporting (DLR): 0.5 µg/L Public Health Goal (PHG): 3 µg/L
	Connecticut - (Conn. Agencies Regs. § 19-13-B102)	State Action Level: 25 µg/L
	Florida (Fla. Admin. Code R. Chap. 62-550)	MCL/Health Advisory Level: 70 µg/L
	Massachusetts (310 Code Mass. Regs. § 22.00)	Office of Research and Standards Guidelines: 0.07 mg/L
	Michigan (Mich. Admin. Admin. Code r.299.44 and r.299.49, 2017)	Residential Drinking Water Criteria (DWC): 800 ppb Nonresidential DWC: 2,500 ppb Groundwater Surface Water Interface Criteria: 740 ppb
	New Jersey (7:10 N.J. Admin. Code § 5.2)	State MCL: 500.5 µg/L
State Water Pollution Discharge Programs	Illinois has adopted water pollution discharge programs (35 Ill. Adm. Code 307-2406).	1,1-dichloroethane is characterized as an "halogenated organic chemical," as applicable to the process wastewater discharges resulting from the manufacture of bulk organic chemicals (accessed May 1, 2025)
State PELs	Alaska (Alaska Administrative Code 8 AAC 61.1100); California [Cal Code Regs. Title 8, § 5155]); Connecticut (Limits for Air Contaminants); Hawaii (Admin. Rules Section 12-60-50); Illinois (Admin. Code 56 IAC part 350); Indiana (Admin. Code 620 article 1–30); Iowa (Admin. Code IAC 10/21/98); Kentucky (Admin. Regs 803 KAR chapter 2); Maine (Admin. Code Title 36, chapter 6); Maryland (Code of Maryland Regulations COMAR 09.12.32); Minnesota (Admin. Rules 5206.0400); Nevada (Admin. Code Chapter 618); New Jersey (Admin. Code 8:59-4.2); New Mexico (New Mexico Admin. Title 20); New York (Rules, Regulations Title 12 §	100 ppm

State Actions[a]		Description of Action
	800.5); North Carolina (Admin. Code 13 NCAC 07F); Oregon (Admin. Rules Chapter 437); South Carolina (Code of Law Title 41 Chapter 15); Tennessee (Admin. Code 0800-01-01-07); Utah (Admin. Code Title R614); Vermont Statues Online (Title 21 Chapter 3, 201-232); Virginia (Admin. Code CVAC25-40-720); Wyoming (Admin. Rule 053-26 Wyo. Code R 26-1)	
	Massachusetts (Massachusetts Department of Environmental Protection; Ambient Air Toxic Guidelines)	Threshold Effects Exposure Limit (24-Hour Average): 100 ug/m^3 Non-Threshold Effects Exposure Limits (Annual Average): 0.5 µg/m^3 Allowable Ambient Limits (Annual Average): 0.5 µg/m^3
	Michigan (Admin. Code R 325.2414);	TWA: 100 ppm STEL: 400 ppm
	Washington (Admin. Code WAC 296-841-20025)	TWA: 100 ppm STEL: 150 ppm
State Right-to-Know Acts		Massachusetts (105 Code Mass. Regs. § 670.000 Appendix A), New Jersey (N.J.A.C. 7:1G) and Pennsylvania (P.L. 734, No. 159 and 34 Pa. Code § 323).
Chemicals of High Concern to Children		Several states have adopted reporting laws for chemicals in children's products containing 1,1-dichloroethane, including Maine's list of Chemical of Concern (38 MRSA Chapter 16-D), Minnesota (Toxic Free Kids Act Minn. Stat. 116.9401 to 116.9407).
Other		California listed 1,1-dichloroethane on Proposition 65 in 1990 due to cancer risk (Cal Code Regs. Title 27, § 27001). 1,1-Dichloroethane is listed as a Candidate Chemical under California's Safer Consumer Products Program established under Health and Safety Code § 25252 and 25253 (California, Candidate Chemicals List; accessed April 18, 2019) (CDTSC, 2017). California lists 1,1-dichloroethane as a designated priority chemical for biomonitoring under criteria established by California SB 1379 (CDPH, 2015) (accessed February 2019). 1,1-Dichloroethane is on the MA Toxic Use Reduction Act (TURA) list of 1994 (301 Code Mass. Regs. § 41.03).

[a] Unless noted otherwise, all hyperlinks accessed June 11, 2025.

A.3 International Laws and Regulations

Table_Apx A-3. International Laws and Regulations

Country/Organization	Requirements and Restrictions[a]	
Canada	1,1-Dichloroethane is on the Non-Domestic Substances List (NDSL). Canada requires notification for 1,1-dichloroethane under the New Substances Notification Regulations (Chemicals and Polymers) so that health and ecological risks can be assessed before the substance is manufactured or imported into Canada above threshold quantities; however, they are subject to fewer information requirements. Canada Gazette Part I, Vol. 142, No. 25, June 21, 2008 (accessed Jan. 28, 2025).	
European Union	1,1-Dichloroethane is registered for use in the EU (European Chemicals Agency (ECHA) database (accessed Jan. 28, 2025).	
Australia	1,1-Dichloroethane can be manufactured or imported into Australia for commercial purposes without notifying the Australian government, provided that the Australian importer/manufacturer is currently registered with the Australian government. 1,1-Dichloroethane was assessed under Human Health Tier II of the Inventory Multi-Tiered Assessment and Prioritisation (IMAP). No specific Australian use, import, or manufacturing information has been identified. (NICNAS, Ethane, 1,1-dichloro-: Human health tier II assessment (accessed Jan. 28, 2025).	
Japan	1,1-Dichloroethane is regulated in Japan under the following legislation: • Act on the Evaluation of Chemical Substances and Regulation of Their Manufacture, *etc.* (Chemical Substances Control Law; CSCL) • Act on Confirmation, etc. of Release Amounts of Specific Chemical Substances in the Environment and Promotion of Improvements to the Management Thereof (PRTR-SDS Law) • Industrial Safety and Health Act (ISHA) • Poisonous and Deleterious Substances Control Act • Act on Prohibition of Chemical Weapons and Control, etc. of Specific Chemicals (Chemical Weapons Prohibition Law) • Act on the Protection of the Ozone Layer through the Control and Other Measures on Specified Substances and Other Substances • Air Pollution Control Act • Water Pollution Prevention Act • Soil Contamination Countermeasures Act • Act on the Control of Household Products Containing Harmful Substances • Food Sanitation Act • High Pressure Gas Safety Act • Explosives Control Act • Fire Service Act • Act on Securing Quality, Efficacy and Safety of Products Including Pharmaceuticals and Medical Devices (PMD Act) Chemical Risk Information Platform [CHRIP] (accessed Jan. 28, 2025).	
Institute for Occupational Safety and Health of the German Social Accident Insurance (IFA) GESTIS (accessed June 11, 2025) International Limit Values for Chemical	Australia, Belgium, Canada, European Union, France, Ireland, Italy, Japan, Latvia, Romania, Singapore, South Korea, Spain, Sweden, United Kingdom	TWA: 100 ppm
	Austria	TWA: 100 ppm STEL: 400 ppm
	Denmark, Switzerland	TWA: 100 ppm

Country/Organization	Requirements and Restrictions[a]	
Agents Database (accessed Apr. 18, 2019).		STEL: 200 ppm
	Finland, New Zealand	TWA: 100 ppm STEL: 250 ppm
	Germany	TWA: 50 ppm STEL: 100 ppm
	Hungary	TWA: 412 mg/m^3
	Norway	TWA: 50 ppm
	Poland	TWA: 400 mg/m^3
	South Africa	TWA: 200 ppm TWA (Mining): 100 ppm STEL (Mining): 200 ppm
	The Netherlands	TWA: 97 ppm STEL: 194 ppm
[a] Unless noted otherwise, all hyperlinks accessed June 11, 2025.		

A.4 Assessment History

Table_Apx A-4. Assessment History of 1,1-Dichloroethane

Authoring Organization	Publication[a]
EPA publications	
U.S. Environmental Protection Agency (EPA), Integrated Risk Information System (IRIS)	IRIS Summary. 1,1-Dichloroethane; CASRN 75-34-3
EPA, National Service Center for Environmental Publications (NSCEP)	Exposure and Risk Assessment {for} Dichloroethanes 1,1-Dichloroethane, 1,2-Dichloroethane
EPA, Office of Chemical Safety and Pollution Prevention (OCSPP)	*Final Scope of the Risk Evaluation for 1,1-Dichloroethane; CASRN 75-34-3* (2020)
EPA, Office of Pollution Prevention and Toxics (OPPT)	ChemView (TSCA submissions – chemical test rule data and substantial risk reports)
EPA, Superfund Health Risk Technical Support Center, National Center for Environmental Assessment, Office of Research and Development	*Provisional Peer Reviewed Toxicity Values for 1,1-Dichloroethane (CASRN 75-34-3)*
Other U.S.-based organizations	
Agency for Toxic Substances and Disease Registry (ATSDR)	*Toxicological Profile for 1,1-Dichloroethane CAS#: 75-34-3, August 2015*
U.S. Centers for Disease Control and Prevention (CDC)	*2015. Fourth National Report on Human Exposure to Environmental Chemicals*
National Cancer Institute (NCI)	NCI 1978. *Bioassay of 1,1-Dichloroethane for Possible Carcinogenicity* (CAS No. 75-34-3). Technical Report Series No. 66 (NCI-CG-TR-66)

Authoring Organization	Publication[a]
	NCI 1977. *Bioassay of 1,1-Dichloroethane for Possible Carcinogenicity*. Bethesda, MD: NCI. National Institutes of Health (NIH) Publication No. 78-1316
National Institute for Occupational Safety and Health (NIOSH)	Current Intelligence Bulletin 27: Chloroethanes Review of Toxicity
	Occupational Health Guidelines for 1,1-Dichloroethane. Occupational Health Guidelines for Chemical Hazards. Washington, DC: U.S. Department of Labor, NIOSH, 1–4. 1978
	1,1-Dichloroethane. NIOSH Pocket Guide to Chemical Hazards. Atlanta, GA: National Institute for Occupational Safety and Health, CDC. 2015
National Toxicology Program (NTP), National Institute of Environmental Health Sciences (NIEHS), NIH	1,1-Dichloroethane: Target Organs and Levels of Evidence for TR-066.
Occupational Safety and Health Administration (OSHA)	*Occupational Exposure to Methylene Chloride* (OSHA, 1997)
International	
ECHA European Union Risk Assessment Report	Information from the Existing Substances Regulation (ESR)
Government of Canada, Environment Canada, Health Canada	Chemicals at a Glance (Fact Sheets) International Resources Assessment or Related Document

[a] All hyperlinks accessed June 11, 2025.

Appendix B LIST OF SUPPLEMENTAL DOCUMENTS

This appendix includes a list and citations for all supplemental documents included in the Risk Evaluation for 1,1-Dichloroethane. See Dockets EPA-HQ-OPPT-2018-0426 and EPA-HQ-OPPT-2024-0114 for all publicly released files associated with this final risk evaluation.

Associated **Systematic Review Protocol and Data Quality Evaluation and Data Extraction** Documents – Provide additional detail and information on systematic review methodologies used as well as the data quality evaluations and extractions criteria and results.

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol (U.S. EPA, 2025b) – In lieu of an update to the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances: A Generic TSCA Systematic Review Protocol with Chemical-Specific Methodologies*, also referred to as the "2021 Draft Systematic Review Protocol" (U.S. EPA, 2021c), this systematic review protocol for the Risk Evaluation for 1,1-Dichloroethane describes some clarifications and different approaches that were implemented than those described in the 2021 Draft Systematic Review Protocol in response to (1) Science Advisory Committee on Chemicals (SACC) comments, (2) public comments, or (3) to reflect chemical-specific risk evaluation needs. This supplemental file may also be referred to as the "1,1-Dichloroethane Systematic Review Protocol."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties (U.S. EPA, 2025ad) – Provides a compilation of tables for the data extraction and data quality evaluation information for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation of physical and chemical properties. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport (U.S. EPA, 2025ab) – Provides a compilation of tables for the data extraction and data quality evaluation information for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation for Environmental Fate and Transport. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure (U.S. EPA, 2025ac) – Provides a compilation of tables for the data extraction and data quality evaluation information for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation of environmental release and occupational exposure. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Quality Evaluation and Data Extraction Information for Environmental Release and Occupational Exposure."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Dermal Absorption (U.S. EPA, 2025aa)

– Provides a compilation of tables for the data extraction and data quality evaluation information for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was extracted and evaluated from a data source that has information relevant for the evaluation for Dermal Absorption. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Quality Evaluation and Data Extraction Information for Dermal Absorption."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure. (U.S. EPA, 2025af) – Provides a compilation of tables for the data quality evaluation information for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was evaluated from a data source that has information relevant for the evaluation of general population, consumer and environmental exposure. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Quality Evaluation Information for General Population, Consumer, and Environmental Exposure."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Extraction Information for General Population, Consumer, and Environmental Exposure (U.S. EPA, 2025z) – Provides a compilation of tables for the data extraction for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was extracted from a data source that has information relevant for the evaluation of general population, consumer, and environmental exposure. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Extraction Information for General Population, Consumer, and Environmental Exposure."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for Human Health Hazard Epidemiology (U.S. EPA, 2025ah) – Provides a compilation of tables for the data quality evaluation information for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was evaluated from a data source that has information relevant for the evaluation of epidemiological information. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Quality Evaluation Information for Human Health Hazard Epidemiology."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for Human Health Hazard Animal Toxicology (U.S. EPA, 2025ag) – Provides a compilation of tables for the data quality evaluation information for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was evaluated from a data source that has information relevant for the evaluation of human health hazard animal toxicity information. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Quality Evaluation Information for Human Health Hazard Animal Toxicology."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Quality Evaluation Information for Environmental Hazard (U.S. EPA, 2025ae) – Provides a compilation of tables for the data quality evaluation information for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was evaluated from a data source that has information relevant for the evaluation of environmental hazard toxicity information. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Quality Evaluation Information for Environmental Hazard."

Risk Evaluation for 1,1-Dichloroethane – Systematic Review Supplemental File: Data Extraction Information for Environmental Hazard and Human Health Hazard Animal Toxicology and Epidemiology (U.S. EPA, 2025y) – Provides a compilation of tables for the data extraction for 1,1-dichloroethane. Each table shows the data point, dataset, or information element that was extracted from a data source that has information relevant for the evaluation of environmental hazard and human health hazard animal toxicology and epidemiology information. This supplemental file may also be referred to as the "1,1-Dichloroethane Data Extraction Information for Environmental Hazard and Human Health Hazard Animal Toxicology and Epidemiology."

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

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment (U.S. EPA, 2025c).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Risk Calculator for Occupational Exposure (U.S. EPA, 2025o).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Laboratory Chemical Occupational Exposure and Environmental Release Modeling Results (U.S. EPA, 2025k).

*Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Repackaging Environmental Release Modeling Results (*U.S. EPA, 2025m*).*

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Occupational Exposure Scenario Mapping Results (U.S. EPA, 2025l).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis (U.S. EPA, 2025r).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD Generic Releases Exposure and Risk Analysis (U.S. EPA, 2025p).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD NEI Exposure and Risk Analysis (U.S. EPA, 2025q).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on Ambient Monitoring Technology Information Center (AMTIC), 1,1-Dichloroethane Monitoring Data 2015 to 2020 (U.S. EPA, 2025e).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis (U.S. EPA, 2025t).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: AERMOD Input Specifications (U.S. EPA, 2025d).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming Central Tendency Exposure Estimates (U.S. EPA, 2025u).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Surface Water Concentration and Fish Ingestion and Swimming High-End Exposure Estimates (U.S. EPA, 2025v).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Drinking Water Exposure Estimates (U.S. EPA, 2025g).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: TRV Calculator (U.S. EPA, 2025w).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Benchmark Dose Modeling (U.S. EPA, 2025f).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on EPI Suite Modeling Results in the Fate Assessment (U.S. EPA, 2025s).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: in vitro Dermal Absorption Study Analysis (U.S. EPA, 2025i).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: in vitro Dermal Absorption Study Calculation Sheet (U.S. EPA, 2025j).

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Human Health Hazard Exposure Response Array Data and Figures (U.S. EPA, 2025h)

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Water Quality Portal Data 2015 to 2020 (U.S. EPA, 2025a)

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Estimates of Number of Workers and ONUs (U.S. EPA, 2025x)

Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Repackaging Occupational Exposure Modeling Results (U.S. EPA, 2025n).

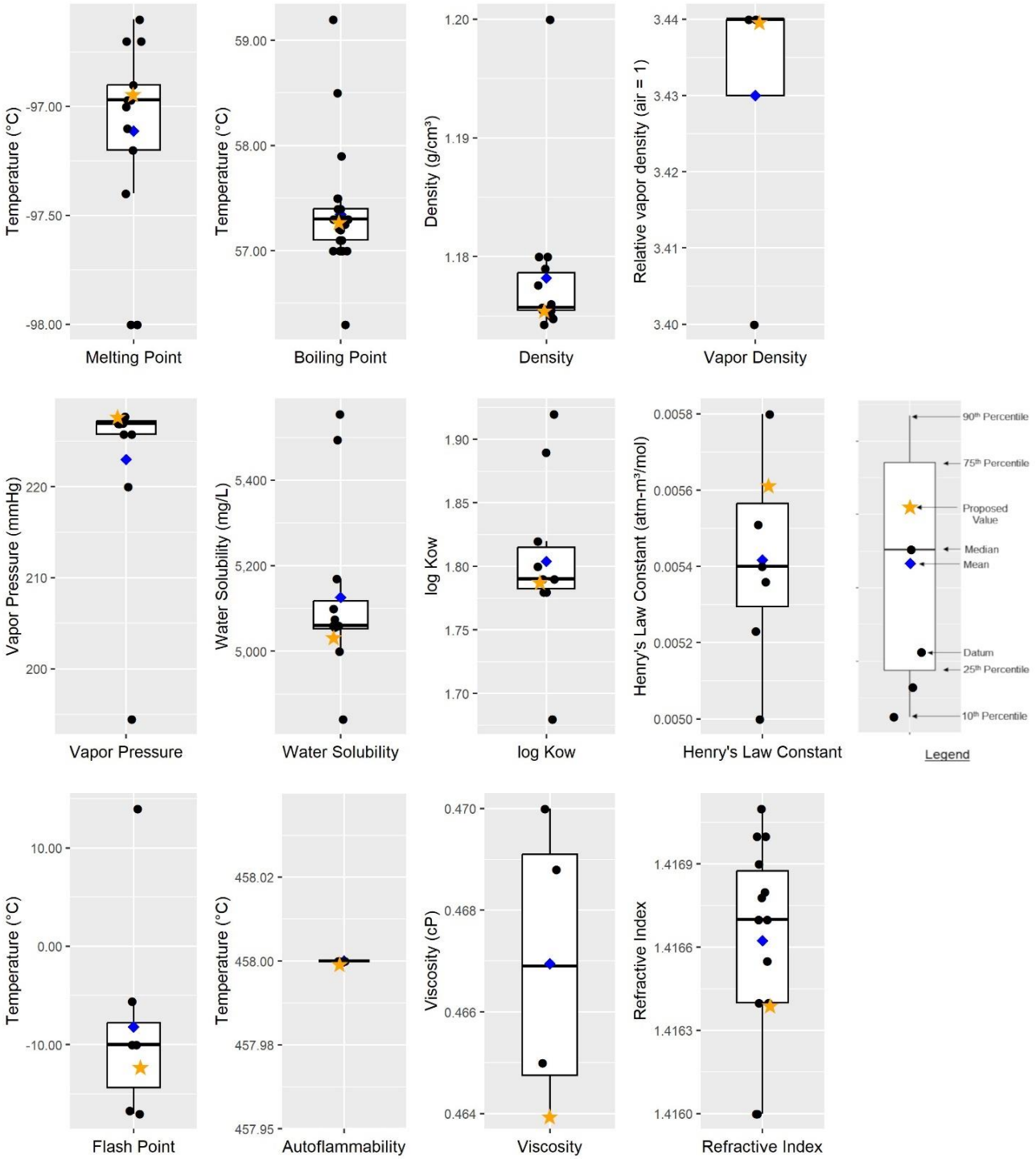
Appendix C PHYSICAL AND CHEMICAL PROPERTIES AND FATE AND TRANSPORT DETAILS

C.1 Physical and Chemical Properties

Selection of a Physical-Chemical Property Value from Multiple High-Quality Sources

The systematic review process identified multiple data with the same quality rating for many physical and chemical properties discussed in this document. Some of these data were duplicates that were initially extracted more than once (*e.g.*, when multiple databases cite the same study), but were later removed during data curation before any further analysis. Much of the remaining data were collected under "standard environmental conditions" (*i.e.*, 20–25 °C and 760 mmHg [average atmospheric pressure]). These data are presented in box and whisker plots in Figure_Apx C-1, which also include descriptive statistics such as the mean and median. Data that were collected under non-standard conditions are also presented in scatter plots, where appropriate, to provide a clear visualization of the temperature- or pressure-dependence of the physical and chemical parameters. It is important to visualize this dependence to illustrate that high data variance may be due to measurements across different experimental conditions and not necessarily high uncertainty in the data. Such visualizations may also allow for the identification of trends that can approximate the parameter under other environmental conditions. Finally, a data point measured under non-standard conditions could better simulate a given scenario for fate assessments or other modeling purposes (*e.g.*, when a temperature other than ≈25 °C would be more relevant for a particular chemical and assessment scenario).

When a specific data point is cited for a given physical and chemical parameter, priority is given to data from expert-curated, peer-reviewed databases that have been identified as "trusted sources" (U.S. EPA, 2021c). If no data were available from trusted databases, second preference was given to measured data from studies that implement experimental measurements according to established test guidelines or that were conducted according to scientific principles with sufficient documentation. Finally, estimated, or calculated data are only presented in the instance that no measured data were available.



Figure_Apx C-1. Physical and Chemical Property Data for 1,1-Dichloroethane Under Standard Conditions

Standard conditions are 20 to 25 °C and 760 mmHg; data collected through systematic review.

Key Sources of Uncertainty of Physical and Chemical Property Values

The physical and chemical property data discussed in this document were the product of a systematic review of reasonably available information. The data analyses, therefore, consider only a subset of all physical-chemical data—not an exhaustive acquisition of all potential data. Due to cross-referencing between many of the databases identified and assessed through the systematic review process, there is

potential for data from one primary source to be collected multiple times resulting in duplication within the dataset. This duplication should be considered as a potential source of uncertainty in the data analyses; however, data-collection procedures and expert judgement were used to minimize this possibility whenever possible.

Overall, there is little uncertainty in the physical and chemical data and analyses presented. The analyses below present the average and standard deviation of all data collected through the systematic review process for each physical-chemical parameter. The standard deviation is reported as uncertainty in the form of tolerance limits (± range) on the average value. Data extracted as a range of values were excluded from the calculations unless expert judgement could identify precise data points within the range. These statistical analyses may be indicative of the amount of uncertainty related to different instrumental techniques or other experimental differences between the studies used to generate the data. Additional sources of uncertainty in these reported physical and chemical values may be inherent to the measurement of the data point itself (*e.g.*, sources of uncertainty or measurement error related to the instrumental method, precision with which a data point is measured and reported in the data source). Finally, all data were assumed to be collected under standard environmental conditions (*i.e.*, 20–25 °C and 760 mmHg) unless otherwise specified. Additional discussions of uncertainty are included within the appropriate subsections below, when necessary.

Molecular Formula: By definition, the molecular formula of 1,1-dichloroethane is $C_2H_4Cl_2$. This parameter was not obtained by systematic review and there is no uncertainty in this value.

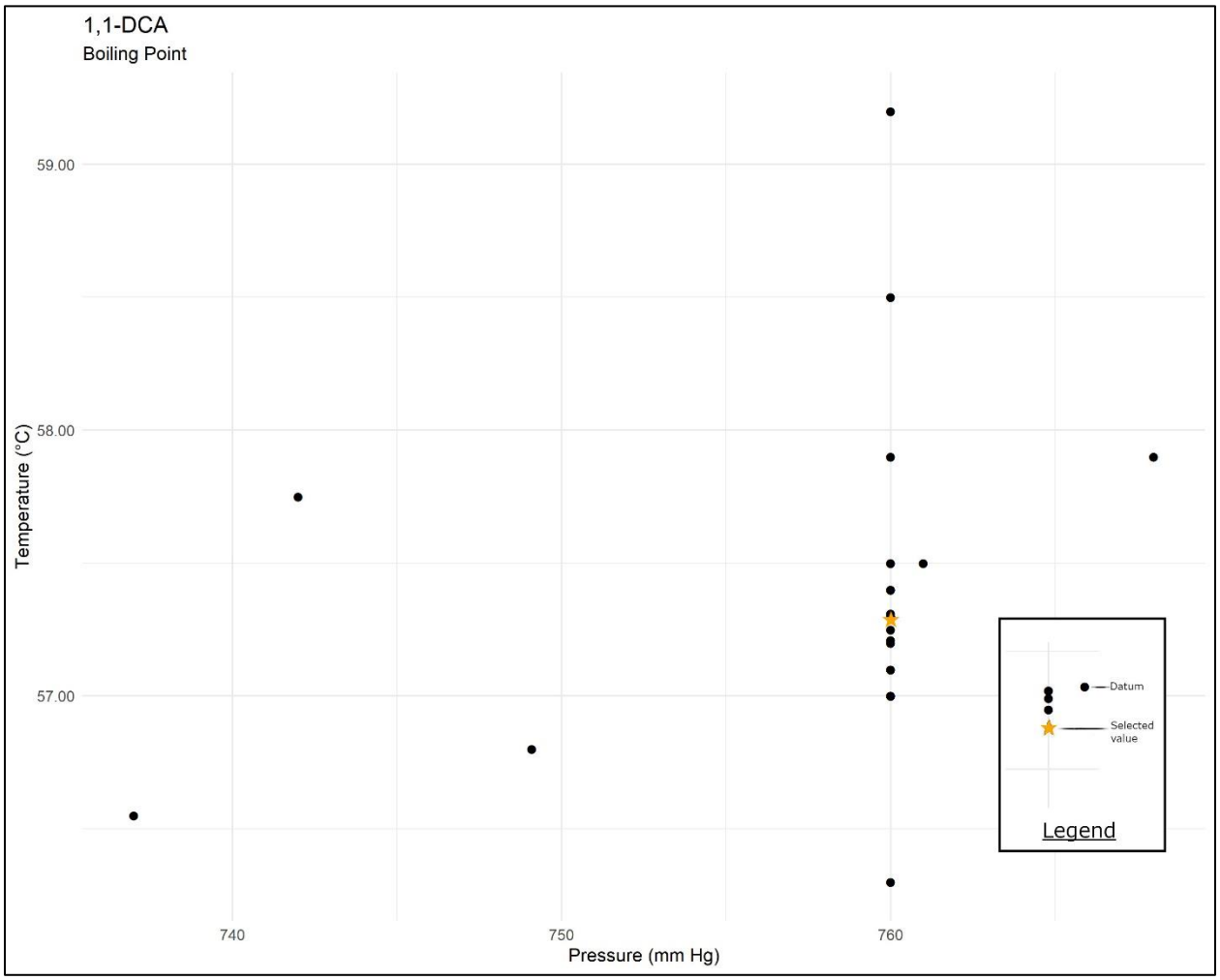
Molecular Weight: By definition, the molecular weight of 1,1-dichloroethane is 98.95 g/mol. This value was not obtained by systematic review, but rather is calculated from the known molecular formula. The uncertainty in this value inherent to molecular weight determination from atomic masses is negligible for the purpose of this risk evaluation.

Physical Form: 1,1-Dichloroethane is a liquid under ambient conditions (*i.e.*, at ≈20 °C and 760 mmHg) (Government of Canada, 2021). It is qualitatively described as being colorless, oily, and having a chloroform- or ether-like odor (NLM, 2018; NIOSH, 2007). These descriptions agree with the qualitative descriptions identified in the *Final Scope of the Risk Evaluation for 1,1-Dichloroethane; CASRN 75-34-3* (also referred to as the "final scope for 1,1-dichloroethane") (U.S. EPA, 2020a).

Melting Point: Systematic review identified 13 melting point data points that cover the range −98 to −96.6 °C. The average melting point was −97.1 ± 0.4 °C. The value −96.93 °C (NLM, 2018) was selected as the melting point of 1,1-dichloroethane for this risk evaluation because it is in close agreement with the average of all data identified, has a high level of precision, was independently reported in multiple high-quality experimental studies and aligns with the value reported in the final scope. The standard deviation of the collected data is relatively low, indicating that the value of this parameter is well-defined.

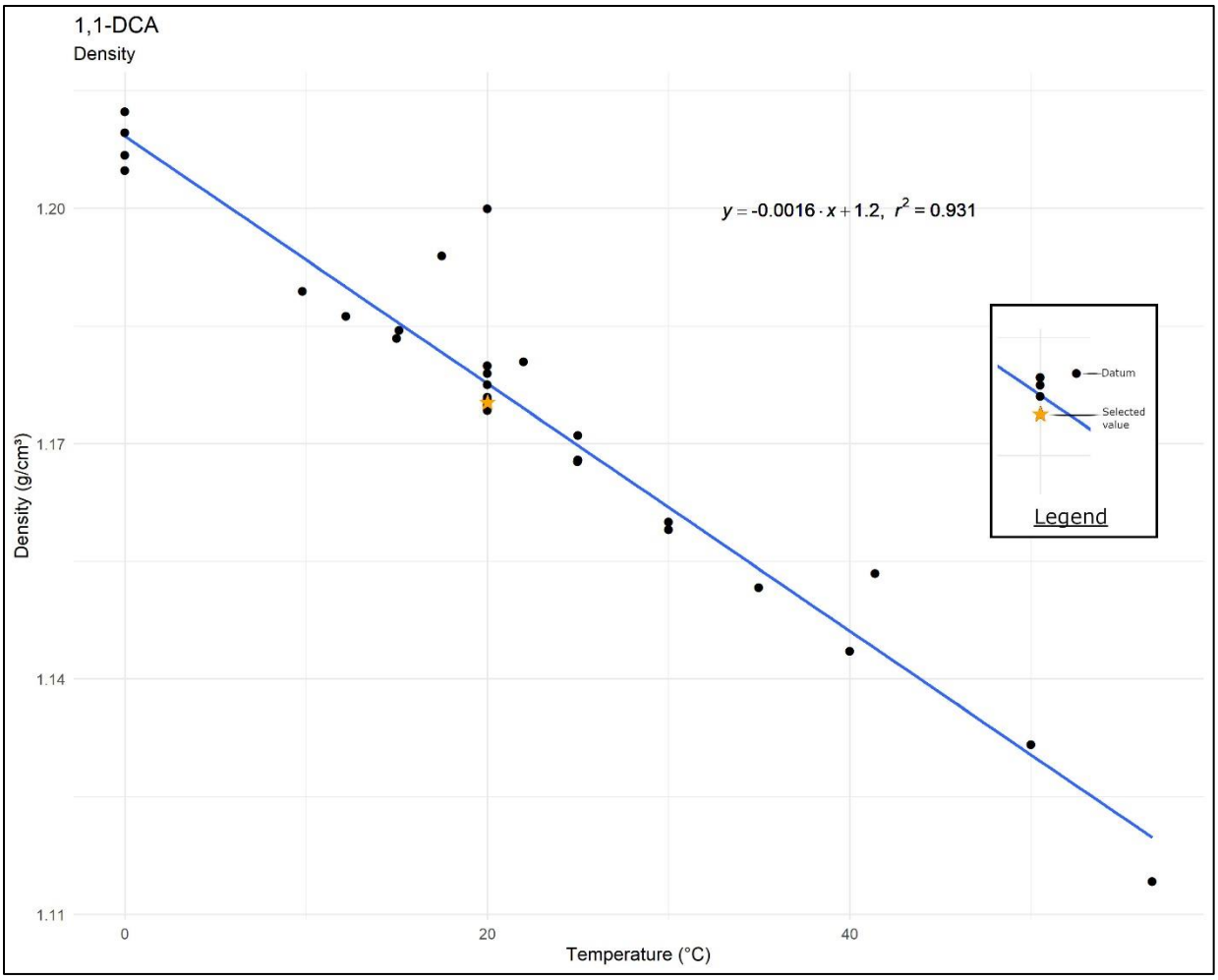
Boiling Point: Systematic review identified 34 boiling point data, including 29 data points collected at 760 mmHg. The data collected under standard conditions cover the range 56.3 to 83.6 °C. Excluding statistical outliers, the range condenses to 28 data points covering 56.3 to 59.2 °C. The average boiling point was 57.3 ± 0.5 °C. The variation of boiling point as a function of pressure is visualized in Figure_Apx C-2. The value 57.3 °C (O'Neil, 2013) was selected as the boiling point of 1,1-dichloroethane for this risk evaluation because it is in close agreement with the average of all the data identified and it was independently reported in multiple high-quality studies. The selected value differs minimally from the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). The

standard deviation of the collected data is relatively low, indicating that the value of this parameter is well-defined.



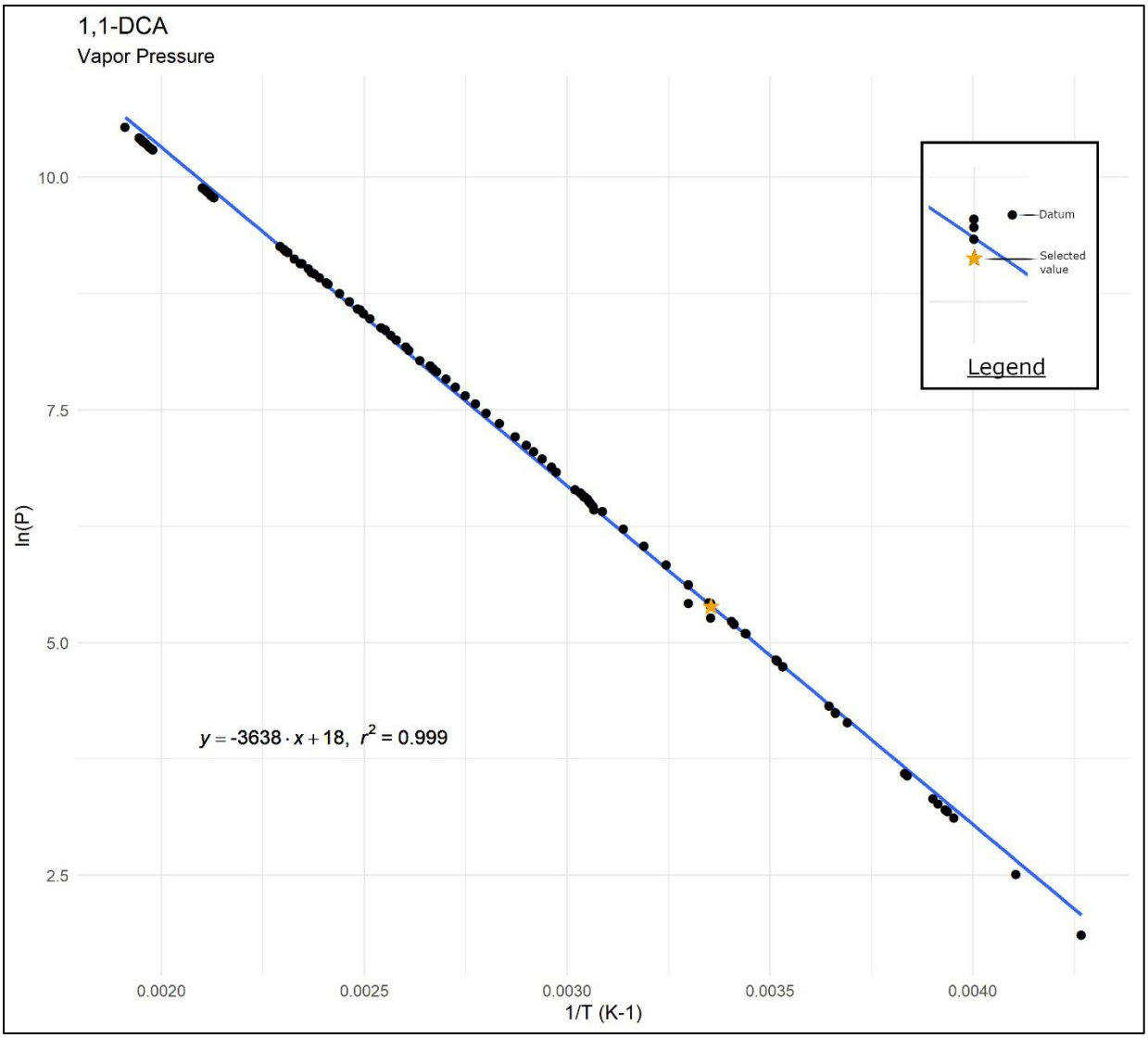
Figure_Apx C-2. Boiling Point of 1,1-Dichloroethane as a Function of Pressure

Density: Systematic review identified 37 density data, including 14 data points collected at 20 °C. The data collected under standard conditions cover the range 1.1743 to 1.2 g/cm^3 (specific gravity and density were assumed to be equal). The average density was 1.1782 ± 0.0066 g/cm^3. The variation of density as a function of temperature is visualized in Figure_Apx C-3. The value 1.1757 g/cm^3 at 20 °C (O'Neil, 2013) was selected as the density of 1,1-dichloroethane for this risk evaluation because it is in close agreement with the average of the data identified, has a high level of precision, and was independently reported in multiple high-quality experimental studies. The selected value differs slightly from the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). The standard deviation of the collected data is relatively low, indicating that the value of this parameter is well-defined.



Figure_Apx C-3. Density of 1,1-Dichloroethane as a Function of Temperature

Vapor Pressure: Systematic review identified 108 vapor pressure data points, including 10 data points collected at 25 °C. The data collected under standard conditions cover the range 194.49 to 228 mmHg at 25 °C. The average vapor pressure was 223 ± 10.3 mmHg at 25 °C. The variation of vapor pressure as a function of temperature, which is governed by the Clausius-Clapeyron relationship, is visualized in Figure_Apx C-4. The value 228 mmHg at 25 °C (Rumble, 2018b) was selected as the vapor pressure of 1,1-dichloroethane for this risk evaluation because it is in close agreement with this analysis, and it was independently reported in multiple high-quality studies. The selected value differs minimally from the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). The standard deviation of the collected data is relatively low, indicating that the value of this parameter is well-defined. Additionally, the vapor pressure at non-standard temperatures can be determined using the results of the systematic review and Figure_Apx C-4, although there is increasing uncertainty at high temperatures and data should not be extrapolated outside of −50 to 250 °C.

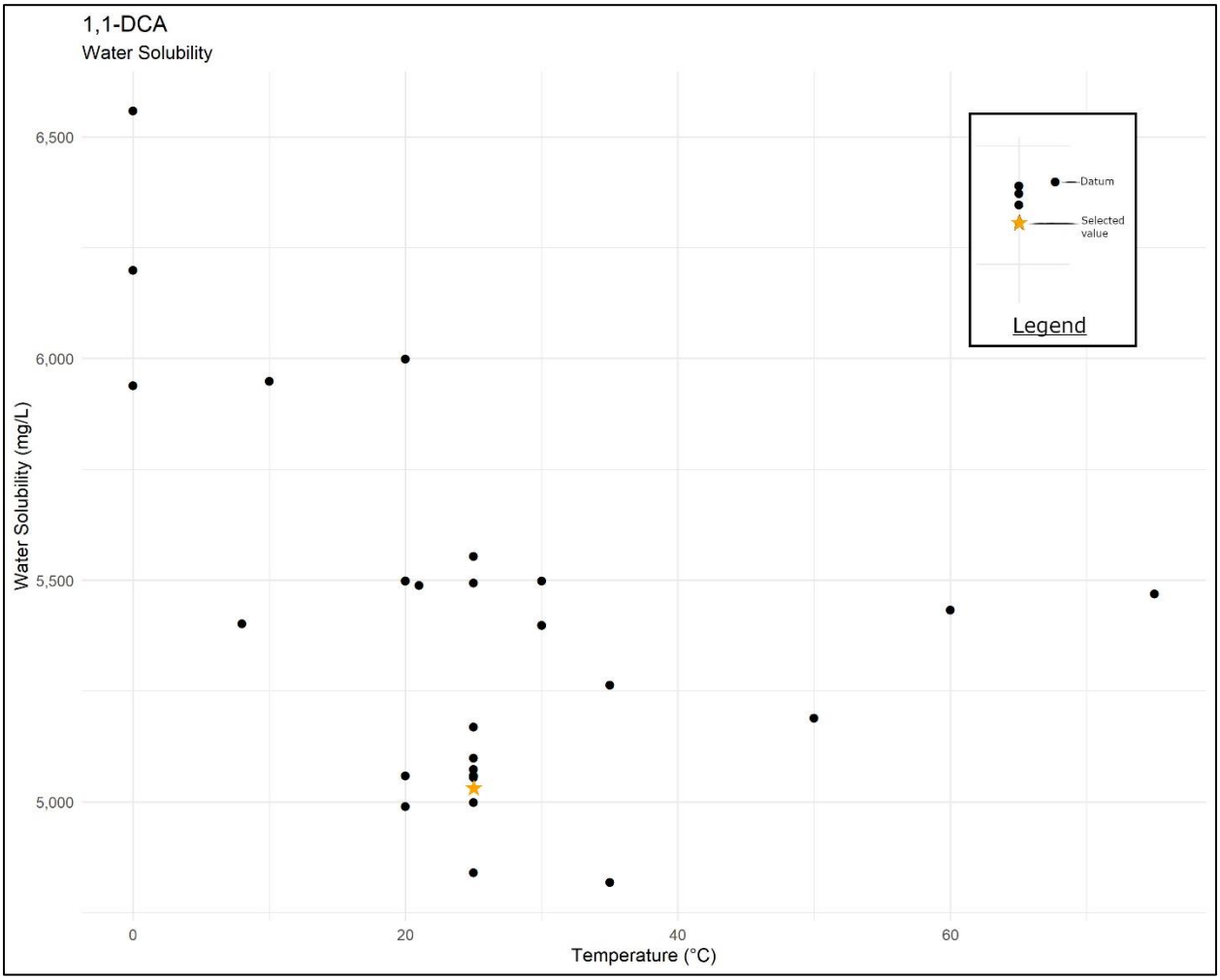


Figure_Apx C-4. Vapor Pressure of 1,1-Dichloroethane as a Function of Temperature

Vapor Density: Systematic review identified four vapor density data points that cover the range 3.4-3.44 (relative to air = 1 g/cm^3). The average vapor density was 3.43 ± 0.02. The value 3.44 (NCBI, 2020) was selected as the vapor density of 1,1-dichloroethane for this risk evaluation because it is in close agreement with the average of all the data identified, it has a high level of precision, was independently reported in multiple high-quality studies, and aligns with the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). The standard deviation of the collected data is relatively low, indicating that the value of this parameter is well-defined.

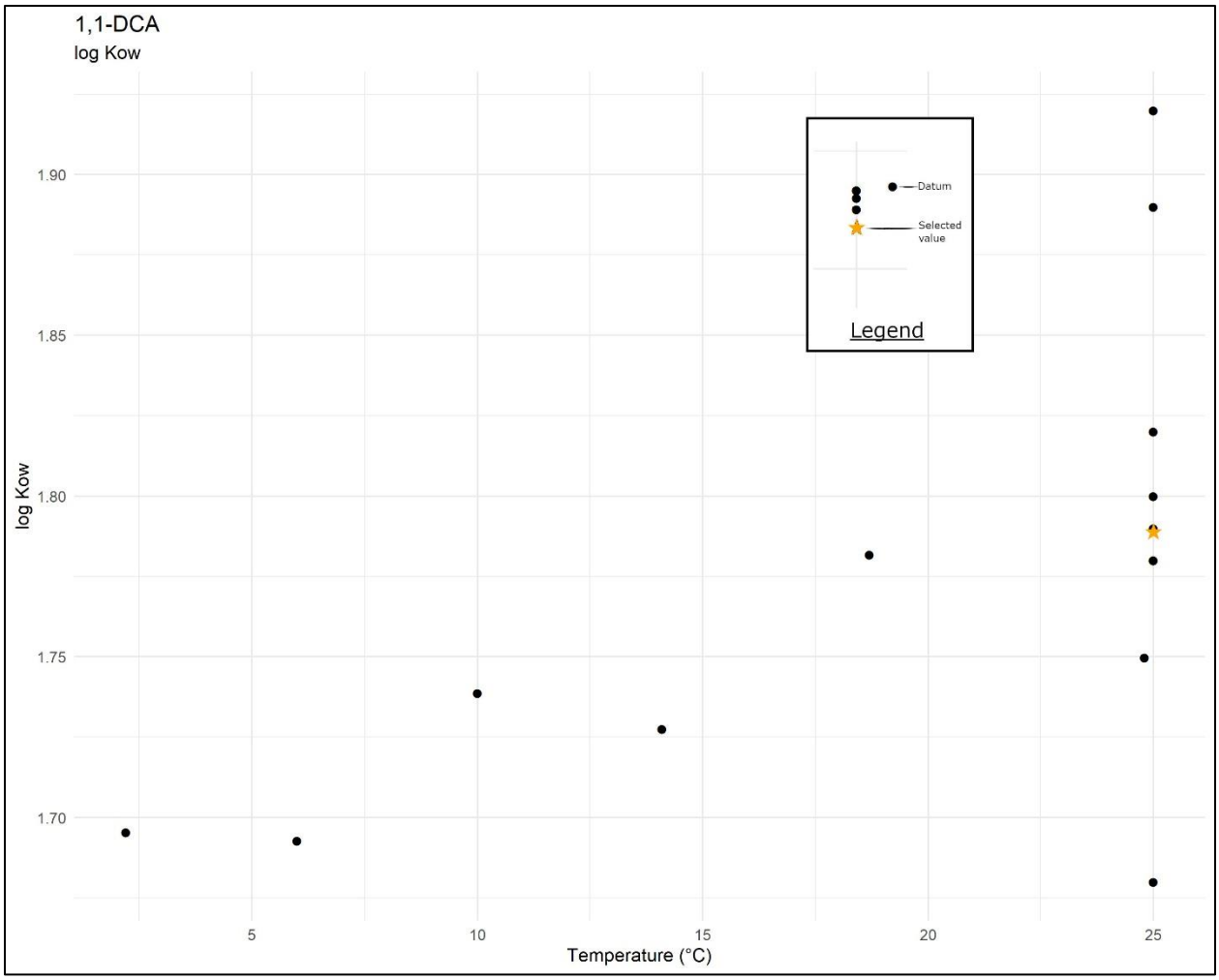
Water Solubility: Systematic review identified 32 water solubility data points, including 12 data points collected at 25 °C. The data collected under standard conditions cover the range 4,842 to 5,555 mg/L at 25 °C. The average water solubility of the 12 data points was 5,126 ± 202 mg/L at 25 °C. The variation of water solubility as a function of temperature is visualized in Figure_Apx C-5. The value 5,040 mg/L at 25 °C (NLM, 2018) was selected as the water solubility of 1,1-dichloroethane for this risk evaluation because it is in rough agreement with the mean and median of all the date identified, has a high level of precision, was independently reported in multiple high-quality studies, and aligns with the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). However, due to the spread of the

data identified and the inconsistencies between data reported at the same temperature, there is non-negligible uncertainty in this selected value. Alternative water solubility values could be appropriate at environmentally relevant conditions.



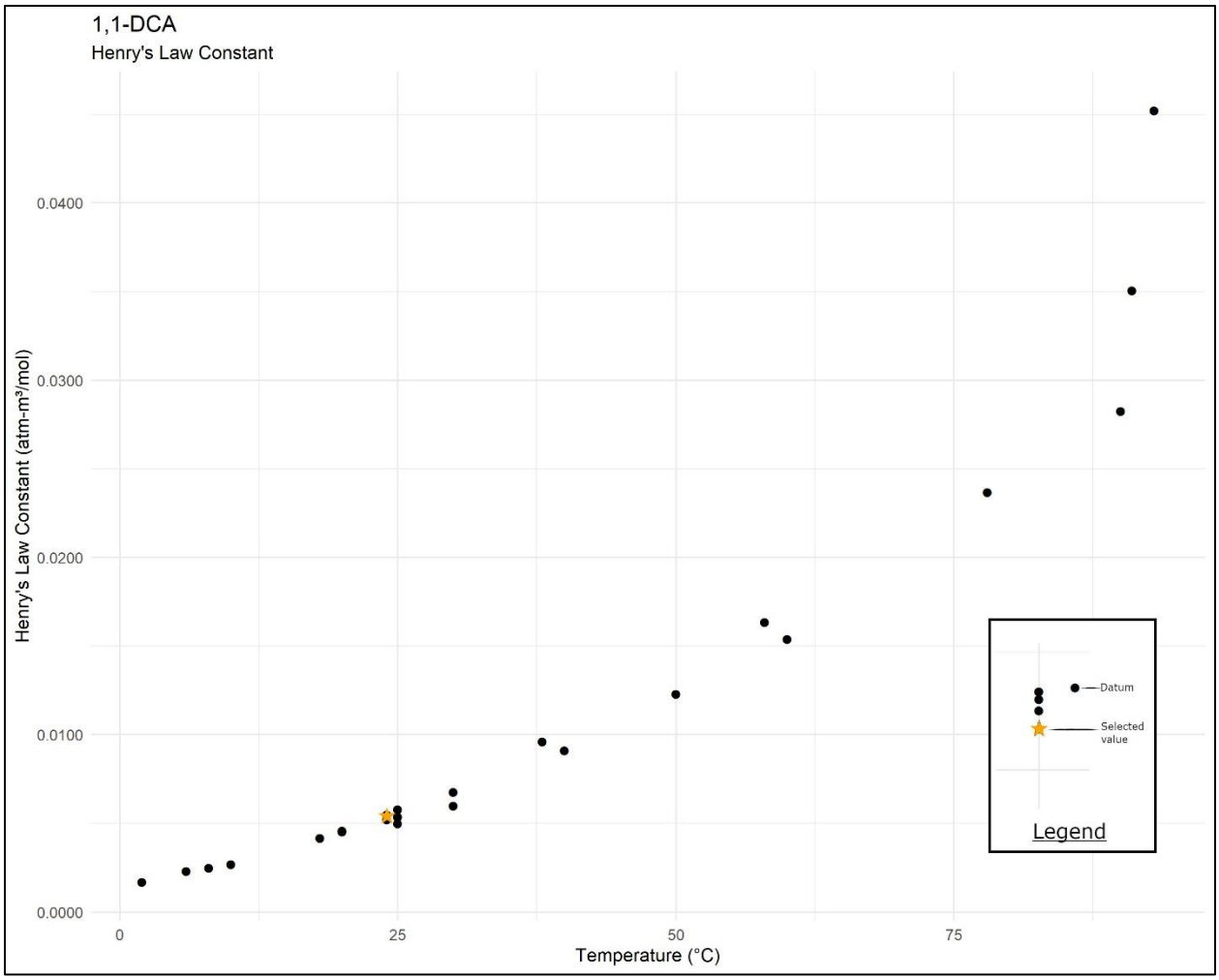
Figure_Apx C-5. Water Solubility of 1,1-Dichloroethane as a Function of Temperature

Octanol/Water Partition Coefficient (log K_{OW}): Systematic review identified 16 log K_{OW} data points, including 10 data points collected at 25 °C. The data collected under standard conditions cover the range of 1.68 to 1.92 at 25 °C. The average log K_{OW} was 1.80 ± 0.07 at 25 °C. The variation of low K_{OW} as a function of temperature is visualized in Figure_Apx C-6. The value 1.79 at 25 °C (Elsevier, 2019) was selected as the log K_{OW} of 1,1-dichloroethane for this risk evaluation because it is in close agreement with the data identified, was independently reported in multiple high-quality studies, and aligns with the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). The standard deviation of the collected data is relatively low, indicating this parameter is well-defined.



Figure_Apx C-6. Octanol/Water Partition Coefficient (log Kow) of 1,1-Dichloroethane as a Function of Temperature

Henry's Law Constant: Systematic review identified 25 Henry's Law constant (HLC) data points, including 7 data points collected at 24 to 25 °C. The data collected under standard conditions cover the range 0.005 to 0.0058 at 24 to 25 °C. The average HLC was 0.00542 ± 0.00026 at 24 to 25 °C. The variation of HLC as a function of temperature is visualized in Figure_Apx C-7. The value 0.00562 atm m^3/mol at 24 °C (NLM, 2018) was selected as the HLC of 1,1-dichloroethane for this risk evaluation because it is in close agreement with this analysis, was independently reported in multiple high-quality studies, and aligns with the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). The standard deviation of the collected data is relatively low, indicating that the value of this parameter is well-defined. Additionally, the HLC at non-standard temperatures can be determined using the results of the systematic review and Figure_Apx C-7—though there is increasing uncertainty at high temperatures and data should not be extrapolated outside of 0 to 100 °C.

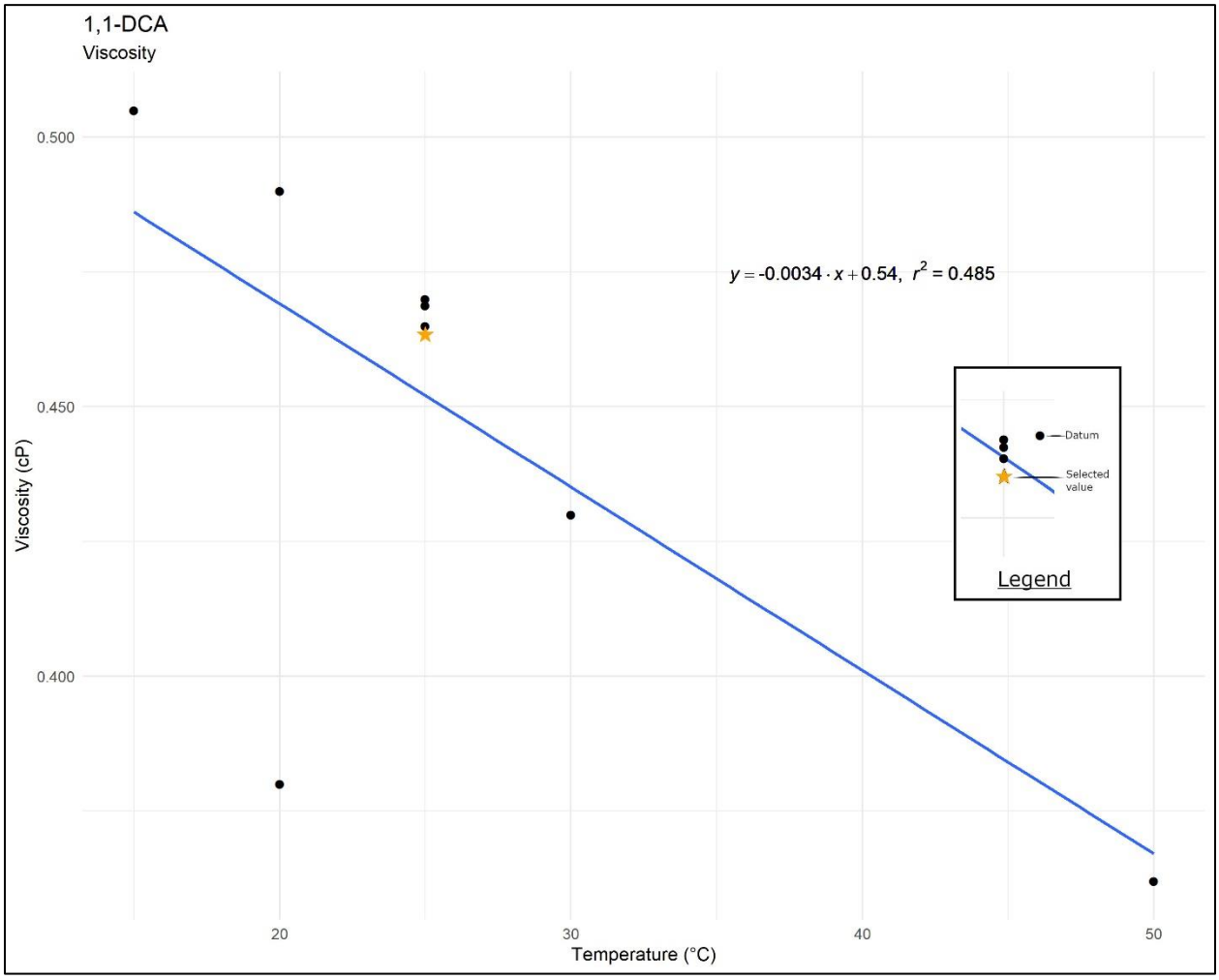


Figure_Apx C-7. Henry's Law Constant of 1,1-Dichloroethane as a Function of Temperature

Flash Point: Systematic review identified seven flash point data points that cover the range −17 to 14 °C. The flash point data collected include values measured using both closed cup and open cup techniques, with some sources reporting values for both techniques, and some sources not indicating the technique used. Closed and open cup measurement techniques generally result in a different value for flash point, and so for each reported value it is important to note the measurement technique used. The average flash point of the seven data was −8.2 ± 10.6 °C. The value −12 °C (Dreher et al., 2014) was selected as the flash point of 1,1-dichloroethane for this risk evaluation because it is in rough agreement with the data identified and was independently reported in multiple high-quality studies. Due to the multiple experimental methods for quantifying flash point (*e.g.*, open cup and closed cup), there is considerable variance in the data collected.

Autoflammability: Systematic review identified four autoflammability data points. All four data points were equal at 458 °C. The value 458 °C (Rumble, 2018b) was selected as the autoflammability of 1,1-dichloroethane for this risk evaluation because it is in absolute agreement with all identified data, is reported in multiple high-quality studies, and aligns with the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a).

Viscosity: Systematic review identified nine viscosity data points, including four data points collected at 25 °C. The data collected under standard conditions cover the range 0.464 to 0.47 cP at 25 °C. The average viscosity was 0.467 ± 0.003 cP at 25 °C. The variation of viscosity as a function of temperature is visualized in Figure_Apx C-8. The value 0.464 cP at 25 °C (Rumble, 2018c) was selected as the viscosity of 1,1-dichloroethane for this risk evaluation because it is in close agreement with the identified data, is reported in multiple high-quality studies, and aligns with the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). The standard deviation of the collected data is relatively low, indicating that this parameter is well-defined.



Figure_Apx C-8. Viscosity of 1,1-Dichloroethane as a Function of Temperature

Refractive Index: Systematic review identified 14 refractive index data points that cover the range 1.416-1.4171. The average refractive index was 1.4166 ± 0.0003. The value 1.4164 (Rumble, 2018a) was selected as the refractive index of 1,1-dichloroethane for this risk evaluation because it is in close agreement with the average of all data identified, was independently reported in multiple high-quality experimental studies, and aligns with the value reported in the final scope for 1,1-dichloroethane (U.S. EPA, 2020a). The standard deviation of the collected data is relatively low, indicating that the value of this parameter is well-defined.

Other Physical and Chemical Properties: Systematic review identified other physical and chemical properties for 1,1-dichloroethane of relevance for this risk evaluation. The following values were

selected for the indicated physical-chemical property of 1,1-dichloroethane for this risk evaluation; however, there is potential uncertainty for these selected values because systematic review did not identify a significant amount of data for these properties:

- Dielectric constant: 10.9 at 20 °C (NLM, 2018; Dreher et al., 2014) (n = 2); and
- Heat of evaporation: 30.8 kJ/mol at 25 °C (Dreher et al., 2014) (n = 1).

C.2 Fate and Transport

C.2.1 Approach and Methodology

EPA conducted a Tier I assessment to identify the environmental compartments (*i.e.*, surface water, sediment, biosolids, soil, groundwater, air) of major and minor relevance to the fate and transport of 1,1-dichloroethane. EPA then conducted a Tier II assessment to identify the fate pathways and media most likely to cause exposure as a result of environmental releases. Media-specific fate analyses were performed as described in Section 2.2.

C.2.1.1 EPI Suite™ Model Inputs

Measured values for bioconcentration and bioaccumulation factors for 1,1-dichloroethane were not found in the literature. As an alternative, these values were estimated using the BCF/BAF model in EPI Suite™. To set up EPI Suite™ for estimating these properties, the "Search CAS" function was used. The octanol-water partition coefficient (K_{OW}) used to estimate BCF and BAF was the recommended value in Table 2-1 in the physical and chemical properties section of the risk evaluation to conduct Level III fugacity modeling discussed in Appendix C.2.1.2 below, EPI Suite™ was run using default settings (*i.e.*, no other parameters were changed or input), with the following exceptions: measured K_{OC}, half-lives estimated from literature values, and emission rates from the Toxics Release Inventory reporting year 2020.

C.2.1.2 Fugacity Modeling

To inform how environmental releases of 1,1-dichloroethane partition between environmental compartments (air, water, sediment, and soil) the approach described by (Mackay et al., 1996) using the Level III fugacity model in EPI Suite™ was employed. The model predicts the partitioning of a substance released to an evaluative environment between air, water, soil, and sediment and identifies important intermedia transfer processes. The Level III Fugacity model is described as a steady-state, non-equilibrium model that includes the processes of degradation, advection (flow out of the evaluative environment) and intermedia transfer. The Level III Fugacity model requires fate assessor input for 1,1-dichloroethane physical and chemical properties, releases to each compartment of the evaluative environment, and half-lives in each compartment. Physical and chemical properties were taken directly from Table 2-1. Environmental degradation half-lives were taken from acceptable studies identified through systematic review as well as additional studies identified after the completion of systematic review. Where environmental degradation half-lives could not be found, they were estimated using EPI Suite™. All other input variables were left at their default settings. Release information was collected from the TRI and the NEI for the year 2020.

Table_Apx C-1 below lists release and half-life inputs for the Level III Fugacity model runs.

Table_Apx C-1. Inputs and Results or Level III Fugacity Modeling for 1,1-Dichloroethane

Environmental Releases (kg/yr TRI 2020)		Compartment Half-Lives (hours)	Data Source	Level III Results Percent Mass Distribution
Air	15,813	936	(U.S. EPA, 2012b)	85
Water	961	2,760[a]	(Washington and Cameron, 2001)	15
Soil	1	2,760	(Washington and Cameron, 2001)	<1
Sediment	N/A	2,760	(Washington and Cameron, 2001)	<1
[a] V acquired through modeling of a mixed contaminant plume under sulfate reducing conditions at a landfill.				

The results of the Level III Fugacity model using the reported releases indicate that emissions of 1,1-dichloroethane will primarily partition to air (85%) and water (15%) with less than 1 percent partitioning to soil and sediment. Thus, air and to a lesser extent water are expected to be important environmental compartments for 1,1-dichloroethane released to the environment.

C.2.1.3 Evidence Integration

The Draft Systematic Review Protocol (U.S. EPA, 2021c) states that during evidence integration, a determination of confidence in the range of fate endpoint(s) are made based on the study quality of contributing data point. The evaluations of the available studies of fate endpoints inform interpretations about the extent to which the data support a conclusion as interpreted from relevant fate and transport parameters determined from systematic review. Interpretations of the strength of a study, model, or data point that contributes to a fate endpoint for a chemical are judged and considered together. This culminates in a final conclusion about the extent to which the available evidence supports the environmental fate endpoint. The following summarizes the data availability, data quality, and data gap filling methods used to address environmental fate endpoints for evidence integration.

Fate in Air

No measured data on 1,1-dichloroethane atmospheric ·OH radical oxidation rates, overall environmental persistence, long-range transport or partitioning between environmental compartments were found in the literature search conducted as part of systematic review. Because no high-quality measured data were available for these endpoints, EPA relied on high quality physical-chemical properties data described in Section 2.1 of the risk evaluation (HLC, vapor pressure [VP], water solubility), EPI Suite™, and the OECD Pov and LRTP [long-range transport potential] Screening Tool to estimate key fate parameters used to assess the fate of 1,1-dichloroethane in air. EPI Suite™ has undergone peer review by EPA's Science Advisory Board (SAB, 2007).

Fate in Aquatic Environments (Surface Water, Sediments)

No data directly applicable to the fate of 1,1-dichloroethane in surface water were found in the literature search conducted as part of systematic review for the chemical. Because no high-quality measured data were available, EPA relied on high quality physical-chemical properties data described in Section 2.1 of this risk evaluation (*e.g.*, HLC, VP, WS, K_{OW}, K_{OC}), EPI Suite™ and the Point Source Calculator (PSC; see also Appendix J.1) Models (discussed further in the Section 3.3.3.2.4) to inform 1,1-dichloroethane partitioning to sediments and volatilization from water. EPI Suite™ has undergone peer review by the EPA Science Advisory Board (SAB, 2007). Conclusions on the biodegradation rates of 1,1-dichloroethane in aquatic environments (aerobic surface water and anaerobic sediments) were informed by the results of OECD Ready Biodegradability tests conducted on analogous chlorinated ethanes,

propanes, and butanes as well as aerobic groundwater biodegradation studies—the majority of which demonstrated slow biodegradation of 1,1-dichloroethane in aerobic aquatic environments. A single high-quality aerobic biodegradation study (Tabak et al., 1981) showing rapid biodegradation in the presence of added amendments was not considered to be representative of releases to pristine environments. Two microcosm studies of 1,1-dichloroethane biodegradation in anaerobic sediments collected from contaminated sites were identified after systematic review was completed and informed conclusions on aquatic sediment half-lives for 1,1-dichloroethane.

Fate in Terrestrial Environments

Limited data directly applicable to the fate of 1,1-dichloroethane in soil were found in the literature search conducted as part of systematic review. High and medium quality studies on the sorption of 1,1-dichloroethane to soil and sediment were used in combination with high quality physical-chemical properties data described in Section 2.1 of this risk evaluation (*e.g.*, HLC, VP, WS, K_{OW}), EPI Suite™, and the Hazardous Waste Delisting Risk Assessment Software (DRAS) to inform the fate assessment of 1,1-dichloroethane in soil. EPI Suite™ has undergone peer review by the EPA Science Advisory Board (SAB, 2007).

Conclusions on the biodegradation rates of 1,1-dichloroethane in aerobic and anaerobic soils were informed by studies identified after systematic review. Because data on the biodegradation of 1,1-dichloroethane in surface soils were not found, studies on the biodegradation of 1,1-dichloroethane conducted in laboratory groundwater systems and sediments were used to inform the potential rates of biodegradation in soils. The majority of the studies demonstrated slow biodegradation of 1,1-dichloroethane in anaerobic groundwater and sediment environments. Assumptions were therefore made that the rates of 1,1-dichloroethane biodegradation in soil will be similar. The groundwater and sediment biodegradation studies are discussed further in Appendices C.2.4.2 and C.2.3.2.

Conclusions on the fate of 1,1-dichloroethane drew from multiple studies identified after the completion of the systematic review literature search. These consisted of studies that determined biodegradation rates in groundwater from field studies, laboratory microcosm studies, and groundwater monitoring studies. The majority of the studies demonstrated slow biodegradation of 1,1-dichloroethane in groundwater. The groundwater biodegradation studies are discussed further in Appendix C.2.4.2.

Limited data directly applicable to the fate of 1,1-dichloroethane in landfills and landfill leachate plumes were found in the literature search conducted as part of systematic review. High- and medium-quality studies on the sorption of 1,1-dichloroethane to soil and sediment were used in combination with high-quality physical and chemical properties data described in Section 2.1 of the risk evaluation (*e.g.*, HLC, VP, WS, K_{OW}, K_{OC}) as well as the Hazardous Waste Delisting Risk Assessment Software (DRAS) to inform the fate assessment of 1,1-dichloroethane in landfills, landfill leachate plumes, and potential impacts on groundwater. Conclusions on the biodegradation rates of 1,1-dichloroethane in landfills and landfill leachate plumes were further informed by studies identified after systematic review. Because data on the biodegradation of 1,1-dichloroethane in landfills and landfill leachate plumes were not found, studies on the biodegradation of 1,1-dichloroethane conducted in sediments and laboratory groundwater systems were used to inform the potential rates of biodegradation. The studies are discussed further in Appendices C.2.4.1, C.2.4.2, and C.2.4.3 below. The majority of the studies demonstrated slow biodegradation of 1,1-dichloroethane. Assumptions were therefore made that the rates of 1,1-dichloroethane biodegradation in landfills and landfill leachate plumes will be similar.

No data directly applicable to the fate of 1,1-dichloroethane in biosolids were found in the literature search conducted as part of systematic review for the chemical. Because no high-quality measured data

were available, EPA relied on high-quality physical and chemical properties data described in Section 2.1 (*e.g.*, HLC, VP, WS, K_{OW}, K_{OC}), and the Office of Water Biosolids Tool to inform the fate and transport of 1,1-dichloroethane in land-applied biosolids and potential impacts on groundwater. The use of the Biosolids Tool is discussed further in Section G.1.2.5.

Environmental Persistence
EPA integrated the results of studies identified and evaluated during and after the systematic review to assess the environmental persistence of 1,1-dichloroethane. The studies are discussed in Appendix D and Section 2.2.

Removal in Wastewater Treatment
A high-quality study was used to inform the fate of 1,1-dichloroethane in publicly owned treatment works (POTWs). The study was conducted by EPA and monitored the fate of Priority Pollutants in 40 representative wastewater treatment plants across the United States. The results from 11 POTWs with data showed a wide range of removal of 1,1-dichloroethane but most values indicated greater than 50 percent removal. The evidence was supplemented with wastewater treatment plant monitoring studies for 1,1-dichloroethane identified after completion of systematic review that showed higher values and estimated removal rates from the Sewage Treatment Plant (STP) Model in EPI Suite™. EPI Suite™ has undergone peer review by the EPA Science Advisory Board (SAB, 2007). This information further informed conclusions regarding a range of removal of 1,1-dichloroethane in POTWs. The studies are discussed in Appendix C.2.5.2.

Bioconcentration/Bioaccumulation
No data were found on the bioaccumulation/bioconcentration potential of 1,1-dichloroethane. In the absence of data, EPA relied on high quality physical-chemical properties data described in Section 2.1 of the risk evaluation (K_{OW}), EPI Suite™, and the Office of Water BCF/BAF estimation methodology described in *Ambient Water Quality for the Protection of Human Health* (U.S. EPA, 2003c) to estimate the values. Estimated BCF/BAF values were compared to available measured values for similar halogenated ethanes and propanes to inform the reliability of the estimated values for 1,1-dichloroethane. EPI Suite™ has undergone peer review by the EPA Science Advisory Board (SAB, 2007). The selection of BCF and BAF values for 1,1-dichloroethane is discussed in Appendix C.2.6.

C.2.2 Air and Atmosphere

1,1-dichloroethane is not expected to undergo significant direct photolysis because it does not absorb radiation in the environmentally available region of the electromagnetic spectrum that has the potential to cause molecular degradation (HSDB, 2008). 1,1-Dichloroethane in the vapor phase will be degraded by reaction with photochemically produced hydroxyl radicals in the atmosphere. A half-life of 39 days was calculated from an estimated rate constant of 2.74×10^{-13} cm^3/molecules-second at 25 °C, assuming an atmospheric hydroxyl radical concentration of 1.5×10^6 molecules/cm^3 and a 12-hour day (U.S. EPA, 2012b). Based on an estimated octanol air partition coefficient (Koa) of 269, 1,1-dichloroethane is not expected to associate strongly with airborne particulates. The results of the Level III Fugacity Model in EPI Suite™ using environmental releases of 1,1-dichloroethane reported in the 2020 Toxics Release Inventory discussed in Appendix C.2.1.2 indicate that at steady state, greater than 75 percent of the mass of 1,1-dichloroethane released to the environment will partition to the air compartment.

With an expected atmospheric half-life of 39 days, significant vapor pressure (228 mmHg at 25 °C) and reported releases to air, the potential for long-range transport was assessed using the OECD Pov and LRTP Screening Tool. The tool includes features that are recommended by the OECD expert group on multimedia modeling. It incorporates a fugacity based steady state multimedia mass balance model of a

global evaluative environment representing soil, water, and the troposphere. In addition to calculating overall environmental persistence (Pov), the model provides two other indicators of long-range transport potential, characteristic travel distance (CTD) and transfer efficiency (TE). CTD is the distance from the point of release of the chemical to the point at which the concentration of the chemical has dropped to 1/*e*, or about 37 percent of its initial value. CTDs are calculated for emissions to air and water and only transport in the medium that receives the release is considered. Because soil is not considered mobile, no CTD is calculated for emissions to soil. The tool considers multiple emission modes to air, water and soil and reports maximum values for Pov, CTD (with the exception of soil), and TE. Transfer efficiency is the ratio of the mass flux of a substance into an environmental compartment and the emissions mass flux. TE is calculated for emissions to air, water, and soil, and is an indicator of how much of an emission reaches a distant target.

The 1,1-dichloroethane chemical properties required as input for the model were taken from Table 2-1, and media-specific half-lives were derived after consideration of the range of half-life values reported in the respective environmental fate discussions for the medium. The tool estimated an overall environmental persistence of 129 days, a characteristic travel distance of 19,031 km and a transfer efficiency of 1.9 percent. These results suggest 1,1-dichloroethane may travel long distances, but a low percentage of the release will reach a distant target. Relative to the Pov and long-rang transport (LRTP) of 10 reference POP chemicals in the tool's database, 1,1-dichloroethane has lower overall environmental persistence and characteristic travel distance.

C.2.2.1 Key Sources of Uncertainty in the Fate Assessment for Air and the Atmosphere

The assessment of the fate of 1,1-dichloroethane in air relied on estimated OH radical oxidation half-lives from the AOPWIN™ model and the Level III Fugacity model in EPI Suite™. The assumptions, applicability domain and accuracy of the AOP model are discussed in the EPI Suite™ help menus. Accurate inputs are critical for fugacity modeling. Inputs to the level III fugacity model include half lives in various media, physical chemical properties, and emissions to air, water and soil. Model results are significantly impacted by emissions assumptions. Thus, for optimal use of the model, accurate emissions data and, if possible, complete emissions inventories should be used.

C.2.3 Aquatic Environments

1,1-Dichloroethane has a hydrolysis half-life of approximately 61 years (Jeffers et al., 1989), therefore hydrolysis is not expected to be an important fate process for 1,1-dichloroethane in aquatic environments. Based on a measured K_{OC} of 31 (Poole and Poole, 1999), partitioning from the water column to suspended and benthic sediments is not expected to be an important process for 1,1-dichloroethane. An HLC constant of 0.00562 atm·m^3/mol at 25 °C, calculated based on a vapor pressure of 228 mmHg at 25 °C and a water solubility of 5,040 mg/L, indicates that 1,1-dichloroethane may volatilize from water surfaces. Biodegradation in water is not expected to be an important loss process for 1,1-dichloroethane. based on aerobic aquatic biodegradation studies on 1,1-dichloroethane and other chlorinated ethanes, propanes and butanes. Overall evidence suggests that biodegradation of 1,1-dichloroethane in the water column may be possible, but rates are expected to be slow and volatilization from water will occur more rapidly than biodegradation.

C.2.3.1 Surface Water

1,1-Dichloroethane released to surface water will be subject to loss primarily via volatilization to air. Biodegradation and sorption to suspended and benthic sediments will be minor removal processes. A half-life for the volatilization from a model river was estimated using the WVol Model in EPI Suite™ (U.S. EPA, 2012b), which follows a two-film concept for estimating the flux of volatiles across the air-

water interface (Liss and Slater, 1974). For a model river 1 m deep with a current velocity of 1 meter per second (m/s) and wind velocity of 5 m/s, a volatilization half-life of approximately 1 hour was calculated. Although volatilization is expected to be rapid, some of the substance will remain in water due to its water solubility (5,040 mg/L) and depending on where its continuous releases to water are occurring. Biodegradation in water is not expected to be an important loss process for 1,1-dichloroethane based on a single aerobic aquatic biodegradation study on 1,1-dichloroethane as well as ready biodegradability studies on other chlorinated ethanes and chlorinated propanes and chlorinated butanes. A study using multiple inoculum subculture transfers promoting acclimation resulted in up to 91 percent biodegradation with loss by volatilization also observed (Tabak et al., 1981). However, these results do not appear to be representative of releases of 1,1-dichloroethane to the environment. The Japanese National Institute of Technology and Evaluation (NITE) collected OECD method "301C Ready Biodegradability data for several chlorinated ethanes" (chloroethane (NITE, 2023g), 1,2-dichloroethane (NITE, 2023b), chloropropanes (2-chloropropane (NITE, 2023f), 1,2-dichloropropane (NITE, 2023c), 1,2,3-trichloropropane (NITE, 2023d)), chlorobutanes (1-chlorobutane (NITE, 2023a), and 1,4-dichlorobutane (NITE, 2023e)). The study results indicated that 0 to 8 percent biodegradation occurred in up to 4 weeks. Overall, these studies suggest that aerobic biodegradation of 1,1-dichloroethane in the water column may be possible, but rates are expected to be slow and volatilization from water will occur more rapidly than biodegradation.

Based on a measured K_{OC} value of 31 (Poole and Poole, 1999), 1,1-dichloroethane is not expected to bind strongly to sediment or suspended organic matter in the water column.

C.2.3.2 Sediments

1,1-Dichloroethane released to water is not expected to significantly partition to organic matter in suspended and benthic sediments based on its measured K_{OC} of 31 (Poole and Poole, 1999). K_{OC} represents the ratio of the concentration of 1,1-dichloroethane sorbed to organic carbon in sediment or soil to the concentration of 1,1-dichloroethane in the overlying water at equilibrium. For comparison, highly hydrophobic chemicals known to partition to and accumulate in sediments such as polychlorinated biphenyls (PCBs) have measured K_{OC} values of in the range of 10,000 to 100,000 or greater. Biodegradation of 1,1-dichloroethane has been shown to occur in freshwater sediment microcosms isolated from contaminated sites. Hamonts et al. (2009) constructed anaerobic microcosms from sediments collected from Zenne River near Brussels, Belgium, with a history of chlorinated aliphatic hydrocarbon exposure. The source of exposure was the infiltration of contaminated groundwater into the river. Reduction of 1,1-dichloroethane within 13 to 46 days was observed for 9 of the 12 sampling sites with conversion from 1,1-dichloroethane to chloroethane and ethane. High organic matter content of the sediments was associated with the most rapid biodegradation with the organic matter perhaps serving as an electron donor for the dechlorination of 1,1-dichloroethane.

Şimşir et al. (2017) observed biodegradation of 1,1-dichloroethane in microcosms using contaminated anaerobic sediment samples collected from the interface of contaminated groundwater from a fractured bedrock aquifer and surface water in Third Creek, a Tennessee River tributary in Knoxville, Tennessee. 1,1-Dichloroethane and lactate were added to the microcosms that were then incubated. After 20 months, 75 to 100 percent of the added 1,1-dichloroethane had been converted to chloroethane. Analysis of the microbial populations present showed a relatively uniform distribution over the 300 m site. It was noted that at some sites, members of the bacteria family *Methylococcaceae* were found in low abundance, suggesting the possibility of aerobic co-metabolic biodegradation of 1,1-dichloroethane at the aerobic-anaerobic transition zone. The distribution of microorganisms capable of aerobic co-metabolism of 1,1-dichloroethane is uncertain. Kuhn et al. (2009) used compound stable isotope analysis for *cis*-dichloroethylene and vinyl chloride to confirm the occurrence and determine the extent of

biodegradation of the compounds in the contaminated aquifer and river sediments of the Zenne River in Belgium also studied by Hamonts et al. (2009). The study identified some zones where indigenous microorganisms biodegraded the substances and other zones where significant biodegradation did not occur. This suggests that even at a relatively small scale, biodegradation of chlorinated alkanes and alkenes may not be uniformly distributed and may or may not occur.

C.2.3.3 Key Sources of Uncertainty in the Fate Assessment for Aquatic Environments

Uncertainty in rates of biodegradation and volatilization are key sources of uncertainty in the fate assessment for aquatic environments. There is limited evidence on the aerobic and anaerobic biodegradation of 1,1-dichloroethane in uncontaminated aquatic environments under environmental conditions. The majority of the studies consist of laboratory microcosm studies or field studies with microbial populations which have developed and acclimated to biodegrade 1,1-dichloroethane through addition of electron donors and/or acceptors over extended periods of exposure. As such, extrapolating rates of biodegradation observed in the laboratory study to environmental biodegradation rates introduces uncertainty. The WVol Model in EPI Suite™ is a screening level model that estimates the rate of volatilization of a chemical from a model river and lake. The program's default parameters for a model river were selected to yield a half-life that may be indicative of relatively fast volatilization from environmental waters due to default current velocity, river depth, and wind velocity. The default parameters for the lake yield a much slower volatilization rate. The low wind velocity and current speed are indicative of a pond (or very shallow lake) under relatively calm conditions. These default parameters were selected to specifically model a body of water under calm conditions. Although physical chemical properties of the modeled substance and wind speed, water flow velocity and water depth can be modified by the user; however, the model does not employ all site-specific environmental parameters that effect the rates of volatilization. Therefore, rates of volatilization at a specific location under specific environmental conditions could be over or underestimated by the model.

C.2.4 Terrestrial Environments

The measured organic carbon partition coefficient of 31 (Poole and Poole, 1999) for 1,1-dichloroethane indicates it will have a low affinity for organic matter in terrestrial environments and thus be subject to transport processes including migration with water through surface soil and unlined landfills to groundwater. 1,1-Dichloroethane releases to soil surfaces may also be subject to volatilization based on its vapor pressure (228 mmHg at 25 °C) and Henry's Law constant (0.00526 atm-m^3/mol). 1,1-Dichloroethane is expected to be bioavailable in soil pore water and groundwater due to its water solubility of 5,040 mg/L. 1,1-Dichloroethane has been detected in groundwater and landfill leachate; however, because 1,1-dichloroethane can be formed from the anaerobic biodegradation of 1,1,1-trichloroethane (1,1,1-trichloroethane), there is uncertainty whether its presence results from the release and anaerobic biodegradation of 1,1,1- trichloroethane or the release of 1,1-dichloroethane itself.

C.2.4.1 Soil

When released to land, 1,1-dichloroethane may migrate from the surface downward due to its density and relatively low affinity for soil organic matter. Volatilization from soil surfaces may also occur. Once below the soil surface. The zone between land surface and the water table within which the moisture content is less than saturation contains soil pore space that typically contains air or other gases. 1,1-Dichloroethane will partition between four phases in the unsaturated (vadose) zone, soil solids, soil water, interstitial air, and if present at sufficiently high concentrations—nonaqueous phase liquid.

If released to land in sufficient quantities, 1,1-dichloroethane could be present and persist as a non-aqueous phase liquid (NAPL) and more specifically as a dense non-aqueous phase liquid (DNAPL) due to its greater density relative to water. 1,1-Dichloroethane as DNAPL can migrate through the vadose

zone under the influence of gravity and then vertically downward through groundwater until it reaches an impermeable layer where it subsequently becomes a continuous source of contamination in the aquifer (Poulsen and Kueper, 1992). However, at the concentrations expected to result from releases to soil from the conditions of use (COUs) under TSCA consideration, 1,1-dichloroethane is not expected to be present as DNAPL but rather in the dissolved phase only. Dissolved 1,1-dichloroethane moves with soil water; however, the rate at which it moves may be slower than soil water due to its sorptive interaction with soil and other factors. Although 1,1-dichloroethane has a relatively low organic carbon: water partition coefficient (K_{OC} = 31), some will be partitioned into organic matter on soil particle surfaces in the vadose zone and in groundwater. Particulate-bound 1,1-dichloroethane generally has a lower potential to migrate to groundwater because particles may be retained in soil due to a physical filtering effect. 1,1-Dichloroethane has a relatively high vapor pressure (228 mmHg at 25 °C) and can exist as a vapor in subsurface voids. This vapor is mobile and can spread through diffusion. Vapor phase transport can also result in releases from the subsurface to the atmosphere.

Biotic and abiotic processes have been shown to degrade 1,1-dichloroethane in soil; however, a number of environmental conditions appear to be necessary for degradation to occur. For biotic degradation (biodegradation) to occur, the presence of microorganisms with the capability of degrading the compound is required as well as favorable environmental conditions that impact biodegradation including temperature, pH, salinity and water content, redox potential, and availability of nutrients. Where high concentrations of 1,1-dichloroethane or other contaminants exhibit toxicity to microorganisms, or 1,1-dichloroethane is present at concentrations too low to induce degradative enzymes, biodegradation may not occur.

1,1-Dichloroethane has been shown to biodegrade slowly in soil under both aerobic and anaerobic conditions but by different microbial populations and different mechanisms. 1,1-Dichloroethane can be biodegraded under aerobic conditions by means of co-metabolic transformation reactions. These are reactions that are catalyzed by microbial oxygenase enzymes, molecular oxygen, and a source of reducing equivalents that yield no carbon or energy benefits to the biodegrading microorganisms (Alvarez-Cohen and Speitel, 2001; Horvath, 1972). The chlorinated solvent oxidation products of the oxygenase reaction may react and be further degraded to CO_2 by microorganisms. These reactions can be carried out by a wide range of oxygenase-expressing microorganisms, including those that utilize a range of nonchlorinated aliphatics and some aromatics, as energy and/or carbon source. (Alvarez-Cohen and Speitel, 2001).

Soils can become anaerobic as microorganisms consume oxygen as a terminal electron acceptor to biodegrade soil organic matter and when soil is saturated or flooded. Whether anaerobic biodegradation occurs, and the rate and extent of anaerobic biodegradation, are influenced primarily by the microorganisms present and the oxidation-reduction (redox) reactions that occur. As oxygen in soils becomes depleted and the soil becomes anaerobic, microbial processes shift generally in a sequence from aerobic respiration to nitrate reduction (denitrification), manganese reduction, iron (III) reduction, sulfate reduction, and finally methanogenesis. Several of these processes can occur at the same time in close proximity, or one process may be relatively dominant. The anaerobic biodegradation of 1,1-dichloroethane is carried out by microorganisms mediating oxidation-reduction reactions where soil organic matter or organic contaminants act as electron donors and 1,1-dichloroethane acts as an electron acceptor. This process is known as reductive dechlorination and is an important biodegradation pathway for 1,1-dichloroethane. Generally, the reduction involves the replacement of a chlorine substituents by hydrogen (hydrogenolysis).

No studies were found on the anaerobic biodegradation of 1,1-dichloroethane in surface soils (upper soil horizons). However, anaerobic biodegradation pathways may be similar for anaerobic soil, aquifers, and sediments, as well as anaerobic digestion waste treatment where similar microbial populations and conditions are present. Studies on the anaerobic biodegradation on 1,1,1-trichloroethane are useful in informing the pathway for 1,1-dichloroethane anaerobic biodegradation as it is known is known to undergo reductive dehalogenation to 1,1-dichloroethane where degradation pathways converge.

A critical review of anaerobic degradation of 1,1,1-trichloroethane and its degradation products identified several studies demonstrating the microbially mediated sequential reductive dechlorination of 1,1,1-trichloroethane to 1,1-dichloroethane and chloroethane (Scheutz et al., 2011). The process has been observed in laboratory experiments with marine sediments, methanogenic biofilm reactors, pure cultures, in batch reactors, and aquifer microcosms. In some of these studies, 1,1-dichloroethane was the primary product of trichloroethane dechlorination, while in other studies chloroethane was the observed terminal dechlorination product presumably forming as a result of sequential dechlorination from 1,1,1-trichloroethane to 1,1-dichloroethane to chloroethane.

Overall, the results of these studies show that (1) biological reductive dechlorination of trichloroethane to chloroethane occurs in anaerobic systems; (2) dechlorination of 1,1-dichloroethane occurs more slowly than dechlorination of trichloroethane; and (3) 1,1-dichloroethane or chloroethane can form as terminal products of the dechlorination reaction, depending on the microbiology and/or redox chemistry of the system.

Vogel (1987) studied the biotic and abiotic transformations ^{14}C 1,1,1-trichloroethane and related compounds including ^{14}C 1,1-dichloroethane under methanogenic conditions. ^{14}C 1,1-dichloroethane was incubated with a mixed methanogenic culture and the addition of acetate as a primary substrate (electron donor) in a small, fixed film reactor with a liquid detention time of 4 days. The reactor had been previously dosed with ^{14}C 1,1,1-trichloroethane. ^{14}C 1,1-dichloroethane was also added to anaerobic batch fermenters containing an inoculum from an anaerobic column and sampled for $^{14}CO_2$ over time. 1,1-Dichloroethane fed to the small, fixed film reactors was partially mineralized to $^{14}CO_2$. About 20 percent mineralization of 1,1-dichloroethane also occurred in the batch fermenters over 84 days.

Sun (2002) observed the reductive dechlorination of 1,1-dichloroethane by a microorganism isolated from a sediment microcosm capable of dechlorinating trichloroethane. Sequential dechlorination from trichloroethane to 1,1-dichloroethane was observed, with some accumulation, followed by conversion to chloroethane. Acetate, trichloroethane and hydrogen or formate were required for growth. When the microorganism was added to anoxic aquifer sediments from sites contaminated with PCE, trichloroethane, and dichloroethane, trichloroethane was completely converted to chloroethane within 2 months—presumably via sequential dechlorination involving transient 1,1-dichloroethane.

Grostern (2006) followed the biodegradation of 1,1,1-trichloroethane, and 1,1-dichloroethane by a mixed anaerobic microbial culture derived from the groundwater and solids of a 1,1,1-trichloroethane contaminated site. In part of the experiment, anaerobic microcosms were established with the cultures. Methanol, ethanol, acetate, and lactate were added as the electron donors and 1,1-dichloroethane as the electron acceptor. Dechlorination in the 1,1-dichloroethane treatment bottles started with no lag and was complete in 12 days. Methanogenesis occurred throughout 1,1-dichloroethane degradation.

U.S. EPA (2013a) compiled first-order biodegradation rate constants for 1,1-dichloroethane from the literature. Most of the data were collected from contaminated sites. The type of study, biogeochemical conditions, and rate constant statistics for multiple values were reported.

Table_Apx C-2. First-Order Biodegradation Rate Constants for 1,1-Dichloroethane

Type of Study	Biogeochemical Conditions	First-Order Rate Constants (day^{-1})						Number of Studies	Reference
		Min	25th	Median	75th	Max	Mean		
Field	Reductive dechlorination	0.0005	0.0005	0.0008	0.0019	0.0033	0.0014	3	(Aziz et al., 2000)
Lab	Not specified	0.0044				0.0096			(Aziz et al., 2000)
Lab and Field	All studies	0	0	0.001	0.014	0.131	0.017	25	(Suarez and Rifai, 1999)
Lab	Aerobic cometabolism	0.014	0.019	0.047	0.123	0.131	0.067	5	(Suarez and Rifai, 1999)
Field	Reductive dechlorination	0				0.011	0.002	16	(Suarez and Rifai, 1999)
Lab	Reductive dechlorination	0.028				0.044	0.036	2	(Suarez and Rifai, 1999)
Field	Reductive dechlorination: sulfate-reducing	0	0	0	0.001	0.028	0.003	13	(Suarez and Rifai, 1999)
Field	Reductive dechlorination: methanogenesis						0.006	3	(Suarez and Rifai, 1999)

When converted to 1,1-dichloroethane, biodegradation half-lives assuming first-order kinetics with the reported rate constants spanning from 72 days to 3.8 years.

C.2.4.2 Groundwater

Releases of 1,1-dichloroethane to land (*e.g.*, landfills without adequate leachate controls or land application of contaminated biosolids) may migrate through soil and reach groundwater. The measured organic carbon partition coefficient of 31 for 1,1-dichloroethane indicates it will have a low affinity for organic matter and will not significantly sorb to suspended solids in groundwater. At the groundwater concentrations expected to result from releases of 1,1-dichloroethane COUs, 1,1-dichloroethane will likely behave as a freely soluble substance. 1,1-Dichloroethane has a hydrolysis half-life of approximately 61 years (Jeffers et al., 1989). Therefore, losses of 1,1-dichloroethane from groundwater are most likely due to biodegradation, which is expected to be slow. A single study was found on the rates of biodegradation of 1,1-dichloroethane in groundwater. Washington (2001) developed an analytical solution for first-order degradation coupled with advective losses and adsorption to solve for degradation constants for perchloroethene, trichloroethene, 1,1,1-trichloroethane, 1,1-dichloroethane, and chloroethane under sulfate reducing conditions at a landfill field site in southeastern Pennsylvania. Samples were collected 4 times yearly from 13 monitoring wells that were spaced to include water from the upper watershed boundary to the most down-gradient discharge location. A degradation half-life of 115 days was calculated for 1,1-dichloroethane. It is important to note that conditions at the site modeled were much more conducive to biodegradation of 1,1-dichloroethane relative to other more aerobic and less contaminated sites. At less contaminated sites, where reducing conditions might not exist or where organic electron donors might not be adequately present, 1,1-dichloroethane biodegradation half-lives can be on the order of years. Huff (2000) calculated first-order decay constants using the BIOCHLOR

Model and changes in 1,1-dichloroethane concentrations up gradient and down gradient from monitoring wells along an apparent groundwater path at a contaminated petrochemical reclamation site in Texas. Redox conditions ranged from sulfate reducing to methanogenic, as indicated by the presence of methane in groundwater and the range of molecular hydrogen concentrations. An increased ratio of 1,2-dichloroethane to 1,1,2-trichloroethane downgradient from the assumed contaminant source area supported the conclusion that reductive dechlorination was occuring. Reductive dechlorination of chlorinated ethanes apparently occurred to a lesser extent than chlorinated ethenes, indicating relatively less potential for natural attenuation of chlorinated ethanes. Apparent first-order decay constants, which yielded simulated concentrations in best agreement with observed changes in concentrations along the segments of the approximate groundwater flowpath, were slightly greater than literature values and resulted in half-lives ranging from 1.5 to 6.9 years.

The possible groundwater concentrations resulting from releases of 1,1-dichloroethane to land under the COUs are discussed in detail in Section G.1.1.

C.2.4.3 Landfills

Releases of 1,1-dichloroethane to land via disposal to landfills (TRI 2015–2020 average 1 kg/year, EPA estimated <22,682 kg/year to RCRA Subtitle C Hazardous Waste Landfills) may occur across as many as 138 sites under the TSCA COUs. The required design and operating procedures of Subtitle C landfills minimize the movement of leachate from the landfill. The combination of the expected waste management practices and the relatively low and disperse quantity of 1,1-dichloroethane disposed of in landfill suggests that the contamination of groundwater by 1,1-dichloroethane released to Subtitle C landfill will not be an important pathway. However, releases of 1,1-dichloroethane to landfills without adequate leachate controls may migrate through soil and reach groundwater.

Two studies that measured the concentration of 1,1-dichloroethane in landfill leachate in the United States were found through systematic review. Concentrations ranged from not detected to 46,000 ng/L from 11 samples collected between 1984 and 1993. 1,1-Dichloroethane is a dense liquid with a low affinity for soil organic carbon and water solubility of approximately 5,040 mg/L. Landfill leachate is generated by excess rainwater percolating through the waste layers of a landfill. Pollutants such as 1,1-dichloroethane can be transferred from the landfilled waste material to the percolating leachate through combined physical, chemical, and microbial processes (Christensen et al., 2001). Compounds in leachate entering an aquifer will be subject to dilution as the leachate mixes with the groundwater. 1,1-Dichloroethane does not appreciably bind to aquifer suspended solids and biodegradation can be slow; thus, dilution may be the only attenuating factor. Due in part to slow groundwater flow rates and complex (tortuous) flow paths, contaminants such as 1,1-dichloroethane can form plumes. Concentrations in a plume can vary but are generally highest in the center of the plume and closest to the source and decrease with distance from the source.

When a landfill leachate plume reaches groundwater, its dissolved organic carbon can significantly impact the native groundwater microbial communities and might lead to an increase in microbial populations and activity. Microorganisms capable of carrying out a variety of processes, mostly reductive (denitrification, manganese, iron, and sulfate reduction, methanogenesis), have been found in leachate plumes (Ludvigsen et al., 1999; Beeman and Suflita, 1990, 1987), and under some conditions, may be able to partially biodegrade 1,1-dichloroethane to chloroethane. However, the rates of biodegradation are expected to be slow.

Migration of 1,1-dichloroethane disposed of in landfills under the COUs to groundwater is not expected to be a significant exposure pathway. To support this conclusion, range-finding estimates were made

using the Hazardous Waste Delisting Risk Assessment Software (DRAS) (U.S. EPA, 2020b). DRAS performs a multi-pathway and multi-chemical risk assessment to evaluate the acceptability of a petitioned waste to be disposed in a Subtitle D landfill or surface impoundment instead of under RCRA Subtitle C requirements. For landfills, DRAS models a mismanagement scenario at an unlined Subtitle D landfill where releases to groundwater are not controlled and 30 days of waste is always left uncovered at the surface and subject to air emission and runoff. DRAS uses leachate analysis of the waste to model exposure of nearby residents to impacted groundwater via ingestion, shower-inhalation, and dermal exposure. Using totals analysis of the waste, DRAS models exposure of nearby residents to surface water and fish ingestion impacted by runoff, inhalation of particulate and volatile emissions from the uncovered waste, and incidental ingestion of residential soil contaminated by settled particulate emissions from the waste.

For the assessment of 1,1-dichloroethane, EPA used the estimated 1,1-dichloroethane groundwater concentrations resulting from leachate contamination to make an initial determination of the importance of the landfill leachate groundwater exposure pathway. Further discussion and details of the modeling are provided in Section G.1.2.3.

C.2.4.4 Biosolids

Chemical substances in wastewater undergoing biological wastewater treatment can be removed from the wastewater by processes including biodegradation, sorption to wastewater solids, and volatilization. As discussed in Section C.2.5.2, 1,1-dichloroethane is expected to be removed in wastewater treatment primarily by volatilization with little removal by biodegradation or sorption to solids. Chemicals removed by sorption to sewage sludge can enter the environment when sewage sludge is land-applied following treatment to meet standards. The treated solids are known as biosolids.

The removal of a nonbiodegradable neutral organic chemical present in WWTP influent via sorption to sludge is evaluated by considering its partitioning to the organic carbon in suspended solids. Because organic substances predominantly partition to organic carbon, the measured sorption coefficient is normalized to the fraction of organic carbon (f_{OC}) present in the solid to yield the chemical's organic carbon:water partition coefficient (K_{OC}).

The organic carbon:water partition coefficient is the expressed as follows:

$$K_{oc} = K_d / f_{oc}$$

Where:

K_d = solids:water partition coefficient
f_{oc} = fraction of organic carbon

As the organic carbon:water partition coefficient (K_{OC}) increases, more of the chemical will be found associated with the suspended solids.

Based on its K_{OC} value of 31, 1,1-dichloroethane is not expected to significantly partition to sewage sludge. Based on the amounts of 1,1-dichloroethane undergoing wastewater treatment, land application of biosolids from 1,1-dichloroethane wastewater treatment is not expected to be a significant exposure pathway.

Section 405(d) of the CWA requires EPA to promulgate regulations for pollutants that can be present in sewage sludge to protect public health and the environment. In 1996, EPA released *Technical Support*

for the Round Two Sewage Sludge Pollutants, which provides information on how both the candidate list and the final list of pollutants for the Round Two sewage sludge regulation were derived. Candidates for Round Two were chosen that were frequently detected in sewage sludge in the 1988 National Sewage Sludge Survey. The NSSS sampled 208 representative POTWs. The survey pollutants with a frequency of detection of less than 10 percent were dropped from further consideration. 1,1-Dichloroethane had a 0 percent detection frequency in the National Sludge Survey and not considered further.

To assess soil concentrations resulting from biosolid applications, EPA relied upon modeling work conducted in Canada (2011), which used Equation 60 of the *European Commission Technical Guidance Document* (TGD) (ECB, 2003). The equation in the TGD is as follows:

Equation_Apx C-1.

$$PEC_{soil} = (C_{sludge} \times AR_{sludge})/(D_{soil} \times BD_{soil})$$

Where:

PEC_{soil}	=	Predicted environmental concentration (PEC) for soil (mg/kg)
C_{sludge}	=	Concentration in sludge (mg/kg)
AR_{sludge}	=	Application rate to sludge amended soils (kg/m^2/year); default = 0.5 from Table A-11 of TGD
D_{soil}	=	Depth of soil tillage (m); default = 0.2 m in agricultural soil and 0.1 m in pastureland from Table A-11 of TGD
BD_{soil}	=	Bulk density of soil (kg/m^3); default = 1,700 kg/m^3 from Section 2.3.4 of TGD

The concentration in sludge was set to 20 mg/kg dry weight based on the combined sludge concentration estimated by SimpleTreat 4.0. Using these assumptions, the estimated 1,1-dichloroethane soil concentrations after the first year of biosolids application were 29.4 ug/kg in tilled agricultural soil and 58.8 µg/kg in pastureland. See Section G.1.2.5 for discussion of the estimation of biosolids concentrations.

The method assumes complete mixing of the chemical in the volume of soil it is applied to as well as no losses from transformation, degradation, volatilization, erosion, or leaching to lower soil layers. Additionally, it is assumed there is no input of 1,1-dichloroethane from atmospheric deposition and there are no background 1,1-dichloroethane accumulations in the soil.

To estimate soil pore water concentrations for 1,1-dichloroethane in soil receiving biosolids for exposures to ecological species, EPA used a modified version of the equilibrium partitioning (EqP) equation developed for weakly adsorbing chemicals such as 1,1-dichloroethane and other volatile organic carbons (VOCs). The modified equation accounts for the contribution of dissolved chemical to the total chemical concentration in soil or sediment (Fuchsman, 2003). The equation assumes that the adsorption of chemical to the mineral components of sediment particles is negligible:

Equation_Apx C-2.

$$C_{total} = C_{dissolved} \times \left[(f_{OC} \times K_{OC}) + \frac{1 - f_{solids}}{f_{solids}}\right]$$

Where:

C_{total}	=	Total chemical concentration in soil (µg/kg)
$C_{dissolved}$	=	Chemical concentration dissolved in pore water (µg/L)

f_{OC}	=	Fraction of sediment present as organic carbon
K_{OC}	=	Organic carbon-water partition coefficient
f_{solids}	=	Fraction of soil solids

Using Equation_Apx C-1 and estimating $C_{dissolved}$ from the K_{OC} for 1,1-dichloroethane assuming a soil organic carbon fraction (f_{OC}) of 0.02, and a soil solids fraction of 0.5, the estimated pore water concentrations are 18.2 µg/L in tilled agricultural soil and 36.6 µg/L in pastureland.

C.2.4.5 Key Sources of Uncertainty in the Fate Assessment for Terrestrial Environments

Uncertainty in rates of biodegradation and volatilization are key sources of uncertainty in the fate assessment for terrestrial environments. The majority of the studies consist of laboratory microcosm studies or field studies with microbial populations that have acclimated to biodegrade 1,1-dichloroethane during long periods of exposure. Therefore, extrapolating biodegradation rates observed in laboratory studies to environmental biodegradation rates introduces uncertainty. Volatilization of 1,1-dichloroethane from soil, landfills, and land-applied biosolids is a complex process. Although the importance of the process is qualitatively addressed, quantitative estimates were not made. As a result, there is uncertainty regarding the estimated concentrations of 1,1-dichloroethane in terrestrial environments; values may have been overestimated because volatilization was not quantitatively addressed.

C.2.5 Persistence Potential

Based on the studies described in Appendix C.2.2, 1,1-dichloroethane is expected to be persistent in air based on its atmospheric oxidation half-life of 39 days. It is likely to be persistent in soil, surface water, and groundwater, where biodegradation half-lives of months to years are expected depending on environmental conditions.

C.2.5.1 Destruction and Removal Efficiency

Disposal of 1,1-dichloroethane may include incineration of up to 1,200 kg/year. Environmental release scenarios include Processing – repackaging for laboratory chemicals and Commercial use as a laboratory chemical (see Section 3.2.1.4 for details). Incineration of 1,1-dichloroethane from these activities is expected to occur at hazardous waste incinerators at a Destruction and Removal Efficiency (DRE) of greater or equal to 99.99 percent.

The CAA 40CFR Part 63, Subpart EEE—National Emission Standards for Hazardous Air Pollutants from Hazardous Waste Combustors requires all hazardous waste combustors—hazardous waste incinerators, hazardous waste cement kilns, hazardous waste lightweight aggregate kilns, hazardous waste solid fuel boilers, hazardous waste liquid fuel boilers, and hazardous waste hydrochloric acid production furnaces—to achieve a DRE of 99.99 percent for each principle organic hazardous constituent (POHC). Organic constituents that represent the greatest degree of difficulty of incineration will be those most likely to be designated as POHCs. If the dioxin-listed hazardous wastes F020, F021, F022, F023, F026, or F027 are burned 99.9999 percent DRE is required.

C.2.5.2 Removal in Wastewater Treatment

1,1-Dichloroethane is a volatile liquid with a vapor pressure of 228 mmHg at 25 °C, water solubility of 5,040 mg/L, log octanol/water partition coefficient of 1.79, and a Henry's Law constant of 0.00562 atm·m^3/mol. 1,1-Dichloroethane is not readily biodegradable and degrades slowly in most aerobic biodegradation studies identified through systematic review.

Based on these properties, the removal of 1,1-dichloroethane in activated sludge wastewater treatment is expected to be by volatilization due to its high vapor pressure and Henry's Law constant. However, 1,1-dichloroethane also has appreciable water solubility. Therefore, although volatilization from wastewater will occur, a portion of 1,1-dichloroethane can remain in the wastewater and be discharged with the effluent.

The removal of 1,1-dichloroethane from wastewater was measured in 11 wastewater treatment plants using activated sludge treatment in the EPA 40 POTW study (U.S. EPA, 1982). The minimum observed removal was 33 percent, maximum 100 percent, and the median was 64 percent. Hannah (1986) compared the removal of 1,1-dichloroethane across four pilot scale biological treatment system types acclimated for 30 days prior to measurement of removal of the chemical. Activated sludge wastewater treatment, commonly used to treat wastewater in the United States, achieved 94 percent removal of 1,1-dichloroethane.

For comparison, the Sewage Treatment Plant (STP) Model in EPI Suite™ (U.S. EPA, 2012b) was run using the physical and chemical properties reported in Section 2.1 of this risk evaluation and assuming no biodegradation of the chemical during treatment. The model predicted 69 percent overall removal with 68 percent attributable to volatilization and less than 1 percent by sorption to activated sludge and biodegradation.

Based on its K_{OC} value of 31, 1,1-dichloroethane is not expected to significantly partition to sewage sludge. Releases of 1,1-dichloroethane to wastewater treatment are expected to be low and disperse across many sites; therefore, land application of biosolids containing 1,1-dichloroethane is not expected to be a significant exposure pathway. To support this conclusion, range-finding estimates were made to evaluate the concentrations of 1,1-dichloroethane in biosolids, in soil receiving biosolids, and soil pore water concentrations resulting from biosolids application.

C.2.5.3 Key Sources of Uncertainty in the Persistence Assessment

A high-quality study indicated 1,1-dichloroethane has a long hydrolysis half-life of approximately 60 years under environmental conditions. 1,1-Dichloroethane biodegradation has been shown to occur slowly in under most environmental conditions with reported half-lives on the order of months or greater. Although other degradation processes can occur, they are not considered to be important in the overall environmental degradation of 1,1-dichloroethane. Thus, uncertainty regarding the environmental persistence of 1,1-dichloroethane is considered to be low.

C.2.6 Bioaccumulation Potential

No data were found on the bioaccumulation/bioconcentration potential of 1,1-dichloroethane. In the absence of data, the EPI Suite™ BCF/BAF Model (Version 4.1) (U.S. EPA, 2012b) was used to estimate bioaccumulation and bioconcentration factors. A full discussion of the performance of the BCF/BAF estimation methods used in EPI Suite™ is available in the help files. Based on estimated BCF and BAF values of 7 and 6.8, respectively, bioaccumulation and bioconcentration in aquatic and terrestrial organisms are not expected to be major environmental processes for 1,1-dichloroethane.

An alternative to estimating BCF and BAF values with EPI Suite™ is the use of the Office of Water methodology for deriving bioaccumulation factors intended to develop BAFs for setting national water quality criteria (U.S. EPA, 2003c). Procedure #3 for chemicals classified in the Office of Water methodology as nonionic organic chemicals with low hydrophobicity (log K_{OW} <4) and low metabolism was used to calculate BAF values for upper trophic level fish of 2.6 L/kg tissue. This value is in general agreement with the EPI Suite™ predicted BAF value of 6.8 and suggests low concern for

bioaccumulation of 1,1-dichloroethane. The differences are due in part to consideration of particulate and dissolved organic carbon levels in water (which impact the bioavailability) and the octanol water partition coefficient (K_{OW}) used in the Office of Water methodology to derive the upper trophic level (TL 4) BAF.

C.2.6.1 Key Sources of Uncertainty in the Bioaccumulation Assessment

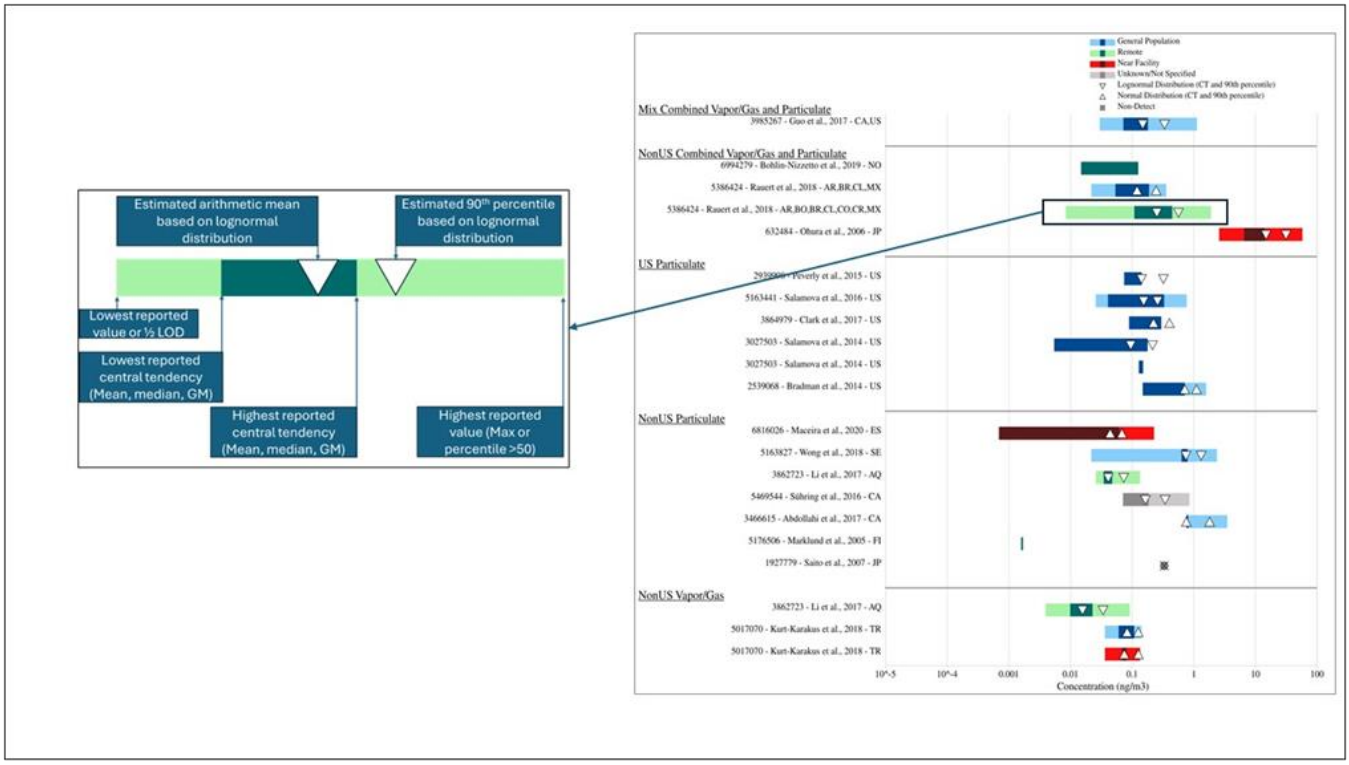
There is uncertainty associated with the EPI Suite™ BCF/BAF model estimates of BCF and BAF values for 1,1-dichloroethane. To address the uncertainty in the estimated BCF values, EPA compared measured BCF values for a series of halogenated ethanes and propanes and EPI Suite™ estimated BCF values. Log BCFs for the chemicals ranged from 0.7 to 1.1 The BCF/BAF model overestimated all BCF values and the largest observed error for BCF estimation was 1.5 log units. Thus, even if the log BCF estimate for 1,1-dichloroethane of 0.85 was subject to the maximum observed error, its log BCF would not be expected to exceed 2.3, indicating low bioconcentration potential (BCF <1,000).

C.3 Measured Data in Literature for Environmental Media

A literature search was conducted to identify peer-reviewed or other sources of 1,1-dichloroethane measured and reported modeled data. A summary of the measured and reported modeled data for the various environmental media is provided below. Detail information can also be found in the *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025ai).

C.3.1 Example Tornado Plot

EPA used tornado plots to display exposure data from studies identified during EPA's systematic review. An example is provided in Figure_Apx C-9 below. The plots provide the range of media concentrations in monitoring various studies. The plots show U.S. and non-U.S. data, fraction *(e.g.*, vapor, gas, particle, and the studies are ordered from top-to-bottom from newer-to-older data. The plots are colored to indicate general population, remote, near facility, and unknown population information.



Figure_Apx C-9. Example Tornado Plot

Exposure data is classified into a variety of location types as described below.

Near Facility
Near facility samples are not strictly contaminated sites and may be site-specific or not site-specific.

General Population
General population exposures are ambient measurements taken in areas near residential populations with no known near facility sources nearby. The data often represents widely distributed releases to the environment.

Remote
Remote exposures are measurements taken in areas away from residential and industrial activity and have no known sources of contamination beyond long-range transport. Examples of remote exposures include samples collected from polar regions, samples from oceans (not including ports), and sample locations specifically described as remote.

Indoor Media
Indoor air and dust samples will have indications in the legend based on sampling location such as commercial buildings, residential homes, public buildings, and vehicles. If studies report more than one of these micro-environments, then they are classified as mixed use.

Wastewater
Wastewater samples will indicate their sampling location at the wastewater processing facility.

There is one tornado plot for every media type where chemical concentrations are plotted on a logarithmic scale. The y-axis of the tornado plot is a list of each study representing a media sampled in a

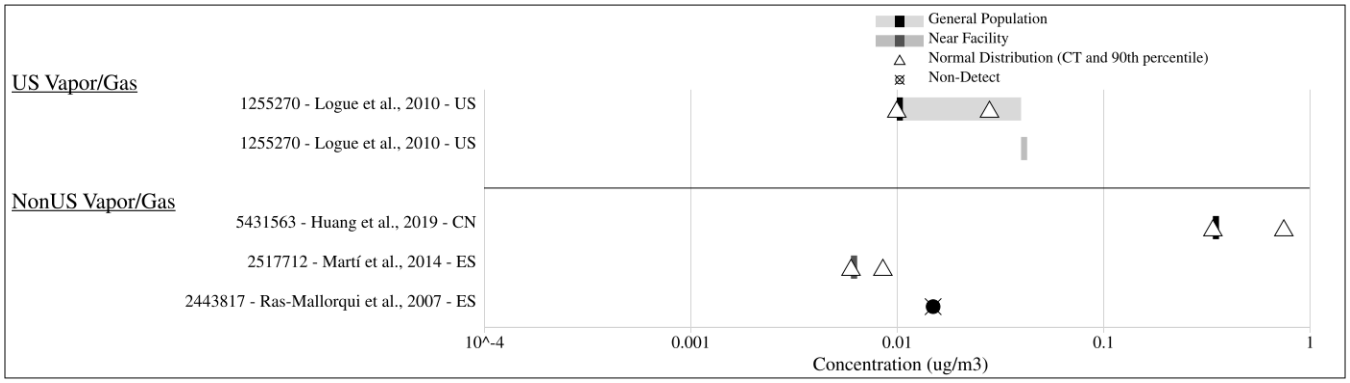
similar micro-environment and location and reported on the same unit/weight basis. A study may have more than one representation. For example, if a study reports exposure data collected at two different locations, the data would be plotted as two separate entries.

Each study on the y-axis is reported with its HERO ID, a short citation, and the country abbreviation of data collection. Additional details on tissue type or metabolite might also be reported. The studies are grouped by "US", combined with "US", or "non-US" data by unit/weight basis, and sorted in descending order by latest data collection year. Every study has a colored bar stretching across the x-axis. The color of the bar corresponds to the location type of the exposure data. The lighter bar represents the range of the reported concentrations, and the darker bar represents the range of reported central tendencies. A study with only dark bars indicates that the only data reported was a measure of central tendency.

Using the reported exposure data, EPA represent the arithmetic mean and 90th percentile. If sufficient central tendency and variance data were reported, the mean and 90th percentile were calculated directly from the study values assuming data were normally or lognormally distributed. When at least a central tendency and percentile value were provided, they were estimated by fitting the data to a lognormal distribution to all available data within the study aggregate. When fitting a lognormal distribution was not possible, a normal distribution was fit. The central tendency and 90th percentile of each distribution are plotted as triangles. Lognormal values are shown as upside-down triangles, while normal values are shown as right-side up. A study with no triangles indicates that there was insufficient data to fit a distribution. A study may not have reported concentrations because all data is below the limit of detection. In these circumstances, the plot will show a circle with an X at half the reported limit of detection. The color of the symbol will correspond to the color of the data's location type such as near facility, general population, and wastewater.

C.3.2 Ambient Air

Measured concentrations of 1,1-dichloroethane in ambient air extracted from four studies are summarized in Figure_Apx C-10 and supplemental information is provided in Table_Apx C-3. Overall, concentrations ranged from not detected to 0.34 µg/m^3 from 472 samples collected between 2005 and 2017 in 3 countries (Canada, Spain, and United States). Location types were categorized as either "General Population" or "Near Facility". Detection frequencies ranged from 0 to not reported.



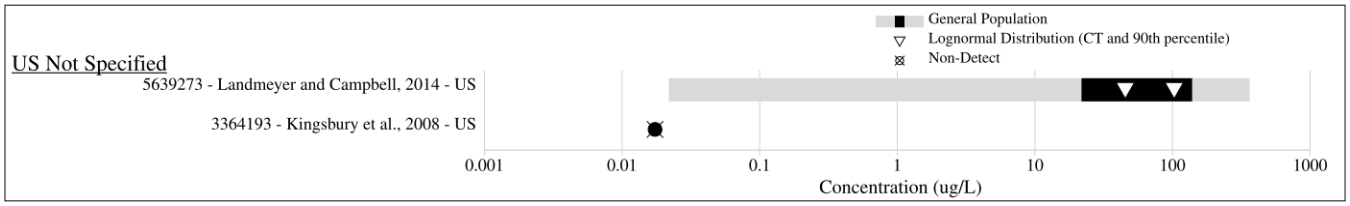
Figure_Apx C-10. Concentrations of 1,1-Dichloroethane (µg/m^3) in the Vapor/Gas Fraction of Ambient Air from U.S.-Based and International Studies, 2005–2017

Table_Apx C-3. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/m^3) Levels in the Vapor/Gas Fraction of Ambient Air from U.S.-Based and International Studies, 2005–2017

Citation	Country	Location Type	Sampling Year(s)	Sample Size (Frequency of Detection)	Detection Limit (µg/m^3)	Overall Quality Level
Logue et al. (2010)	U.S.	General Population	2006–2008	244 (N/R)	N/R	High
Logue et al. (2010)	U.S.	Near Facility	2006–2008	122 (N/R)	N/R	High
Huang et al. (2019)	China	General Population	2016–2017	37 (N/R)	N/R	High
Martí et al. (2014)	Spain	Near Facility	2014	36 (N/R)	N/R	Medium
Ras-Mallorqui et al. (2007)	Spain	General Population (Background)	2005–2006	33 (0)	30	High

C.3.3 Drinking Water

Measured concentrations of 1,1-dichloroethane in drinking water extracted from two studies are summarized in Figure_Apx C-11 and supplemental information is provided in Table_Apx C-4. Overall, concentrations ranged from not detected to 367 µg/L from 170 samples collected between 2002 and 2012 in United States. Location types were categorized as "General Population." Reported frequency of detection ranged from 0 to 0.17.



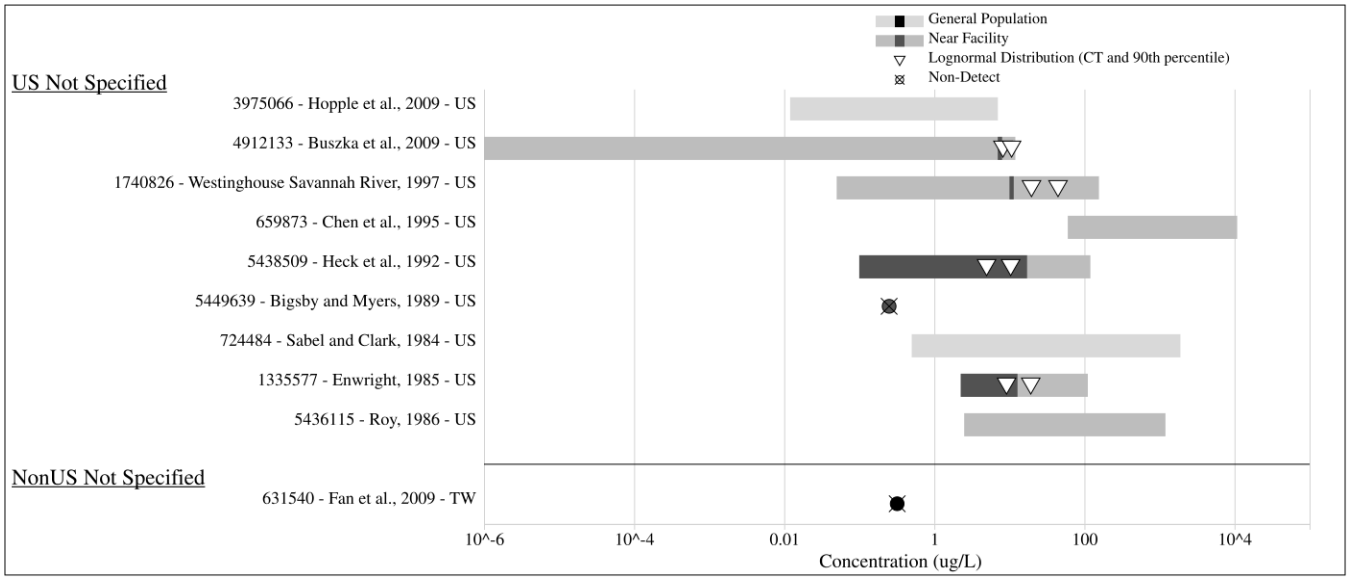
Figure_Apx C-11. Concentrations of 1,1-Dichloroethane (µg/L) in Drinking Water from a U.S.-Based Study, 2002–2012

Table_Apx C-4. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/L) Levels in Drinking Water from a U.S.-Based Study, 2002–2012

Citation	Country	Location Type	Sampling Years	Sample Size (Frequency of Detection)	Detection Limit (µg/L)	Overall Quality Level
Landmeyer and Campbell (2014)	U.S.	General Population	2010–2012	23 (0.17)	44	High
Kingsbury et al. (2008)	U.S.	General Population	2002–2004	147 (0)	35	High

C.3.1 Groundwater

Measured concentrations of 1,1-dichloroethane in groundwater extracted from nine studies are summarized in Figure_Apx C-12 and supplemental information is provided in Table_Apx C-5. Overall, concentrations ranged from not detected to 10,800 µg/L from 497 samples collected between 1984 and 2005 in Taiwan and United States. Location types were categorized as "General Population" and "Near Facility." Reported frequency of detection ranged from 0 to 0.86.



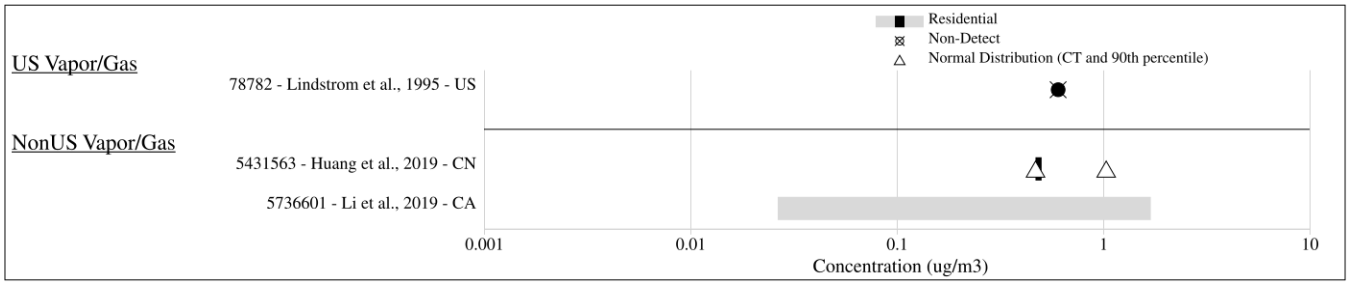
Figure_Apx C-12. Concentrations of 1,1-Dichloroethane (µg/L) in Groundwater from U.S.-Based and International Studies, 1984–2005

Table_Apx C-5. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/L) Levels in Groundwater from U.S.-Based and International Studies, 1984–2005

Citation	Country	Location Type	Sampling Year(s)	Sample Size (Frequency of Detection)	Detection Limit (µg/L)	Overall Quality Level
Hopple et al. (2009)	U.S.	General Population	2002–2005	292 (0.07)	24	High
Buszka et al. (2009)	U.S.	Near Facility	2000–2002	7 (0.86)	N/R	Medium
Westinghouse Savannah River Company (1997)	U.S.	Near Facility	1995–1996	136 (0.19)	20,000	Medium
Chen and Zoltek (1995)	U.S.	Near Facility	1989–1993	8 (0.62)	N/R	Medium
Heck et al. (1992)	U.S.	Near Facility	1990	13 (0.23)	200	Medium
Bigsby and Myers (1989)	U.S.	Near Facility	1988	7 (0)	500	Medium
Sabel and Clark (1984)	U.S.	General Population	1984	20 (0.35)	N/R	Medium
Roy F. Weston Inc (1986)	U.S.	Near Facility	1984	8 (0.25)	5000	Medium
Fan et al. (2009)	Taiwan	Near Facility	2005	6 (0.83)	640	Medium

C.3.2 Indoor Air

Measured concentrations of 1,1-dichloroethane in indoor air extracted from three studies are summarized in Figure_Apx C-13 and supplemental information is provided in Table_Apx C-6. Overall, concentrations ranged from not detected to 1.700 from 3,602 µg/m^3 samples collected between 1992 and 2017 in three countries (Canada, China, and the United States). Location types were categorized as residential and the reported frequency of detection was zero.



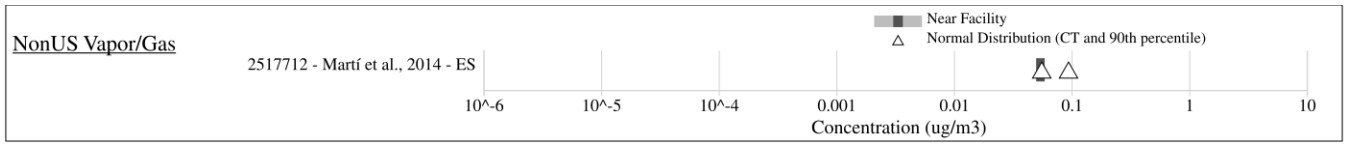
Figure_Apx C-13. Concentrations of 1,1-Dichloroethane (µg/m^3) in the Vapor/Gas Fraction in Indoor Air from U.S.-Based and International Studies, 1992–2017

Table_Apx C-6. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/m^3) Levels in the Vapor/Gas Fraction in Indoor Air from U.S.-Based and International Studies, 1992–2017

Citation	Country	Location Type	Sampling Years	Sample Size (Frequency of Detection)	Detection Limit (µg/m^3)	Overall Quality Level
Lindstrom et al. (1995)	U.S.	Residential	1992–1993	34 (0)	1,210	Medium
Huang et al. (2019)	China	Residential	2016–2017	44 (N/R)	N/R	High
Li et al. (2019)	Canada	Residential	2012–2013	3,524 (0)	53	High

C.3.3 Soil and Soil-Water Leachate

Measured concentrations of 1,1-dichloroethane in soil extracted from one study are summarized in Figure_Apx C-14 and supplemental information is provided in Table_Apx C-7. Overall, concentrations ranged from 0.050 to 0.060 µg/m^3 from seven samples collected between 2012 and 2014 in Spain. Location types were categorized as "Near Facility." Reported frequency of detection was not reported.



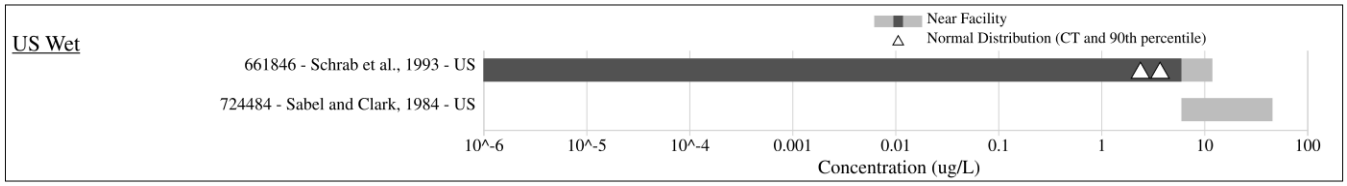
Figure_Apx C-14. Concentrations of 1,1-Dichloroethane (µg/m^3) in the Vapor/Gas Fraction of Soil from International Studies, 2012–2014

Table_Apx C-7. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/m^3) Levels in the Vapor/Gas Fraction of Soil from International Studies, 2012–2014

Citation	Country	Location Type	Sampling Years	Sample Size (Frequency of Detection)	Detection Limit (µg/m^3)	Overall Quality Level
Martí et al. (2014)	Spain	Near Facility	2012–2014	7 (N/R)	0.0011	Medium

Measured concentrations of 1,1-dichloroethane in soil-water leachate extracted from two sources are summarized in Figure_Apx C-15 while supplemental information is provided in Table_Apx C-8. Overall, concentrations ranged from not detected to 46 µg/L from 11 samples collected between 1984 and 1993 in the United States. Location types were categorized as "Near Facility." Reported frequency

of detection ranged from 0.2 to 0.83.



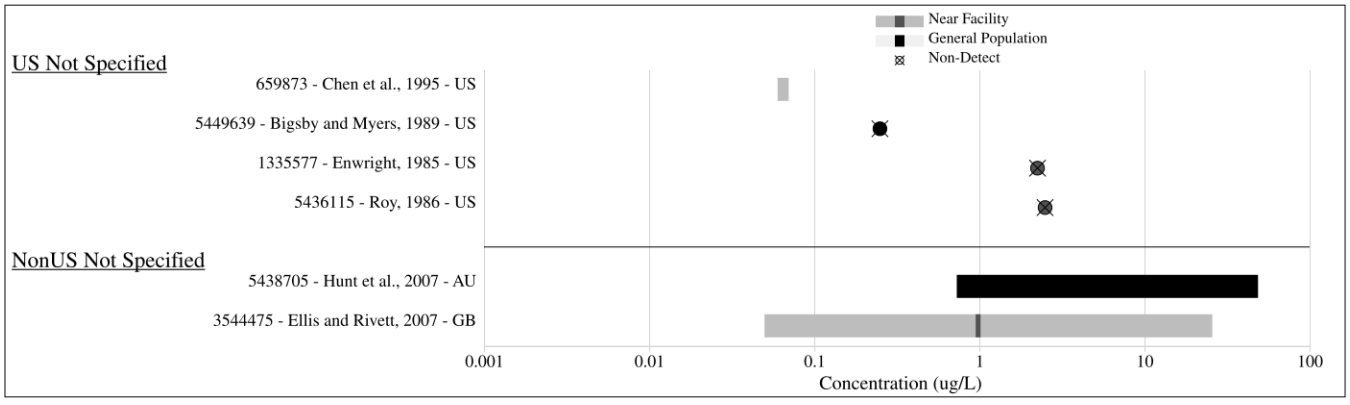
Figure_Apx C-15. Concentrations of 1,1-Dichloroethane (µg/L) in the Soil-Water Leachate from U.S.-Based Studies for Locations near Facility Releases, 1984–1993

Table_Apx C-8. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/L) Levels in the Soil-Water Leachate from U.S.-Based Studies for Locations near Facility Releases, 1984–1993

Citation	Country	Location Type	Sampling Year	Sample Size (Frequency of Detection)	Detection Limit (µg/L)	Overall Quality Level
Schrab et al. (1993)	U.S.	Near Facility	1993	5 (0.20)	N/R	Medium
Sabel and Clark (1984)	U.S.	Near Facility	1984	6 (0.83)	N/R	Medium

C.3.4 Surface Water

Measured concentrations of 1,1-dichloroethane in surface water extracted from six studies are summarized in Figure_Apx C-16 and supplemental information is provided in Table_Apx C-9. Overall, concentrations ranged from not detected to 48.7 µg/L from 155 samples collected between 1984 and 2005 in three countries (Australia, Great Britain, and United States). Location types were categorized as "General Population" and "Near Facility." Reported frequency of detection ranged from 0 to 0.5.



Figure_Apx C-16. Concentrations of 1,1-Dichloroethane (µg/L) in Surface Water from U.S.-Based and International Studies, 1984–2005

Table_Apx C-9. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/L) Levels in Surface Water from U.S.-Based and International Studies, 1984–2005

Citation	Country	Location Type	Sampling Year(s)	Sample Size (Frequency of Detection)	Detection Limit (µg/L)	Overall Quality Level
Chen and Zoltek (1995)	U.S.	Near Facility	1989–1993	12 (0.50)	N/R	Medium
Bigsby and Myers (1989)	U.S.	General Population	1988	3 (0)	500	Medium
Enwright Associates (1985)	U.S.	Near Facility	1984	6 (0)	4,500	Medium
Roy F. Weston Inc (1986)	U.S.	Near Facility	1984	6 (0)	5,000	Medium
Hunt et al. (2007)	Australia	General Population	2004–2005	93 (N/R)	N/R	High
Ellis and Rivett (2007)	Great Britian	Near Facility	2001	35 (0.37)	100	Medium

C.3.5 Wastewater

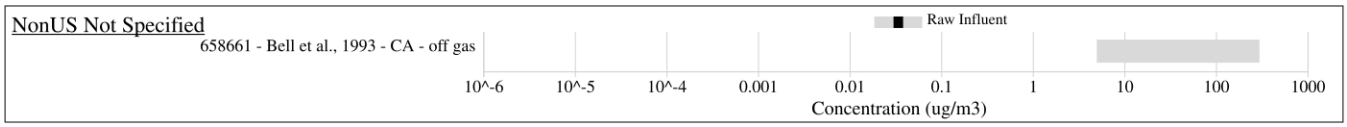
Measured concentrations of 1,1-dichloroethane in wastewater untreated effluent extracted from two sources are summarized in Figure_Apx C-17 and supplemental information is provided in Table_Apx C-10. Overall, concentrations ranged from not detected to 594 µg/L from 29 samples collected between 1981 and 1984 in the United States. Location types were categorized as "Untreated Effluent" at "Discharge Origin." Reported frequency of detection ranged from 0 to 0.25.

Figure_Apx C-17. Concentrations of 1,1-Dichloroethane (µg/L) in Wastewater Untreated Effluent from U.S.-Based Studies, 1981–1984

Table_Apx C-10. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/L) Levels in Wastewater Untreated Effluent from U.S.-Based Studies, 1981–1984

Citation	Country	Location Type	Sampling Year(s)	Sample Size (Frequency of Detection)	Detection Limit (µg/L)	Overall Quality Level
Enwright Associates (1985)	U.S.	Untreated Effluent at Discharge Origin	1984	21 (0)	4,500	Medium
Ghassemi et al. (1984)	U.S.	Untreated Effluent at Discharge Origin	1981–1983	8 (0.25)	N/R	Low

Measured concentrations of 1,1-dichloroethane in wastewater raw influent extracted from one source are summarized in Figure_Apx C-18 and supplemental information is provided in Table_Apx C-11. Overall, concentrations were not detected from eight samples collected in 1993 in California; U.S. Location types were categorized as "Raw Influent." Reported frequency of detection was not reported.



Figure_Apx C-18. Concentrations of 1,1-Dichloroethane (µg/m^3) in Wastewater in Raw Influent from a U.S.-Based Study in 1993

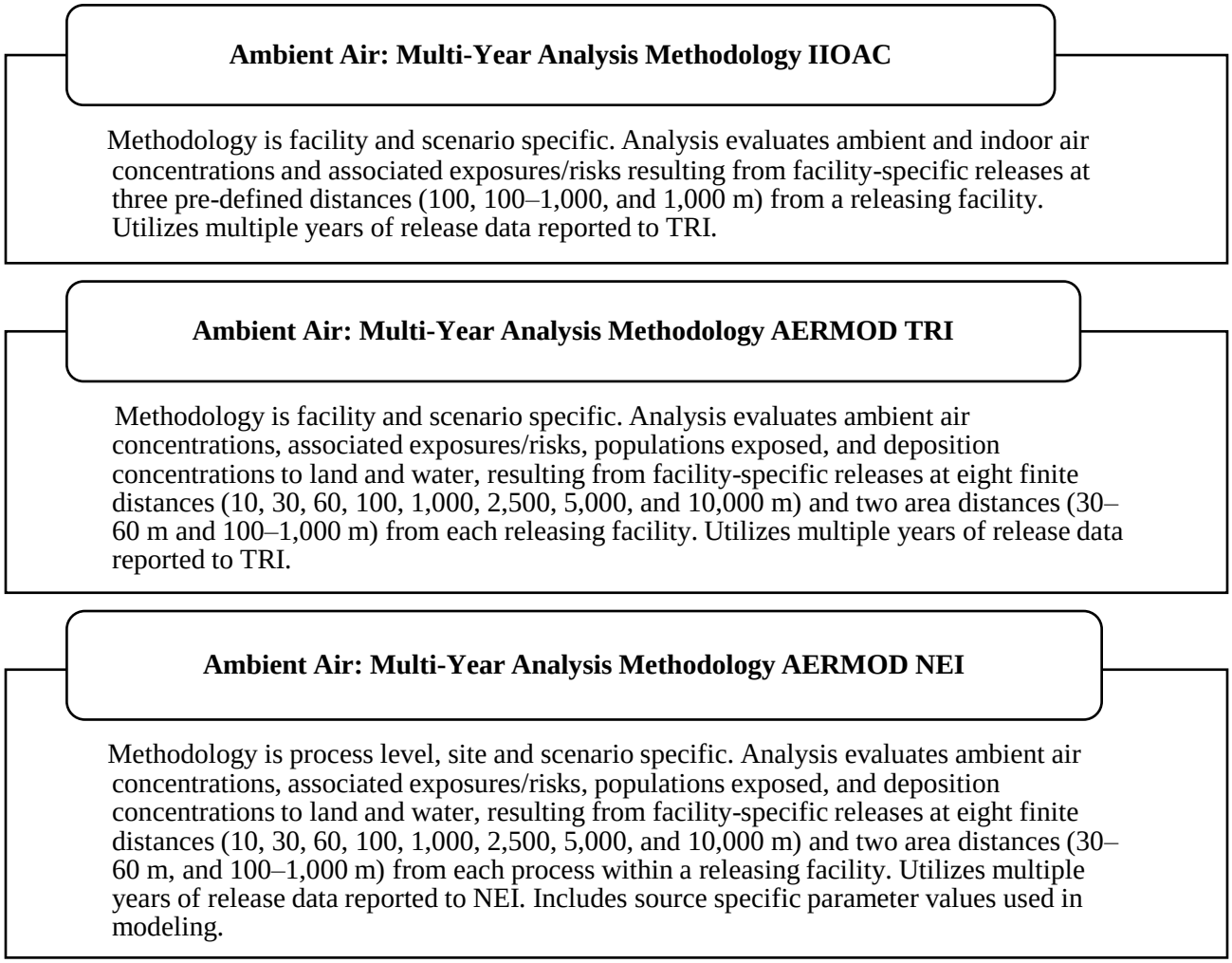
Table_Apx C-11. Summary of Peer-Reviewed Literature that Measured 1,1-Dichloroethane (µg/m^3) Levels in Wastewater in Raw Influent from a U.S.-Based Study in 1993

Citation	Country	Location Type	Sampling Year	Sample Size (Frequency of Detection)	Detection Limit (µg/m^3)	Overall Quality Level
Bell et al. (1993)	U.S.	Raw Influent	1993	8 (N/R)	1,000	Medium

Appendix D AIR EXPOSURE PATHWAY

D.1 Modeling Approach for Estimating Concentrations of 1,1-Dichloroethane in Air and Deposition to Land and Water

EPA applied a tiered approach to estimate ambient air concentrations and exposures for members of the general population that are in proximity (between 10–10,000 m) to emissions sources, emitting the chemical being evaluated to the ambient air (Figure_Apx D-1.). All exposures were assessed for the inhalation route only.



Figure_Apx D-1. Brief Description of Methodologies and Analyses Used to Estimate Air Concentrations and Exposures

D.1.1 Multi-Year Analysis Methodology IIOAC

The Multi-Year Analysis Methodology IIOAC identifies, at a high level, if there are inhalation exposures to select populations from a chemical undergoing risk evaluation that indicates a potential risk. This methodology inherently includes both estimates of exposures as well as estimates of risks to inform the need, or potential need, for further analysis. If findings from the Multi-Year Analysis Methodology IIOAC 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 Multi-Year

Analysis Methodology IIOAC do not indicate any potential risks for a given chemical above (or below as applicable) typical agency benchmarks, EPA would not expect a risk would be identified with higher tier analyses, but may still conduct a limited higher tier analysis at select distances to ensure potential risks are not missed (*e.g.*, at distances <100 m to ensure risks do not appear very near a facility where human populations may be exposed).

D.1.1.1 Model

The Multi-Year Analysis Methodology IIOAC utilizes EPA's Integrated Indoor/Outdoor Air Calculator (IIOAC) Model[1] to estimate high-end and central tendency (mean) exposures for members of the general population at three pre-defined distances from a facility releasing a chemical to the ambient air (100, 100–1,000, 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 American Meteorological Society/Environmental Protection Agency Regulatory Model (AERMOD). As such, IIOAC is limited by the parameterizations utilized for the pre-run scenarios within AERMOD (meteorologic data, stack heights, distances, 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, 2019d).

D.1.1.2 Releases

EPA modeled exposures using the release data developed as described in Section 3.3.1. Release data was provided (and modeled) on a facility-by-facility basis using facility-specific chemical releases (fugitive and stack releases) as reported to the TRI.

D.1.1.3 Exposure Scenarios

EPA evaluated the most "conservative exposure scenario" of the 16 scenarios in the *Draft TSCA Screening Level Approach for Assessing Ambient Air and Water Exposures to Fenceline Communities*, referred to herein as the "2022 Fenceline Report."[2] This most 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.

EPA selected 1 of the 14 climate regions to represent a high-end (South [Coastal]) climate region. This selected climate region represents the meteorological data set that tended to provide high-end concentration estimates relative to the other stations within IIOAC. The meteorological data within the IIOAC Model are from years 2011 to 2015 as that is the meteorological data utilized in the suite of pre-run AERMOD exposure scenarios during development of the IIOAC Model (see IIOAC Users Guide (U.S. EPA, 2019d)). 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.

For complete input parameters, including release scenarios, refer to the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on IIOAC TRI Exposure and Risk Analysis* (U.S. EPA, 2025t).

[1] The IIOAC website is available at https://www.epa.gov/tsca-screening-tools/iioac-integrated-indoor-outdoor-air-calculator (accessed June 11, 2025).
[2] Additional information about 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 (accessed June 11, 2025).

D.1.2 Multi-Year Analysis Methodology AERMOD (TRI or NEI)

The Multi-Year Methodology AERMOD (TRI or NEI) was developed to allow EPA to conduct a higher-tier analysis of releases, exposures, and associated risks to members of the general population around releasing facilities at multiple finite distances and area distances when EPA has site-specific data like reported releases, facility locations (for local meteorological data), and source attribution. This methodology can incorporate additional process level, site- and scenario-specific information like stack parameters (stack height, stack temperature, plume velocity, etc.), building characteristics, release patterns, different terrains, and other parameters when reasonably available. The Multi-Year Methodology AERMOD can be performed independent of the Multi-Year Analysis Methodology IIOAC described above, can include wet and dry deposition estimates and in conjunction with process level-, site-, and scenario-specific information, provides a more refined analysis that allows EPA to fully characterize risks for chemicals undergoing risk evaluation.

D.1.2.1 Model

The Multi-Year Methodology AERMOD (TRI or NEI) utilizes EPA's AERMOD to estimate exposures to members of the general population at multiple finite distances and area distances from a facility releasing a chemical to the ambient air. 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, 2018).

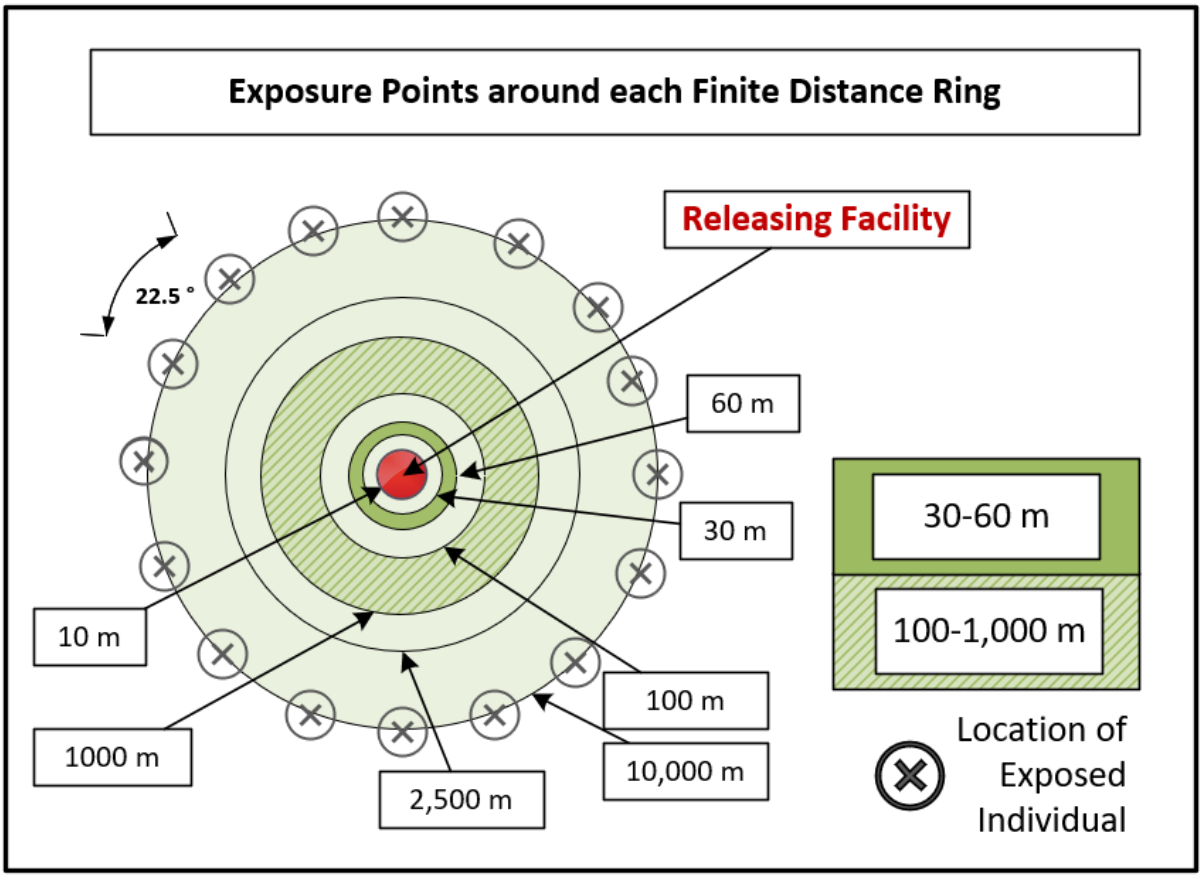
D.1.2.2 Releases

EPA modeled exposures using the release data developed as described in Section 3.2 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 TRI or NEI, where available.
2. Alternative release estimates where facility specific data were not available.

D.1.2.3 Exposure Scenarios

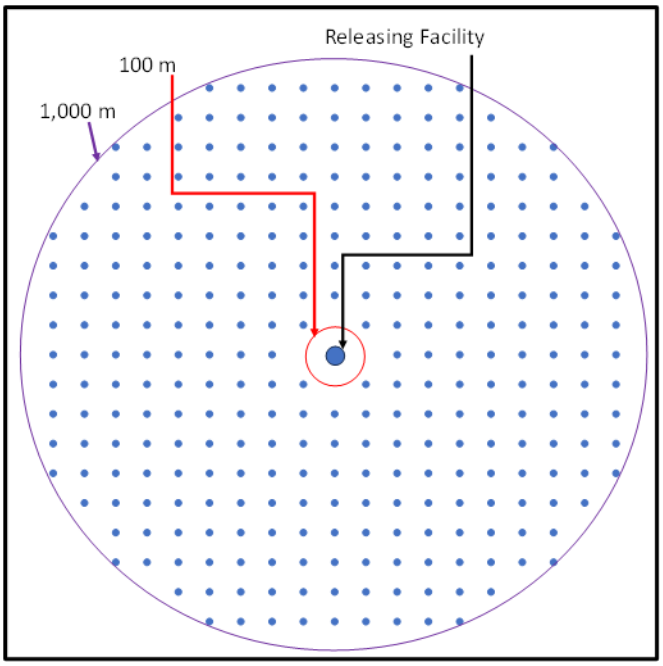
The Multi-Year Methodology AERMOD (TRI or NEI) evaluated exposures to members of the general population at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and two area distances (30–60 m and 100–1,000 m) from each TRI or NEI releasing facility for each occupational exposure scenario (OES; or generic facility for alternative release estimates). Human populations for each of the eight finite distances were placed in a polar grid every 22.5 degrees around the respective distance ring. This results in a total of 16 modeled exposure points around each finite distance ring for which exposures are modeled. Figure_Apx D-2 provides a visual depiction of the placement of exposure points around a finite distance ring. Although the visual depiction only shows exposure point locations around a single finite distance ring, the same placement occurred for all eight finite distance rings.



Figure_Apx D-2. Modeled Exposure Points for Finite Distance Rings for Ambient Air Modeling (AERMOD)

Modeled exposure points for the area distance 30 to 60 m evaluated were placed in a cartesian grid at equal distances between 30 and 60 m around each releasing facility. Exposure points were placed at 10-meter increments. This results in a total of 80 points for which exposures are modeled. Modeled exposure points for the area distance 100 to 1,000 m evaluated were placed in a cartesian grid at equal distances between 100 and 1,000 m around each releasing facility. Exposure points were placed at 100-meter increments. This results in a total of 300 points for which exposures are modeled.

Figure_Apx D-3 provides a visual depiction of the placement of exposure points (each dot) around the 100 to 1,000 m area distance ring. All exposure points were at 1.8 m above ground as an approximation for breathing height for ambient air concentration estimations. A duplicate set of exposure points was at ground level (0 m) for deposition estimations.



Figure_Apx D-3. Modeled Exposure Point Locations for Area Distance for Ambient Air Modeling (AERMOD)

D.1.2.4 Meteorological Data

Meteorological data for TRI reporting facilities was obtained using the same AERMOD-ready meteorological data that EPA's Risk and Technology Review (RTR) Program uses for multimedia, multipathway-risk modeling in review of NESHAPs. The 2019 meteorological data[3] that the RTR program currently uses, includes 838 hourly stations with data mostly from the year 2019. For 47 stations (mainly in Alaska and West Virginia), EPA utilized data from 2016, 2017, or 2018 to fill notable spatial gaps. The 2016 meteorological data (no longer available for download from the EPA website) covers 824 hourly stations in the 50 States, District of Columbia, and Puerto Rico. The 2019 meteorological data was used to model 2018, 2019, and 2020 air emission releases. The 2016 meteorological data was used to model air emission releases reported from 2014 through 2017. The 2016 meteorologic data was processed with version 16216 of AERMOD's meteorological preprocessor (AERMET) and the 2019 meteorologic data was processed with version 19191 of AERMET. Following EPA guidance, all processing utilized sub-hourly wind measurements (to calculate hourly-averaged wind speed and wind direction; see Section 8.4.2 of that guidance). The processing for the 2016 and 2019 data also used the "ADJ_U*" option for mitigating modeling issues during light-wind, stable conditions. Facility coordinates, in the form of latitude/longitude coordinates, were used to match the facility to the closest available meteorological station. All processing also used automatic substitutions for small gaps in data for cloud cover and temperature. Each facility was matched to its closest surface meteorological station.

For NEI facilities, where the latitude/longitude can vary by individual source, EPA consolidated each facility around a single latitude/longitude by averaging the individual source latitudes and longitudes. The average latitude/longitude was used to determine the meteorological station closest to the NEI facility, the urban/rural designation, and surrounding land cover setting for the deposition modeling.

[3] 2019 meteorological data is available at https://www.epa.gov/fera/download-human-exposure-model-hem (accessed June 11, 2025).

Meteorological data for the EPA estimated releases (2 OESs where there was no site-specific data available for modeling; Commercial use as a laboratory chemical, and Processing – repackaging for laboratory chemicals) were modeled with two meteorological stations, Sioux Falls, South Dakota, for central-tendency meteorology, and Lake Charles, Louisiana, for higher-end meteorology. These two meteorological stations represent meteorological datasets that tended to provide high-end and central tendency concentration estimates relative to the other stations within IIOAC based on a sensitivity analysis of the average concentration and deposition predictions conducted in support of IIOAC development. These two meteorological stations are based on 5 years of data (2011–2015) and provide high-end and central tendency exposure concentrations utilized for risk calculation purposes to identify potential risks. All processing used sub-hourly wind measurements to calculate hourly-averaged wind speed and wind direction. The "ADJ_U*" option was not used for the 2011 to 2015 data as this could lead to model overpredictions of ambient concentrations during those conditions. All processing also used automatic substitutions for small gaps in data for cloud cover and temperature.

D.1.2.5 Urban/Rural Designations

Urban/rural designations of the area around a facility are relevant when considering possible boundary layer effects on concentrations. Air emissions taking place in an urbanized area are subject to the effects of urban heat islands, particularly at night. When sources are set as urban in AERMOD, the model will modify the boundary layer to enhance nighttime turbulence, often leading to higher nighttime air concentrations. AERMOD uses urban-area population as a proxy for the intensity of this effect.

EPA utilized a population density analysis to identify facilities warranting an urban designation for the AERMOD runs. Specifically, EPA considered a facility to be in an urban area if it had a population density exceeding 750 people per square kilometer (km^2) within a 3-kilometer radius of the facility (see Section 7.2.1.1 of the guidance referenced in footnote 4 below) and set the relevant inputs to urban within AERMOD. For facilities set for urban modeling, AERMOD requires an estimate of the urban population count. EPA estimated the urban-area population by identifying a proxy for the area of urbanization. The urban-area proxy was the largest radius around the facility (out to a limit of 15 km) having a population density greater than 750 people per km^2. EPA identified the population within that radius and applied it for modeling purposes. The Agency used U.S. Census data at the level of block groups for these analyses (with geographies from the 2019 census TIGER/Line shapefiles[4] and population counts from the American Community Survey[5] [2015–2019], 5-year estimates-detailed tables [table B01003]). For the NEI facility mentioned earlier (EIS Facility ID 16206511) that did not have latitude/longitude, EPA assumed its locations were not urban.

For the EPA-estimated releases where TRI or city data were not available for a facility requiring modeling (Commercial use as a laboratory chemical, and Processing – repackaging for laboratory chemicals) the Agency modeled each such facility once as urban and once as not urban.[6] Because there is no recommended default urban population for AERMOD modeling, for these facilities EPA assumed an urban population of 1 million people, which is consistent with the estimated populations used with IIOAC. Although slightly higher, the assumed urban population is close to the average of all the urban populations used for the TRI reporting facilities, which was 847,906 people.

[4] 2019 census TIGER/Line shapefiles are available at https://www.census.gov/geographies/mapping-files/timE−series/geo/tiger-linE−file.2019.html (accessed June 11, 2025).
[5] American Community Survey website: https://www.census.gov/programs-surveys/acs (accessed June 11, 2025).
[6] Although this may be viewed as a potential double counting of these releases, EPA only utilized the highest estimated releases from a single exposure scenario from the suite of exposure scenarios modeled for surrogate/estimated facility releases as exposure estimates and for associated risk calculations.

D.1.2.6 Physical Source Specifications for TRI Release Facilities and Alternative Release Estimates

Source-specific physical characteristics like actual release location, stack height, exit gas temperature, etc. are generally not reported as part of the TRI dataset but can affect the plume characteristics and associated dispersion of the plume. TRI release facilities and EPA estimated releases (where TRI or city data were not available) were modeled centering all emissions on one location and using IIOAC default physical parameters. Stack emissions were modeled from a point source at 10 m above ground from a 2-meter inside diameter, with an exit gas temperature of 300 Kelvin and an exit gas velocity of 5 m/sec (Table 6 of the IIOAC User Guide). Fugitive emissions were modeled at 3.05 m above ground from a square area source of 10 m on a side (Table 7 of the IIOAC User Guide).

D.1.2.7 Temporal Emission Patterns

TRI and NEI Release Facilities

Temporal emission patterns are another factor that can affect the overall modeled concentration estimates. The release assessments for this work included information on temporal emission patterns—release duration (across the hours of a day, or intraday) and release pattern (across the days of a year, or inter-day)—stratified by OES. When release duration was "unknown," EPA assumed releases occurred each hour of the day. EPA's assumptions for intraday release duration are provided in Table_Apx D-1. The hours shown conform to AERMOD's notation scheme of using hours 1 to 24, where hour 1 is the hour ending at 1 a.m. and hour 24 is the final hour of the same day ending at midnight.

Table_Apx D-1. Assumptions for Intraday Emission-Release Duration

Hours per Day of Emissions	Assumed Hours of the Day Emitting (Inclusive)
Unknown	All (hours 1–24)
1	Hour 13 (hour ending at 1 p.m.; *i.e.*, 12–1 p.m.)
2	Hours 13–14 (hour ending at 1 p.m. through hour ending at 2 p.m.; *i.e.*, 12–2 p.m.)
3	Hours 13–15 (hour ending at 1 p.m. through hour ending at 3 p.m.; *i.e.*, 12–3 p.m.)
4	Hours 13–16 (hour ending at 1 p.m. through hour ending at 4 p.m.; *i.e.*, 12–4 p.m.)
5	Hours 13–17 (hour ending at 1 p.m. through hour ending at 5 p.m.; *i.e.*, 12–5 p.m.)
8	Hours 9–16 (hour ending at 9 a.m. through hour ending at 4 p.m.; *i.e.*, 8 a.m. to 4 p.m.)
12	Hours 9–20 (hour ending at 9 a.m. through hour ending at 8 p.m.; *i.e.*, 8 a.m. to 8 p.m.)
14	Hours 7–20 (hour ending at 7 a.m. through hour ending at 8 p.m.; *i.e.*, 6 a.m. to 8 p.m.)

EPA's assumptions for inter-day release pattern are provided in Table_Apx D-2. The Agency started with the assumption that emissions took place every day of the year. Next, EPA turned emissions off for certain days of the year as needed to achieve the desired number of emission days: assumptions such as no emissions on Saturday and Sunday, no emissions on the days around New Year's Day, no emissions at regular patterns like the first Monday of every month, and so on.

Table_Apx D-2. Assumptions for Inter-Day Emission-Release Pattern

Provided Language for Release Pattern	Implemented Release Pattern: Days When Emissions Are on (Format of Month Number/Day Number)
Release pattern: **365 days/year** assumes year-round operations	All days
Release pattern: **350 days/year** assumes emitting operations **7 days/week and 50 weeks/year**	All days except 1/1–1/4 and 12/21–12/31 (and 1/5 for years 2016 and 2020)
Release pattern: **260 days/year**	All Monday through Friday, except 1/1 in years 2015, 2016, 2018, 2019, and 2020, and except 12/25 in year 2020
Release pattern: **258 days/year**	All Monday through Friday, except 12/24–12/26, and except 12/27 in years 2011, 2014, 2015, 2016, and 2020, and except 12/28 in 2015, 2016, and 2020, and except 12/29 in 2020
Release pattern: **250 days/year** assumes emitting operations **5 days/week and 50 weeks/year**	All Monday through Friday, except 1/1–1/4 and 12/21–12/31 (and 1/5 for years 2016 and 2020)
Release pattern: **235 days/year**	All Monday through Friday, except 1/1–1/8, 4/1–4/7, 7/1–7/7, 10/1–10/7, and 12/25–12/31, and except 12/24 in 2012 and 2020
Release pattern: **129 days/year**	The first 10 days of each month, plus the 11th of January through September
Release pattern: **26 days/year**	The first and 15th of each month, plus the 25th of June and December
Note: Some of the "Provided Language for Release Pattern" is specific to an OES.	

Alternative Release Estimates

EPA's assumptions for intraday release duration for the EPA estimated releases (Commercial use as a laboratory chemical, and Processing – repackaging for laboratory chemicals) are provided in Table_Apx D-3. The hours shown conform to AERMOD's notation scheme of using hours 1 to 24, where hour 1 is the hour ending at 1 a.m. and hour 24 is the final hour of the same day ending at midnight.

Table_Apx D-3. Assumptions for Intraday Emission-Release Duration

Hours per Day of Emissions	Assumed Hours of the Day Emitting (Inclusive)
1	Hour 13 (hour ending at 1 p.m.; *i.e.,* 12–1 p.m.)
2	Hours 13–14 (hour ending at 1 p.m. through hour ending at 2 p.m.; *i.e.*, 12–2 p.m.)
4	Hours 13–16 (hour ending at 1 p.m. through hour ending at 4 p.m.; *i.e.*, 12–4 p.m.)
5	Hours 13–17 (hour ending at 1 p.m. through hour ending at 5 p.m.; *i.e.*, 12–5 p.m.)
8	Hours 9–16 (hour ending at 9 a.m. through hour ending at 4 p.m.; *i.e.,* 8 a.m.to 4 p.m.)
24	All hours

EPA's assumptions for inter-day release frequency are provided in Table_Apx D-4.

Table_Apx D-4. Assumptions for Inter-Day Emission-Release Pattern

Days of Emissions per Year	Implemented Release Pattern: Days When Emissions Are on (Format of Month Number/Day Number)
28	All Monday through Friday, except 12/24–12/26, and except 12/27 in years 2011, 2014, and 2015, and except 12/28 in 2015
235	All Monday through Friday, except 1/1–1/8, and except 4/1–4/7, and 7/1–7/7, and 10/1-10/7, and 12/25-12/31, and 12/24 in 2012
129	The first 10 days of each month, plus the 11th of January through September
26	The first and 15th of each month, plus the 25th of June and December

D.1.2.8 Emission Rates

The release assessments included emission rates for each facility in pounds per year for TRI reporting facilities, tons per year for NEI reporting facilities, and kilograms per year for each scenario for the EPA estimated releases (Commercial use as a laboratory chemical, and Processing – repackaging for laboratory chemicals), for fugitive and stack sources as appropriate. Emission rates included in the release assessments were converted to units needed by AERMOD (g/s for stack sources; $g/s/m^2$ for fugitive sources). The conversion from per-hour to per-second utilized the number of emitting hours per year based on the assumed temporal release patterns (see Section D.1.2.7). The conversion to per m^2 for fugitive sources utilized length and width values outlined in Section D.1.2.6.

D.1.2.9 Deposition Parameters

AERMOD was used to model daily (g/m^2/day) and annual (g/m^2/year) deposition rates from air to land and water at eight finite distances (10, 30, 60, 100, 1,000, 2,500, 5,000, and 10,000 m) and two area distances (30–60 m, and 100–1,000 m) from each releasing facility. Concentrations of 1,1-dichloroethane in soil from total (wet and dry) air deposition was estimated to assess exposures of 1,1-dichloroethane to terrestrial species. AERMOD can model both gaseous and particle deposition. Based on physical and chemical properties of 1,1-dichloroethane (see Section 2.1), EPA considered only gaseous deposition. Input parameter values for AERMOD deposition modeling are shown in Table_Apx D-5.

Table_Apx D-5. Settings for Gaseous Deposition

Parameter	Value	Source(s)
Diffusivity in air	8.36E−02 cm^2/s	
Diffusivity in water	1.06E−05 cm^2/s	
Henry's Law constant	569.4 Pa m^3/mol	Table 2-1
r_{cl}: Cuticular resistance to uptake by lipids for individual leaves	1.82E05 s/cm	Based on Method 1: Approximation of R_{cl} value as a function of vapor pressure (Welke et al., 1998; Kerler and Schoenherr, 1988) (see below)
Seasons	DJF = winter with no snow; MAM = transitional spring with partial green coverage or short annuals; JJA = midsummer with lush vegetation; SON = autumn with unharvested cropland	Assumption
Land cover	Site-specific in 36 directions around the source, utilizing the 2019 version of the National Land Cover Database (supplemented with the 2011 version for Hawaii and 2001 version for Puerto Rico)	National Land Cover Database (accessed June 11, 2025)
Pa = Pascal; mol = mole; log = logarithm base 10; DJF = December–February; MAM = March–May; JJA = June–August; SON = September–November		

Cuticular Resistance

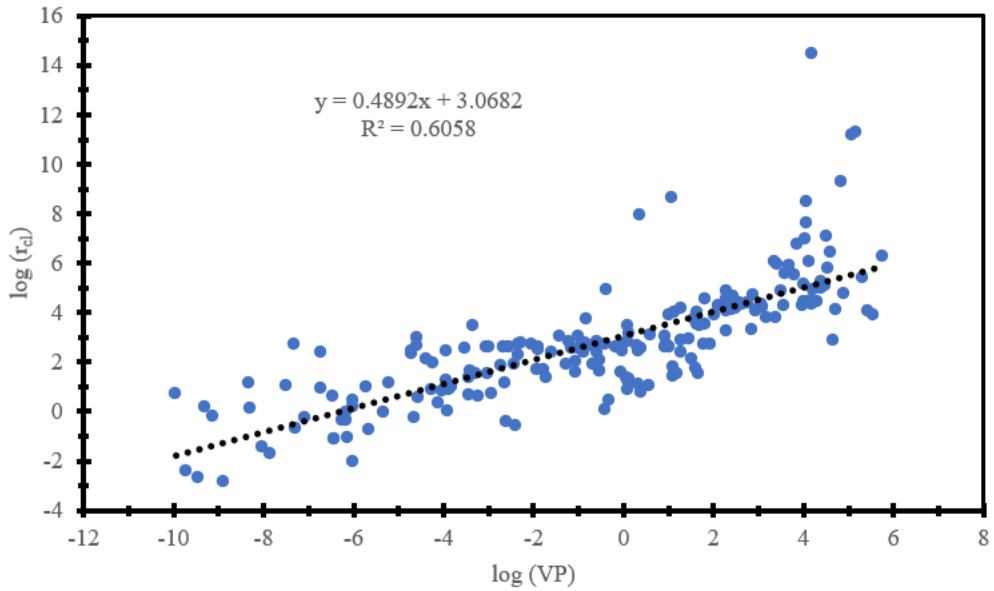
The cuticular resistance (r_{cl}) value represents the resistance of a chemical to uptake by individual leaves in a vegetative canopy. For chemicals, for which the r_{cl} value is not readily available in literature, EPA developed three methods to estimate the r_{cl} value. For 1,1-dichloroethane, the Agency used r_{cl} value estimated using Method 1, as described below. After additional review of information, EPA did identify a reported r_{cl} value of 1.16×10^5 (Wesely et al., 2002). Due to the similarity between the two values, the Agency is presenting results using the calculated r_{cl} value.

Method 1 – Approximation of R_{cl} Value as a Function of Vapor Pressure: Data from the literature indicate that r_{cl} value varies as a function of the vapor pressure (VP, units of Pa) of a chemical (Welke et al., 1998; Kerler and Schoenherr, 1988). A high VP indicates that chemical has a high propensity for the vapor phase relative to the condensed phase, and therefore, would have high resistance to uptake from the atmosphere into leaves (*i.e.*, high r_{cl}). Furthermore, Wesely (2002) provides a large database of VP and r_{cl} values.

Analysis of the Wesely (2002) data reveals that there is a linear correlation between log(VP) and log(r_{cl}), as illustrated in Figure_Apx D-4 and Equation_Apx D-1 below. Linear regression *yields* r_{cl} *as* a function of VP ($R^2 = 0.606$):

Equation_Apx D-1.

$$log(r_{cl}) = 0.489log(VP) + 3.068$$
$$\therefore\ r\_cl = 1170 \times VP^{0.498}$$



Figure_Apx D-4. Cuticular Resistance as a Function of Vapor Pressure

Method 2 – Empirical Calculation of Cuticular Resistance: Method 2 estimates r_{cl} value using various empirical equations found in literature. This method assumes the vapor pressure of the chemical at 20 to 25 °C is equal to the saturation vapor pressure. For VOCs, using the equations collectively provided under equation below (Welke et al., 1998) the polymer matrix-air partition coefficient (K_{Mxa}) can be calculated as follows:

$$\log\left(K_{MXa}\right) = 6.290 - 0.892\log(\mathrm{VP})$$

Next, K_{Mxa} can be converted to the cuticular membrane-air partition coefficient, K_{Cma}:

$$K_{CMa} = 0.77K_{MXa}$$

Welke et al. (1998) also provide an empirical relationship between the polymer matric-water partition coefficient and the air-water partition coefficient, K_{Mxw}. Recognizing the air-water partition coefficient is the Henry's Law constant, HLC (unitless), yields the following:

$$K_{MXw} = K_{MXa} \times HLC$$

This relationship can be generalized from the polymer matrix to the cuticular membrane, as follows:

$$K_{CMw} = K_{CMa} \times HLC$$

In a separate study, Kerler (1988) have developed an empirical relationship that equates K_{CMw} to the permeance coefficient for cuticular membranes, P_{CM}. However, this relationship was developed using data for non-volatile chemicals. Consequently, applying it to volatile organic chemicals introduces a large amount of uncertainty to the analysis and may not be scientifically justifiable.

$$\log(P_{CM}) = 238\left(\frac{\log(K_{CMw})}{MV}\right) - 12.48$$

In the above equation, MV is the molecular volume of the chemical in question, which can be calculated from the molar mass, m (units of g/mol), and density, d (units of g/cm^3), as follows:

$$MV = \ m/d$$

Finally, r_{cl} is understood to be the inverse of P_{CM}. The above relationships can be put together and simplified to yield a single equation for r_{cl} as a function of vapor pressure, molar mass, and density:

$$r_{cl} = \left(\frac{HLC \times 1.51 \times 10^6}{VP^{0.892}}\right)^{\frac{-238d}{m}} \times 10^{12.48}$$

Method 3 – Read-Across of Cuticular Resistance from an Analog: This method assumes that chemicals that have structural similarity, physical and chemical similarity, and exhibit similar vapor pressures will also exhibit similar r_{cl} values. Available data in literature (Wesely et al., 2002) can be used as a crosswalk for read-across determination of r_{cl}. The unknown r_{cl} value is then assumed to be equal to the r_{cl} of the analog.

D.1.2.10 Other Model Settings

EPA assumed flat terrain for all modeling scenarios.

D.1.2.11 Ambient Air Exposure Concentration Outputs

Hourly-average air concentration and total (wet and dry) deposition rate outputs were provided from AERMOD for each exposure point around each distance ring (*i.e.*, each of 16 exposure points around a finite distance ring or each exposure point 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 exposure point around each ring. Daily averages for the area distance ring were calculated as the arithmetic average of the hourly data for each day modeled across all exposure points within the area distance ring. This results in the following number of daily average concentrations at each distance modeled.

1. Daily averages for TRI and NEI reporting facilities (using 2016 calendar year meteorological data): One daily average concentration for each of 366 days for each of 16 exposure points 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 TRI reporting facilities (using 2019 calendar year meteorological data): One daily average concentration for each of 365 days for each of 16 exposure points around each finite distance ring. This results in a total of 5,840 daily average concentration values for each finite distance modeled (365 × 16 = 5,840).

Period averages were calculated by averaging all the hourly values at each exposure points for each distance ring over 1 year. This results in a total of 16 period average concentration values for each finite distance ring. Additionally, period averages across all years were calculated by averaging all hourly values at each exposure points for each distance ring across all multiple years.

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 ($\mu g/m^3$) for ambient air concentrations and grams per square meter (g/m^2) for deposition rates.

Post-processing scripts were used to extract and summarize the output concentrations for each facility, release, and exposure scenario. The following statistics for daily- and period-average concentrations

were extracted or calculated from the results for each of the modeled distances (*i.e.*, each ring or grid of exposure points) and scenarios (also see Table_Apx D-6):

- minimum;
- maximum;
- average;
- standard deviation; and
- 10th, 25th, 50th, 75th, and 95th percentiles.

The above equations assume instantaneous mixing with no degradation or other means of chemical reduction in soil over time and that 1,1-dichloroethane loading in soil is only from direct air-to-surface deposition (*i.e.*, no runoff).

Table_Apx D-6. Description of Daily or Period Average and Air Concentration Statistics

Statistic	Description
Minimum	The minimum daily or period average concentration estimated across all exposure points at the modeled distance.
Maximum	The maximum daily or period average concentration estimated across all exposure points at the modeled distance.
Average	Arithmetic mean of all daily or period average concentrations estimated across all exposure points 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 across all exposure points 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.

Using the modeled 95th percentile maximum daily deposition rates described in Table 3-10, the concentration of 1,1-dichloroethane in soil was calculated using the following equations:

Equation_Apx D-2.

$$Daily_{Dep} = Tot_{Dep} \times Ar \times CF$$

Where:

Ann_{Dep} = Total daily deposition to soil (µg)
Tot_{Dep} = Daily deposition flux to soil (g/m^2)
Ar = Area of soil (m^2)
CF = Conversion of g to µg

Equation_Apx D-3.

$$Soil_{Conc} = Daily_{Dep}/(Ar \times Mix \times Dens)$$

Where:

$Soil_{Conc}$ = Daily-average concentration in soil (µg/kg)
Ann_{Dep} = Total daily deposition to soil (µg)
Mix = Mixing depth (m); default = 0.1 m from the European Commission Technical Guidance Document (ECB, 2003)

Ar	=	Area of soil (m^2)
$Dens$	=	Density of soil; default = 1,700 kg/m^3 from the European Commission Technical Guidance Document (ECB, 2003)

The above equations assume instantaneous mixing with no degradation or other means of chemical reduction in soil over time and that 1,1-dichloroethane loading in soil is only from direct air-to-surface deposition (*i.e.*, no runoff).

D.1.2.12 Physical Source Specifications: NEI Release Facilities

EPA modeled each NEI emission source in its own model run, even for facilities with multiple sources. Site-specific parameter values were used in modeling, when available. When parameters were not available and/or values were reported outside of normal bounds, reported values were replaced using procedures that EPA uses in its AirToxScreen (see Section 2.1.3 of the AirToxScreen Technical Support Document[7] and Section D.1.2.6 herein). For some stack parameters, a default values based on the source classification code (SCC) of the emission source (as reported in the NEI) was used. If there was no default value for the source's SCC, a global default value was used.

EPA used replacement values for release height, length, and width for most fugitive sources. For 2,453 NEI fugitive sources that had release heights, length, and width values that were missing or reported as zero, the Agency set their release heights to 3.048 m. For 62 NEI fugitive sources that had values above zero for length and width, but the release heights value that were missing or reported as zero, EPA set their release heights to 0 m. Values were missing or reported as 0 m for length for 2,641 sources and for width for 2,630 sources. The Agency replaced these values with a value of 10 m. For any missing values of angle (1,584 sources), EPA replaced them with zero degrees. There were 6,889 regular vertical sources (modeled as "POINT" sources in AERMOD), while 129 were vertical sources with rain caps (modeled as "POINtrichloroethaneP"), 95 were horizontal sources (modeled as "POINTHOR"), and 9 were downward-facing vents (also modeled as "POINTHOR"). These source-type designations in AERMOD engage distinct algorithms regarding how the releases initially disperse when leaving the sources. SCCs were provided for each point source.

EPA used the NEI-provided values for most point sources, but replacement values were needed for exit gas temperature and/or exit gas velocity for over 1,000 point sources. For 17 sources that had reported exit gas temperature of 0 °F, EPA replaced the value with the default values by SCC. One of the sources that was not in the SCC default file. EPA used a global default value of 295.4 K for the exit gas temperature. All point sources had in-bounds values for release heights and inside stack diameters, so no replacements were required for those parameters. Three sources that had exit gas velocity values slightly above the maximum bounding value of 1,000 feet per second (ft/s), were replaced with the maximum in-bounds value of 1,000 ft/s (304.8 m/s). For sources that had values for exit gas velocity that were missing or zero (1,344 sources), the values of inside stack diameter and exit gas flow rate was used to calculate exit gas velocity as shown in Table_Apx D-7. Minimum or maximum in-bounds values were used for those calculated exit gas velocity values that were out of bounds (15 sources).

[7] Technical Support Document: EPA's Air Toxics Screening Assessment 2018 AirToxScreen TSD (accessed June 11, 2025).

Table_Apx D-7. Procedures for Replacing Values Missing, Equal to Zero, or Out of Normal Bounds for Physical Source Parameters for NEI Sources

Parameter	Bounds	Condition			
		Value Missing or 0			Value Out of Normal Bounds
		First Pass	Second Pass (First Pass Unsuccessful)	Third Pass (First Two Passes Unsuccessful)	
Stack height	1–1,300 ft (0.3048–396 m)	Use default value by SCC (pstk file)	Use global default: 3 m	N/A	Use the minimum or maximum in-bound value if below or above bounds, respectively
Stack inside diameter	0.001–300 ft (0.0003048–91.4 m)	Use default value by SCC (pstk file)	Use global default: 0.2 m	N/A	Use the minimum or maximum in-bound value if below or above bounds, respectively
Stack exit gas temp. [a]	>0–4,000 °F (>255.4–2,477.6 K)	Use default value by SCC (pstk file)	Use global default: 295.4 K	N/A	Use the minimum or maximum in-bound value if below or above bounds, respectively
Stack exit gas velocity	0.001–1,000 ft/s (0.0003048–304.8 m/s)	Calculate from existing exit gas flow rate and inside diameter: (4 × flow) / (pi × diameter^2)	Use default value by SCC (pstk file)	Use global default: 4 m/s	Use the minimum or maximum in-bound value if below or above bounds, respectively
Fugitive height	N/A	0 m if length and width are not missing and are above 0; 3.048 m if length or width are missing or 0	N/A	N/A	N/A
Fugitive length	N/A	10 m	N/A	N/A	N/A
Fugitive width	N/A	10 m	N/A	N/A	N/A
Fugitive angle	N/A	0 degrees	N/A	N/A	N/A

K = Kelvin; SCC = source classification code

[a] For exit gas temperatures, AirToxScreen's bounds were set so that values must exceed 0 °F.

Notes: pstk file = file of default stack parameters by source classification code (SCC) from EPA's SMOKE emissions kernel: pstk_13nov2018_v1.txt, retrieved on 28 September 2022 from https://cmascenter.org/smoke/ (accessed June 11, 2025)

D.2 Inhalation Exposure Estimates for Fenceline Communities

Acute and chronic inhalation exposures were estimated based on air concentrations estimated in Section 3.3.1 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).

The equations used to calculate each of the exposure values provided below:

Equation_Apx D-4.

$$AC = (DAC \times ET)/AT$$

$$ADC = (AAC \times ET \times EF \times ED)/AT$$

$$LADC = (AAC \times ET \times EF \times ED)/AT$$

Where:

AC	=	Acute concentration (µg/m^3)
DAC	=	Daily Average Air Concentration, model output reflecting average concentrations over a 24-hour period (µg/m^3)
ET	=	Exposure time (24 hours/day)
AAC	=	Annual Average Air Concentration, model output reflecting average concentrations over a year (µg/m^3)
EF	=	Exposure frequency (365 days/year)
ED	=	Exposure duration (1 year for non-cancer ADC; 78 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

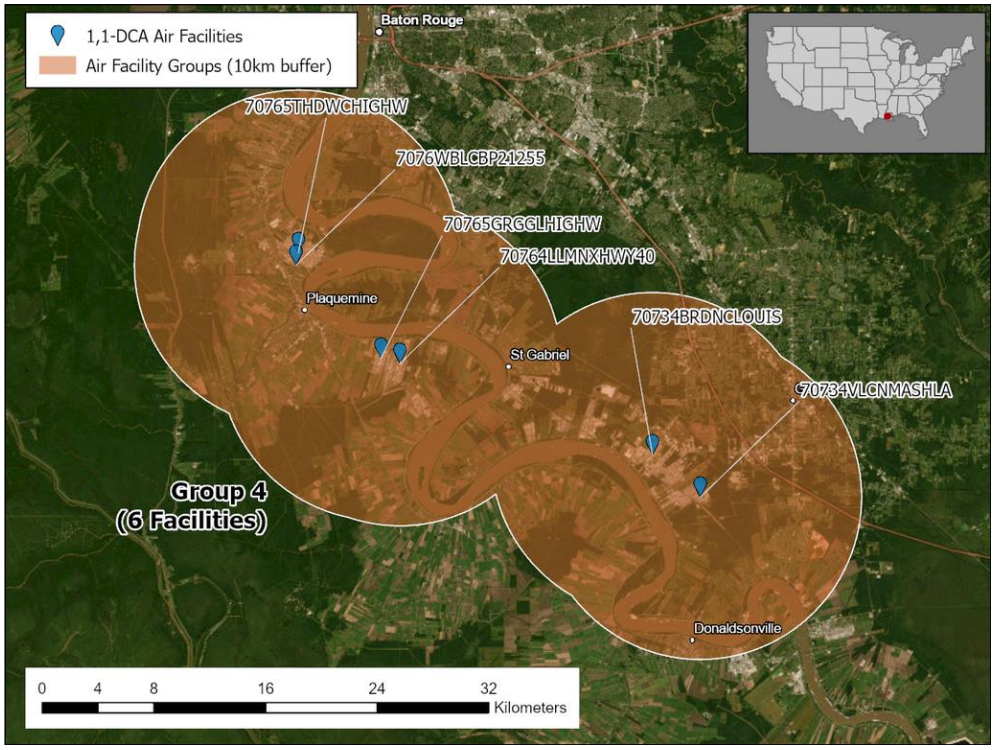
For fenceline communities, all exposure estimates assume continuous exposure (24 hours/day) throughout the duration of exposure. The exposure duration used to calculate the LADC is based on the 95th percentile of the expected duration at a single residence, 78 years and the averaging time is based on a 78-year lifetime.

Detailed reporting of modeled air concentrations and corresponding AC, ADC, and LADC estimates are provided in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025r), *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD Generic Releases Exposure and Risk Analysis* (U.S. EPA, 2025p), and in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Supplemental Information on AERMOD NEI Exposure and Risk Analysis* (U.S. EPA, 2025q).

D.3 Aggregate Analysis Across TRI 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 methodology for this analysis is consistent with what was previously described in the *Draft Supplement to the Risk Evaluation for 1,4-Dioxane* (U.S. EPA, 2023b).

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-kilometer 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 D-5).



Figure_Apx D-5. Example of Group of Air Releasing Facilities with Overlapping 10-Kilometer Buffers for Aggregate Air Risk Screening

EPA combined modeled air concentrations from each facility in the group 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 this analysis, the facilities are treated as if they are all releasing from the same point. This is a conservative approach, since the facilities within 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 Equation_Apx D-5 below for each distance interval.

Equation_Apx D-5.

$$MOE_{total} = 1/(1/(MOE_1) + 1/(MOE_2) + 1/(MOE_3) + \cdots)$$

Where:

MOE_{total} = Aggregated MOE value for the group
$MOE_{1,2,3}$ ... = 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

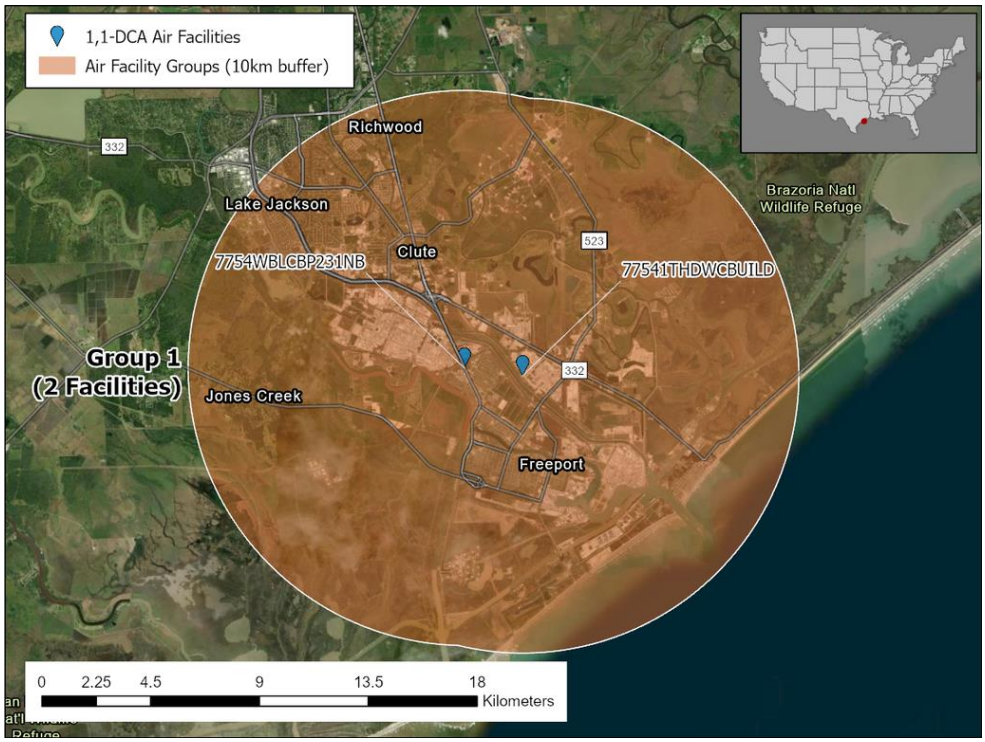
only one or more facilities. The resulting aggregate risk calculations were reviewed to determine where the numerical results suggested a concern for aggregate air risk that had not been represented by the individual facility risk analysis. Where this additional risk was flagged, the mapped locations 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.

The grouping analysis for 1,1-dichloroethane resulted in four groups of nearby facilities, ranging from two to six facilities per group (Table_Apx D-8). No additional aggregate air risk relative to benchmarks was identified for each of the four groups. For one of the groups (Group 2) there is an additional distance interval (100 m) showing risk from the aggregate calculation greater than 1×10^{-6}, but not from the individual facilities. However, the inspection of the mapped locations of the facilities within Group 2 shows that the contributing facilities are greater than 1 km apart, so this aggregate scenario would not occur. Therefore, further inspection and additional land use analysis were not warranted for Group 2. While Groups 3 and 4 each contained one or more facilities showing risk out to some distance, there was no additional distance interval showing risk from the aggregate calculation greater than 1×10^{-6}. Although 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 result in noticeable increased risk estimates (*i.e.*, aggregation did not increase cancer risk levels beyond individual facility risk levels), so any determinations of risk are already accounted for by the individual facility analysis. No cancer risk estimates in Group 1 exceeded 1 in 1 million benchmark.

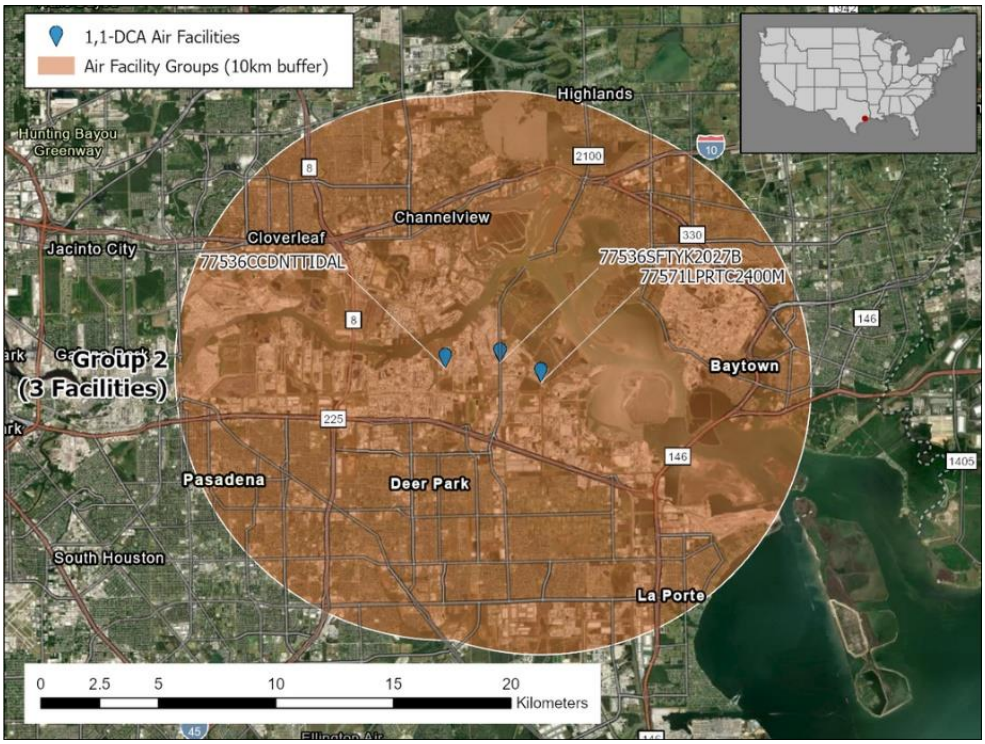
Table_Apx D-8. Summary of Aggregate Analysis for TRI Facilities

Total Air Facilities with TRI Release Data	Number of Facilities in Groups	Number of Groups	Number of Groups with Additional Aggregate Risk
23	13	4	0

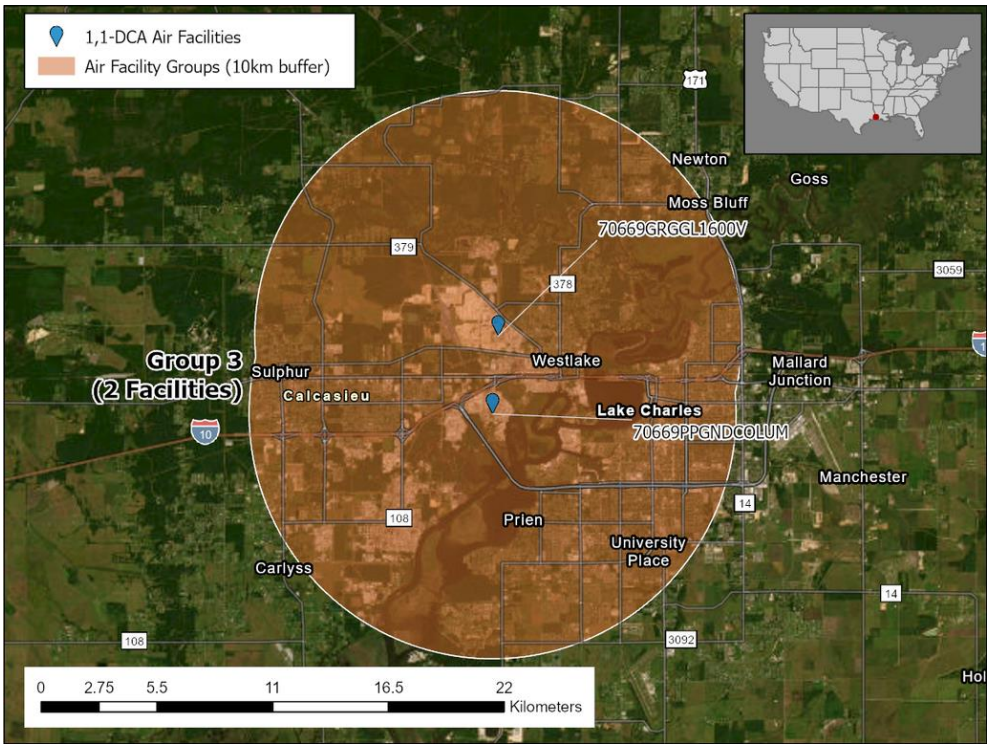
Maps of the four facility groups with the 10-kilometer buffers used to define them are provided below in Figure_Apx D-6 through Figure_Apx D-9. Results of the aggregate analysis are presented in the *Risk Evaluation for 1,1-Dichloroethane — Supplemental Information File: Supplemental Information on AERMOD TRI Exposure and Risk Analysis* (U.S. EPA, 2025r).



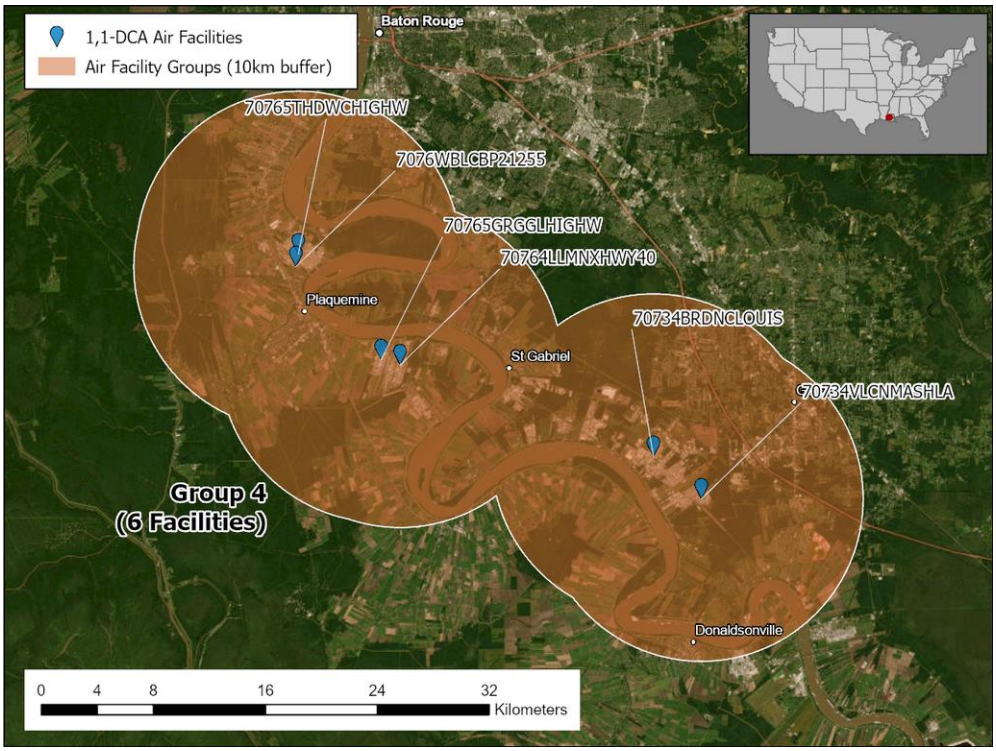
Figure_Apx D-6. Map of Aggregated Air Facilities, Group 1



Figure_Apx D-7. Map of Aggregated Air Facilities, Group 2



Figure_Apx D-8. Map of Aggregated Air Facilities, Group 3



Figure_Apx D-9. Map of Aggregated Air Facilities, Group 4

D.4 Ambient Air Exposure to Population Evaluation

TRI Population Evaluation

This evaluation aimed to quantify population exposure around a subset of AERMOD TRI release sites where estimates of non-cancer risk or cancer risk exceed minimum benchmarks for human health, and thus reflect high-end exposures of 1,1-dichloroethane. The 95th percentile (p95) of AERMOD average daily modeled results were used in order to remain conservative with the scenario modeled. Average daily p95 air concentrations (ADC) and life-time average daily p95 concentrations (LADC) of 1,1-dichloroethane were estimated prior to this evaluation. Cancer risk (CR) values were then estimated from LADC values. Of the 23 TRI facility releases modeled using AERMOD, 10 resulted in CR values that exceeded the minimum CR value of 1×10^{-6} whereas none resulted in modeled air concentrations that exceeded the minimum non-cancer risk (NCR), which would include a margin of exposure (MOE) calculation below the benchmark of 300. These 10 AERMOD TRI release sites thus became the focus of the population characterization because of the ability to capture high-end exposures of 1,1-dichloroethane in ambient air.

The goal of population characterization was to quantify population density and PESS groups. Nearby environments and community infrastructure of interest were identified, and distances between the subset of AERMOD TRI air release sites and population census blocks and community locations were estimated to understand the likelihood that these populations experience high-end exposures of 1,1-dichloroethane.

Analysis Assumptions and Uncertainties

There is an inherent uncertainty associated with the TRI coordinates that are meant to represent sites of 1,1-dichloroethane release to ambient air. For instance, in some cases the TRI coordinates may be located at the edge of the facility complex, such as at an entrance to the facility, a mailbox address, or a road leading up to the facility, which may not capture the actual site of emission. The accuracy of the facility's release site coordinates is thus strictly tied to the accuracy of the AERMOD results at the various distances modeled, and that were considered in this evaluation. This degree of uncertainty should be considered when interpreting the population results.

The population metrics and distances estimated as a part of the analysis also relies on computed centroid coordinates from the boundaries of U.S. census (polygon shapefile) blocks. Because the size of census blocks is determined by population, rural areas tend to have larger census block polygons compared to densely populated urban or suburban areas. This "centroid effect" is also a factor that affects the distances estimated between facility release sites and the surrounding census blocks, and thus as with the modeled AERMOD distances, the distances relative to census blocks and community infrastructure that are being calculated should not be overinterpreted.

In some cases, CR values greater than or equal to 1×10^{-6} are found at 1,000 m, but not 2,500 m, so it cannot be ruled out that CR does not exceed 1×10^{-6} between 1,000 and 2,500 m away from the AERMOD TRI release site. Since it is unlikely that populations beyond 2,500 m are exposed to CR values greater than 1×10^{-6}, only census block centroids within 2,600 m were considered for this evaluation. It is important to note, however, that there is a possibility that census block areas exist within 2,600 m, but are not included in this evaluation because their centroids are positioned just beyond 2,600 m.

Methods

Overview of Approach: After identifying which AERMOD TRI release sites to focus on for this evaluation (*i.e.*, those with CR values $>1\times10^{-6}$ that reflect a high-end exposure), the next step involved a

visualization of the surrounding landscape and community infrastructure using Google Earth/Maps to inform which kinds of population, household, and community location data to obtain and analyze. 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*. However, radial distance measurements were not made in Google Earth because these measurements were made a later step with more precision. An internal decision framework document to aid in identifying PESS groups was used to help identify which environments and community infrastructure to examine. Specific population densities, environment and community locations of interest, and distances between the TRI release sites and census blocks and spatial boundaries of these environments/infrastructure were quantitated using GIS and R computing software. Input data was obtained from external sources and imported into R. New results generated as a part of this evaluation were compared with AERMOD results and their associated modeled distances to identify the likelihood that these populations experience high-end exposures to 1,1-dichloroethane.

Site Selection and Visualization: LADC results from all 23 AERMOD TRI release sites were used to estimate cancer risk values at the following discrete or areal modeled distances: 10, 30, 30 to 60, 60, 100, 100 to 1,000, 1,000, 2,500, 5,000, and 10,000 m. Ten TRI facilities with LADC levels and calculated cancer risk values greater than 1×10^{-6} were identified. Site characteristics of these 10 TRI facilities are included below in Table_Apx D-9.

Table_Apx D-9. Facilities Reporting TRI Emission Included in General Population Characterization

OES	Facility Name	City	State	TRI ID
Manufacturing	Occidental Chemical Holding Corp – Geismar Plant	Geismar	LA	70734VLCNMASHLA
	Oxy Vinyls LP La Porte VCM Plant	La Porte	TX	77571LPRTC2400M
Processing as a reactant	Westlake Vinyls Inc	Calvert City	KY	42029WSTLK2468I
	Westlake Lake Charles North	Westlake	LA	70669GRGGL1600V
	Eagle US 2 LLC	Westlake	LA	70669PPGNDCOLUM
	Shintech Plaquemine Plant	Plaquemine	LA	70764LLMNXHWY40
	Blue Cube Operations LLC – Plaquemine Site	Plaquemine	LA	7076WBLCBP21255
	Freeport_Olin BC	Freeport	TX	7754WBLCBP231NB
Waste handling, disposal, treatment, and recycling	Axiall LLC	Plaquemine	LA	70765GRGGLHIGHW
	Ash Grove Cement	Foreman	AR	71836SHGRVPOBOX

Google Earth/Google Maps was used to conduct a preliminary (visual) analysis of the areas surrounding these 10 TRI facilities to identify residential neighborhoods and environments or community infrastructure of interest that may include a PESS group. For example, homes, parks, childcare centers, schools, places of worship, hospitals and clinics were among the types of environments and community infrastructure being considered and that were visually inspected.

Population and Household Data Selection

Population data associated with census block groups was gathered from the American Community Survey (ACS) 2017 to 2021, which includes 5-year estimates of community member characteristics. These data and the 2021 census block polygon (shapefile) dataset were obtained from data.census.gov

and TIGER/Line Shapefile (accessed June 11, 2025), respectively. Data for the locations of childcare centers, public schools, private schools, colleges and universities, places of worship, and healthcare facilities (hospitals, urgent cares, VA health facilities, and dialysis centers) were obtained from the Department of Homeland Security's Homeland Infrastructure Foundation-Level Data Geoportal (accessed June 11, 2025).

Data Pre-Processing

Much of the data analysis in this evaluation was performed using R computing software. The census block dataset contains over 8 million rows, which is an impractical size to perform complex geospatial operations with. To make the dataset more manageable to work with in R, the census block dataset was clipped to 2,600 m of the subset of AERMOD TRI release sites. The 2,600 m distance was chosen because 1,000 m is the furthest distance in which a CR great or equal to 1×10^{-6} was observed, but it cannot be ruled out that CR does not exceed 1×10^{-6} between 1,000 and 2,500 m in those instances. The clipping area was extended an additional 100 m to account for small changes in the geospatial area that can result when transforming spatial data from one projection system to another. Only census block centroids within 2,600 m of the subset of AERMOD TRI release sites were included for the next steps in the analysis.

The ACS database containing population and household-level information is available at the census block group level, which may contain one of more individual census blocks. EPA's goal was to estimate population and household metrics for each individual census block and then evaluate block-level results at relevant distances to the subset of AERMOD TRI release sites. Thus, it was necessary to downscale the ACS population and household data from the census block group level to the level of individual blocks. To do this, the proportion of individual blocks within a block group was used with population and household data at the block group level to estimate the expected results scaled down to individual blocks.

Identifying Sites with a General Population

Prior to performing any weighted statistics, individual census blocks without a population based on the population column of the census block group centroid dataset were removed. This column describes the 2020 Census population count for the census block. However, to protect the privacy of survey respondents, these population counts were subjected to random noise, which means that a small amount may have been added or subtracted to the population count to slightly obscure the original population value. Although this pre-processing step may be less conservative than assuming every census block has a population, it likely removes census blocks in non-residential areas and so was the preferred step to take. All census block centroids within 1,000 and 2,600 m of each facility were first grouped by their census block group ID. Then, the number of populated census blocks per block group located within 1,000 or 2,600 m of the facility was calculated. The block group's population was then multiplied by the number of populated census blocks within 1,000 or 2,600 m of the facility and then divided by the total number of census blocks in the block group. The weighted populations for each of the census block groups were then summed together to provide the estimated weighted population size around each facility.

When adding population metrics together for a given OES, it is important to identify where potential overlap between facilities and populations exist to avoid double counting. None of the census blocks within 1,000 m of the facilities overlapped with each other, so all the facility populations were simply added to find the population by OES. Some census blocks were within 2,600 m of multiple facilities. One census block was within 2,600 m of the Shintech Plaquemine Plant site (OES: Processing as a reactant), Blue Cube Operations LLC Plaquemine Site (OES: Processing as a reactant), and the Axiall

LLC site (OES: Waste handling, disposal, treatment, and recyling). Additionally, two more census blocks were located within 2,600 m of both the Westlake Lake Charles North site and the Eagle US 2 LLC site (both of which have an OES of Processing as a reactant).

To account for these population overlaps and avoid double counting populations when summing population totals by OES, the census blocks associated with more than one TRI facility were first identified. The maximum weighted population of these block groups was then calculated. When adding the populations for each OES together, the non-maximum weighted population(s) for the same census blocks were then subtracted. This avoids double counting populations, while still allowing for a conservative estimate of the total population by OES.

Characterizing Exposure

AERMOD models air concentrations at eight discrete distances ranging from 10 to 10,000 m and two areal-averaged distances at 30 to 60 m and 100 to 1,000 m. This means if high levels of 1,1-dichloroethane in ambient air are modeled at 1,000 m, EPA cannot rule out that distances between 1,000 to 2,500 m do not also experience high levels of 1,1-dichloroethane in air. Comparing estimated distances of the general population to both the maximum AERMOD modeled distance that reflect high-end exposure, as well as the next modeled distance, allows us to evaluate the possibility of exposure at and in between these two distances. However, given that air concentrations decrease linearly with distance, a possible exposure may not be a likely exposure if the general population lives well beyond the AERMOD modeled distance that CR was found. Unreasonable risk determinations based on high-end exposures should consider these relevant distances between modeled concentrations and where populations are expected as well as the magnitude of distances being evaluated. This is important given the uncertainty surrounding distance estimates is greater at shorter distances than longer distances since TRI coordinates may not necessarily reflect the true air release sites of 1,1-dichloroethane.

NEI Population Evaluation

The methods taken for the NEI population evaluation were very similar to those taken for the TRI population evaluation, and so much of the goals, assumptions and uncertainties, methods, site/data selection, and exposure characterization applies equally. There were a few notable differences in how the AERMOD NEI results were analyzed, which are outlined below.

The NEI data include releases from multiple emission units for a given facility. These units may be fugitive and/or stack type emissions, each of which may be assigned a different OES designation. This data was obtained for 2014 and 2017. It is important to note that the facility release sites, number of emission units per site, their type of emissions, and their subsequent OES designation can change between 2014 and 2017. Because concentrations from multiple emission units were modeled using AERMOD, it was desirable to account for their aggregate release and exposure. This was done by adding calculated CR values for each AERMOD modeled distance across emission units of a given facility. This step was taken separately for 2014 and 2017. These facility total CR values were then used to identify a subset of AERMOD NEI release sites to focus on for the population evaluation by selecting on those facility CR totals that exceed the minimum CR value of 1×10^{-6}.

The population and household data were collected using the same approach for the TRI population evaluation with one notable exception. Although the TRI evaluation considered only a single site (coordinate) for the geospatial analysis, EPA's NEI evaluation accounted for all emissions units within a facility. In other words, census blocks and their associated ACS data were geospatially analyzed relative to each emission unit with a given facility complex. The population metrics were obtained for a given emission unit and then summed across all units for a given distance threshold (*e.g.*, 1,000 m from the

emission units). This was done for facility release sites in both the 2014 and 2017 datasets; however, the list of facilities and number of emission units were largely the same between the 2 years.

With respect to exposure characterization, it is important to note using an aggregate approach it is assumed that each population surrounding an individual emission unit is equally exposed to the facility total 1,1-dichloroethane levels and CR values. Although this may overestimate exposure and CR values for a given population around an emission unit, this conservation step was preferred over underestimating exposure that may result by assuming that emission units are not aggregating with one another.

EPA determined that 517 facility release sites have estimated CR values that exceed the minimum CR value of 1×10^{-6}. In an effort to refine the focus on those sites that pose a likely exposure to these CR values, the Agency evaluated the population for only those AERMOD NEI release sites that have a populated census block that overlaps or is within 100 m of the furthered modeled distances where CR greater than or equal to 1×10^{-6} is expected. For example, if a facility total CR value for the AERMOD modeled 100 to 1,000 m area exceeds 1×10^{-6}, then this site was only considered with a populated census block was measured within 1,100m of any individual emission unit. This subset of AERMOD NEI release sites were evaluated specifically to interpret population results that have a greater confidence of true exposure to the estimated CR values. It should not preclude, however, that there are additional AERMOD release sites that have a likely exposure to estimated CR values if a populated census block was measured beyond the 100-m threshold. That is, EPA cannot rule out that exposure is not occuring a distance from 100 m to a few hundred meters or greater from the emission units because of the uncertainties in where populations may be living that come with performing a proximity analysis based on census block centroids.

Another notable difference between the NEI and TRI population evaluations is that (at present), only populations within 1,000 m of the emission units were considered for the NEI evaluation. In addition, proximity to community locations and infrastructure of interest have not yet been evaluated.

Appendix E SURFACE WATER CONCENTRATIONS

E.1 Surface Water Monitoring Data

E.1.1 Monitoring Data Retrieval and Processing

The complete set of 1,1-dichloroethane monitoring results stored in the Water Quality Portal (WQP) was downloaded in March 2023 (NWQMC, 2022) using the *dataRetrieval* package in R (R Core Team, 2022) and imported directly into the R computing platform console. Specifically, the *readWQPdata* and *whatWQPsites* functions were used to acquire all WQP sample results and site data with a "1,1-Dichloroethane" characteristic name. No additional arguments were used with both functions. The downloaded dataset is large and comprehensive, where only certain data fields were desired for EPA's intended use in the 1,1-dichloroethane risk evaluation. The WQP dataset was subsequently filtered for only surface water sample types with the following "MonitoringLocationTypeName":

- Spring
- Stream
- Wetland
- Lake
- Great Lake
- Reservoir
- Impoundment
- Stream: Canal
- Stream: Ditch
- Facility Other
- Floodwater Urban
- River/Stream
- River/Stream Ephemeral
- River/Stream Intermittent
- River/Stream Perennial

Sample results identified as below the detection limit or non-detects (*i.e.*, "ResultMeasureValue" indicated with an N/A) were replaced with values at one-half the quantitation limit ("DetectionQuantitationLimitMeasure.MeasureValue" ÷ 2). All rows without a sample result value or reported detection quantitation limit were subsequently removed. The sample result values of any replicate samples collected on the same day at the same time were averaged. Rows with an "ActivityYear" between 2015 and 2020 were kept, representative samples collected during this time period. Samples flagged as QC blanks in the "ActivityTypeCode" column were removed. Only dissolved aqueous samples were kept as indicated by a "µg L^{-1}" or "mg L^{-1}" unit identifier in the "ResultMeasure.MeasureUnitCode" column. Sample units were adjusted to µg L^{-1} if needed. All sample results less than zero were forced to equal zero. Because one-half of the detection quantitation limit was used to replace below detection or non-detection sample result values, an appropriate detection quantitation limit cutoff was determined. The 95th quantile, 99th quantile, and max detection quantitation limits were examined to identify that less than or equal to 5 µg L^{-1} is a reasonable detection quantitation limit. Any adjusted sample result values greater than 5 µg L^{-1} was removed.

Monitoring data from drinking water systems were acquired from the Third Unregulated Contaminant Monitoring Rule (UCMR3) database (U.S. EPA, 2017b). The UCMR3 dataset includes PWSs serving more than 10,000 people and 800 of the nation's PWSs that serve 10,000 or fewer people. The complete

history of 1,1-dichloroethane measurements in the UCMR3 finished drinking water dataset was acquired. Sample result values below the Minimum Reporting Limit (MRL) as indicated by a "<" sign in the "AnalyticalResultsSign" column were replaced with the MRL. In this case, the highest reported MRL for all 1,1-dichloroethane drinking water measurements is 0.03 $\mu g L^{-1}$, which is low enough where the full MRL as opposed to one-half of the MRL can be used. Sample details were reviewed and screened to remove those indicating that they were collected from groundwater (*i.e.,* those including "Well" in the "SamplePointName" column) and select for those only including surface water source types (*i.e.,* those including "SW" in the "FacilityWaterType").

E.2 Surface Water Concentration Modeling

E.2.1 Hydrologic Flow Data Assimilation

The joint U.S. Geological Survey (USGS) and EPA National Hydrography Dataset (NHDPlus V2.1) national seamless flowline network database was used to obtain modeled stream or river (hereby referred to as stream) hydrologic flow data. The NHD dataset is one of the largest national hydrologic datasets, containing geospatially delineated flowline stream networks, information on the sequential linkages between flowline reach segments (*i.e.,* to-node and from-node identifiers), and modeled flow values for greater than 2.7 million stream segments nationwide (U.S. EPA, 2016b). The NHD dataset is comprehensive at the nation scale and has been used for numerous regional and national hydrologic modeling studies since its creation. The NHD dataset contains mean annual and monthly stream flows for nearly all individual stream segments in the national flow network. Stream flows were determined by the Enhanced Runoff Method (EROM) Flow Estimate model, which determines flow values through from multi-step estimation and calibration process with each step designed to incrementally improve the stream flow estimate. The first step involves accumulating runoff based on flow balance grids from a 30-year period from 1971 to 2000. The last step involves correcting flows at a distance upstream and downstream of an observed gage flow. The modeled EROM flow data fields are labeled with a leading "QE." The dataset is incorporated into recordkeeping and modeling across EPA programs that require knowledge of a national stream network, providing consistency and compatibility with projects across the EPA. Pertaining to our efforts in this risk evaluation, the EPA's Enforcement and Compliance History Online (ECHO) Database uses facility-linkages to the 14-digit Hydrologic Unit Classification (HUC) reach codes associated with the NHD flowline network.

A list of facilities releasing 1,1-dichloroethane to surface waters were obtained from the ECHO Pollutant Load Tool "Custom Search" tab as outlined in *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Environmental Releases and Occupational Exposure Assessment*. These facilities include those that directly discharge into surface waters, compiled from their parent TRI and Discharge Monitoring Reports (DMR) database. None of the facilities indirectly discharge to a surface water body; for example, which may arise from the transfer of 1,1-dichloroethane to a disposal facility. For each facility, the NPDES identifier was used to retrieve a corresponding 14-digit NHDPlusV2 reach code using the ECHO DMR API wrapper ("dmr_rest_services.get_facility_report"). This step was repeated for each year between 2015 to 2020 to obtain reach codes that correspond to the year that wastewater discharge data was collected. Note, all NPDESs pulled from TRI are also represented in the DMR database.

Values of modeled EROM mean annual stream flow (QE_MA) and monthly annual stream flow (*e.g.,* QE_01, QE_02, QE_03, etc.) were retrieved from the seamless NHDPlusV2 flowline network database for all acquired reach codes. Since individual reach codes may include one or more flowline segments (*i.e.,* a unique COMID identifier) and thus multiple modeled flow values, the lowest flow value for a given reach code was kept. Although most NHD flowlines represent streams, some may represent

coastal water bodies, where the mean annual stream flow values are reported as an N/A or as zero. Flow values reported as N/A or zero were subsequently flagged as possible coastlines. In some cases, a reach code was not returned through the ECHO DMR API wrapper. When this occurred, a calculated facility effluent flow was used instead of a NHD modeled flow value, thus reflecting the effluent flow at the facility outfall instead of the receiving water body. Facility effluent flow was also used when a reach code was returned, but the value was reported as an N/A or zero. EPA decided this was a more conservative and efficient approach than to identify where the true outfall and receiving water body is for a given facility NPDES that did not return a reach code. Because DMR reach codes were assigned using the NHD flowline database, instances when a reach code is not returned could reflect a reporting error or an instance where the receiving water body was a lentic system such as a lake or pond. Thus, through this approach, a calculated facility effluent flow was also used in the event the receiving water body is a lake, pond, or reservoir, which would require detailed information of the lentic water body's volume to estimate the aqueous concentration. An average annual facility effluent flow (in millions of liters) was calculated by dividing the annual pollutant load (kg yr^{-1}) by the average concentration (mg L^{-1}), derived from the Pollutant Load Tool estimation function. This value was then divided by 365 to obtain an average facility effluent flow in units of millions of liters per day (MLD).

To estimate an aqueous concentration of 1,1-dichloroethane in a receiving stream, the annual pollutant load (kg yr^{-1}) was divided by a hydrologic flow value (in MLD) originating from the NHD EROM dataset and the units adjusted accordingly. Several different hydrologic flow metrics were estimated, which detailed in the next section. For each of the metrics, stream flow was compared to the calculated facility effluent flow, and the lower of the two flow values was kept. When NHD-based flow could not be estimated, the calculated facility effluent flow was chosen. The Pollutant Loading Tool returns a continuous dataset of annual pollutant load and average concentrations, so a calculated facility effluent flow value can always be used, allowing for a continuous record of flow metrics to choose from to estimate an aqueous concentration of 1,1-dichloroethane.

E.2.2 Facility-Specific Release Modeling

In previous TSCA risk evaluations, EPA applied the E-FAST 2014 tool (U.S. EPA, 2014) to estimate aqueous chemical concentrations and exposure resulting from individual facility discharges to surface waters. To make the calculations more flexibility, efficient, and repeatable, many of the underlying calculations that EPA uses were translated to an excel workbook format. Without the need to use the E-FAST software directly, which can be cumbersome and time consuming, facility pollutant loads, associated flow data, and facility release schedules can be used with the nimbler E-FAST-style excel workbook. This refinement in methodology allows an assessor to manual enter and adjust inputs parameters as needed, but more importantly, provides an opportunity to enter newer and more relevant hydrologic flow information than what was included in the older, underlying, E-FAST software (the EPA original Reach File 1 dating back to 1984). With this improved approach, facility-specific modeling can be conducted using similar methodology and logic of the E-FAST 2014 tool but with update hydrologic flow data and an overall improved confidence in the accuracy of the estimated aqueous concentrations and linkages between the facility releases and their true receiving water body. This updated approach was first employed in EPA's risk evaluation of 1,4-dioxane. This risk evaluation of 1,1-dichloroethane has adopted a similar approach herein.

Several different types of metrics were estimated using either the annual or monthly mean modeled EROM flow values: arithmetic mean flow, harmonic mean flow, the lowest 30-day average flow occurring in a 5-year period (30Q5), and the lowest 7-day average flow occurring in a 10-year period (7Q10). The harmonic mean and 30Q5 flow metrics have been used in previous risk evaluations for exposures from drinking water consumption, dermal contact, and fish ingestion that affect human health.

The 7Q10 flow metric has previously been used to evaluate exposures to aquatic ecological species. Of these flow metrics, only the arithmetic mean can be acquired from the NHDPlusV2 EROM dataset. The other flow metrics (harmonic mean, 30Q5, and 7Q10) have historically required an extensive, costly, and generally inefficient modeling procedure, which is impractical to do in a timely manner for a large list of new sites until the procedure is made more efficient. Thus, an alternative approach to estimating these flow metrics was taken, consistent with how they are calculated in the underlying E-FAST Probabilistic Dilution Model (PDM).

Regression equations from the E-FAST user manual (Versar, 2014) were applied as detailed below. NHD EROM mean annual and lowest monthly flow values serve as the foundation for these calculations, where the mean annual flow served as the arithmetic mean and the lowest monthly average flow (*i.e.*, lowest of the monthly series: QE_1, QE_2, QE_3, etc.) was used as a proxy for 30Q5 flow. Since the modeled EROM flow metrics represent averages across a 30-year timeframe, the lowest of the monthly means for a given reach is a close representation of the lowest 30-day average flow occurring in a 30-year time period (*i.e.*, 30Q30), and thus reflects a longer term average in comparison to 30Q5 flow. The arithmetic mean and "30Q30" were entered into the regression equations below to solve for the harmonic mean and 7Q10 flow metrics:

Equation_Apx E-1.

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

Where:

$7Q10$ = Modeled 7Q10 flow, in MLD
$30Q5$ = Lowest monthly average flow from NHD, in MLD

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

Where:

HM = Modeled harmonic mean flow, in MLD
AM = Annual average flow from NHD, in MLD
$7Q10$ = Modeled 7Q10 flow from the previous equation, in MLD

These different calculated stream flow metrics were then compared to the calculated facility effluent flow. When facility effluent flow exceeded a given stream flow metric (*i.e.*, facility flow > *HM*, *30Q5*, or *7Q10*), then facility effluent flow replaced the stream flow metric value. When a stream flow metric could not be estimated for the reasons outlined above, then the facility effluent flow value was also used.

For each facility, the highest annual load during the 2015 to 2020 time period was used to estimate aqueous 1,1-dichloroethane concentration. 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: 1 day, 30 days, and the maximum expected days of operation listed in Table 3-3. 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.

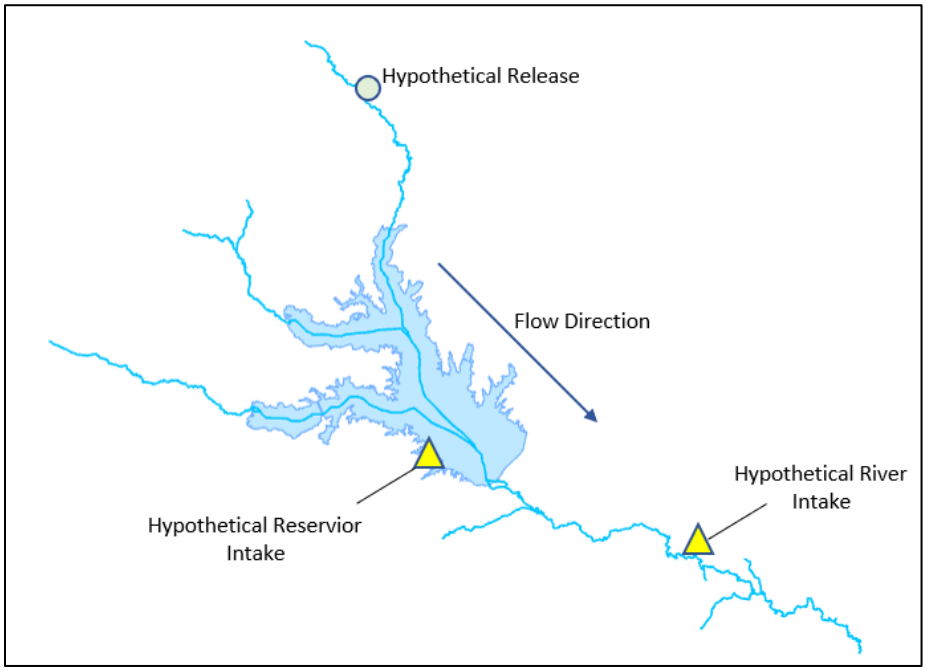
For each scenario, the aqueous concentration was calculated using the equation below:

Equation_Apx E-2.

$$Concentration\ (\mu g/L) = \frac{Daily\ Load\ \left(\frac{kg}{day}\right) * \ 10^{9}\ \left(\frac{\mu g}{kg}\right)}{Flow\ (MLD) * \ 10^{6}\ \left(\frac{L}{ML}\right)}$$

E.2.3 Modeling at Drinking Water Intakes

To estimate aqueous 1,1-dichloroethane concentrations in drinking water, surface water intake locations downstream of the facilities releasing 1,1-dichloroethane were identified. The coordinates of surface water intake locations for public water systems (PWS) were obtained from the Safe Drinking Water Information (SDWIS) Federal Data Warehouse. The site coordinates and associated NHDPlusV2 reach codes associated with facilities releasing 1,1-dichloroethane to surface waters were already obtained in the steps outlined in Section E.2.1. To obtain the reach codes associated with drinking water intake locations, the nearest neighboring flowline or waterbody from the NHDPlusV2 dataset was identified using the "Near" tool in ArcGIS Pro software. In addition, flowlines and their reach codes that intersect with standing water bodies were identified. This can occur when reservoirs are constructed from dammed rivers, which may have intake locations at the bank of the reservoir as opposed to the center link of the river (Figure_Apx E-1).



Figure_Apx E-1. Generic Schematic of Hypothetical Release Point with Surface Water Intakes for Drinking Water Systems Located Downstream

An R script was developed to search for and identify reach codes with intake locations that exist downstream of each reach code with a facility release site by using the "to-node" and "from-node" reach code sequence identifiers as a part of the NHDPlusV2 database. For each facility, the script functions by starting with the facility-linked reach code and incrementally stepping downstream to the next reach

code, recording the length of the stream segment (in km) and whether the reach has a drinking water intake. When a reach with a drinking water intake is identified, the PWS details and the total distance traveled is recorded in a separate data file. The script then continues to search downstream until hitting a terminal reach code (*i.e.,* where no subsequent reach codes can be search, such as is the case with a coastline) or when the maximum search distance is realized. For this assessment, a maximum search stream length of 250 km was applied.

The search function creates a separate data file that includes all possible combinations of PWS intakes downstream of the facility release sites. Thus, a given facility release site may encounter multiple PWSs, which each may have multiple intake locations during the search 250 km downstream. For each intake, the accompanying reach code was used to acquire modeled EROM flow data from the NHD flowline database using the approach outlined in Section 3.3.3.6.1. Because a PWS may have multiple intakes, the most upstream intake location was kept while all others were removed for the next step. Aqueous concentrations of 1,1-dichloroethane were then estimated at each intake location using a dilution factor that was calculated by dividing the stream flow of the reach or the facility effluent plant flow at the facility release site (*i.e.,* start flow) by the stream flow of the reach at the drinking water intake location (*i.e.,* end flow). If the end flow was greater than the start flow, the dilution factor was made equal to 1. The concentration estimated at the site of facility discharge was multiplied by the dilution factor to estimate an aqueous concentration of 1,1-dichloroethane at the site of the drinking water intake. For each PWS, additional information was obtained from the SDWIS Federal Reporting System (U.S. EPA, 2022b). The "PWS_TYPE_CODE" column was used to select only sites representing Community Water Systems (CWS) and Non-Transient Non-Community Water Systems (NTNCWS) for exposure analysis. In some cases, PWSs draw water from sources other than surface water, including groundwater or purchased water from another location. In a prior step, site information from SDWIS was used to select for only those PWSs that draw from surface waters as the primary source (*i.e.,* those with identified as "SW" for surface water in the "PrimarySourceCode" Column).

Appendix F GROUNDWATER CONCENTRATIONS

F.1 Groundwater Monitoring Data

F.1.1 Monitoring Data Retrieval and Processing

The complete set of 1,1-dichloroethane monitoring results stored in the Water Quality Portal (WQP) was downloaded in March 2023 (NWQMC, 2022) using the *dataRetrieval* package in R (R Core Team, 2022) and imported directly into the R computing platform console. Specifically, the *readWQPdata* and *whatWQPsites* functions were used to acquire all WQP sample results and site data with a "1,1-Dichloroethane" characteristic name. No additional arguments were used with both functions. The downloaded dataset is large and comprehensive, where only certain data fields were desired for EPA's intended use in the 1,1-dichloroethane risk evaluation. The WQP dataset was subsequently filtered for only groundwater sample types with the following "MonitoringLocationTypeName:"

- Well;
- Subsurface;
- Subsurface: Groundwater Drain; and
- Well: Multiple Wells.

Sample results identified as below the detection limit or non-detects (*i.e.,* "ResultMeasureValue" indicated with an N/A) were replaced with values at one-half the quantitation limit ("DetectionQuantitationLimitMeasure.MeasureValue" ÷ 2). All rows without a sample result value or reported detection quantitation limit were subsequently removed. The sample result values of any replicate samples collected on the same day at the same time were averaged. Rows with an "ActivityYear" between 2015 and 2020 were kept, representative of samples collected during this time period. Samples flagged as QC blanks in the "ActivityTypeCode" column were removed. Only dissolved aqueous samples were kept as indicated by a "µg L^{-1}" or "mg L^{-1}" unit identifier in the "ResultMeasure.MeasureUnitCode" column. Sample units were adjusted to µg L^{-1} if needed. All sample results less than zero were forced to equal zero. Since ½ the detection quantitation limit was used to replace below detection or non-detection sample result values, an appropriate detection quantitation limit cutoff was determined. The 95th quantile, 99th quantile, and max detection quantitation limits were examined to identify that less than or equal to 20 µg L^{-1} is a reasonable detection quantitation limit. Any adjusted sample result values exceeding 20 µg L^{-1} were removed.

Appendix G LAND PATHWAY CONCENTRATIONS IN SOIL GROUNDWATER AND BIOSOLIDS

G.1 Land Pathway (Soils, Groundwater, and Biosolids)

G.1.1 Air Deposition to Soil

EPA used AERMOD to estimate air deposition from facility releases and calculate the resulting soil concentrations near the 1,1-dichloroethane emitting facility. AERMOD modeling methodology is detailed in Appendix C.3. The highest 95th percentile maximum daily air deposition rates for each OES generally occurred at 10 m from the facility (Table_Apx O-1). For this reason, 1,1-dichloroethane soil concentrations that could result from maximum daily air deposition were estimated for each OES at a distance of 10 m from facility for determining dietary exposure of terrestrial ecological receptors. Appendix D.1.2.9 presents details and equations and details in estimating 1,1-dichloroethane in soil from air deposition.

Table_Apx O-1 presents the resulting calculated 95th percentile maximum 1,1-dichloroethane soil concentrations 10 m from facility corresponding to the applicable exposure scenarios. Across exposure scenarios, the exposure scenario for Manufacturing of 1,1-dichloroethane as an isolated intermediate resulted in the highest estimated 1,1-dichloroethane soil concentrations which could result from air deposition. These 1,1-dichloroethane soil concentrations that could result from air deposition were then used to estimate soil pore water concentrations 10 m from facility (Table_Apx O-1) according to the methodology described in Section C.2.4.4.

Table_Apx G-1. Soil Catchment and Soil Catchment Pore Water Concentrations Estimated from 95th Percentile Maximum Daily Air Deposition Rates 10 m from Facility for 1,1-Dichloroethane Releases Reported to TRI

OES	Number of Facilities	Maximum Daily Air Deposition ($g/m^2/day$) [a]	Soil Concentrations (µg/kg)	Soil Pore Water Concentrations (µg/L)
Manufacturing of 1,1-dichloroethane as an isolated intermediate	9	4.02E−02	2.36E02	1.46E02
Processing as a reactive intermediate	6	8.90E−04	5.24	3.23
Waste handling, treatment and disposal (non-POTW)	8	2.10E−05	1.24E−01	7.63E−02
[a] Estimated via AERMOD within 10 m of releasing facilities.				

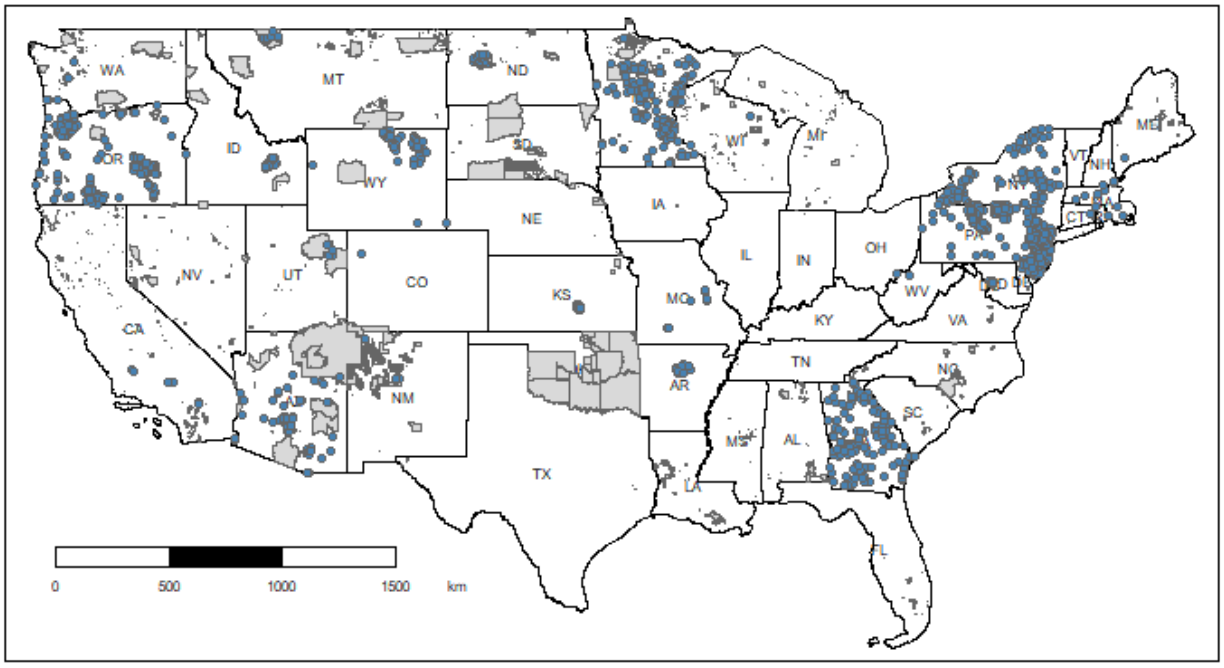
To help determine the significance of the air deposition to the groundwater exposure pathway, annual air deposition loading rates of 1,1-dichloroethane to soil were input to the Pesticide in Water Calculator (PWC) (U.S. EPA, 2020b) model to estimate groundwater concentrations. PWC simulates chemical substance applications to land surfaces and the chemical substance's subsequent transport to and fate in water bodies, including surface water bodies as well as simple ground water aquifers. Scenarios with six sandy soils containing a relatively low fraction of organic carbon and shallow groundwater were modeled. The loading of 1,1-dichloroethane to the soil surface was estimated by taking the 95th

percentile air deposition rate at 1,000 m from the emission source for the largest OES emission (Processing as a reactive intermediate) and estimating the mass deposited on soil per hectare. From this loading the model estimated post breakthrough average groundwater concentrations ranging from approximately 2.7 to 8.0 µg/L, suggesting that the air deposition to groundwater pathway is not an important source of general population exposure to 1,1-dichloroethane. No additional analysis of the air deposition to groundwater pathway was conducted.

G.1.2 Measured Concentrations in Groundwater

G.1.2.1 Ambient Groundwater Monitoring

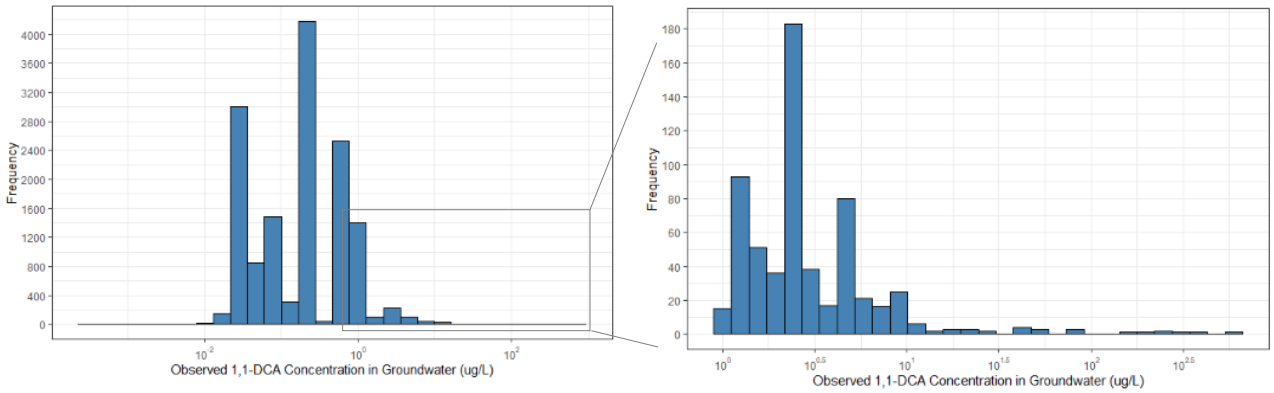
Concentrations of 1,1-dichloroethane measured from groundwater monitoring wells are collated by the National Water Quality Monitoring Council and stored in the WQP (NWQMC, 2022). Groundwater 1,1-dichloroethane concentration results were acquired between 2015 to 2020 from the WQP. Figure_Apx G-1 shows the spatial distribution of measured concentrations of 1,1-dichloroethane in groundwater across the contiguous United States. Groundwater was measured at a much higher frequency in Oregon, Georgia, Minnesota, New York, and New Jersey in comparison to the remaining states. The distribution of the groundwater sample concentrations is shown in Figure_Apx G-2. The process for identifying this data is provided in Appendix F. This analysis is intended to characterize the observed ranges of 1,1-dichloroethane concentrations in groundwater irrespective of the reasons for sample collection and to provide context for the modeled groundwater concentrations presented in Section G.1.2.3.



Figure_Apx G-1. Locations of 1,1-Dichloroethane Measured in Groundwater Monitoring Wells Acquired from the WQP, 2015–2020

AIANNH tribal boundaries are shaded in gray.

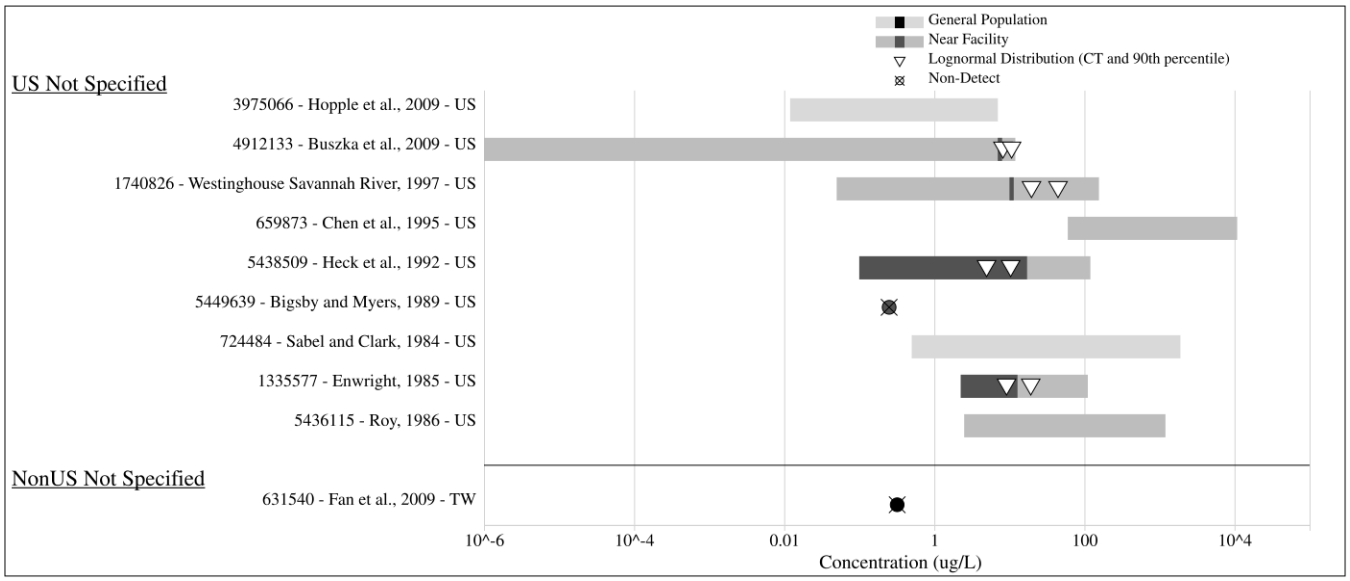
Note: Alaska, American Samoa, Guam, Hawaii, N. Mariana Islands, Puerto Rico, and the U.S. Virgin Islands are not shown because they do not contain groundwater monitoring data within the WQP.



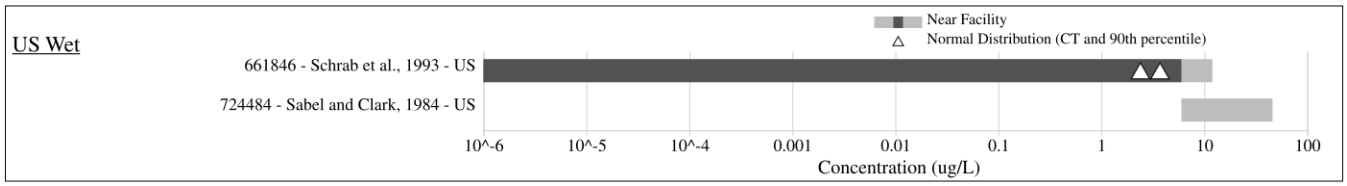
Figure_Apx G-2. Distribution of 1,1-Dichloroethane Concentrations from Groundwater Monitoring Wells (N = 14,483) Acquired from the Water Quality Portal, 2015–2020

Concentrations of 1,1-dichloroethane in groundwater ranged from 0 to 650 µg/L for samples collected between 2015 and 2020. The 50th and 95th percentile of groundwater concentrations of 1,1-dichloroethane was 0.25 and 1 µg/L. There were 602 groundwater samples with concentrations of 1,1-dichloroethane that exceeded 1 µg/L (Figure_Apx G-2, right inset). For this subset of results greater than 1 µg/L, the 50th and 95th percentile was 2.5 and 12 µg/L, respectively. There were 33 (≈0.3% of the total) groundwater monitoring wells that exceeded 1,1-dichloroethane concentrations of 10 µg/L for samples collected between 2015 and 2020.

A small amount of groundwater and soil-water leachate 1,1-dichloroethane concentration data was collected through EPA's systematic review of published literature. A summary of the individual studies is shown in Figure_Apx G-3 for groundwater data and Figure_Apx G-4 for leachate data. A review of published literature resulted in nine studies reporting measured concentrations of 1,1-dichloroethane in groundwater. Concentrations ranged from not detected to 1,900,000 ng/L (Sabel and Clark, 1984) in 400 samples collected between 1984 and 2005 in the United States.



Figure_Apx G-3. Concentrations of 1,1-Dichloroethane (µg/L) in Groundwater from U.S.-Based and International Studies, 1984–2005



Figure_Apx G-4. Concentrations of 1,1-Dichloroethane (µg/L) in the Soil-Water Leachate from U.S.-Based Studies for Locations near Facility Releases, 1984–1993

G.1.2.2 Measured Concentrations in Groundwater Sourced Drinking Water

The UCMR3 dataset was used to gather concentrations of 1,1-dichloroethane found in finished drinking water from PWSs that draw primarily from groundwater sources. Of 2,046 systems considered in the groundwater sampling effort between 2013 and 2015, only 137 detected 1,1-dichloroethane above the 0.03 µg/L MRL in drinking water. The maximum concentration of 1,1-dichloroethane measured in groundwater sourced finished drinking water was 8.1 µg/L. Similar for surface water derived sources, these results indicate that 1,1-dichloroethane in finished drinking water derived from groundwater was measured in relatively low amounts across the nation between 2013 to 2015 (see Tables 5-15 and 5-17 in the EPA Regulatory Determination Support Document (U.S. EPA, 2021b).

G.1.2.3 Modeled Concentrations in Groundwater

EPA found reported releases of 1,1-dichloroethane to land (TRI 2015–2020, average 1 kg/year) and used Generic Scenarios or Emission Scenario Documents to model releases of less than 22,682 kg/year to Hazardous Waste Landfills under the TSCA COUs. The groundwater concentrations resulting from the range of expected releases, making the conservative assumption that the releases go to non-hazardous waste landfills, are predicted to be less than 9.17×10^{-4} mg/L (Table_Apx G-2).

Table_Apx G-2. Estimated Groundwater Concentrations (mg/L) of 1,1-Dichloroethane Found in Wells Within 1 Mile of a Disposal Facility Determined by the DRAS Model

Leachate Concentration (mg/L)	Loading Rate				
	0.1 kg/year	1.0 kg/year	10 kg/year	100 kg/year	1,000 kg/year
1.0 E−05	1.11E−14	1.06E−13	1.01E−12	9.62E−12	9.17E−11
1.0 E−04	1.11E−13	1.06E−12	1.01E−11	9.62E−11	9.17E−10
1.0 E−03	1.11E−12	1.06E−11	1.01E−10	9.62E−10	9.17E−09
1.0 E−02	1.11E−11	1.06E−10	1.01E−09	9.62E−09	9.17E−08
1.0 E−01	1.11E−10	1.06E−09	1.01E−08	9.62E−08	9.17E−07
1.0	1.11E−09	1.06E−08	1.01E−07	9.62E−07	9.17E−06
10	1.11E−08	1.06E−07	1.01E−06	9.62E−06	9.17E−05
100	1.11E−07	1.06E−06	1.01E−05	9.62E−05	9.17E−04
Concentrations organized by potential loading rates (kg) and potential leachate concentrations (mg /L).					

Disposal to Landfills and Method to Model Groundwater Concentrations

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., 2015a; Li et al., 2013; Mohr and DiGuiseppi, 2010). This contamination may be attributed to perforations in the liners, failure of the leachate capturing system, or improper management of the landfills. 1,1-Dichloroethane can migrate away from landfills in leachate to groundwater. If communities rely on this

groundwater as their primary drinking water source, there is a potential for exposure via ingestion if that water is contaminated with 1,1-dichloroethane 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 RCRA: 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 1,1-dichloroethane, meeting the U076 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 TRI and are reported for both onsite and offsite facilities. 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. 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 beyond 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 Municipal Solid Waste (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 2 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 can still occur due to imperfections introduced during construction or that form over time (Li et al., 2015a; 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 no known potential sources of 1,1-dichloroethane to Subtitle D landfills. Waste generators that produce less than 100 kg per month of non-acutely hazardous waste, including 1,1-dichloroethane meeting the U076 waste code, may dispose of this waste in these landfills. Nonhazardous industrial

wastes also have the potential to contain 1,1-dichloroethane at variable concentrations, but due to its limited use as a laboratory chemical, concentrations in waste are expected to be low. EPA did not identify any consumer or commercial products that contain 1,1-dichloroethane; therefore, release of 1,1-dichloroethane to Subtitle D nonhazardous waste landfills as part of municipal solid waste is expected to be negligible. In addition, landfilled 1,1-dichloroethane will only reach groundwater from landfills that do not have an adequate liner and leachate control systems. Based on the previous information, EPA concludes the potential for exposure to general populations to 1,1-dichloroethane via ingestion of leachate contaminated groundwater is negligible. To support this conclusion, an assessment was conducted to evaluate the potential for groundwater contamination by 1,1-dichloroethane in leachate in the absence of landfill controls.

This assessment was completed using the Hazardous Waste DRAS (U.S. EPA, 2020b). DRAS was specifically designed to address the Criteria for Listing Hazardous Waste identified in Title 40 CFR 261.11(a)(3), a requirement for evaluating proposed hazardous waste delistings. In this assessment, DRAS was used to determine potential groundwater concentrations of 1,1-dichloroethane after disposal into a non-hazardous waste landfill. The results of this assessment are provided in Table_Apx G-2. Because measured loading rates of 1,1-dichloroethane to individual landfills are unknown, multiple DRAS runs were conducted, which included the estimated ranges of waste loading per site (see Section 3.2.1.3 for loading estimates). The assessment relied on the default values for 1,1-dichloroethane 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 calculation was used to convert the DAF to a potential concentration that receptors located within 1 mile of a landfill might be exposed if the release was not controlled.

Summary of Disposal to Landfills and Groundwater Concentrations

EPA determined through modeling that groundwater concentration of 1,1-dichloroethane 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 substance with 1,1-dichloroethane's physical and chemical properties (water solubility, Henry's Law constant) and fate characteristics (biodegradability, half-life in groundwater). 1,1-Dichloroethane migrates in groundwater at approximately the rate of hydraulic flow and can persist with a half-life of greater than 150 days in anaerobic environments (Adamson et al., 2014; Mohr and DiGuiseppi, 2010). Thus, these concentrations are likely to represent the range of exposure concentrations for individuals living within a 1-mile radius of a poorly managed landfill who rely on groundwater as their primary source of drinking water.

EPA also determined that the modeled concentrations are within the range of concentrations of 1,1-dichloroethane found in groundwater monitoring studies. Monitoring data from the WQP dataset reported 1,1-dichloroethane concentrations in groundwater ranging from near detection limit to 650 µg/L. Although the corresponding sites in these monitoring surveys may not be specifically tied to the disposal of 1,1-dichloroethane to landfills, they provide context for what concentrations may be expected when contamination occurs. These concentrations support the conclusion that the low concentrations modeled by EPA are common in groundwater aquifers nationwide.

G.1.2.4 Measured Concentrations in Biosolids and Sludge

Biosolids are a primarily organic solid product produced by wastewater treatment processes that can be beneficially recycled via land application. The EPA published The Standards for the Use or Disposal of Sewage Sludge (40 CFR, Part 503) in 1993 to protect public health and the environment from any reasonably anticipated adverse effects of certain pollutants that might be present in sewage sludge biosolids. Municipal wastewater treatment systems mainly treat biosolids to ensure pathogen and vector attraction (*e.g.*, rats) reduction and limits in metals concentrations; however, other chemicals are monitored as well.

Data regarding 1,1-dichloroethane measured concentrations in biosolids has not been identified in public databases or published literature particularly for those facilities that treat wastes and report discharges of 1,1-dichloroeethane. EPA did refer to the 1988 Sewage Sludge Survey and found 0 percent detection frequency for 1,1-dichloroethane (see Appendix C.2.4.4). In addition, EPA identified a 2004 published report by the King County Department of Natural Resources and Parks (King County DNRP), Wastewater Treatment Division characterizing two municipal wastewater treatment facilities that monitored biosolids for 135 chemicals, including 1,1-dichloroethane (King County DNRP, 2004). In reviewing the 2004 report, EPA concluded that 1,1-dichloroethane was not detected in these biosolids. In subsequent annual reports, King County DNRP did not list 1,1-dichloroethane levels in biosolids and noted in those reports that 1,1-dichloroethane is not detected in biosolids. However, data on the 125 POTWs (see Table 3-4) reporting releases of 1,1-dichloroethane and that generate biosolids that are either disposed or used for land application, are not available.

G.1.2.5 Modeled Concentrations in Groundwater Resulting from Land Application of Biosolids

Although there is no literature data of 1,1-dichloroethane in biosolids, EPA estimated 1,1-dichloroethane in biosolids because 125 POTWs treat and release 1,1-dichloroethane to surface water and generate biosolids in the process.

The Biosolids Tool (BST) (U.S. EPA, 2023a) was used to assess the importance of the biosolids land application to groundwater pathway. The BST is a multimedia, multipathway, multireceptor deterministic, problem formulation, and screening level model that can estimate high-end human and ecological hazards based on potential exposures associated with land application of biosolids or placement of biosolids in a surface disposal unit. The BST was peer reviewed by the EPA Science Advisory Board in 2023 (EPA-SAB-24-001). A default annual biosolids land application rate of 1 kg/m^2/year and a 1,1-dichloroethane biosolids concentration of 20 mg/kg, estimated using the SimpleTreat 4.0 wastewater treatment plant model, were used as input to the BST. The model predicted groundwater concentrations of 3.2 µg/L suggesting the biosolids land application containing 1,1-dichloroethane with migration to groundwater is not an important source of general population exposure. However, soil and pore water exposures to 1,1-dichloroethane from biosolids land application could occur to ecological species and is presented in the subsequent sections below.

G.1.2.6 Modeled Concentrations in Wastewater Treatment Plant Sludge

Chemical substances in wastewater undergoing biological wastewater treatment may be removed from the wastewater by processes including biodegradation, sorption to wastewater solids, and volatilization. As discussed in Appendix C.2.5.2, 1,1-dichloroethane is expected to be removed in wastewater treatment primarily by volatilization with little removal by biodegradation or sorption to solids. Chemicals removed by sorption to sewage sludge may enter the environment when sewage sludge is land-applied following treatment to meet standards. The treated solids are known as biosolids. The removal of a nonbiodegradable neutral organic chemical present in WWTP influent via sorption to

sludge is evaluated by considering its partitioning to sludge organic carbon.

Based on its K_{OC} value of 31, 1,1-dichloroethane is not expected to significantly partition to sewage sludge. Releases of 1,1-dichloroethane to wastewater treatment are expected to be low and disperse across many sites; therefore, land application of biosolids containing 1,1-dichloroethane is not expected to be a significant exposure pathway. To support this conclusion, range-finding estimates were prepared to evaluate the concentrations of 1,1-dichloroethane in biosolids, in soil receiving biosolids, and soil pore water concentrations resulting from biosolids application. Releases from wastewater treatment plants with DMRs for 1,1-dichloroethane were reviewed to identify those plants discharging the highest amount of 1,1-dichloroethane annually. The two highest releasing facilities were not chosen due to errors or uncertainties in their release estimates. The site with the third largest estimated releases of 1,1-dichloroethane to water was chosen and it was assumed that all biosolids generated at that facility were land-applied over a year at a single site. The releases from the facility were used to back-calculate input to the SimpleTreat 4.0 wastewater treatment plant model to estimate the concentration of 1,1-dichloroethane in biosolids. EPA assumed that the modeled site used activated sludge wastewater treatment and that SimpleTreat 4.0 defaults were a reasonable representation of the activated sludge treatment at the site. Using this loading data, the model predicted 1,1-dichloroethane concentration in combined sludge of 20 mg/kg. Details on the procedure are provided in Appendix C.2.4.4.

Modeled Concentrations of 1,1-Dichloroethane in Soil Receiving Biosolids

No information on the concentration of 1,1-dichloroethane in soil receiving biosolids was found. To assess soil concentrations resulting from biosolid applications, EPA relied upon modeling work conducted in Canada (EC/HC, 2011), which used Equation 60 of the European Commission TGD (ECB, 2003). The concentration in sludge was set to 20 mg/kg dry weight based on the combined sludge concentration estimated by SimpleTreat 4.0. Using these assumptions, the estimated 1,1-dichloroethane soil concentrations after the first year of biosolids application were 29.4 ug/kg in tilled agricultural soil and 58.8 µg/kg in pastureland. See Section G.1.2.5 for discussion of the estimation of biosolids concentrations.

The method assumes complete mixing of the chemical in the volume of soil it is applied to as well as no losses from transformation, degradation, volatilization, erosion, or leaching to lower soil layers. Additionally, it is assumed there is no input of 1,1-dichloroethane from atmospheric deposition and there are no background 1,1-dichloroethane accumulations in the soil.

Modeled Concentrations of 1,1-Dichloroethane in Soil Pore Water Receiving Biosolids

To estimate soil pore water concentrations for 1,1-dichloroethane in soil receiving biosolids for ecological species' exposures, EPA used a modified version of the equilibrium partitioning (EqP) equation developed for weakly adsorbing chemicals such as 1,1-dichloroethane and other VOCs. The modified equation accounts for the contribution of dissolved chemical to the total chemical concentration in soil or sediment (Fuchsman, 2002). The equation assumes that the adsorption of chemical to the mineral components of sediment particles is negligible.

Using Equation_Apx C-1 and estimating $C_{dissolved}$ from the K_{OC} for 1,1-dichloroethane assuming a soil organic carbon fraction (f_{OC}) of 0.02, and a soil solids fraction of 0.5, the estimated pore water concentrations are 18.2 µg/L in tilled agricultural soil and 36.6 µg/L in pastureland (Table_Apx G-3).

Table_Apx G-3. Soil and Soil Pore Water Concentrations Estimated from Annual Application of Biosolids

Exposure Scenario	Combined Sludge Concentration (µg/kg)	Soil Type	Soil Concentration (µg/kg)	Soil Pore Water Concentration (mg/L)
Waste Handling, Treatment and Disposal (POTW)	20,000	Tilled agricultural	29.2	18.2
		Pastureland	58.8	36.6
[a] Modeled using SimpleTreat 4.0 wastewater treatment plant model.				

Appendix H DRINKING WATER EXPOSURE ESTIMATES

Levels of acute and chronic exposure from the consumption of 1,1-dichloroethane in drinking water were estimated using the surface water concentrations estimated in Sections 3.3.3.2.2 and groundwater concentrations estimated in Appendix G.1.2.3. Additional information on these drinking source-waters are described in Sections H.1 and H.2 below.

Acute and chronic drinking water exposures used to evaluate non-cancer risks were estimated as an ADR or ADD, respectively. Lifetime exposures used to evaluate cancer risks were estimated as a LADD. The following equations were used to calculate each of these exposure values:

Equation_Apx H-1.

$$ADR = (SWC \times (1 - DWT/100) \times IR_{dw} \times RD \times CF1)/(BW \times AT)$$

Equation_Apx H-2.

$$ADD = (SWC \times (1 - DWT/100) \times IR_{dw} \times ED \times RD \times CF1)/(BW \times AT \times CF2)$$

Equation_Apx H-3.

$$LADD = (SWC \times (1 - DWT/100) \times IR_{dw} \times ED \times RD \times CF1)/(BW \times AT \times CF2)$$

Where:

SWC	=	Surface water concentration (ppb or µg/L)
DWT	=	Removal during drinking water treatment (%)
IR_{dw}	=	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)

The same inputs for body weight, averaging time (AT), and exposure duration were applied across the evaluations of drinking water, incidental oral exposure, and incidental dermal exposure. For all calculations, mean body weight data were derived from Chapter 8, Table 8-1 in EPA's *Exposure Factors Handbook* (EFH) (U.S. EPA, 2011a). To align with the age groups of interest, weight averages were calculated for the infant age group (birth to <1 year) and toddlers (1–5 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 1,1-dichloroethane release are assumed to be 1 according to the methodology used in E-FAST 2014 (U.S. EPA, 2014). Thus, exposure levels are derived from aqueous concentration estimates that assume the entire annual load of 1,1-dichloroethane is released from the facility at single time. 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, 2011a). 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 levels were estimated for the following age groups: adult (21+ years), youth (16–20 years), youth (10–15 years), child (6–10 years), toddler (1–5 years), and infant (birth to <1 year). Drinking water intake rates are provided in the 2019 update of Chapter 3 of the EFH (U.S. EPA, 2019a). Weighted averages were calculated for acute and chronic drinking water intakes for adults 21 years or older and toddlers aged 1 to 5 years. From Table 3-17 in the EFH, 95th percentile consumer data were used for acute drinking water intake rates. From Table 3-9 in the EFH, mean per capita data were used for chronic drinking water intake rates.

H.1 Surface Water Sources of Drinking Water

Exposure levels resulting from the contamination of 1,1-dichloroethane in drinking water sourced from surface waters was estimated from aqueous concentrations generated at individual PWS intake locations as described in Section E.2.3. It is important to note that aqueous concentrations of 1,1-dichloroethane were not estimated in still water bodies, such as lakes, ponds, or reservoirs, even if PWS draws from these surface water bodies. Rather, in these cases, modeled EROM stream flow values or the facility effluent plant flow (*e.g.*, when upstream flow exceeds downstream flow) served as the basis for estimate aqueous concentrations at the PWS intake location. Given the difficulty of determining lake volume for many sites and the uncertainty around applying generic dilution factors was avoided.

The aqueous concentrations derived from a modeled 30Q5 stream flow, or from the facility effluent flow, were used to estimate an ADR or acute exposure level. The aqueous concentrations derived from the modeled harmonic mean stream flow, or from the facility effluent flow, were used to estimate an ADD, LADD, and LADC or chronic exposure levels. Prior to estimating exposure levels, information on the treatment processes for each PWS was obtained from SDWIS. For PWSs that treat raw source water using packed tower aeration, aqueous concentration estimates at those drinking water intakes were adjusted to account for 80 percent drinking water treatment removal. For all other sites and their corresponding treatment processes, drinking water treatment removal was set to 0 percent to represent a conservative estimate of possible drinking water exposures.

It is important to note that 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, 2006). In assessing drinking water exposures, the ability to treat and remove or transform chemicals in possible drinking water supplies should be considered. Because of the wide range of treatment processes that inconsistently remove 1,1-dichloroethane from ambient surface water and groundwater prior to possible general population consumption as drinking water, EPA assumes zero removal except for PWSs that utilize packed tower aeration processes to provide a conservative estimate of general population drinking water exposures (further details are described in Section C.2.3.1).

H.2 Groundwater Sources of Drinking Water

Exposure levels resulting from the contamination of 1,1-dichloroethane in drinking water sourced from groundwater was estimated from aqueous concentrations generated from the DRAS model as described in Section G.1.1.

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. Drinking water treatment removal (DWT) was set to 0 percent for groundwater under the assumption that home wells are unlikely to remove 1,1-dichloroethane.

H.3 Removal Through Drinking Water Treatment

Removal of 1,1-dichloroethane in drinking water treatment is expected to be primarily due to its volatility and potential to be adsorbed to activated carbon where activated carbon treatment is in place. The effectiveness of treatment such as air stripping for the removal of volatile chemicals can be predicted by physical and chemical properties such as the Henry's Law constant (HLC). Removal of chemicals in granular activated carbon (GAC) treatment systems are more difficult to predict from physical and chemical properties, but information on the adsorption capacity of GAC for chemicals helps inform the effectiveness and feasibility of GAC treatment for the removal of the chemical from water.

1,1-Dichloroethane can be removed by GAC (U.S. EPA, 2021a). To achieve high removal, a GAC system would have to incorporate design and operating parameters that account for the 1,1-dichloroethane sorptive capacity of GAC. In conclusion, a GAC treatment system could be designed and operated to achieve high removal of 1,1-dichloroethane, but without performance data there is high uncertainty estimating its treatment efficiency.

Appendix I GENERAL POPULATION ORAL EXPOSURE ESTIMATES FROM SOIL

I.1 Incidental Oral Ingestion from Soils (Biosolids)

No current information on the concentration of 1,1-dichloroethane in wastewater treatment sludge or biosolids was found. In the absence of measured data, EPA estimated the maximum amount of 1,1-dichloroethane entering wastewater treatment from the releases reported for any facility in its DMR. The releases were converted to daily loading rates and used as input to the SimpleTreat 4.0 wastewater treatment plant model (RIVM 2014; accessed June 11, 2025). It was assumed that the modeled site used activated sludge wastewater treatment and that SimpleTreat 4.0 defaults were a reasonable representation of the activated sludge treatment at the site. Using this loading data, the model predicted 1,1-dichloroethane concentration in combined sludge of 20 mg/kg.

To assess soil concentrations resulting from biosolid applications, EPA relied upon modeling work conducted in Canada (EC/HC 2011), which used Equation 60 of the *European Commission Technical Guidance Document* (TGD) (ECB 2003). The equation in the TGD is provided in Equation_Apx I-1 below:

Equation_Apx I-1.

$$PEC_{soil} = (C_{sludge} \times AR_{sludge})/(D_{soil} \times BD_{soil})$$

Where:

PEC_{soil}	=	Predicted environmental concentration (PEC) for soil (mg/kg)
C_{sludge}	=	Concentration in sludge (mg/kg)
AR_{sludge}	=	Application rate to sludge amended soils (kg/m^2/year); default = 0.5 from Table A-11 of TGD
D_{soil}	=	Depth of soil tillage (m); default = 0.2 m in agricultural soil and 0.1 m in pastureland from Table A-11 of TGD
BD_{soil}	=	Bulk density of soil (kg/m^3); default = 1,700 kg/m^3 from Section 2.3.4 of TGD

Using Equation_Apx I-2 above, the concentration of 1,1-dichloroethane in pastureland soil receiving an annual application of biosolids was estimated to be 58.8 µg/kg. See Section G.1.2.3 for details on the estimation of 1,1-dichloroethane biosolids concentrations.

ADDs for children ingesting soil receiving biosolids were calculated for 1,1-dichloroethane using Equation_Apx I-2 below.

Equation_Apx I-2.

$$ADD = (C \times IR \times EF \times ED \times CF)/(BW \times AT)$$

Where:

ADD	=	Average Daily Dose (mg/kg/d)
C	=	Soil concentration (mg/kg)
IR	=	Intake rate of contaminated soil (mg/d)
EF	=	Exposure frequency (d)
CF	=	Conversion factor (1.0×10^{-6} kg/mg)
BW	=	Body weight (kg)
AT	=	Averaging time (non-cancer: ED × EF, cancer: 78 years × EF)

The recommended intake rate for children aged 3 to 6 years for soil pica (soil ingestion) is 1,000 mg/d. (U.S. EPA, 2017c). Mean body weight (18.6 kg) for 3- to 6-year-olds was taken from EPA's *Exposure Factors Handbook* (U.S. EPA, 2011a).

Table_Apx I-1. Modeled Exposure to 1,1-Dichloroethane in Land-Applied Biosolids for Children

OES	Average Daily Dose (mg/kg-day)
Disposal	3.16E−06

Thus, at the estimated 1,1-dichloroethane soil concentration of 58.8 ug/kg, the ADD for a 3- to 6-year-old child ingesting 1,000 mg/day of contaminated soil would be 3.16×10^{-6} mg/kg/day (Table_Apx I-1).

An alternate approach to estimating the concentration of 1,1-dichloroethane in soil from land-applied biosolids and subsequent childrens exposure employed the use of the BST (U.S. EPA, 2023a). The BST is a peer reviewed, multimedia, multipathway, multireceptor deterministic, problem formulation, and screening level model that can estimate high-end human and ecological hazards based on potential exposures associated with land application of biosolids or placement of biosolids in a surface disposal unit. A default annual biosolids land application rate of 1 kg/m^2/year and a 1,1-dichloroethane biosolids concentration of 20 mg/kg, estimated using the SimpleTreat 4.0 wastewater treatment plant model, were used as input to the BST. The model predicted a maximum soil concentration of approximately 1.6 µg/kg corresponding to an average daily dose of 8.6×10^{-8} mg/kg-day using the described assumptions above. Because this acute dose estimate of 1,1-dichloroethane exposure is very low compared to oral hazard values, acute and chronic risk of oral exposures from ingestion of soil were not expected and were not estimated.

I.2 Incidental Oral Ingestion from Soils (Air Deposition)

No information on the concentration of or exposure to 1,1-dichloroethane in soil from air deposition was found. Estimates of 1,1-dichloroethane air deposition to soil are discussed in detail in Section G.1.1. The deposition rates and soil concentrations of 1,1-dichloroethane were calculated with Equation_Apx I-3 and Equation_Apx I-4 below.

Equation_Apx I-3.

$$Ann_{Dep} = Tot_{Dep} \times Ar \times CF$$

Where:

Ann_{Dep}	=	Total annual deposition to soil (µg)
Tot_{Dep}	=	Annual deposition flux to soil (g/m^2)
Ar	=	Area of soil (m^2)
CF	=	Conversion of g to µg

Equation_Apx I-4.

$$Soil_{Conc} = Ann_{Dep}/(Ar \times Mix \times Dens)$$

Where:

$SoilConc$	=	Annual-average concentration in soil (µg/kg)
$AnnDep$	=	Total annual deposition to soil (µg)
Mix	=	Mixing depth (m); default = 0.1 m from the European Commission

TGD (ECB, 2003)

Ar = Area of soil (m^2)

$Dens$ = Density of soil; default = 1,700 kg/m^3 from the European Commission TGD (ECB, 2003)

The above equations assume instantaneous mixing with no degradation or other means of chemical reduction in soil over time and that 1,1-dichloroethane loading in soil is only from direct air-to-surface deposition (*i.e.*, no runoff).

Section G.1.1 presents the range of calculated soil concentrations corresponding to the emission scenarios considered. From Equation_Apx I-4, the highest estimated 95th percentile soil concentration amongst all exposure scenarios was for the processing as a reactant (OES) scenario:

- 4.91×10^3 µg/kg at "fenceline" populations (100 m from the source); and
- 6.29×10^1 µg/kg at "community" populations (1,000 m from the source).

ADDs were calculated for air deposited 1,1-dichloroethane ingestion via soil using Equation_Apx I-5:

Equation_Apx I-5.

$$ADD = (C \times IR \times EF \times ED \times CF)/(BW \times AT)$$

Where:

ADD = Average daily dose (mg/kg/d)

C = Soil concentration (mg/kg)

IR = Intake rate of contaminated soil (mg/d)

EF = Exposure frequency (d)

CF = Conversion factor (10×10^{-6} kg/mg)

BW = Body weight (kg)

AT = Averaging time (non-cancer: ED × EF, cancer: 78 years × EF)

Modeled soil concentrations were calculated from 95th percentile air deposition (Section G.1.1) concentrations for 100 and 1,000 m from a facility. These calculations were conducted for the Processing as a reactant OES (Table_Apx I-2).

The recommended intake rate for children aged 3 to 6 years for soil pica is 1,000 mg/d (U.S. EPA, 2017c). Mean body weight (18.6 kg) for 3- to 6-year-olds was taken from the EHF (U.S. EPA, 2011a).

Table_Apx I-2. Modeled Soil Ingestion Doses for the Processing as a Reactant OES, for Children

OES	Distance (m)	95th Percentile Soil Concentration (µg/kg)	Average Daily Dose (mg/kg-day)
Processing as a reactant	100	4.91E3	2.64E−04
	1,000	6.29E1	3.72E−06

Because this average daily dose estimate of 1,1-dichloroethane exposure is very low compared to oral hazard values, acute and chronic risk of oral exposures from ingestion of soil were not expected and were not estimated.

Appendix J ECOLOGICAL EXPOSURE ESTIMATES

Estimated aqueous concentrations at the facility release sites were compared to their respective acute and chronic concentration of concern (COC). Initial surface water (water column) concentrations were estimated by dividing the annual load for a given facility by the number of ecological exposure days that correspond to the acute or chronic scenario for the water column and benthic pore water. Details on how the COCs for aquatic ecological species were determined can be found in Section 4. Concentrations that exceeded their respective acute and chronic water column and benthic pore water COCs were kept for a second modeling step using the Point Source Calculator (PSC).

J.1 Point Source Calculator

J.1.1 Description of Point Source Calculator

The PSC is a tool designed to estimate acute and chronic concentrations of chemicals directly released to surface water bodies. It is a proposed potential refinement to E-FAST for estimating exposures from wastewater discharges to surface waters. In addition to calculating aqueous concentrations (in the water column) based on the chemical loading release rate and receiving water body streamflow as E-FAST does, the PSC accounts for several key physicochemical processes that can affect levels of a released chemical during transport. More specifically, the PSC allows for chemical removal through sorption to sorption to sediment, volatilization, and transformation processes (*i.e.*, aerobic and anaerobic metabolism, hydrolysis, and photolysis), thus providing a higher tiered model that produces a potentially less conservative estimates of concentration and exposure compared to E-FAST. In addition, the PSC provides estimates of the chemical concentration in the benthic pore water and bulk sediment of a receiving water body. Because of these additional processes, PSC requires a number of chemical-specific input parameters, including chemical partitioning (sediment, air, water) and degradation rates. PSC also requires specific release site parameters, such as waterbody dimensions, baseflow, and meteorological data as well as a group of water column and benthic porewater/sediment biogeochemical parameters. A description of the PSC input parameters can be found in Section 4 of the *Point Source Calculator: A Model for Estimating Chemical Concentration in Water Bodies* document (U.S. EPA, 2019c).

The PSC is particularly useful for estimating benthic pore water concentrations for assessing benthic organism exposures, but was designed for use on a site-specific basis, thus requiring a number of assumptions about release site parameters before applying to national-scale exposure assessments. Because the PSC has more input parameters and requires default assumptions for national-scale assessments, EPA's Office of Pesticides Program (OPP) performed a thorough sensitivity analysis to identify a standard set of assumptions for PSC runs that can be applied nationally. This sensitivity analysis informed our use of the PSC Model and choice of input parameters, which are detailed below. Of the additional parameters considered to effect chemical concentration in the water column—benthic porewater and benthic bulk sediment—the most are the user's selection of the meteorological file, water body dimensions, and water body baseflow. Although the baseflow should be included for each individual site, without sufficient information on the meteorology or receiving water body dimensions, it is recommended to use the following standard input parameters: the 90th percentile meteorological file (*i.e.*, w24027) and water body dimensions of 5 m × 1 m × 40 m (width × depth × length).

J.1.2 Point Source Calculator Input Parameters

Table_Apx J-1 to Table_Apx J-4 include the standard set of input parameters used with the PSC, excluding the mass release and constant flow rate parameters, which changed for each site and scenario (acute or chronic). A new list of facility release sites were created from those releases that resulted in an

estimated aqueous (water column) concentration of 1,1-dichloroethane exceeding a water column and benthic pore water acute COC (1,769 and 1,769 µg/L, respectively) or water column and benthic pore water chronic COC (93 and 6,800 µg/L, respectively). For either scenario, the constant flow rate remained the same. Here the estimated 7Q10 flow value created in Section E.2 was used. For those facility release sites with estimated concentrations exceeding the respective acute COC, the mass release parameter equaled the annual load, thus reflecting a 1-day maximum release scenario. For those facility release sites with estimated concentrations exceeding the respective chronic COC, the mass release parameters equaled the annual load divided by 21 (water column chronic) or 15 (benthic pore water chronic), thus reflecting a 21- or 15-day release schedule where the annual load was released in equal amounts over 21 or 15 consecutive days. The default Water Column and Benthic compartment PSC input parameters were used as well as the default Mass Transfer Coefficient.

The respective water column and benthic acute and chronic COCs were used for each of the water column and benthic pore water toxicity options. For example, for the chronic water column scenario, a user defined "21-Day Avg" scenario was included. For those sites that exceeded the benthic pore water chronic COC with initial (water column) concentrations, they were then modeled with PSC to estimate their benthic chronic sediment concentration and compared to the respective COC (2,900 µg/L). It is important to note that initial estimates of aqueous concentration in the water column were used to create a new list of facilities to model in PSC for benthic water pore and sediment concentrations. Thus, it is assumed that if an initial water column concentration did not exceed the benthic pore water COC than it would not exceed the benthic pore water COC post-PSC modeling. This is expected to be the case for 1,1-dichloroethane because benthic pore water concentrations are not expected to exceed the water column concentrations from which they were derived using the PSC Model.

Table_Apx J-1. 1,1-Dichloroethane Chemical-Specific PSC Input Parameters

Physiochemical PSC Input Parameters	
Sorption coefficient K_{OC} (ml/g)	30.20
Water column half-life (days)	365 at 25 ˚C
Photolysis half-life (days)	365 at 0 °Lat.
Hydrolysis half-life (days)	365 at 25 °C
Benthic half-life (days)	365 at 25 °C
Volatilization (yes/no)	Yes – Use Henry's Law constant
Molecular weight	98.95
Henry's Law constant (atm m3/mol)	0.00562
Heat of Henry (J/mol)	0
Reference Temperature (° C)	24

Table_Apx J-2. 1,1-Dichloroethane PSC Mass Release Schedule for an Acute Exposure Scenario

Mass Release Schedule	
Offset (# of lead days before release begins)	0
Days on (# of consecutive release days)	1
Days off (# of consecutive days without release)	364
Mass release (kg/day)	Site annual load

Table_Apx J-3. 1,1-Dichloroethane PSC Mass Release Schedule for a Chronic Exposure Scenario

Mass Release Schedule	
Offset (# of lead days before release begins)	0
Days on (# of consecutive release days)	21, 15, or 35
Days off (# of consecutive days without release)	344, 350, or 330
Mass release (kg/day)	Site annual load ÷ # of days off

Table_Apx J-4. Meteorologic and Hydrologic PSC Input Parameters

Meteorologic and Hydrologic Input Parameters	
Meteorologic data file	w24027
Water body dimensions (width × depth × length)	5 m × 1 m × 40 m
Constant flow rate (m^3/day)	Site 7Q10 flow

J.1.3 Water Column, Pore Water, and Benthic Sediment Results

The PSC estimates daily concentrations of the chemical in the water column, benthic pore water, and bulk benthic sediment for a given year, and repeats the simulation for 30 consecutive years. The main Results tab of the PSC software includes a time series graph of these daily simulations repeated for 30 years. The Results tab also provides concentration estimates on a daily sliding average (*i.e.*, "1-Day Avg", "7-Day Avg", "28-Day Avg"). These averages reflect the maximum of the entire times series for the period of days indicated, meaning a "1-Day Avg" is the maximum estimated daily concentration for the entire time series and a "21-Day Avg" is the maximum average of 21 consecutive daily estimated concentrations. However, these average metrics do not necessarily correspond to the first group of that might be indicates by the metric. For example, the "35-Day Average" may not include the first 35 days of each year's simulation. Concentration results for the water column (µg/L), benthic pore water (µg/L), and total benthic sediment (µg/kg) were retrieved from either the "1-Day Avg", "21-Day Avg", "15-Day Avg", or "35-Day Avg" to coincide with the acute and chronic release toxicity scenarios.

The PSC also estimates the number of days that the chemical concentration exceeds a user-defined concentration of concern for each of the water column, pore water, and benthic bulk sediment compartments. Because a sediment toxicity COC was not applied, this data was not included. The days of exceedance was estimated by multiplying the "1-Day Avg" "Days > COC" fraction by 10,957 (the total number of days in the time series) and then divided by 30 (the total number of years in the simulation). This metric aligns with the daily concentration output file. Note, through this approach the user's mass release schedule bounds the days of exceedance metric in the water column primarily because of washout (*i.e.*, replacement of "clean water" from downstream water transport) that occurs immediately following the last day of chemical mass release in the model. The days of exceedance metric should be interpreting with caution for this reason.

J.2 Exposures to Terrestrial Species

J.2.1 Measured Concentrations in the Terrestrial Environment

No reasonably available data on 1,1-dichloroethane concentrations in terrestrial biota were identified. One study of urban rats in Oslo, Norway, tested for but did not detect any related chlorinated solvents such as 1,2-dichloroethane in the livers of rats (detection limit of 20 ng/g dry weight) (COWI AS, 2018).

J.2.2 Modeled Concentrations in the Terrestrial Environment

In general, for terrestrial mammals and birds, relative contribution to total exposure associated with inhalation is secondary in comparison to exposures by diet and indirect ingestion (EFSA, 2023). EPA has quantitatively evaluated the relative contribution of inhalation exposures for terrestrial mammals and birds in previous peer-reviewed *Guidance for Developing Ecological Soil Screening Levels (Eco-SSLs)* (U.S. EPA, 2003a, b). For 1,1-dichloroethane, other factors that guided EPA's decision to qualitatively assess 1,1-dichloroethane inhalation exposure to terrestrial receptors at a population level were (1) limited facility releases, and (2) the lack of 1,1-dichloroethane inhalation hazard data in terrestrial mammals for ecologically relevant endpoints. Air deposition to soil modeling is described in Section G.1.1. EPA determined the primary exposure pathway for terrestrial organisms is through soil via dietary uptake and incidental ingestion. As described in Section G.1.1, IIOAC and subsequently AERMOD were used to assess the estimated release of 1,1-dichloroethane to soil via air deposition 10 m from the facility (Table 3-10) from fugitive emissions reported to TRI. Air deposition of 1,1-dichloroethane to soil based on fugitive and/or stack emissions reported to NEI or modeled in generic scenarios was assessed qualitatively for exposure to terrestrial receptors since the modeled annual maximum 95th percentile (NEI) or high-end (generic scenario) air concentrations of 1,1-dichloroethane at 10 m from these sources were less than or approximately equal to that of the modeled 1,1-dichloroethane annual maximum 95th percentile air concentrations resulting from TRI-reported fugitive emissions at 10 m from releasing facilities (Tables 3-9, 3-12, and 3-13). Annual application of biosolids were also considered as a potential source of 1,1-dichloroethane in soil (Table_Apx G-3). Resulting soil pore water concentrations from daily air deposition or annual biosolids land application were also calculated.

Terrestrial plants were assessed for exposure to 1,1-dichloroethane soil pore water concentrations as described in Section L.2, and 1,1-dichloroethane soil and soil pore water concentrations were used for estimating dietary exposure through trophic transfer as described in Section L.3. For trophic transfer, EPA assumed 1,1-dichloroethane concentrations in dietary species *Trifolium* sp. as equal to the 1,1-dichloroethane maximum soil pore water concentrations for daily air deposition to soil (Table_Apx J-8) or biosolids land application of 1,1-dichloroethane (Table_Apx J-11), and in earthworms as equal to the aggregate of maximum soil and soil pore water concentrations from daily air deposition of 1,1-dichloroethane (Table_Apx J-8) or biosolids land application of 1,1-dichloroethane (Table_Apx J-11). The highest concentrations of 1,1-dichloroethane resulting from air deposition to soil in *Trifolium* sp. and earthworms were 0.15 mg/kg and 0.38 mg/kg, respectively, for the Manufacturing OES. The highest concentrations of 1,1-dichloroethane resulting from biosolids application to pastureland in *Trifolium* sp. and earthworms were 3.7×10^{-2} mg/kg and 9.5×10^{-2} mg/kg, respectively, for the Waste handling, treatment, and disposal (POTW) OES—which was the only OES with this environmental release pathway.

J.3 Trophic Transfer Exposure

J.3.1 Trophic Transfer (Wildlife)

Trophic transfer is the process by which chemical contaminants can be taken up by organisms through dietary and media exposures and be transferred from one trophic level to another. EPA has assessed the available studies collected in accordance with the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021c) and *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025ai) relating to the biomonitoring of 1,1-dichloroethane.

1,1-Dichloroethane is released to the environment by multiple exposure pathways (see Figure 2-1). The primary exposure pathway for terrestrial mammals and birds is through diet. On land, deposition of 1,1-

dichloroethane from air to soil and application of biosolids are the primary exposure pathways for dietary exposure to terrestrial mammals, whereas the primary exposure pathway for water is releases from facilities. Benthic pore water 1,1-dichloroethane concentrations determined by VVMW-PSC modeling based on the COU/OES-specific number of operating days per year (Table 3-3) are approximately equal to surface water concentrations across all COUs (see Section 3.3.3.4.2), indicating that the exposure to 1,1-dichloroethane through the aquatic dietary exposure pathway for higher trophic levels will occur from consumption of organisms in the water column or in the sediment.

Representative mammal species are chosen to connect the 1,1-dichloroethane transport exposure pathway via terrestrial trophic transfer. Uptake of contaminated soil pore water is connected by the representative plant *Trifolium* sp. to the representative herbivorous mammal meadow vole (*Microtus pennsylvanicus*). The meadow vole was selected to represent herbivores as the majority of its diet consists of plant matter, it is a native North American species, and it is a similar size to the small mammals used to derive the TRV. *Trifolium* sp. was selected as the representative plant because plants of this genus comprise a significant portion of the meadow vole diet (Lindroth and Batzli, 1984). Uptake of aggregated contaminated soil and soil pore water is connected by the representative soil invertebrate earthworm (*Eisenia fetida*) to the representative insectivorous mammal, short-tailed shrew (*Blarina brevicauda*). The short-tailed shrew was selected to represent insectivores as it is highly insectivorous, is a native North American species, and is a similar size to the small mammals used to derive the TRV. The earthworm was selected as the representative soil invertebrate because earthworms and other annelids comprise a significant portion of the short-tailed shrew diet (U.S. EPA, 1993).

Meadow voles primarily feed on plant shoots with a preference for dicot shoots in the summer and fall. When green vegetation is not available, meadow voles will feed on other foods such as seeds and roots. Thus, they are representative herbivorous terrestrial mammals for use in trophic transfer. Depending on the location and season, dicot shoots may comprise 12 to 66 percent of the meadow vole's diet (U.S. EPA, 1993). Short-tailed shrews primarily feed on invertebrates with earthworms comprising approximately 31 percent (stomach volume) to 42 percent (frequency of occurrence) of their diet; therefore, they are representative insectivorous terrestrial mammals for use in trophic transfer. The calculations for assessing 1,1-dichloroethane exposure from soil uptake by plants and earthworms and the transfer of 1,1-dichloroethane through diet to higher trophic levels are presented in Section 4.3.1.1 as well as and biota concentrations shown in Table_Apx J-8 and Table_Apx J-11. Because surface water sources for wildlife water ingestion are typically ephemeral, the trophic transfer analysis for terrestrial organisms assumed 1,1-dichloroethane exposure concentration for wildlife water intake are equal to soil concentrations for each corresponding exposure scenario.

The representative semi-aquatic terrestrial species is the American mink (*Mustela vison*), which has a highly variable diet depending on their habitat. In a riparian habitat, American mink derive 74 to 92 percent of their diet from aquatic organisms, including fish, crustaceans, birds, mammals, and vegetation (Alexander, 1977). Similar to soil concentrations used for terrestrial organisms, the highest modeled surface water and benthic pore water 1,1-dichloroethane concentration across exposure scenarios were used as surrogates for the 1,1-dichloroethane concentration found in the American mink's diet. Both in the form of water intake and a diet of either fish (bioconcentration from surface water) or crayfish (bioconcentration from benthic pore water). For trophic transfer, fish and crayfish concentrations shown in Table_Apx J-6 and Table_Apx J-7, respectively, were used in conjunction with trophic transfer calculations provided below in Section 4.3.1.1.

J.3.2 Trophic Transfer (Dietary Exposure)

EPA conducted screening level approaches for aquatic and terrestrial risk estimation based on exposure via trophic transfer using conservative assumptions for factors such as AUF as well as 1,1-dichloroethane absorption from diet, soil, sediment, and water. This chlorinated solvent has releases to aquatic and terrestrial environments as shown in Figure 2-1 and Table 3-6. Due to lack of reasonably available measured data, a BCF of 7 for 1,1-dichloroethane was estimated using EPI Suite™ (U.S. EPA, 2012b). Section 4.1.2.2 reports estimated concentrations of 1,1-dichloroethane within representative fish and crayfish tissue based the estimated BCF. A screening level analysis was conducted for trophic transfer, which employs a combination of conservative assumptions (*i.e.*, conditions for several exposure factors included within Equation_Apx J-1 below) and utilization of the maximum values obtained from modeled and/or monitoring data from relevant environmental compartments.

Following the basic equations as reported in Chapter 4 of the *U.S. EPA Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a), wildlife receptors can be exposed to contaminants in soil by two main pathways—incidental ingestion of soil while feeding and ingestion of food items that have become contaminated due to uptake from soil. The general equation used to estimate dietary exposure via these two pathways is provided below (Equation_Apx J-1). It was adapted to include consumption of water contaminated with 1,1-dichloroethane and for use with semi-aquatic mammals, including incidental ingestion of sediment instead of soil (see also Table_Apx J-5).

Exposure factors for food intake rate (FIR) and water intake rate (WIR) were sourced from the EPA's *Wildlife Exposure Factors Handbook* (also called "Wildlife EFH") (U.S. EPA, 1993); the exposure factor for sediment intake rate (SIR) was sourced from the EPA's *Second Five Year Review Report Hudson River PCBs Superfund Site Appendix 11 Human Health and Ecological Risks* (U.S. EPA, 2017a). The proportion of total food intake that is soil (P_s) is represented at the 90th percentile for representative taxa (short-tailed shrew and meadow vole) and was sourced from calculations and modeling in EPA's *Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a). The proportion of total food intake, which is sediment (P_s) for representative taxa (American mink), was calculated by dividing the sediment ingestion rate (SIR) by food consumption, which was derived by multiplying the FIR by the body weight of the mink (sourced from EPA's Wildlife EFH) (U.S. EPA, 1993). The SIR for American mink was sourced from calculations in EPA's *Second Five Year Review Report Hudson River PCBs Superfund Site Appendix 11 Human Health and Ecological Risks* (U.S. EPA, 2017a).

Equation_Apx J-1.

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

Where:

DE_j	=	Dietary exposure for contaminant (j) (mg/kg-body weight [bw]/day)
S_j	=	Concentration of contaminant (j) in soil or sediment (mg/kg dry weight)
P_s	=	Proportion of total food intake that is soil or sediment (kg soil/kg food; SIR/[(FIR)(bw)])
SIR	=	Sediment intake rate (kg of sediment [dry weight] per day)
FIR	=	Food intake rate (kg of food [dry weight] per kg body weight per day)
AF_{sj}	=	Absorbed fraction of contaminant (j) from soil or sediment (s) (for screening purposes, set to 1)
W_j	=	Concentration of contaminant (j) in water (mg/L); assumed to equal soil pore water concentrations for the purposes of terrestrial trophic transfer

AF_{wj} = Absorbed fraction of contaminant (j) from water (w) (for screening purposes, set to 1)
WIR = Water intake rate (kg of water per kg body weight per day)
N = Number of different biota type (i) in diet
B_{ij} = Concentration of contaminant (j) in biota type (i) (mg/kg dry weight)
P_i = Proportion of biota type (i) in diet
AF_{ij} = Absorbed fraction of contaminant (j) from biota type (i) (for screening purposes, set to 1)
AUF = Area use factor (for screening purposes, set to 1)

Table_Apx J-5. Terms and Values Used to Assess Potential Trophic Transfer of 1,1-Dichloroethane for Terrestrial and Semi-Aquatic Receptors

Term	Earthworm (*Eisenia fetida*)	Short-Tailed Shrew (*Blarina brevicauda*)	*Trifolium* sp.	Meadow Vole (*Microtus pennsylvanicus*)	American Mink (*Mustela vison*)
P_s	1	0.03 [a]	1	0.032 [a]	5.35E−04 [b]
FIR	1	0.555 [c]	1	0.325 [c]	0.22 [c]
AF_{sj}	1	1	1	1	1
P_i	1	1	1	1	1
WIR	1	0.223 [c]	1	0.21 [c]	0.105 [c]
AF_{wj}	1	1	1	1	1
AF_{ij}	1	1	1	1	1
SIR	N/A	N/A	N/A	N/A	1.20E−04 [d]
bw	N/A	N/A	N/A	N/A	1.0195 kg [e]
N	1	1	1	1	1
AUF	1	1	1	1	1
Highest values based on air deposition					
S_j [f]	0.382 mg/kg [g] 1,1-dichloroethane	0.382 mg/kg [g] 1,1-dichloroethane	0.146 mg/kg [h] 1,1-dichloroethane	0.382 mg/kg [g] 1,1-dichloroethane	N/A
W_j	0.382 mg/kg [g] 1,1-dichloroethane	0.382 mg/kg [g] 1,1-dichloroethane	0.146 mg/kg [h] 1,1-dichloroethane	0.382 mg/kg [g] 1,1-dichloroethane	N/A
B_{ij}	0.382 mg/kg [g] 1,1-dichloroethane (soil and soil pore water)	0.382 mg/kg 1,1-dichloroethane (worm)	0.146 mg/kg [h] 1,1-dichloroethane (soil pore water)	0.146 mg/kg 1,1-dichloroethane (plant)	N/A
Highest values based on biosolid land application					
S_j [f]	0.095 mg/kg [g] 1,1-dichloroethane	0.095 mg/kg [g] 1,1-dichloroethane	0.037 mg/kg [h] 1,1-dichloroethane	0.095 mg/kg [g] 1,1-dichloroethane	N/A
W_j	0.095 mg/kg [g] 1,1-dichloroethane	0.095 mg/kg [g] 1,1-dichloroethane	0.037 mg/kg [h] 1,1-dichloroethane	0.095 mg/kg [g] 1,1-dichloroethane	N/A
B_{ij}	0.095 mg/kg [g] 1,1-dichloroethane (soil and soil pore water)	0.095 mg/kg 1,1-dichloroethane (worm)	0.037 mg/kg [h] 1,1-dichloroethane (soil pore water)	0.037 mg/kg 1,1-dichloroethane (plant)	N/A
Highest values based on release to surface water					
S_j [f]	N/A	N/A	N/A	N/A	0.12 mg/kg [i] 1,1-dichloroethane
W_j	N/A	N/A	N/A	N/A	0.085 mg/L [j] 1,1-dichloroethane
B_{ij}	N/A	N/A	N/A	N/A	0.59 mg/kg [k] 1,1-dichloroethane (fish)

Term	Earthworm (*Eisenia fetida*)	Short-Tailed Shrew (*Blarina brevicauda*)	*Trifolium* sp.	Meadow Vole (*Microtus pennsylvanicus*)	American Mink (*Mustela vison*)
					0.55 mg/kg [l] 1,1-dichloroethane (crayfish)

[a] Soil ingestion as proportion of diet represented at the 90th percentile sourced from EPA's *Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a)
[b] Sediment ingestion as proportion of diet, calculated by dividing the SIR by kg food, where kg food = FIR × body weight (*bw*) of the mink
[c] Exposure factors (*FIR* and *WIR*) sourced from EPA's Wildlife EFH (U.S. EPA, 1993)
[d] Exposure factor (*SIR*) sourced from EPA's *Second Five Year Review Report Hudson River PCBs Superfund Site Appendix 11 Human Health and Ecological Risks* (U.S. EPA, 2017a)
[e] Mink body weight used to calculate P_s sourced from EPA's Wildlife EFH (U.S. EPA, 1993)
[f] 1,1-Dichloroethane concentration in aggregated soil and soil pore water for earthworm, short-tailed shrew, and meadow vole; 1,1-Dichloroethane concentration in soil pore water for *Trifolium* sp.; 1,1-Dichloroethane concentration in sediment for mink
[g] Highest modeled aggregated soil and soil pore water concentration of 1,1-dichloroethane calculated based on AERMOD modeling (daily deposition) for fugitive air 1,1-dichloroethane releases reported to TRI for the COU/OES Manufacturing of 1,1-dichloroethane. Concentration of contaminant in water assumed to be equal to this concentration
[h] Highest modeled soil pore water concentration of 1,1-dichloroethane calculated based on AERMOD modeling (daily deposition) for fugitive air 1,1-dichloroethane releases reported to TRI for the COU/OES Manufacturing of 1,1-dichloroethane. Concentration of contaminant in water assumed to be equal to this concentration
[i] Highest sediment concentration of 1,1-dichloroethane obtained using PSC modeling
[j] Highest surface water concentration of 1,1-dichloroethane obtained using PSC modeling
[k] Highest fish concentration (mg/kg) calculated from highest surface water concentration of 1,1-dichloroethane (PSC) and estimated BCF of 7 (U.S. EPA, 2012b)
[l] Highest crayfish concentration (mg/kg) calculated from highest benthic pore water concentration of 1,1-dichloroethane (PSC) and estimated BCF of 7 (U.S. EPA, 2012b)

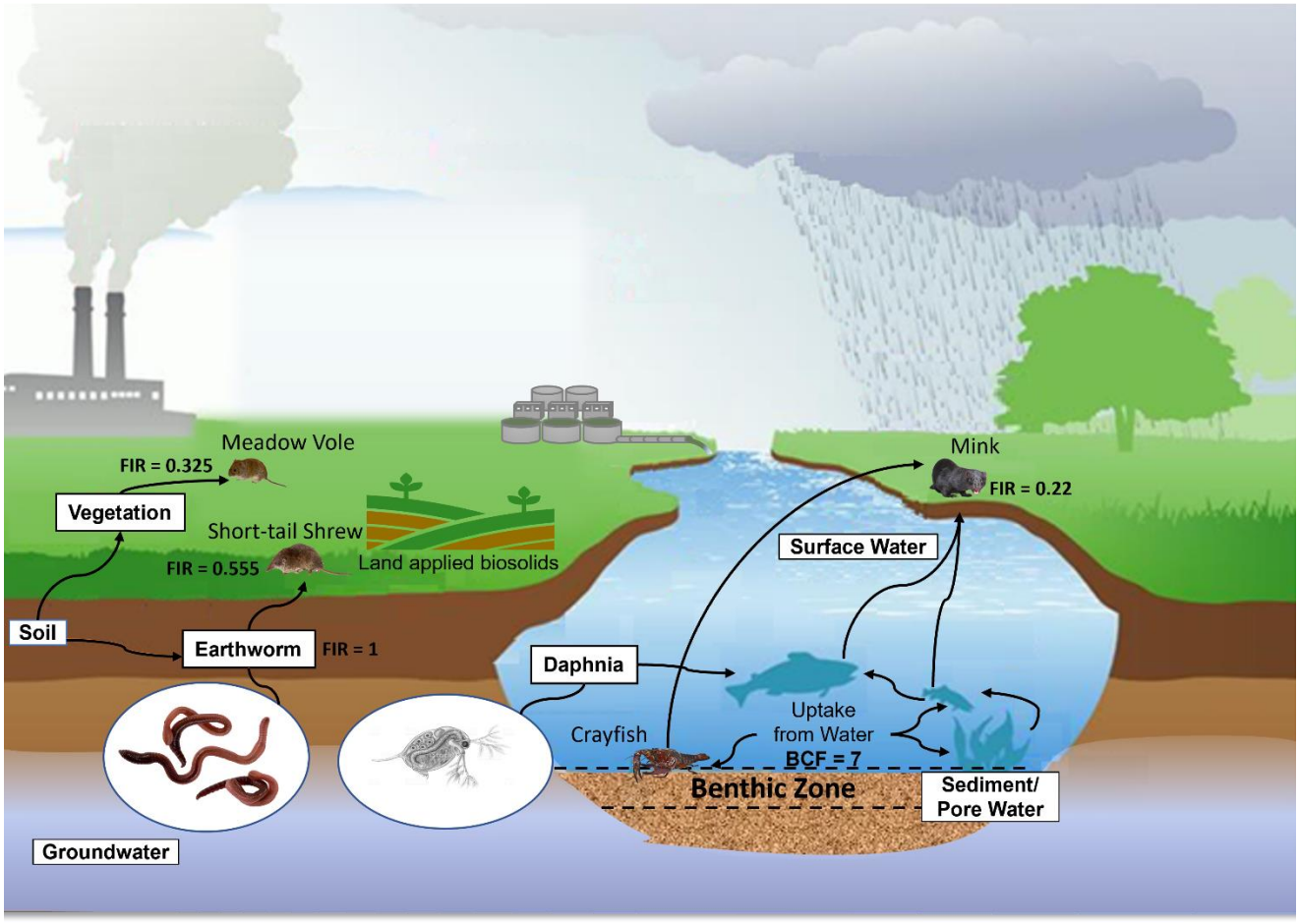
As illustrated in Figure_Apx J-1, representative mammal species were chosen to connect (1) the 1,1-dichloroethane transport exposure pathway via trophic transfer of 1,1-dichloroethane uptake from contaminated soil and soil pore water to earthworm followed by consumption by an insectivorous mammal (short-tailed shrew); and (2) 1,1-dichloroethane uptake from contaminated soil pore water to plant (*Trifolium* sp.) followed by consumption by an herbivorous mammal (meadow vole). For semi-aquatic terrestrial species, a representative mammal (American mink) was chosen to connect the 1,1-dichloroethane transport exposure pathway via trophic transfer from fish or crayfish uptake of 1,1-dichloroethane from contaminated surface water and benthic pore water.

At the screening level, one conservative assumption is that the invertebrate diet for the short-tailed shrew comprises 100 percent earthworms from contaminated soil. Similarly, the dietary assumption for the meadow vole is 100 percent *Trifolium* sp. from contaminated soil. For mink, in one scenario 100 percent of the American mink's diet is predicted to come from fish, and in the second scenario 100 percent of the American mink's diet is predicted to come from crayfish. Additionally, the screening level analysis uses the highest modeled 1,1-dichloroethane soil, soil pore water, surface water, or benthic pore water contaminate levels based on daily air deposition or annual biosolids land application (soil and soil pore water) as well as the COU/OES-specific number of operating days per year for surface water releases (surface water, benthic pore water, and sediment) to determine whether a more detailed assessment is required. Because surface water sources for terrestrial wildlife water ingestion are typically ephemeral, the trophic transfer analysis for the short-tailed shrew and meadow vole assumed 1,1-dichloroethane exposure concentration for wildlife water intake are equal to aggregated soil and soil pore water concentrations for each corresponding exposure scenario.

The highest soil and soil porewater concentrations calculated based on AERMOD daily air deposition for the COU/OES described in Table_Apx J-8 or annual biosolids land application for the COU/OES described in Table_Apx J-11 were used to represent 1,1-dichloroethane concentrations in media for terrestrial trophic transfer. Similarly, the highest PSC-modeled surface water and sediment concentrations over the operating days per year for the COU/OES described in Table_Apx J-6 and Table_Apx J-7 were used to represent 1,1-dichloroethane concentrations in media for trophic transfer to a semi-aquatic mammal (mink). Additional assumptions for this analysis have been considered to represent conservative screening values (U.S. EPA, 2005a). Within this model, incidental oral soil or sediment exposure is added to the dietary exposure (including water consumption) resulting in total oral exposure to 1,1-dichloroethane. In addition, EPA assumes that 100 percent of the contaminant is absorbed from both the soil (AF_{sj}), water (AF_{wj}) and biota representing prey (AF_{ij}). The proportional representation of time an animal spends occupying an exposed environment is known as the AUF and has been set at 1 for all biota within this equation (Table_Apx J-5). Values for calculated dietary exposure by COU are shown in Table_Apx J-12 and Table_Apx J-13 for trophic transfer to shrew and vole from air deposition of 1,1-dichloroethane to soil; Table_Apx J-14 and Table_Apx J-15 for trophic transfer to shrew and vole from biosolids land application of 1,1-dichloroethane to soil; and Table_Apx J-8 and Table_Apx J-9 for trophic transfer to mink consuming fish and crayfish.

In each trophic transfer scenario for concentrations resulting from air deposition to soil, the manufacturing OES results in the highest biota concentrations and dietary exposure (Appendix J.3.3). The Waste handling, treatment, and disposal (POTW) OES was the only OES with releases to soil via biosolid land application. In each trophic transfer scenario for this pathway, the pastureland pathway resulted in the highest biota concentrations and dietary exposure (Appendix J.3.3). In each trophic transfer scenario for concentrations resulting from releases to surface water, the Manufacturing OES results in the highest biota concentrations and dietary exposure (Appendix J.3.3). The highest dietary exposure across all scenarios results from the Manufacturing OES surface water releases and consumption of fish by mink and is 0.14 mg/kg/day (Table_Apx J-8). Earthworm and *Trifolium* sp. concentrations (mg/kg) were conservatively assumed equal to aggregated soil and soil pore water concentrations (earthworm) or soil pore water concentrations only (*Trifolium* sp.). Fish and crayfish concentrations (mg/kg) were calculated using surface water and benthic pore water concentrations of 1,1-dichloroethane, respectively, from PSC and an estimated BCF of 7 (U.S. EPA, 2012b). A comparison of fish consumption in mink is also provided using actual measured concentrations of 1,1-dichloroethane in Lake Pontchartrain oysters (Ferrario et al., 1985) and the maximum measured surface water concentration of 1,1-dichloroethane as reported in Section 3.3.3.1. The estimated exposure for mink consuming fish based on these reported values is 7.5×10^{-3} mg/kg/day as compared to the highest and lowest COU/OES-based dietary exposure estimates of 0.14 mg/kg/day and 1.0×10^{-3} mg/kg/day for the Manufacturing COU/OES and Use as a laboratory chemical COU/OES, respectively.

The trophic transfer of 1,1-dichloroethane from media to biota is illustrated in Figure_Apx J-1 with the movement of 1,1-dichloroethane through the food web indicated by black arrows. Within the aquatic environment, the benthic zone is bounded by dashed black lines from the bottom of the water column to sediment surface and subsurface layers. The depth that the benthic environment extends into subsurface sediment is site-specific. Figure_Apx J-1 illustrates the 1,1-dichloroethane BCF for aquatic organisms and food intake rates (FIRs) for the representative terrestrial organisms.



Figure_Apx J-1. Trophic Transfer of 1,1-Dichloroethane in Aquatic and Terrestrial Ecosystems
FIR = food ingestion rate.

J.3.3 Concentrations in Biota and Associated Dietary Exposure Estimates

Table_Apx J-6. 1,1-Dichloroethane Fish Concentrations Calculated from PSC-Modeled Industrial and Commercial 1,1-Dichloroethane Releases

COU (Life Cycle/Category/Subcategory)	OES	Facility	Receiving Waterbody	SWC (µg/L) [a]	Fish Concentration (ng/g)
Manufacture/ Domestic manufacturing/ Domestic manufacturing	Manufacturing	LA0000761	Bayou D'Inde & Bayou Verdine	85	590
Processing/As a reactant/ Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	TX0119792	Unnamed ditch, San Jacinto Bay	13	90
Processing/As a reactant/ Intermediate in all other chemical product and preparation manufacturing					
Processing/Recycling/Recycling					
Processing/Processing – repackaging/Processing – repackaging	Processing – repackaging	IL0064564	Rock River	7.0E−01	4.9

COU (Life Cycle/Category/Subcategory)	OES	Facility	Receiving Waterbody	SWC (µg/L) [a]	Fish Concentration (ng/g)
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	IL0034592	Sawmill Creek	6.4E−01	4.5
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	NE0043371	Stevens Creek	12	87
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	KY0022039	Valley Creek	8.2	57
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (remediation)	CA0064599	South Fork of Arroyo Conejo Creek	31	210
Distribution in Commerce/ Distribution in commerce/ Distribution in commerce	Distribution in commerce	N/A[b]			

[a] Max daily average represents the maximum surface water concentration (SWC) over the COU/OES-specific operating days per year (Table 3-3).
[b] Distribution in commerce does not result in surface water releases (Table 3-6).

Table_Apx J-7. 1,1-Dichloroethane Crayfish Concentrations Calculated from PSC-Modeled Industrial and Commercial 1,1-Dichloroethane Releases

COU (Life Cycle/Category/Subcategory)	Scenario Name	Facility	Receiving Waterbody	PWC (µg/L)[a]	Crayfish Concentration (ng/g)
Manufacture/domestic manufacturing/domestic manufacturing	Manufacturing	LA0000761	Bayou D'Inde & Bayou Verdine	78	550
Processing/As a reactant/ intermediate in all other basic organic chemical manufacture	Processing as a Reactive Intermediate	TX0119792	Unnamed ditch, San Jacinto Bay	12	87
Processing/As a reactant/ Intermediate in all other chemical product and preparation manufacturing					
Processing/Recycling/Recycling					
Processing/Processing – repackaging/Processing – repackaging	Processing – Repackaging	IL0064564	Rock River	6.1E−01	4.3
Commercial Use/Other use/ Laboratory chemicals	Commercial Use as a Laboratory Chemical	IL0034592	Sawmill Creek	5.5E−01	3.8
Disposal/Disposal/Disposal	General Waste Handling, Treatment and Disposal	NE0043371	Stevens Creek	12	83
Disposal/Disposal/Disposal	Waste Handling, Treatment and Disposal (POTW)	KY0022039	Valley Creek	7.9	55
Disposal/Disposal/Disposal	Waste Handling, Treatment, and Disposal (Remediation)	CA0064599	South Fork of Arroyo Conejo Creek	29	210

COU (Life Cycle/Category/Subcategory)	Scenario Name	Facility	Receiving Waterbody	PWC (µg/L)[a]	Crayfish Concentration (ng/g)
Distribution in Commerce/ Distribution in commerce/ Distribution in commerce	Distribution in Commerce	N/A[b]			

[a] Max daily average represents the maximum benthic pore water concentration (PWC) over the COU/OES-specific operating days per year (Table 3-3).
[b] Distribution in Commerce does not result in surface water releases (Table 3-6).

Table_Apx J-8. Dietary Exposure Estimates Using EPAs Wildlife Risk Model for Eco-SSLs for Screening Level Trophic Transfer of 1,1-Dichloroethane to the American Mink from Consumption of Fish

COU (Life Cycle Stage/Category/Subcategory)	OES	Fish Concentration (mg/kg) [a]	1,1-Dichloroethane Dietary Exposure (mg/kg/day) [b]
Manufacture/Domestic manufacturing/Domestic manufacturing	Manufacturing	5.9E−01	1.4E−01
Processing/As a reactant/Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	9.0E−02	2.1E−02
Processing/As a reactant/intermediate in all other chemical product and preparation manufacturing			
Processing/Recycling/Recycling			
Processing/Processing – repackaging/Processing – repackaging	Processing – repackaging	4.9E−03	1.2E−03
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	4.5E−03	1.0E−03
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	8.7E−02	2.0E−02
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	5.7E−02	1.3E−02
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (remediation)	2.1E−01	5.1E−02
Distribution in Commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce	N/A[c]	
Published data			
Lake Pontchartrain oysters (Ferrario et al., 1985)		3.3E−02	7.5E−03

[a] Whole fish concentrations were calculated using the highest modeled max daily average surface water concentrations for 1,1-dichloroethane (via PSC modeling based on total number of operating days) and a BCF of 7.
[b] Dietary exposure to 1,1-dichloroethane includes consumption of biota (fish), incidental ingestion of sediment, and ingestion of water.
[c] Distribution in Commerce does not result in surface water releases (Table 3-6).

Table_Apx J-9. Dietary Exposure Estimates Using EPAs Wildlife Risk Model for Eco-SSLs for Screening Level Trophic Transfer of 1,1-Dichloroethane to the American Mink from Consumption of Crayfish

COU (Life Cycle Stage/Category/Subcategory)	OES	Crayfish Concentration (mg/kg) [a]	1,1-Dichloroethane Dietary Exposure (mg/kg/day) [b]
Manufacture/Domestic manufacturing/Domestic manufacturing	Manufacturing	5.5E−01	1.3E−01
Processing/As a reactant/intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	8.7E−02	2.0E−02
Processing/As a reactant/intermediate in all other chemical product and preparation manufacturing			
Processing/Recycling/Recycling			
Processing/Processing – repackaging/ Processing – repackaging	Processing – repackaging	4.3E−03	1.0E−03
Commercial Use/Other use/ Laboratory chemicals	Commercial use as a laboratory chemical	3.8E−03	9.1E−04
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	8.3E−02	1.9E−02
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	5.5E−02	1.3E−02
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (remediation)	2.1E−01	4.8E−02
Dstribution in Commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce	N/A[c]	

[a] Whole crayfish concentrations were calculated using the highest modeled max daily average benthic pore water concentrations for 1,1-dichloroethane (via PSC modeling based on total number of operating days) and a BCF of 7.
[b] Dietary exposure to 1,1-dichloroethane includes consumption of biota (crayfish), incidental ingestion of sediment, and ingestion of water.
[c] Distribution in Commerce does not result in surface water releases (Table 3-6).

Table_Apx J-10. 1,1-Dichloroethane *Trifolium* sp. and Earthworm Concentrations Calculated from AERMOD Modeled Industrial and Commercial Releases Reported to TRI

COU (Life Cycle Stage/Category/Subcategory)	OES	Soil (mg/kg) [a]	Soil Pore Water Concentration (mg/L) [a]	Plant Concentration (mg/kg)	Earthworm Concentration (mg/kg)
Manufacture/Domestic manufacturing/Domestic manufacturing	Manufacturing	2.4E−01	1.5E−01	1.5E−01	3.8E−01
Processing/As a reactant/ intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	5.2E−03	3.2E−03	3.2E−03	8.4E−03
Processing/As a reactant/ intermediate in all other chemical product and preparation manufacturing					

COU (Life Cycle Stage/Category/Subcategory)	OES	Soil (mg/kg) [a]	Soil Pore Water Concentration (mg/L) [a]	Plant Concentration (mg/kg)	Earthworm Concentration (mg/kg)
Processing/Recycling/Recycling					
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	1.2E−04	7.6E−05	7.6E−05	2.0E−04
[a] Soil catchment and soil catchment pore water concentrations estimated from 95th percentile maximum daily air deposition rates 10 m from facility for fugitive air 1,1-dichloroethane releases reported to TRI.					

Table_Apx J-11. 1,1-Dichloroethane *Trifolium* sp. and Earthworm Concentrations Calculated from Land Application of 1,1-Dichloroethane in Biosolids

COU (Life Cycle Stage/Category/ Subcategory)	OES	Pathway	Soil (mg/kg) [a]	Soil Pore Water Concentration (mg/L)[a]	Plant Concentration (mg/kg)	Earthworm Concentration (mg/kg)
Disposal/Disposal/ Disposal	Waste handling, treatment, and disposal (POTW)	Tilled Agricultural	2.9E−02	1.9E−02	1.9E−02	4.8E−02
		Pastureland	3.7E−02	5.9E−02	3.7E−02	9.5E−02
[a] Soil and soil pore water concentrations estimated from annual application of biosolids.						

Table_Apx J-12. Dietary Exposure Estimates Using EPAs Wildlife Risk Model for Eco-SSLs for Screening Level Trophic Transfer of 1,1-Dichloroethane to the Short-Tailed Shrew that Could Result from Air Deposition to Soil for 1,1-Dichloroethane Releases Reported to TRI

COU (Life Cycle Stage/Category/Subcategory)	OES	Earthworm Concentration (mg/kg) [a]	1,1-Dichloroethane Dietary Exposure (mg/kg/day) [b]
Manufacture/Domestic manufacturing/ Domestic manufacturing	Manufacturing	3.8E−01	2.5E−01
Processing/As a reactant/Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	8.5E−03	5.6E−03
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing			
Processing/Recycling/Recycling			
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	2.0E−04	1.3E−04
[a] Estimated 1,1-dichloroethane concentration in representative soil invertebrate, earthworm, assumed equal to aggregated highest calculated soil and soil pore water concentration via air deposition of 1,1-dichloroethane in fugitive air releases reported to TRI to soil. [b] Dietary exposure to 1,1-dichloroethane includes consumption of biota (earthworm), incidental ingestion of soil, and ingestion of water.			

Table_Apx J-13. Dietary Exposure Estimates Using EPAs Wildlife Risk Model for Eco-SSLs for Screening Level Trophic Transfer of 1,1-Dichloroethane to the Meadow Vole that Could Result from Air Deposition to Soil for 1,1-Dichloroethane Releases Reported to TRI

COU (Life Cycle Stage/Category/Subcategory)	OES	Plant Concentration (mg/kg) [a]	1,1-Dichloroethane Dietary Exposure (mg/kg/day) [b]
Manufacture/Domestic manufacturing/ Domestic manufacturing	Manufacturing	1.5E−01	8.2E−02
Processing/As a reactant/Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	3.2E−03	1.8E−03
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing			
Processing/Recycling/Recycling			
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	7.6E−05	4.3E−05

[a] Estimated 1,1-dichloroethane concentration in representative terrestrial plant *Trifolium* sp., assumed equal to the highest calculated soil pore water concentration via air deposition of 1,1-dichloroethane in fugitive air releases reported to TRI to soil.
[b] Dietary exposure to 1,1-dichloroethane includes consumption of biota (*Trifolium* sp.), incidental ingestion of soil, and ingestion of water.

Table_Apx J-14. Dietary Exposure Estimates Using EPAs Wildlife Risk Model for Eco-SSLs for Screening Level Trophic Transfer of 1,1-Dichloroethane to the Short-Tailed Shrew that Could Result from Land Application of Biosolids

COU (Life Cycle Stage/Category/Subcategory)	OES	Pathway	Earthworm Concentration (mg/kg) [a]	1,1-Dichloroethane Dietary Exposure (mg/kg/day) [b]
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	Tilled agricultural	4.8E−02	3.1E−02
		Pastureland	9.5E−02	6.3E−02

[a] Estimated 1,1-dichloroethane concentration in representative soil invertebrate, earthworm, assumed equal to aggregated highest calculated soil and soil pore water concentration via land application of biosolids to soil.
[b] Dietary exposure to 1,1-dichloroethane includes consumption of biota (earthworm), incidental ingestion of soil, and ingestion of water.

Table_Apx J-15. Dietary Exposure Estimates Using EPAs Wildlife Risk Model for Eco-SSLs for Screening Level Trophic Transfer of 1,1-Dichloroethane to the Meadow Vole that Could Result from Land Application of Biosolids

COU (Life Cycle Stage/Category/Subcategory)	OES	Pathway	Plant Concentration (mg/kg) [a]	1,1-Dichloroethane Dietary Exposure (mg/kg/day) [b]
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	Tilled agricultural	1.9E−02	1.0E−02
		Pastureland	3.7E−02	2.1E−02

[a] Estimated 1,1-dichloroethane concentration in representative terrestrial plant *Trifolium* sp., assumed equal to the highest calculated soil pore water concentration via land application of biosolids to soil.
[b] Dietary exposure to 1,1-dichloroethane includes consumption of biota (*Trifolium* sp.), incidental ingestion of soil, and ingestion of water.

J.3.4 Trophic Transfer Confidence

EPA uses several considerations when weighing the scientific evidence to determine confidence in the dietary exposure estimates. These considerations include the quality of the database, consistency, strength and precision, and relevance (Table_Apx K-2). This approach is in agreement with the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021c) and *Risk Evaluation for 1,1-Dichloroethane – Systematic Review Protocol* (U.S. EPA, 2025ai). Table_Apx J-16 summarizes how these considerations were determined for each dietary exposure threshold. For trophic transfer EPA considered the evidence for (1) insectivorous terrestrial mammals – moderate, (2) herbivorous terrestrial mammals – moderate, (3) fish-consuming semi-aquatic mammals – moderate, and (4) crayfish-consuming semi-aquatic mammals – slight (Table_Apx J-16).

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

Few empirical biomonitoring data in ecological receptors were reasonably available for 1,1-dichloroethane or related chlorinated solvents. These data include one study containing 1,1-dichloroethane measurements in oysters (Ferrario et al., 1985), one study containing fish tissue concentrations in other similar chlorinated solvents (1,1,1-trichloroethane and trichloroethylene) (Roose and Brinkman, 1998) and a third study with non-detect of 1,2-dichloroethane in urban rats (COWI AS, 2018). Thus, the quality of the database was rated slight. For COUs/OESs-based dietary exposure estimates, biota concentrations in representative species and their diet were calculated based on the methodology described in Section 4.3.1.1. The calculated aquatic biota concentrations were of similar range to the reported concentrations of 1,1-dichloroethane and related chlorinated solvents in aquatic biota, which resulted in a moderate confidence for consistency of the aquatic-based dietary exposure estimates for the trophic transfer analyses shown in Table_Apx J-16 whereas this consideration was determined "NA" for terrestrial-based dietary exposure estimates.

No empirical BCF or BAF data were reasonably available; therefore, concentrations in aquatic biota were calculated based on a predicted BCF derived from bioconcentration of a training set of chemicals from water to fish. Because the training set utilized to generate the 1,1-dichloroethane BCF value in EPI Suite™ contains other low-molecular weight chlorinated solvents (U.S. EPA, 2012b), this results in a moderate confidence for strength and precision for the trophic transfer based on fish consumption. Applying this predicted BCF value based on fish to calculate whole crayfish concentrations adds uncertainty to dietary exposures estimates from consumption of sediment-dwelling invertebrates by mink resulting in a slight confidence in the strength and precision of the dietary exposure estimates

based on crayfish consumption. For terrestrial mammal trophic transfer, due to lack of empirical BAF values, it was conservatively assumed that whole earthworm and whole plant concentrations were equal to soil and/or soil pore water concentrations, respectively. However, the use of species-specific exposure factors (*i.e.*, feed intake rate, water intake rate, the proportion of soil or sediment within the diet) from reliable resources assisted in obtaining dietary exposure estimates within the RQ equation (U.S. EPA, 2017a, 1993), thereby increasing the confidence for strength and precision, resulting in an moderate confidence for strength and precision of the dietary exposure estimates in terrestrial trophic transfer.

Relevance (Biological, Physical and Chemical, and Environmental)
The short-tailed shrew, meadow vole, and American mink were selected as representative mammals for the soil invertivore-, soil herbivore-, and aquatic-based trophic transfer analyses, respectively (U.S. EPA, 1993), based on their import in previous trophic transfer analyses conducted by the Agency (U.S. EPA, 2003a, b). Appropriate dietary species (earthworm, plant, fish, crayfish) were selected based on dietary information for shrew, vole, and mink provided in the EPA's Wildlife EFH (U.S. EPA, 1993). The selection of the relevant apex and their representative dietary species in the trophic transfer analyses increases confidence in the biological relevance of the dietary exposure estimates. Modeled concentrations for water and soil used to determine biota concentrations for trophic transfer were based on 1,1-dichloroethane data and not those of an analog; therefore, increasing confidence in physical and chemical relevance of the dietary exposures in the trophic transfer analyses (for information on analog selection see Section 4.2.1.1). The current trophic transfer analysis investigated dietary exposure resulting from 1,1-dichloroethane in biota and environmentally relevant media such as soil, sediment, and water. The screening level analysis for trophic transfer used equation terms (*e.g.*, AUF and the proportion of 1,1-dichloroethane absorbed from diet, and soil or sediment) all set to the most conservative values, emphasizing a cautious approach to risk to 1,1-dichloroethane via trophic transfer.

Assumptions within the trophic transfer equation (Appendix J.3.2) for this analysis have been considered to represent conservative screening values (U.S. EPA, 2005a) and those assumptions were applied similarly for each trophic level and representative species. Applications across representative species included assuming 100 percent 1,1-dichloroethane bioavailability from both the soil (AF_{sj}) and biota representing prey (AF_{ij}). No additional dietary species other than the selected dietary species were included as part of the dietary exposure for the respective terrestrial mammal ($P_i = 1$). The AUF, defined as the home range size relative to the contaminated area (*i.e.*, site ÷ home range = AUF), within this screening level analysis was designated as 1 for all organisms, which assumes a potentially longer residence within an exposed area or a large exposure area. These conservative approaches, which likely overrepresent 1,1-dichloroethane's ability to transfer among the trophic levels, decrease environmental relevance of the dietary exposures within the trophic transfer analyses, resulting in an overall moderate confidence for relevance of the dietary exposure estimates.

Trophic Transfer Confidence
Due to moderate confidence in both the strength and precision and relevance for the dietary exposure estimates to insectivorous and herbivorous terrestrial mammals, the trophic transfer confidence is moderate in both cases. Due to moderate confidence in strength and precision and relevance in dietary exposure estimates to mink based on fish consumption, the trophic transfer confidence is moderate. Due to slight confidence in quality of the database and strength and precision considerations for dietary exposure estimates to mink based on crayfish consumption, the trophic transfer confidence is assigned slight.

Table_Apx J-16. 1,1-Dichloroethane Evidence Table Summarizing Overall Confidence Derived for Trophic Transfer (Dietary)

Types of Evidence	Quality of the Database	Consistency	Strength and Precision	Relevance[a]	Trophic Transfer Confidence
Chronic avian assessment	N/A	N/A	N/A	N/A	Indeterminate
Chronic mammalian assessment (insectivorous)	+	N/A	++	++	Moderate
Chronic mammalian assessment (herbivorous)	+	N/A	++	++	Moderate
Chronic mammalian assessment (fish consumption)	+	++	++	++	Moderate
Chronic mammalian assessment (crayfish consumption)	+	++	+	++	Slight

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

Appendix K ENVIRONMENTAL HAZARD DETAILS

K.1 Approach and Methodology

For aquatic species, EPA estimates hazard by calculating a concentration of concern (COC) for a hazard threshold. COCs can be calculated using a deterministic method by dividing a hazard value by an assessment factor (AF) according to EPA methods (Suter, 2016; U.S. EPA, 2013b, 2012a).

Equation_Apx K-1.

$$COC = toxicity\ value \div AF$$

COCs can also be calculated using probabilistic methods. For example, an SSD can be used to calculate a hazardous concentration for 5 percent of species (HC05). The HC05 estimates the concentration of a chemical that is expected to protect 95 percent of aquatic species. This HC05 can then be used to calculate a COC. For 1,1-dichloroethane, Web-based Interspecies Correlation Estimation (Web-ICE) (Appendix K.2.1.1) followed by the species sensitivity distribution (SSD) probabilistic method (Appendix K.2.1.2) was used to calculate the HC05 on which the acute COC is based. The deterministic method was used to calculate a chronic COC.

Terrestrial receptor groups are simplified to terrestrial plants, soil dwelling invertebrates, mammals, and birds. For terrestrial plants and soil dwelling organisms, EPA estimates hazard by using a hazard value based on hazard information relating soil or soil pore water concentrations to a hazard value. For avian and mammalian toxicity reference values (TRVs) in units of an oral dose in mg/kg/bw-day are identified using a peer reviewed approach used to establish soil screening levels for the Superfund Program. The TRV is expressed as doses in units of mg/kg-bw/day. Although the TRV for 1,1-dichloroethane is derived from mammalian laboratory studies, body weight is normalized, therefore the TRV can be used with ecologically relevant wildlife species to evaluate chronic dietary exposure to 1,1-dichloroethane (U.S. EPA, 2007).

K.2 Hazard Identification

K.2.1 Aquatic Hazard Data

K.2.1.1 Web-Based Interspecies Correlation Estimation (Web-ICE)

Results from the systematic review process assigned an overall quality level of high to six acceptable aquatic toxicity studies for 1,1-dichloroethane, high or medium to six acceptable aquatic studies for analog 1,2-dichloropropane, and high or medium to two acceptable aquatic study for analog 1,1,2-trichloroethane, with one 1,1-dichloroethane and two 1,2-dichloropropane studies producing LC50 (*i.e.*, the concentration of a substance that is lethal to 50% of a test population) concentration endpoint data (Table 4-7). To supplement the empirical data, EPA used a modeling approach, Web-ICE. Web-ICE predicts toxicity values for environmental species that are absent from a dataset and can provide a more robust dataset to estimate toxicity thresholds. Specifically, EPA used Web-ICE to quantitatively supplement empirical data for aquatic organisms for acute exposure durations.

The Web-ICE application was developed by EPA and collaborators to provide interspecies extrapolation models for acute toxicity (Raimondo and Barron, 2010). Web-ICE models estimate the acute toxicity (LC50/LD50 [lethal dose of a substance required to kill 50% of a test population after a specified time]) of a chemical to a species, genus, or family with no test data (the predicted taxon) from the known toxicity of the chemical to a species with test data (the commonly tested surrogate species).

Web-ICE models are log-linear least square regressions of the relationship between surrogate and predicted taxon based on a database of acute toxicity values. It returns median effect or lethal water concentrations for aquatic species (EC50 [concentration of a substance required to achieve a biological response halfway between the baseline, no response, and the maximum possible response]/LC50). Separate acute toxicity databases are maintained for aquatic animals (vertebrates and invertebrates), aquatic plants (algae), and wildlife (birds and mammals)—with 2,286 models for aquatic animals, 58 models for algae, and 560 models for terrestrial wildlife taxa in Web-ICE v4.0 (U.S. EPA, 2024). Open-ended toxicity values (*i.e.*, >100 mg/kg or <100 mg/kg) and duplicate records among multiple sources are not included in any of the databases.

The aquatic animal database within Web-ICE is composed of 48- or 96-hour EC50/LC50 values based on immobility or mortality. This database is described in detail in the Aquatic Database Documentation found on the Download Model Data (accessed June 11, 2025) page of Web-ICE and describes the data sources, normalization, and quality and standardization criteria (*e.g.*, data filters) for data used in the models. Data used in model development adhered to standard acute toxicity test condition requirements of the ASTM International (ASTM, 2014) and EPA's Office of Chemical Safety and Pollution Prevention (OCSPP) (*e.g.*, (U.S. EPA, 2016a).

EPA used the 1,1-dichloroethane 48-hour LC50 data for *Daphnia magna* and the 1,2-dichloropropane 96-hour LC50 toxicity data for fathead minnow and opossum shrimp (Table 4-7) as surrogate species to predict LC50 toxicity values using the Web-ICE application (Raimondo and Barron, 2010). The Web-ICE Model estimated toxicity values for 149 species. For model validation, the model results were then screened by the following quality standards to ensure confidence in the model predictions. If a predicted species did not meet all the quality criteria below, the species was eliminated from the dataset (U.S. EPA, 2024):

- High R^2 (≥0.6)
    - The proportion of the data variance that is explained by the model. The closer the R^2 value is to 1, the more robust the model is in describing the relationship between the predicted and surrogate taxa.
- Low mean square error (MSE; ≤0.95)
    - An unbiased estimator of the variance of the regression line.
- High slope (≥0.6)
    - The regression coefficient represents the change in log10 value of the predicted taxon toxicity for every change in log10 value of the surrogate species toxicity.
    - For models where the predicted value exceeds the model maximum, a stricter slope of 0.66 to 1.33 is applied (Raimondo et al., 2023)
- No more than two orders of magnitude of difference between the upper and lower bounds of the confidence interval of the predicted toxicity.

After screening, 102 acute toxicity values representing 75 species (25 fish, 2 amphibians, and 48 aquatic invertebrates) were added to the chironomid 48-hour LC50, fathead minnow 96-hour LC50, daphnia 48-hour LC50, and opossum shrimp 96-hour LC50 data (Table_Apx K-1). The toxicity data were then used to calculate the distribution of species sensitivity through the SSD Toolbox (Etterson, 2020a), as described in Appendix K.2.1.2.

Table_Apx K-1. Empirical and Web-ICE Predicted Species that Met Model Selection Criteria

Common Name	Scientific Name	Surrogate	Estimated Toxicity (µg/L)	95% CI	CI Difference	R^2	MSE	Slope
Empirical data								
Mysid	*Americamysis bahia*		24,790					
Chironomid	*Chironomus riparius*		150,000					
Daphnia	*Daphnia magna*		34,300					
Fathead minnow	*Pimephales promelas*		133,340					
Web-ICE data								
Copepod	*Acartia clausi*	Daphnid (*Daphnia magna*)	24,277.39	2,441.29–241,426.14	2	0.72	0.32	0.71
Copepod	*Acartia clausi*	Fathead minnow (*Pimephales promelas*)	13,785.74	2,055.01–92,479.52	1	0.8	0.23	0.88
Amphipod	*Allorchestes compressa*	Mysid (*Americamysis bahia*)	25,469.89	5,482.90–118,315.97	2	0.95	0.03	0.66
Amphipod	*Allorchestes compressa*	Fathead minnow (*Pimephales promelas*)	31,057.39	8,571.19–112,535.32	2	0.96	0.02	0.84
Threeridge	*Amblema plicata*	Daphnid (*Daphnia magna*)	7,479.19	2,771.51–20,183.32	1	0.97	0.13	0.94
Threeridge	*Amblema plicata*	Fathead minnow (*Pimephales promelas*)	7,615.83	2,131.29–27,213.96	1	0.97	0.12	1.3
Black bullhead	*Ameiurus melas*	Fathead minnow (*Pimephales promelas*)	243,845.83	19,360.87–3,071,183.09	2	0.96	0.16	1.07
Mysid	*Americamysis bigelowi*	Mysid (*Americamysis bahia*)	106,070.9	48,103.39–233,892.78	1	0.99	0	1.29
Mysid	*Americamysis bigelowi*	Fathead minnow (*Pimephales promelas*)	16,489.37	1,524.40–178,363.83	2	0.89	0.05	1
Isopod	*Asellus aquaticus*	Daphnid (*Daphnia magna*)	583,142.08	119,486.48–2,845,967.89	1	0.91	0.31	0.85
Vernal pool fairy shrimp	*Branchinecta lynchi*	Daphnid (*Daphnia magna*)	24,783.05	12,894.55–47,632.48	0	0.98	0.07	0.94
Isopod	*Caecidotea brevicauda*	Fathead minnow (*Pimephales promelas*)	7,239.52	740.82–70745.92	2	0.72	0.49	0.76
Polychaete	*Capitella capitata*	Mysid (*Americamysis bahia*)	62,057.33	15,969.43–241,155.23	1	0.97	0.04	0.95
Goldfish	*Carassius auratus*	Fathead minnow (*Pimephales promelas*)	157,504.93	97,401.60–254,696.04	1	0.94	0.14	0.97
White sucker	*Catostomus commersonii*	Fathead minnow (*Pimephales promelas*)	330,278.92	41,613.75–2,621,348.84	2	0.92	0.2	1.14
Daphnid	*Ceriodaphnia dubia*	Mysid (*Americamysis bahia*)	12,139.7	3,794.22–38,841.28	1	0.83	0.77	0.97
Daphnid	*Ceriodaphnia dubia*	Daphnid (*Daphnia magna*)	23,536.23	15,719.96–35,238.90	0	0.95	0.24	1.02
Bigscale mullet	*Chelon macrolepis*	Daphnid (*Daphnia magna*)	1,604,020.27	553,455.75–4,648,756.51	1	0.99	0	0.9
Midge	*Chironomus plumosus*	Mysid (*Americamysis bahia*)	14,926.21	2,908.40–76,602.93	1	0.77	0.66	0.73
Midge	*Chironomus tentans*	Daphnid (*Daphnia magna*)	150,361.1	10,582.12–2,136,475.58	2	0.89	0.81	0.97
Water flea	*Chydorus sphaericus*	Daphnid (*Daphnia magna*)	12,792.84	8,333.50–19,638.43	1	0.98	0.05	0.94
Mrigal carp	*Cirrhinus mrigala*	Fathead minnow (*Pimephales promelas*)	122,679.62	30,551.64–492,618.00	1	0.98	0.01	1.1
Common shrimp	*Crangon crangon*	Fathead minnow (*Pimephales promelas*)	91,213.15	40,194.34–206,990.27	1	0.99	0	0.97
Amphipod	*Crangonyx pseudogracilis*	Daphnid (*Daphnia magna*)	24,9124.35	35,153.61–1,765,478.24	2	0.74	0.75	0.91
Eastern oyster	*Crassostrea virginica*	Fathead minnow (*Pimephales promelas*)	30,610.61	9,844.24–95,183.53	1	0.66	0.48	0.84
Sheepshead minnow	*Cyprinodon variegatus*	Fathead minnow (*Pimephales promelas*)	41,874.22	19,570.37–89,597.23	0	0.72	0.33	0.75
Common carp	*Cyprinus carpio*	Fathead minnow (*Pimephales promelas*)	137,533	54,611.32–346,362.71	1	0.84	0.2	0.99

Common Name	Scientific Name	Surrogate	Estimated Toxicity (µg/L)	95% CI	CI Difference	R^2	MSE	Slope
Zebrafish	*Danio rerio*	Daphnid (*Daphnia magna*)	51,951.43	9,678.30–278,866.03	2	0.71	0.61	0.77
Zebrafish	*Danio rerio*	Fathead minnow (*Pimephales promelas*)	93,894.98	34,630.75–254,579.20	1	0.97	0.04	0.9
Zebrafish-embryo	*Danio rerio-embryo*	Daphnid (*Daphnia magna*)	50,733.79	24,905.01–103,349.38	1	0.66	0.86	0.65
Zebrafish-embryo	*Danio rerio-embryo*	Fathead minnow (*Pimephales promelas*)	141,587.99	99,973.45–200,524.84	1	0.93	0.2	0.93
Daphnid	*Daphnia galeata*	Daphnid (*Daphnia magna*)	26,901.53	3,928.12–184,233.58	2	0.96	0.08	0.91
Daphnid	*Daphnia longispina*	Daphnid (*Daphnia magna*)	88,105.03	8,932.55–86,9011.94	2	0.98	0.06	1.21
Daphnid	*Daphnia pulex*	Daphnid (*Daphnia magna*)	26,437.4	17,005.19–41,101.34	0	0.95	0.14	1.01
Daphnid	*Daphnia pulicaria*	Daphnid (*Daphnia magna*)	32,209.26	8,126.32–127,663.74	2	0.94	0.23	1.06
Pink shrimp	*Farfantepenaeus duorarum*	Mysid (*Americamysis bahia*)	27,565.26	4,058.51–187,222.10	2	0.81	0.61	0.98
Banana prawn	*Fenneropenaeus merguiensis*	Mysid (*Americamysis bahia*)	59,018.66	5,213.31–668,135.37	2	0.86	0.16	0.85
Mosquitofish	*Gambusia affinis*	Fathead minnow (*Pimephales promelas*)	74,514.83	11,436.16–485,517.91	1	0.98	0.12	0.94
Amphipod	*Gammarus fasciatus*	Daphnid (*Daphnia magna*)	14,243	5,729.92–35,404.14	1	0.75	0.75	0.81
Amphipod	*Gammarus fasciatus*	Mysid (*Americamysis bahia*)	37,876.26	5,563.20–257,875.19	2	0.69	0.81	0.91
Amphipod	*Gammarus minus*	Fathead minnow (*Pimephales promelas*)	170,929.6	35,314.27–827,340.47	1	0.95	0.04	0.72
Amphipod	*Gammarus pseudolimnaeus*	Daphnid (*Daphnia magna*)	25,033.78	6,325.86–99,067.93	1	0.74	0.75	0.91
Amphipod	*Gammarus pseudolimnaeus*	Mysid (*Americamysis bahia*)	7,052.85	1,501.74–33,123.26	1	0.8	0.55	0.82
Catla	*Gibelion catla*	Fathead minnow (*Pimephales promelas*)	176,204.76	12,309.61–2,522,266.29	2	0.96	0.02	1.09
Amphipod	*Hyalella azteca*	Mysid (*Americamysis bahia*)	27,383.57	7,841.48–95,627.27	1	0.88	0.68	1.03
Channel catfish	*Ictalurus punctatus*	Fathead minnow (*Pimephales promelas*)	117,869.02	69,164.02–200,871.86	1	0.87	0.3	0.97
Flagfish	*Jordanella floridae*	Fathead minnow (*Pimephales promelas*)	57,319.06	8,459.37–388,382.97	2	0.83	0.46	0.9
Neosho mucket	*Lampsilis rafinesqueana*	Daphnid (*Daphnia magna*)	19,227.49	6,346.49–58,252.11	1	0.99	0.02	0.89
Neosho mucket	*Lampsilis rafinesqueana*	Fathead minnow (*Pimephales promelas*)	73,773.62	6,962.15–781,733.67	2	0.98	0.05	1.58
Fatmucket	*Lampsilis siliquoidea*	Daphnid (*Daphnia magna*)	21,565.21	12,840.33–36,218.55	0	0.94	0.18	0.9
Fatmucket	*Lampsilis siliquoidea*	Fathead minnow (*Pimephales promelas*)	25,942.94	4,621.97–145,616.57	2	0.64	0.93	0.93
Green floater	*Lasmigona subviridis*	Daphnid (*Daphnia magna*)	9,271.86	1,031.84–83,314.43	1	0.96	0.07	0.66
Peppered loach	*Lepidocephalichthys guntea*	Fathead minnow (*Pimephales promelas*)	52,826.83	10,867.50–256,790.62	1	0.99	0	0.78
Bluegill	*Lepomis macrochirus*	Mysid (*Americamysis bahia*)	30,626.64	18,704.94–50,146.70	0	0.71	0.54	0.7
Bluegill	*Lepomis macrochirus*	Daphnid (*Daphnia magna*)	26,414.23	18,267.45–38,194.26	0	0.65	0.84	0.71
Bluegill	*Lepomis macrochirus*	Fathead minnow (*Pimephales promelas*)	61,145.18	38,558.83–96,961.80	0	0.81	0.43	0.93
Oligochaete	*Limnodrilus hoffmeisteri*	Daphnid (*Daphnia magna*)	96,032.58	26,854.58–343,414.65	1	0.8	0.35	0.7
Bullfrog	*Lithobates catesbeianus*	Fathead minnow (*Pimephales promelas*)	133,397.53	67,979.29–261,769.46	1	0.98	0.09	0.97
Oligochaete	*Lumbriculus variegatus*	Fathead minnow (*Pimephales promelas*)	126,573.44	29,982.93–534,331.88	1	0.81	0.43	0.94
Swamp lymnaea	*Lymnaea stagnalis*	Daphnid (*Daphnia magna*)	30,158.73	11,036.71–82,411.24	0	0.95	0.22	0.94

Common Name	Scientific Name	Surrogate	Estimated Toxicity (µg/L)	95% CI	CI Difference	R^2	MSE	Slope
Swamp lymnaea	*Lymnaea stagnalis*	Fathead minnow (*Pimephales promelas*)	46,749.32	4,781.25–457,097.31	2	0.84	0.59	1.2
Oriental river shrimp	*Macrobrachium nipponense*	Daphnid (*Daphnia magna*)	23,117.6	9,608.86–55,617.78	1	0.98	0.05	1.14
Western pearlshell	*Margaritifera falcata*	Daphnid (*Daphnia magna*)	18,278.06	9,312.83–35,873.88	1	0.95	0.15	0.87
Washboard	*Megalonaias nervosa*	Daphnid (*Daphnia magna*)	13,795.85	7,896.45–24,102.66	1	0.97	0.12	0.96
Washboard	*Megalonaias nervosa*	Fathead minnow (*Pimephales promelas*)	15,796.25	1,649.01–151,315.74	2	0.73	0.88	1.14
Atlantic silverside	*Menidia menidia*	Mysid (*Americamysis bahia*)	50,284.41	4,290.87–589,278.75	2	0.81	0.41	0.83
Water flea	*Moina macrocopa*	Daphnid (*Daphnia magna*)	23,142.82	5,385.97–99,441.67	1	0.96	0.1	0.72
Cutthroat trout	*Oncorhynchus clarkii*	Fathead minnow (*Pimephales promelas*)	42,565.86	13,136.40–137,926.09	1	0.82	0.34	0.99
Coho salmon	*Oncorhynchus kisutch*	Fathead minnow (*Pimephales promelas*)	28,013.92	4,212.83–186,282.97	2	0.64	0.69	0.91
Rainbow trout	*Oncorhynchus mykiss*	Mysid (*Americamysis bahia*)	18,260.89	11,533.79–28,911.58	0	0.62	0.71	0.66
Rainbow trout	*Oncorhynchus mykiss*	Daphnid (*Daphnia magna*)	18,581.73	13,754.04–25,103.96	0	0.62	0.84	0.71
Rainbow trout	*Oncorhynchus mykiss*	Fathead minnow (*Pimephales promelas*)	58790.82	43,160.06–80,082.39	0	0.87	0.28	0.97
Chinook salmon	*Oncorhynchus tshawytscha*	Fathead minnow (*Pimephales promelas*)	54,013.06	3,744.85–779,045.20	2	0.7	0.82	1.16
Mozambique tilapia	*Oreochromis mossambicus*	Fathead minnow (*Pimephales promelas*)	61,006.21	8,663.77–429,576.81	2	0.72	0.33	0.86
Pheasantshell	*Ortmanniana pectorosa*	Daphnid (*Daphnia magna*)	22,890.49	6518.52–80382.37	1	0.97	0.11	0.96
Pheasantshell	*Ortmanniana pectorosa*	Fathead minnow (*Pimephales promelas*)	56,330.86	4324.40–733780.57	2	0.97	0.07	1.58
Medaka	*Oryzias latipes*	Fathead minnow (*Pimephales promelas*)	101,502.83	60,666.22–169,828.03	1	0.98	0.04	0.85
Mississippi grass shrimp	*Palaemonetes kadiakensis*	Daphnid (*Daphnia magna*)	8,016.94	987.60–65,078.25	2	0.63	0.71	0.75
Daggerblade grass shrimp	*Palaemonetes pugio*	Mysid (*Americamysis bahia*)	114,432.85	36,613.31–357,653.39	1	0.92	0.25	1.09
Midge	*Paratanytarsus dissimilis*	Fathead minnow (*Pimephales promelas*)	241,082.19	51,857.97–1,120,765.44	2	0.84	0.52	0.86
Midge	*Paratanytarsus parthenogeneticus*	Daphnid (*Daphnia magna*)	94,804.83	42,906.77–209,476.37	1	0.98	0.05	0.97
Midge	*Paratanytarsus parthenogeneticus*	Fathead minnow (*Pimephales promelas*)	426,080.11	86,700.59–2,093,921.81	2	0.98	0.09	0.98
Bryozoan	*Pectinatella magnifica*	Daphnid (*Daphnia magna*)	119,205.26	11,569.76–1,228,191.76	2	0.98	0	0.86
Yellow perch	*Perca flavescens*	Fathead minnow (*Pimephales promelas*)	26,687.03	2,944.32–241,888.23	2	0.73	0.54	0.93
Tadpole physa	*Physella gyrina*	Daphnid (*Daphnia magna*)	29,679.01	13,779.45–63,924.42	0	0.95	0.19	0.96
Tadpole physa	*Physella gyrina*	Fathead minnow (*Pimephales promelas*)	46,602.19	6,887.26–315,330.51	2	0.84	0.58	1.22
Guppy	*Poecilia reticulata*	Fathead minnow (*Pimephales promelas*)	51,329.66	20,397.35–129,170.40	1	0.78	0.38	0.84
Water flea	*Pseudosida ramosa*	Daphnid (*Daphnia magna*)	13,607.58	1,697.67–109,070.47	2	0.86	0.56	0.88
Brook trout	*Salvelinus fontinalis*	Fathead minnow (*Pimephales promelas*)	28,506.65	5,024.76–161,724.97	2	0.7	0.74	0.95
Lake trout	*Salvelinus namaycush*	Fathead minnow (*Pimephales promelas*)	9,735.43	1,913.91–49,520.95	1	0.65	0.33	0.76
Daphnid	*Simocephalus serrulatus*	Daphnid (*Daphnia magna*)	170,66.73	4,180.09–69,681.06	1	0.85	0.28	0.93

Common Name	Scientific Name	Surrogate	Estimated Toxicity (µg/L)	95% CI	CI Difference	R^2	MSE	Slope
Beaver-tail fairy shrimp	*Thamnocephalus platyurus*	Daphnid (*Daphnia magna*)	22,887.35	14,699.82–35,635.19	0	0.97	0.12	0.91
Copepod	*Tigriopus japonicus*	Fathead minnow (*Pimephales promelas*)	177,050.22	16,909.45–1,853,802.51	2	0.72	0.13	0.77
Oligochaete	*Tubifex tubifex*	Daphnid (*Daphnia magna*)	263,232.51	38,592.88–1,795,443.63	2	0.79	0.77	0.89
Oligochaete	*Tubifex tubifex*	Fathead minnow (*Pimephales promelas*)	164,095.02	29,492.22–913,026.17	1	0.86	0.52	1.03
Paper pondshell	*Utterbackia imbecillis*	Daphnid (*Daphnia magna*)	20,135.8	13,253.48–30,592.00	0	0.97	0.08	0.88
Paper pondshell	*Utterbackia imbecillis*	Fathead minnow (*Pimephales promelas*)	18,462.17	2,423.45–140,647.31	2	0.65	0.85	0.86
African clawed frog	*Xenopus laevis*	Fathead minnow (*Pimephales promelas*)	46,684.93	10,190.13–213,881.76	1	0.91	0.14	0.76

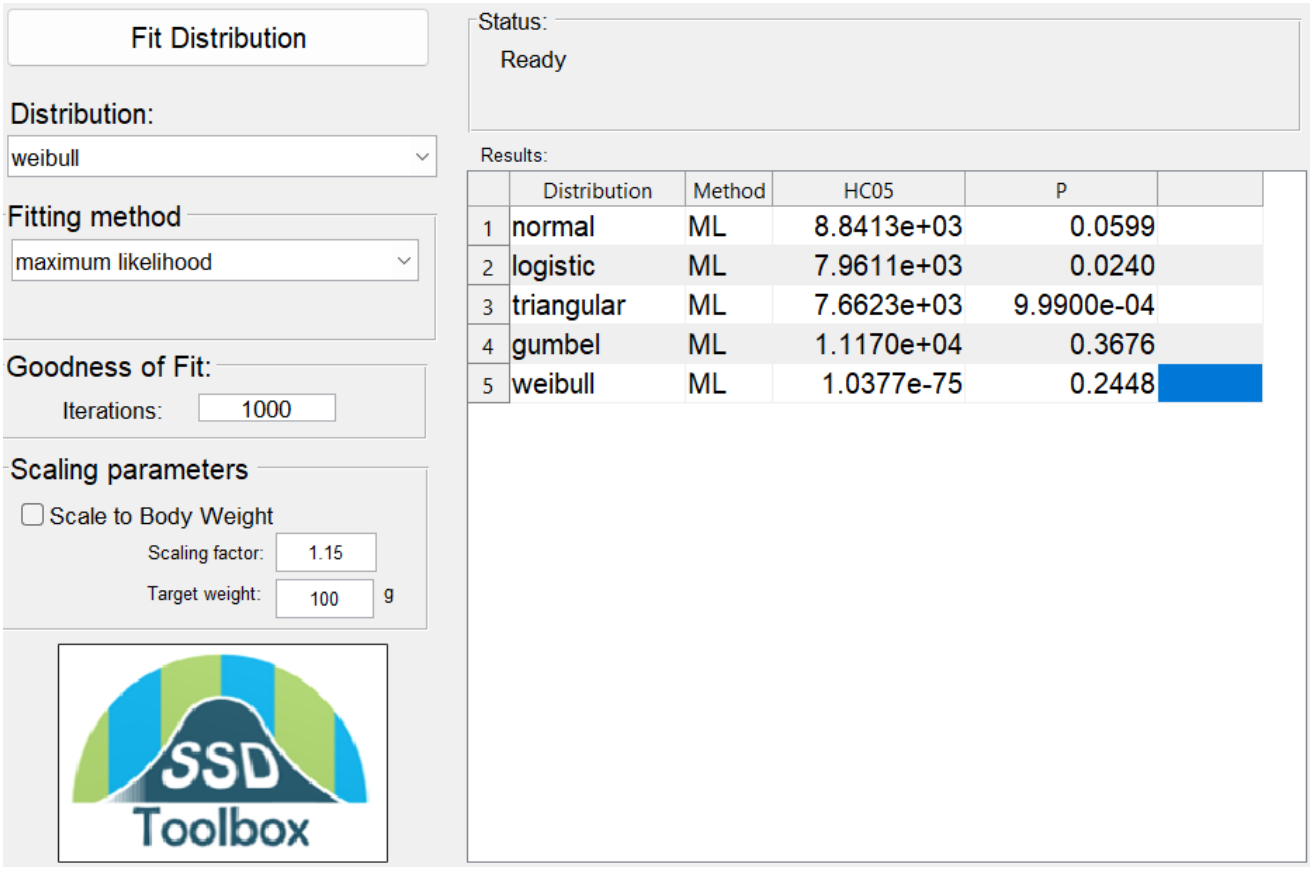
K.2.1.2 Species Sensitivity Distribution (SSD)

The SSD Toolbox is a resource created by EPA's Office of Research and Development (ORD) that can fit SSDs to environmental hazard data (Etterson, 2020a). The SSD Toolbox runs on Matlab 2018b (9.5) for Windows 64 bit. For the 1,1-dichloroethane Risk Evaluation, EPA calculated an SSD with the SSD Toolbox using acute LC50 hazard data for 1,1-dichloroethane and 1,2-dichloropropane from systematic review, and estimated data from the Web-ICE application (Appendix K.2.1.1) that included 25 fish, 2 amphibians, and 48 invertebrate species. The SSD is used to calculate a hazardous concentration for 5 percent of species (HC05). In other words, HC05 estimates the concentration that is expected to be protective for 95 percent of species.

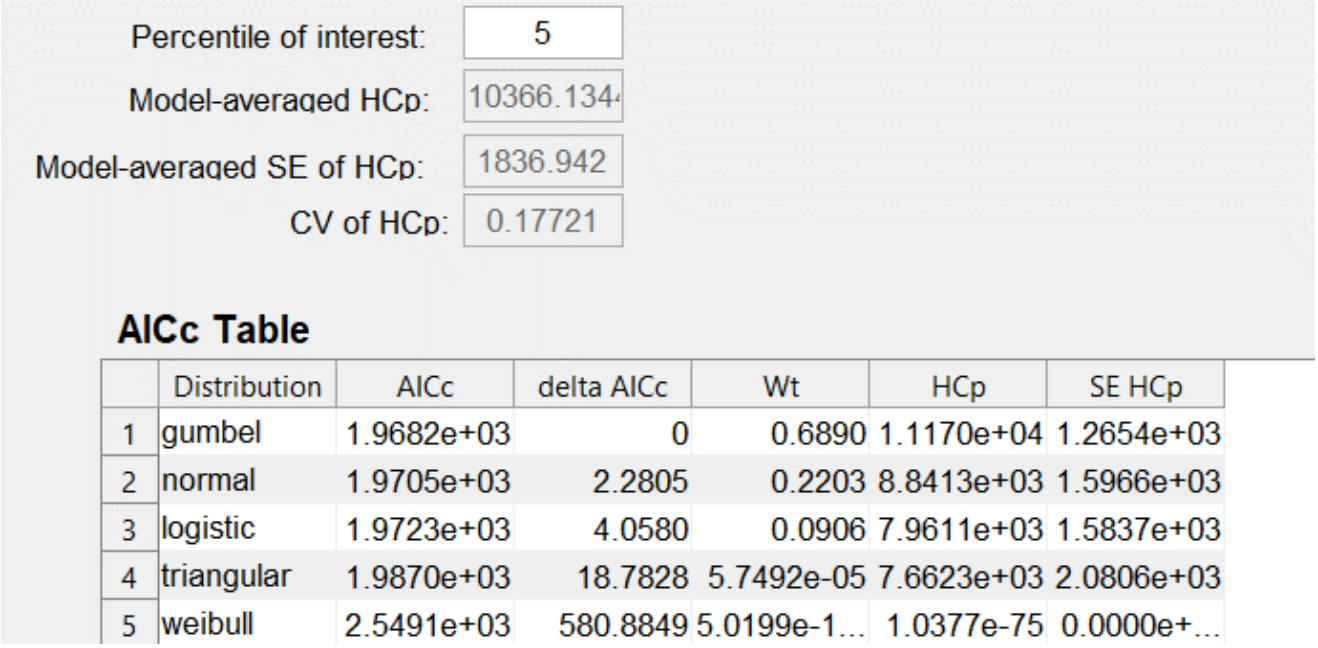
The SSD Toolbox contains functions for fitting up to six distributions (normal, logistic, triangular, Gumbel, Weibull, and Burr) across four model estimation methods (maximum likelihood, moment estimators, graphical methods, and Bayesian methods, in this case the Metropolis-Hastings algorithm). Maximum likelihood was used to model the data for 1,1-dichloroethane due to its general acceptance for fitting SSDs (Etterson, 2020b), its low sampling variance, and the fact that models can also be compared *a posteriori* using information theoretic methods, in this case Akaike's Information Criterion corrected for sample size (AICc). AICc was used along with a comparison of p-values and a visual assessment of Q-Q plots, which are methods available to all model estimation methods, to select the distribution used to calculate the HC05 for this analysis. Based on the guidance documents for use of the SSD Toolbox (Etterson, 2020b), the Burr distribution is provided only for comparison and is not used for modeling, so it was not included.

SSD Toolbox uses a parametric bootstrap method to calculate a p-value to compare goodness-of-fit across distributions. In this type of test, p-values greater than 0.05 are required (Etterson, 2020b). The normal, ($p = 0.06$), Gumbel ($p = 0.37$), and Weibull ($p = 0.24$) distributions all passed this initial screening (Figure_Apx K-1). The sample-size corrected AICc was lowest for the Gumbel distribution (Figure_Apx K-2). Because numerical methods may lack statistical power for small sample sizes, a visual inspection of the data was also used to assess goodness-of-fit, in this case a comparison of Q-Q plots between the three distributions. In a Q-Q plot, the horizontal axis gives the empirical quantiles, and the vertical axis gives the predicted quantiles (from the fitted distribution). A good model fit shows the data points in close proximity to the diagonal line across the data distribution. Comparison of Q-Q plots between the three distributions identified the Gumbel distribution as the best distribution. After examining the Q-Q plots for both distributions, the Gumbel distribution was chosen because the Q-Q plot has a better fit, especially in the lower left quadrant, near where the HC05 is determined (Figure_Apx K-3).

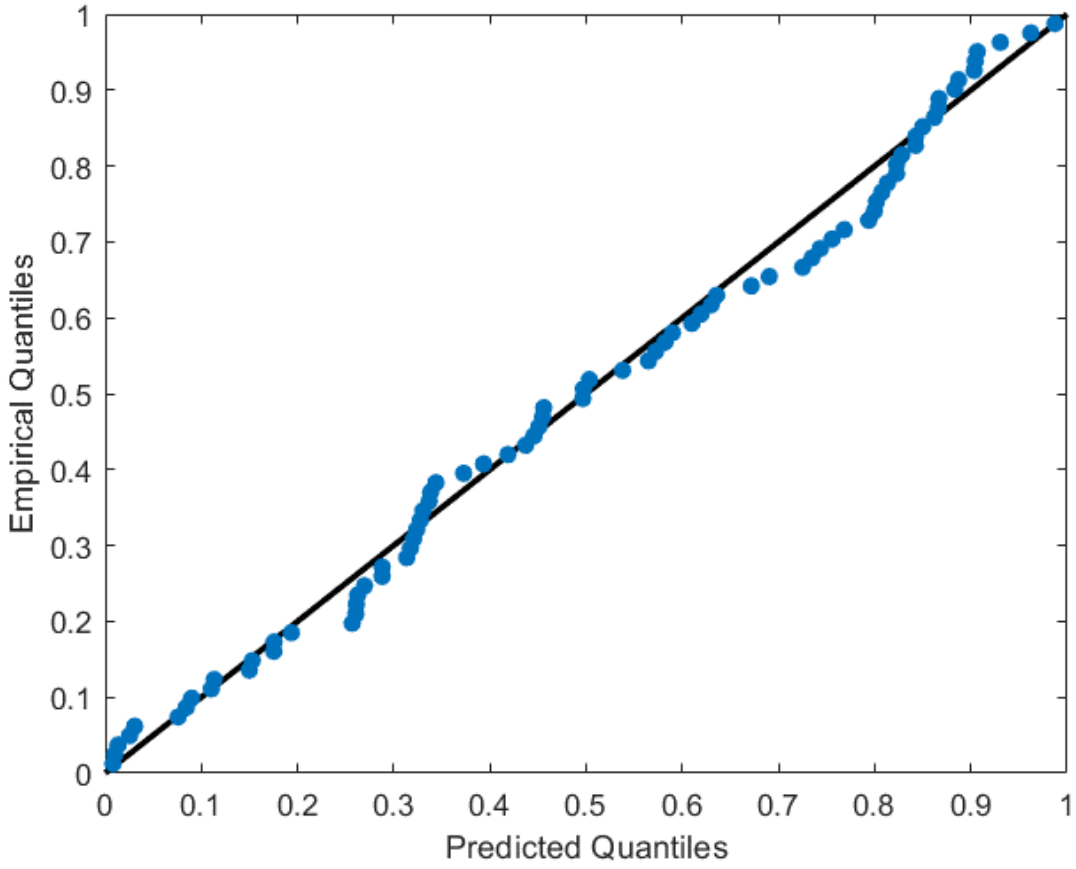
This distribution was then plotted along with data points for both measured and modeled species. Life history information was attached to each species, indicating an even distribution of various life history strategies along the curve (Figure_Apx K-4). The calculated HC05 was 11,170 µg/L (95% CI = 8,931–14,370 µg/L). The lower 95 percent CI of the HC05 (8,931 µg/L) was then used as the acute aquatic COC.



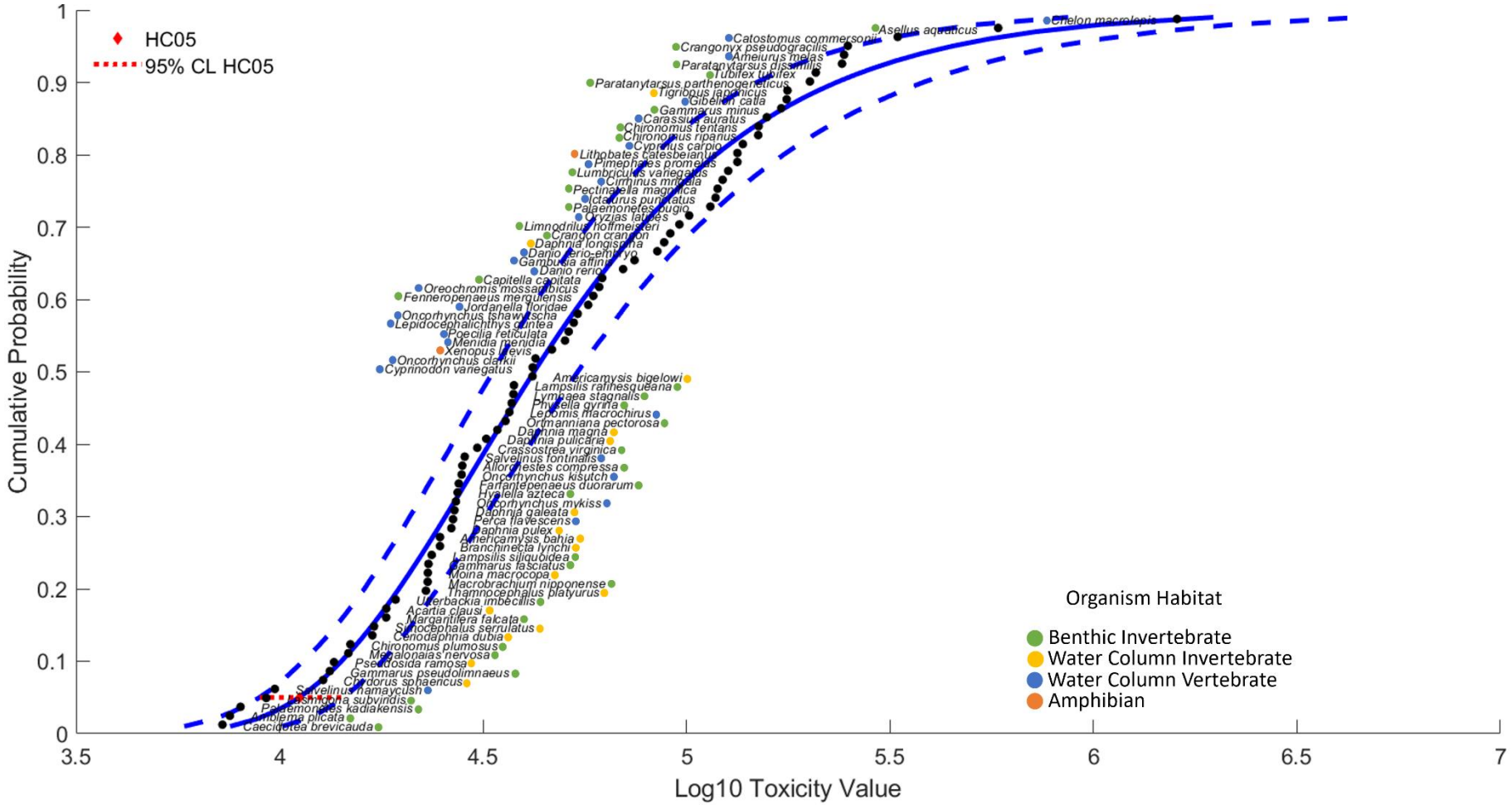
Figure_Apx K-1. SSD Toolbox Interface Showing HC05s and P Values for Each Distribution Using Maximum Likelihood Fitting Method Using 1,1-Dichloroethane's and 1,2-Dichloropropane's Acute Aquatic Hazard Data (Etterson, 2020a)



Figure_Apx K-2. AICc for the Five Distribution Options in the SSD Toolbox for 1,1-Dichloroethane and 1,2-Dichloropropane Acute Aquatic Hazard Data (Etterson, 2020a)



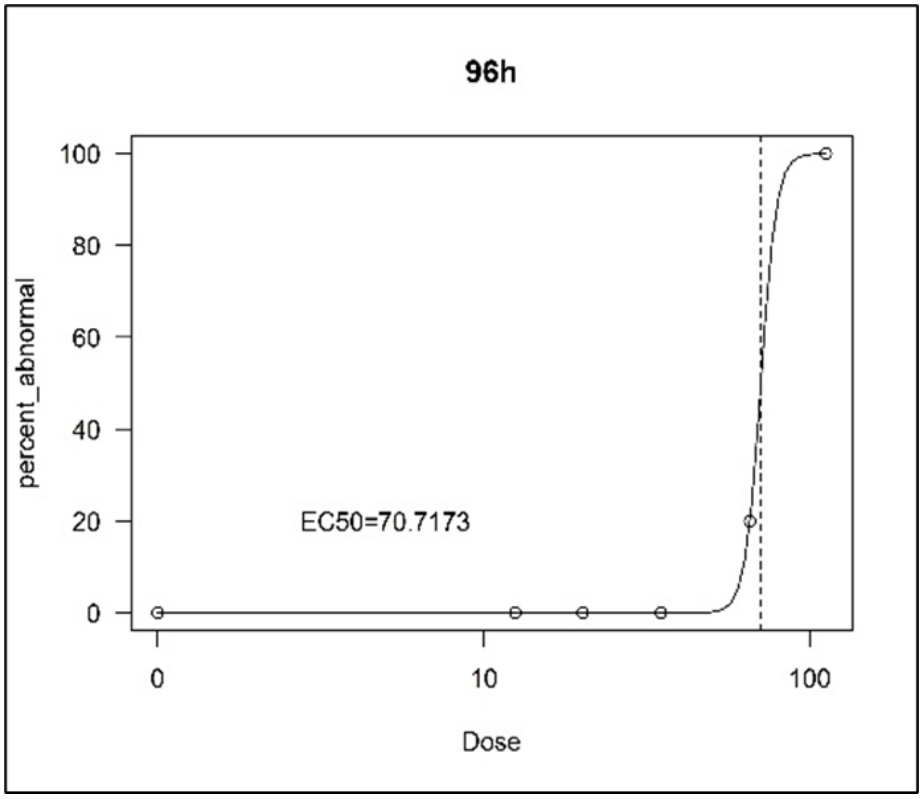
Figure_Apx K-3. Q-Q plot of 1,1-Dichloroethane and 1,2-Dichloropropane Acute Aquatic Hazard Data with the Logistic Distribution (Etterson, 2020a)



Figure_Apx K-4. SSD Distribution for 1,1-Dichloroethane and 1,2-Dichloropropane Acute Hazard Data in µg/L (Etterson, 2020a)

K.2.1.3 Dose-Response Curve Fit Methods

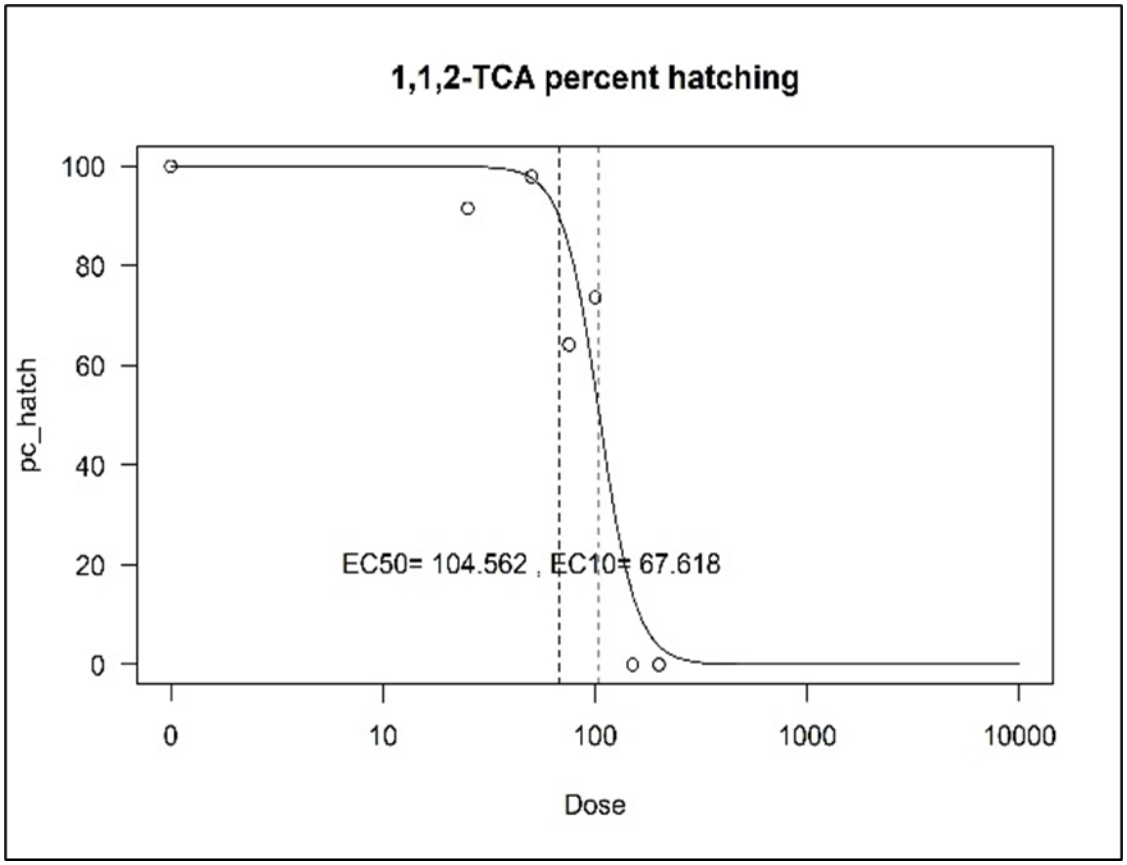
Swimming behavior data for *Oryzias latipes* exposed to 1,1-dichloroethane were further analyzed to derive an EC_{50} value by fitting a dose-response curve. The authors of the original dose-response study (Mitsubishi Chemical Medience Corporation, 2009) recorded number of fish out of 10 fish per treatment concentration with abnormal swimming behavior at 96-hours. For this EC50 derivation, data were first censored for mortalities, then the response was expressed as percent abnormal at each concentration. The control group had zero abnormal swimmers, so there was no need to standardize the response as a percent of control. Preliminary analyses indicated this relationship was well characterized using a log-logistic curve in R v.4.2.1 (R Core Team, 2022; Ritz et al., 2015) with slope and inflection point as the estimated parameters. The lower asymptote was fixed to 0 percent and the upper asymptote to 100 percent to constrain the predicted y value to a realistic range. The inflection point estimated by the curve fit (*i.e.*, the point along the curve halfway between the upper and lower asymptotes) was used to estimate the EC50. Figure_Apx K-5 shows the log-logistic curve for the 96h time point, with a vertical dotted line indicating the EC50.



Figure_Apx K-5. Log-Logistic Curve Fit to 96-Hour Abnormal Swimming Behavior Data from (Mitsubishi Chemical Medience Corporation, 2009) for *Oryzias latipes* Exposed to 1,1-Dichloroethane

The hatching rate endpoint for *Ophryotrocha labronica* exposed to 1,1,2-trichloroethane was further analyzed to derive EC50 and EC10 values by fitting a dose-response curve. The authors of the original dose-response study (Rosenberg et al., 1975) reported for each concentration of 1,1,2-trichloroethane the hatching percent of *O. labronica* eggs. The hatching rate endpoint is expressed as percent relative to control response. Hormetic observations (*i.e.*, treatments having a response exceeding that of the control) were not censored. Characterizing EC50 and EC10 values required defining the 0 percent effect

and 100 percent effect. Estimated between these two thresholds are the EC50, or the 50 percent inhibition of egg hatching, and EC10, 10 percent inhibition of egg hatching. Responses plateaued as concentration increased. Because zero was the minimum possible realistic value, the 100 percent effect (*i.e.*, lower asymptote) was set at zero. The 0 percent effect was defined as the control response; therefore, the upper asymptote was fixed at 100 percent of the control response. Hatching percent followed a decreasing logistic shape. Several functions were tested using R v. 4.2.1, with and without upper and lower asymptotes (R Core Team, 2022; Ritz et al., 2015). A log-logistic curve was ultimately fit to the data with slope and inflection point as the estimated parameters. The EC50 was calculated as the concentration along the curve halfway between 0 and 100 percent control response and the EC10 as the concentration a tenth of the way along the curve. Figure_Apx K-6 shows the log-logistic curve, with vertical dotted lines indicating the EC50 and EC10.



Figure_Apx K-6. Log-logistic Curve Fit to Hatching Percent Data from *Ophryotrocha labronica* Exposed to 1,1,2-Trichloroethane (Rosenberg et al., 1975).

K.2.2 Terrestrial Hazard Data

For mammalian species, EPA estimates hazard by calculating a TRV. The TRV is expressed as doses in units of mg/kg-bw/day. Data from laboratory rat and mouse studies can be used to evaluate chronic dietary exposure in ecologically relevant wildlife species because of this normalization to body weight. For calculation of the mammal TRV, an *a priori* framework for selection of the TRV value based on the results of the no-observed-adverse-effect level (NOAEL) and lowest-observed-adverse-effect level (LOAEL) data (Figure_Apx K-7) was used. The minimum data set required to calculate a TRV consists of three results with NOAEL or LOAEL values for reproduction, growth, or mortality for at least two species. If these minimum results are not available, then a TRV is not calculated.

For mammalian species, EPA estimates hazard by calculating a TRV. The TRV is expressed as doses in units of mg/kg-bw/day. Although the TRV for 1,1-dichloroethane is derived from laboratory mice and rat studies, body weight is normalized, therefore the TRV can be used with ecologically relevant wildlife species to evaluate chronic dietary exposure to 1,1-dichloroethane. Representative wildlife species chronic hazard threshold will be evaluated in the trophic transfer assessments using the TRV. The flow chart in Figure_Apx K-7 was used to select the data to calculate the TRV with NOAEL and/or LOAEL data (U.S. EPA, 2007). The movement through the flowchart used to calculate the TRV for 1,1-dichloroethane is described below and illustrated in Figure 4-2.

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

Yes, 8 results across 2 species (rats and mice) were identified as suitable for use. Endpoints included 10-day, 6-week, 13-week, and 78-week NOAEL/LOAELs in both male and female organisms. These results are summarized in Table 4-8.

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

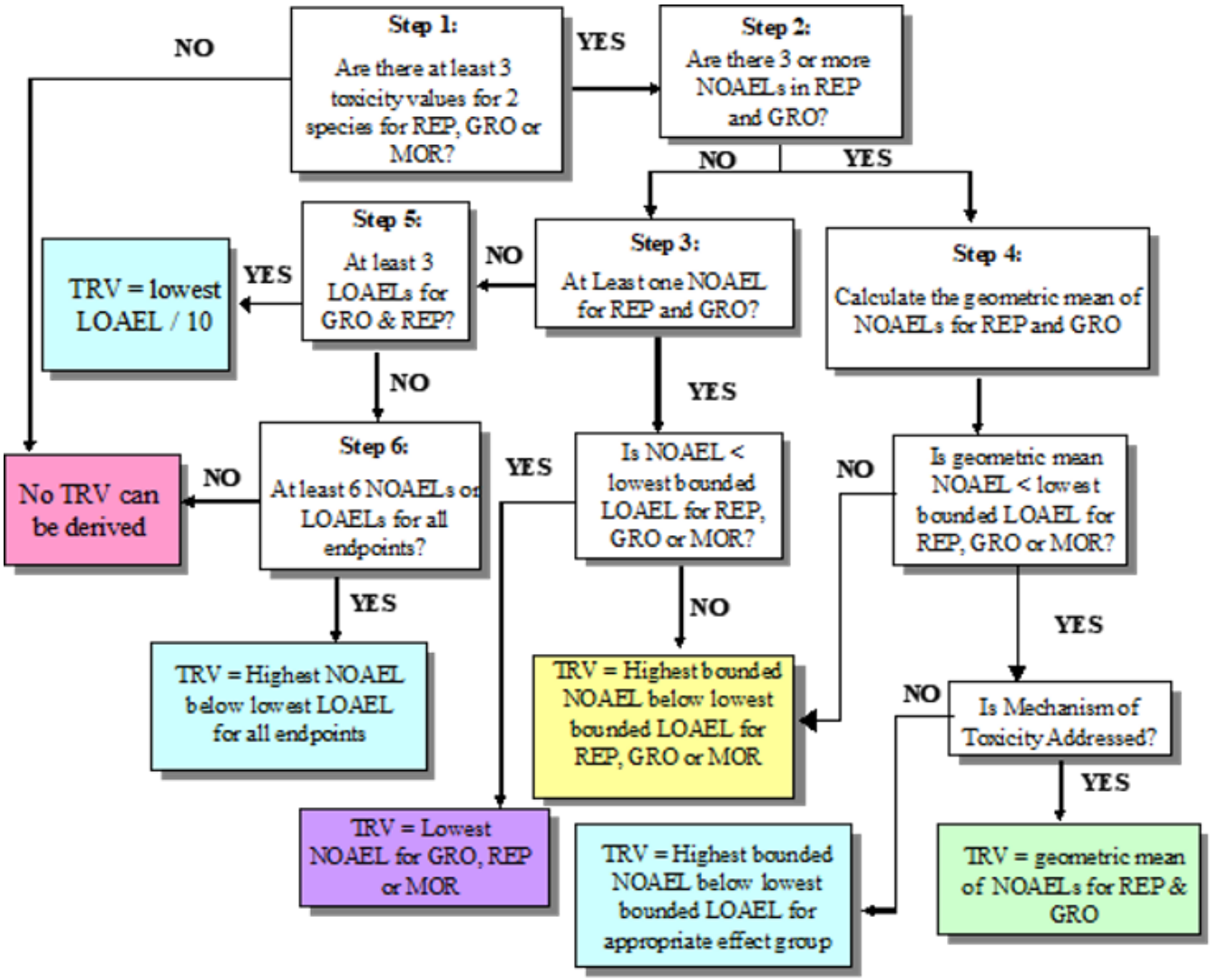
Yes, four of the above-referenced results report a NOAEL in the reproduction or growth effect groups.

Move from Step 2 to Step 4: Calculate a geometric mean of the NOAELs for reproduction and growth. Is this number lower than the lowest bounded LOAEL for reproduction, growth, and mortality?

The geometric mean of the NOAELs for reproduction and growth is 2,236 mg/kg-bw/day. This is greater than 1,429 mg/kg-bw/day, which is the lowest bounded LOAEL for reproduction, growth, and mortality.

TRV = Highest bounded NOAEL below lowest bounded LOAEL for reproduction, growth, and mortality.

The mammalian wildlife TRV for 1,1-dichloroethane is 1,189 mg/kg-bw/day.



Figure_Apx K-7. TRV Flow Chart

K.2.3 Evidence Integration

Data integration includes analysis, synthesis, and integration of information for the risk evaluation. During data integration, EPA considers quality, consistency, relevancy, coherence, and biological plausibility to make final conclusions regarding the weight of scientific evidence. As stated in the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021c), data integration involves transparently discussing the significant issues, strengths, and limitations as well as the uncertainties of the reasonably available information and the major points of interpretation. The general analytical approaches for integrating evidence for environmental hazard is discussed in Section 7.4 of the Draft Protocol.

The organization and approach to integrating hazard evidence is determined by the reasonably available evidence regarding routes of exposure, exposure media, duration of exposure, taxa, metabolism and distribution, effects evaluated, the number of studies pertaining to each effect, as well as the results of the data quality evaluation.

The environmental hazard integration is organized around effects to aquatic and terrestrial organisms as well as the respective environmental compartments (*e.g.*, pelagic, benthic, soil). Environmental hazard

assessment may be complex based on the considerations of the quantity, relevance, and quality of the available evidence.

For 1,1-dichloroethane, environmental hazard data from toxicology studies identified during systematic review have used evidence that characterizes apical endpoints (*i.e.*, endpoints that could have population level effects such as reproduction, growth, and/or mortality). Additionally, mechanistic data that can be linked to apical endpoints will add to the weight of scientific evidence supporting hazard thresholds. EPA also considered predictions from Web-ICE to supplement the empirical data found during systematic review.

K.2.3.1 Weight of Scientific Evidence

After calculating the hazard thresholds that were carried forward to characterize risk, a narrative describing the weight of scientific evidence and uncertainties was completed to support EPA's decisions. The weight of scientific evidence fundamentally means that the evidence is weighed (*i.e.*, ranked), and weighted (*i.e.*, a piece or set of evidence or uncertainty may have more importance or influence in the result than another). Based on the weight of scientific evidence and uncertainties, a confidence statement was developed that qualitatively ranks (*i.e.*, Robust, Moderate, Slight, or Indeterminate) the confidence in the hazard threshold. The qualitative confidence levels are described below and illustrated in Table_Apx K-2.

The evidence considerations and criteria detailed within (U.S. EPA, 2021c) guide the application of strength-of-evidence judgments for environmental hazard effect within a given evidence stream and were adapted from Table 7-10 of the 2021 Draft Systematic Review Protocol (U.S. EPA, 2021c),

EPA used the strength-of-evidence and uncertainties from (U.S. EPA, 2021c) for the hazard assessment to qualitatively rank the overall confidence using evidence for environmental hazard. Confidence levels of Robust, Moderate, Slight, or Indeterminant are assigned for each evidence property that corresponds to the evidence considerations (U.S. EPA, 2021c). The rank of the *Quality of the Database* consideration is based on the systematic review data quality rank (High, Medium, or Low) for studies used to calculate the hazard threshold, and whether there are data gaps in the toxicity dataset. Another consideration in the *Quality of the Database* is the risk of bias (*i.e.*, how representative is the study to ecologically relevant endpoints). Additionally, because of the importance of the studies used for deriving hazard thresholds, the *Quality of the Database* consideration may have greater weight than the other individual considerations. The High, Medium, and Low systematic review ranks correspond to the evidence table ranks of Robust, Moderate, or Slight, respectively. The evidence considerations are weighted based on professional judgement to obtain the *Overall Confidence* for each hazard threshold. In other words, the weights of each evidence property relative to the other properties are dependent on the specifics of the weight of scientific evidence and uncertainties that are described in the narrative and may or may not be equal. Therefore, the overall score is not necessarily a mean or defaulted to the lowest score. The confidence levels and uncertainty type examples are described below.

Confidence Levels

- Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the exposure or hazard estimate.
- Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize exposure or hazard estimates.

- Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.
- Indeterminant corresponds to entries in evidence tables where information is not available within a specific evidence consideration.

Types of Uncertainties

The following uncertainties may be relevant to one or more of the weight of scientific evidence considerations listed above and will be integrated into that property's rank in the evidence table (Table_Apx K-2).

- Scenario uncertainty: Uncertainty regarding missing or incomplete information needed to fully define the exposure and dose.
    - The sources of scenario uncertainty include descriptive errors, aggregation errors, errors in professional judgment, and incomplete analysis.
- Parameter uncertainty: Uncertainty regarding some parameter.
    - Sources of parameter uncertainty include measurement errors, sampling errors, variability, and use of generic or surrogate data.
- Model uncertainty: Uncertainty regarding gaps in scientific theory required to make predictions on the basis of causal inferences.
    - Modeling assumptions may be simplified representations of reality.

Table_Apx K-2 summarizes the weight of scientific evidence and uncertainties, while increasing transparency on how EPA arrived at the overall confidence level for each exposure hazard threshold. Symbols are used to provide a visual overview of the confidence in the body of evidence, while de-emphasizing an individual ranking that may give the impression that ranks are cumulative (*e.g.*, ranks of different categories may have different weights).

Table_Apx K-2. Considerations that Inform Evaluations of the Strength of the Evidence Within an Evidence Stream (*i.e.*, Apical Endpoints, Mechanistic, or Field Studies)

Consideration	Increased Evidence Strength (of the Apical Endpoints, Mechanistic, or Field Studies Evidence)	Decreased Evidence Strength (of the Apical Endpoints, Mechanistic, or Field Studies Evidence)
The evidence considerations and criteria laid out here guide the application of strength-of-evidence judgments for an outcome or environmental hazard effect within a given evidence stream. Evidence integration or synthesis results that do not warrant an increase or decrease in evidence strength for a given consideration are considered "neutral" and are not described in this table (and, in general, are captured in the assessment-specific evidence profile tables).		
Quality of the Database* (risk of bias)	• A large evidence base of *high*- or *medium*-quality studies increases strength. • Strength increases if relevant species are represented in a database.	• An evidence base of mostly *low*-quality studies decreases strength. • Strength also decreases if the database has data gaps for relevant species, *i.e.*, a trophic level that is not represented. • Decisions to increase strength for other considerations in this table should generally not be made if there are serious concerns for risk of bias; in other words, all the other considerations in this table are dependent upon the quality of the database.[a]
Consistency	Similarity of findings for a given outcome (*e.g.*, of a similar magnitude, direction) across independent studies or experiments increases strength, particularly when consistency is observed across species, life stage, sex, wildlife populations, and across or within aquatic and terrestrial exposure pathways.	• Unexplained inconsistency (*i.e.*, conflicting evidence; see (U.S. EPA, 2005b) decreases strength. • Strength should not be decreased if discrepant findings can be reasonably explained by study confidence conclusions; variation in population or species, sex, or life stage; frequency of exposure (*e.g.*, intermittent or continuous); exposure levels (low or high); or exposure duration.
Strength (effect magnitude) and precision	• Evidence of a large magnitude effect (considered either within or across studies) can increase strength. • Effects of a concerning rarity or severity can also increase strength, even if they are of a small magnitude. • Precise results from individual studies or across the set of studies increases strength, noting that biological significance is prioritized over statistical significance. • Use of probabilistic model (*e.g.*, Web-ICE, SSD) may increase strength.	Strength may be decreased if effect sizes that are small in magnitude are concluded not to be biologically significant, or if there are only a few studies with imprecise results.
Biological gradient/dose-response	• Evidence of dose-response increases strength. • Dose-response may be demonstrated across studies or within studies and it can be dose- or duration-dependent. • Dose response may not be a monotonic dose-response (monotonicity should not necessarily be expected, *e.g.*, different outcomes may be expected at low vs. high doses due to activation	• A lack of dose-response when expected based on biological understanding and having a wide range of doses/exposures evaluated in the evidence base can decrease strength. • In experimental studies, strength may be decreased when effects resolve under certain experimental conditions (*e.g.*, rapid reversibility after removal of exposure).

Consideration	Increased Evidence Strength (of the Apical Endpoints, Mechanistic, or Field Studies Evidence)	Decreased Evidence Strength (of the Apical Endpoints, Mechanistic, or Field Studies Evidence)
	of different mechanistic pathways or induction of systemic toxicity at very high doses). • Decreases in a response after cessation of exposure (*e.g.*, return to baseline fecundity) also may increase strength by increasing certainty in a relationship between exposure and outcome (this particularly applicable to field studies).	• However, many reversible effects are of high concern. Deciding between these situations is informed by factors such as the toxicokinetics of the chemical and the conditions of exposure, see (U.S. EPA, 1998), endpoint severity, judgments regarding the potential for delayed or secondary effects, as well as the exposure context focus of the assessment (*e.g.*, addressing intermittent or short-term exposures). • In rare cases, and typically only in toxicology studies, the magnitude of effects at a given exposure level might decrease with longer exposures (*e.g.*, due to tolerance or acclimation). • Like the discussion of reversibility above, a decision about whether this decreases evidence strength depends on the exposure context focus of the assessment and other factors. • If the data are not adequate to evaluate a dose-response pattern, then strength is neither increased nor decreased.
Biological relevance	Effects observed in different populations or representative species suggesting that the effect is likely relevant to the population or representative species of interest (*e.g.*, correspondence among the taxa, life stages, and processes measured or observed and the assessment endpoint).	An effect observed only in a specific population or species without a clear analogy to the population or representative species of interest decreases strength.
Physical/chemical relevance	Correspondence between the substance tested and the substance constituting the stressor of concern.	The substance tested is an analog of the chemical of interest or a mixture of chemicals which include other chemicals besides the chemical of interest.
Environmental relevance	Correspondence between test conditions and conditions in the region of concern.	The test is conducted using conditions that would not occur in the environment.

[a] Database refers to the entire dataset of studies integrated in the environmental hazard assessment and used to inform the strength of the evidence. In this context, database does *not* refer to a computer database that stores aggregations of data records such as the ECOTOX Knowledgebase.

Appendix L ENVIRONMENTAL RISK DETAILS

L.1 Risk Characterization for Aquatic Receptors

Details described in Section 4.3.2.

L.2 Risk Characterization for Terrestrial Receptors

RQs were less than 1 for the five COUs quantitatively assessed for air deposition to soil from TRI-reported fugitive emissions of 1,1-dichloroethane when using the highest AERMOD predictions for daily air deposition to soil at 10 m from facility. EPA expects risk estimates for air deposition to soil from NEI and environmental release modeled stack and/or fugitive emissions to be comparable or less than those developed based on TRI fugitive emissions, therefore, two additional COU/OESs (repackaging of 1,1-dichloroethane and commercial use of 1,1-dichloroethane as a laboratory chemical) were assessed qualitatively for risk to terrestrial organisms. Table_Apx L-1 presents soil pore water concentrations and RQ values for daily air deposition to soil pore water, indicating RQs below 1 for terrestrial plants. The highest 1,1-dichloroethane soil pore water concentration calculated using AERMOD predictions at 10 m from facility is 146 µg/L based on the COU/OES manufacturing 1,1-dichloroethane. EPA expects that the RQs for terrestrial plants exposed to air deposition to soil from NEI-reported fugitive and/or stack emissions of 1,1-dichloroethane (8 COUs) or environmental release-modeled (Monte-Carlo simulated) fugitive and/or stack emissions of 1,1-dichloroethane (2 COUs) would be similar or less than the RQ values for air deposition to soil from TRI-reported fugitive emissions of 1,1-dichloroethane (with the highest RQ value for terrestrial plants = 1.8×10^{-4} based on manufacturing 1,1-dichloroethane). This is because the modeled 1,1-dichloroethane air concentrations at 10 m from releasing facilities resulting from NEI-reported or Monte-Carlo simulated fugitive and stack emissions (Table 3-13 and Table_Apx L-1, respectively) are less than or comparable to modeled 1,1-dichloroethane air concentrations at 10 m from releasing facilities resulting from TRI-reported fugitive emissions of 1,1-dichloroethane (Table 3-9). Therefore, estimates of risk associated with air deposition to soil from NEI-reported or environmental release-modeled (Monte-Carlo simulated) fugitive and/or stack emissions of 1,1-dichloroethane are assessed qualitatively in Table_Apx L-1.

In the case of commercial use of 1,1-dichloroethane as a laboratory chemical, the modeled air concentration at 10 m from releasing facility included both fugitive and stack emissions in the environmental release-model (Monte-Carlo simulation) and could not be attributed to one emission type. However, this modeled air concentration (1.5 mg/m^3) is two orders of magnitude less than the maximum air concentration of 230 mg/m^3 modeled from TRI-reported fugitive emissions from manufacturing 1,1-dichloroethane, the COU/OES with the highest modeled air concentration at 10 m from releasing facility (RQ for terrestrial plants = 1.8×10^{-4} from 1,1-dichloroethane air deposition to soil).

RQs were less than 1 for the disposal COU when using the highest predictions for biosolids land application to tilled agricultural and pastureland soils. Table_Apx L-2 presents soil pore water concentrations and RQ values for waste handling, treatment, and disposal of 1,1-dichloroethane at POTWs, indicating RQs below 1 for terrestrial plants.

Table_Apx L-1. Calculated RQs For Terrestrial Plants Based on Modeled Air Deposition of 1,1-Dichloroethane to Soil from Reported or Modeled Fugitive Emissions

COU (Life Cycle Stage/Category/Subcategory)	OES	Source	Number of Facilities [a]	Soil Pore Water Concentration (µg/L) at 10 m [b]	Hazard Threshold (mg/L) [c]	RQ
Manufacture/Domestic manufacturing/ Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	TRI	0/9	1.50E02	8.00E05	1.8E−04
		NEI	0/9	Assessed qualitatively due to modeled air concentrations < those based on TRI data		
Processing/As a reactant/Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	TRI	0/6	3.2	8.00E05	4.0E−06
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing		NEI	0/50	Assessed qualitatively due to modeled air concentrations ≈ those based on TRI data		
Processing/Recycling/Recycling						
Processing/Processing – repackaging/ Processing – repackaging	Processing – repackaging	Modeled [d]	N/A	Assessed qualitatively due to modeled air concentrations ≈ those based on TRI data		
Distribution in commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce	NEI	0/5	Assessed qualitatively due to modeled air concentrations ≈ those based on TRI data		
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	NEI	0/2	Assessed qualitatively due to modeled air concentrations ≈ those based on TRI data		
		Modeled [d e]	N/A			
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	TRI	0/8	7.6E−02	8.02E05	9.5E−08
		NEI	0/102	Assessed qualitatively due to modeled air concentrations ≈ those based on TRI data		

[a] Number of facilities for a given OES with RQ > 1
[b] Soil pore water concentrations calculated from estimated soil catchment concentrations that could be in soil via maximum daily air deposition (95th percentile) of 1,1-dichloroethane at a distance of 10 m from facility based on releases reported to TRI.
[c] Based on hazard data from Canadian poplar (*Populus x canadensis*) exposed to 1,1-dichloroethane for 2 weeks in growth medium.
[d] COU/OESs for which releases were Monte-Carlo simulated (environmental release-modeled)
[e] Estimates of fugitive air emissions could not be separated from stack emission estimates.

Table_Apx L-2. Calculated RQs For Terrestrial Plants Based on 1,1-Dichloroethane Soil Pore Water Concentrations (µg/L) as Calculated Using Modeled Biosolid Land Application Data

COU (Life Cycle Stage/Category/Subcategory)	Occupational Exposure Scenario	Number of Facilities[a]	Soil Type	Soil Pore Water Concentration (µg/L) [b]	Hazard Threshold (µg/L) [c]	RQ
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	NA	Tilled agricultural	18.5	8.02E05	2.3E−05
			Pastureland	36.6	8.02E05	4.6E−05

[a] In the absence of measured data, EPA estimated the maximum amount of 1,1-dichloroethane entering wastewater treatment from the maximum releases reported for any facility in its DMR.
[b] Soil pore water concentration calculated from estimated concentration of 1,1-dichloroethane in soil receiving an annual application of biosolids.
[c] Based on hazard data from Canadian poplar (*Populus x canadensis*) exposed to 1,1-dichloroethane for 2 weeks in growth medium.

L.3 Risk Characterization Based on Trophic Transfer in the Environment

Trophic transfer of 1,1-dichloroethane and risk to terrestrial species was evaluated using a screening level approach conducted as described in the *EPA's Guidance for Developing Ecological Soil Screening Levels* (U.S. EPA, 2005a). 1,1-Dichloroethane concentrations within biota and resulting RQ values for five relevant COUs represented by three OESs for air deposition to soil 10 m from releasing facilities of TRI-reported fugitive emissions are presented in Table_Apx L-3 for trophic transfer to insectivorous mammals (represented by the short-tailed shrew) and Table_Apx L-4 for trophic transfer to herbivorous mammals (represented by the meadow vole). Table_Apx L-3 and Table_Apx L-4 present biota concentrations and RQ values for the COU/OES with the highest soil and soil pore water concentrations from air deposition 10 m from releasing facilities of TRI-reported fugitive emissions in trophic transfer to insectivorous and herbivorous mammals, respectively (manufacturing of 1,1-dichloroethane as an isolated intermediate). Trophic transfer in soil to insectivorous and herbivorous mammals from 1,1-dichloroethane air deposition 10 m from releasing facilities of NEI-reported or environmental release-modeled (Monte-Carlo simulated) fugitive and/or stack emissions (7 COUs and 2 COUs, respectively) were assessed qualitatively for reasons described in Section L.3.

Briefly, based on maximum air concentrations reported in Table 3-9, Table 3-12, and Table 3-13, air deposition to soil 10 m from releasing facilities of NEI-reported fugitive or stack emissions or environmental release-modeled fugitive and/or stack emissions was anticipated to be comparable or lower than levels quantified for TRI-reported fugitive emissions of 1,1-dichloroethane at the same distance from releasing facilities. Therefore, EPA expects that the RQs for trophic transfer of 1,1-dichloroethane from air deposition to soil from NEI-reported fugitive and/or stack emissions (7 COUs) or environmental release-modeled (Monte-Carlo simulated) fugitive and/or stack emissions (2 COUs) would be similar or less than the RQ values for trophic transfer of 1,1-dichloroethane from air deposition to soil from TRI-reported fugitive emissions (with the highest RQ value for trophic transfer based on air deposition to soil = 2.1×10^{-4} for manufacturing 1,1-dichloroethane).

1,1-Dichloroethane concentrations within biota and resulting RQ values for one COU represented by one OES for biosolids land application to agricultural tilled and pastureland soils are presented in Tables 4-16 and 4-17 for trophic transfer to insectivorous mammals (shrew) and herbivorous mammals (vole), respectively. RQs were below 1 for all soil and soil pore water concentrations and COUs based on the mammalian TRV, calculated using empirical toxicity data with mice and rats.

1,1-Dichloroethane concentrations within biota and resulting RQ values for six relevant COUs represented by seven OESs for releases to surface water and benthic pore water are presented in Table_Apx L-5 for trophic transfer to semi-aquatic mammals (mink) consuming fish and Table_Apx L-6 for trophic transfer to semi-aquatic mammals consuming crayfish. Table_Apx L-5 and Table_Apx L-6 present biota (fish and crayfish, respectively) concentrations and RQ values for the COU/OES with the highest surface water and benthic pore water concentrations via PSC based on total number of operating days, which was the COU/OES manufacture/manufacturing of 1,1-dichloroethane. The chronic TRV, calculated using empirical toxicity data with mice and rats and representing hazard in a semi-aquatic mammal (mink), resulted in RQs less than 1 for all modeled surface water and benthic pore water concentrations.

L.3.1 Trophic Transfer Analysis Results

Table_Apx L-3. RQs for Screening Level Trophic Transfer of 1,1-Dichloroethane that Could Result from Air Deposition (1,1-Dichloroethane Releases Reported to TRI) in Insectivorous Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	Earthworm Concentration (mg/kg) [a]	TRV (mg/kg-bw/day) [b]	Short-Tailed Shrew (*Blarina brevicauda*)	
				1,1-Dichloroethane Dietary Exposure (mg/kg/day)[c]	RQ
Manufacture/Domestic manufacturing/Domestic manufacturing	Manufacturing	7.0E−03	1,189	4.6E−03	3.9E−06
Processing/As a reactant/Intermediate in all other basic organic chemical manufacture	Processing as a reactive Intermediate	0.38	1,189	0.25	2.1E−04
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing					
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	1.1E−03	1,189	6.9E−04	5.8E−07

[a] Estimated 1,1-dichloroethane concentration in representative soil invertebrate, earthworm, assumed equal to aggregated highest calculated soil and soil pore water concentration via air deposition to soil for fugitive air releases of 1,1-dichloroethane reported to TRI.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (earthworm), incidental ingestion of soil, and ingestion of water.

Table_Apx L-4. RQs for Screening Level Trophic Transfer of 1,1-Dichloroethane Which Could Result from Air Deposition (1,1-Dichloroethane Releases Reported to TRI) in Herbivorous Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	Plant Concentration (mg/kg) [a]	TRV (mg/kg-bw/day) [b]	Meadow Vole (*Microtus pennsylvanicus*)	
				1,1-Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Manufacture/Domestic manufacturing/ Domestic manufacturing	Manufacturing	2.7E−03	1,189	1.5E−03	1.3E−06
Processing/As a reactant/Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	0.15	1,189	8.2E−02	6.9E−05
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing					
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	4.0E−04	1,189	2.3E−04	1.9E−07

[a] Estimated 1,1-dichloroethane concentration in representative terrestrial plant *Trifolium* sp., assumed equal to the highest calculated soil pore water concentration via air deposition to soil for fugitive air releases of 1,1-dichloroethane reported to TRI.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (*Trifolium* sp.), incidental ingestion of soil, and ingestion of water.

Table_Apx L-5. RQs Based on Potential Trophic Transfer of 1,1-Dichloroethane from Fish to American Mink (*Mustela vison*) as a Model Aquatic Predator Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	SWC (µg/L) [a]	Fish Concentration (mg/kg)	TRV (mg/kg-bw/day) [b]	American Mink (*Mustela vison*)	
					1,1-Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Manufacture/Domestic manufacturing/ Domestic manufacturing	Manufacturing	85	0.59	1,189	0.14	1.2E−04
Processing/As a reactant/Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	13	9.0E−02	1,189	2.1E−02	1.8E−05
Processing/As a reactant/Intermediate in all other chemical product and preparation manufacturing						
Processing/Processing – repackaging/ Processing – repackaging	Processing – repackaging	0.7	4.9E−03	1,189	1.2E−03	9.7E−07
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	0.64	4.5E−03	1,189	1.0E−03	8.8E−07
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	12	8.7E−02	1,189	2.0E−02	1.7E−05
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	8.2	5.7E−02	1,189	1.3E−02	1.1E−05
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (Remediation)	31	0.21	1,189	5.0E−02	4.2E−05

[a] 1,1-Dichloroethane concentration represents the highest modeled surface water concentration via PSC modeling.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (fish), incidental ingestion of sediment, and ingestion of water.
[d] Distribution in Commerce does not result in surface water releases (Table 3-6).

Table_Apx L-6. Highest RQs Based on Potential Trophic Transfer of 1,1-Dichloroethane from Crayfish to American Mink (*Mustela vison*) as a Model Aquatic Predator Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	Benthic Pore Water (µg/L) [a]	Crayfish Concentration (mg/kg)	TRV (mg/kg-bw/day) [b]	American Mink (*Mustela vison*)	
					1,1-Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Manufacture/Domestic manufacturing/ Domestic manufacturing	Manufacturing	78	0.55	1,189	0.13	1.1E−04
Processing/As a reactant/ Intermediate in all other basic organic chemical manufacture	Processing as a reactive intermediate	12	8.7E−02	1,189	2.0E−02	1.7E−05
Processing/As a reactant/ Intermediate in all other chemical product and preparation manufacturing						
Processing/Processing – repackaging/ Processing – repackaging	Processing – repackaging	6.1E−01	4.3E−03	1,189	1.0E−03	8.5E−07
Commercial Use/Other use/Laboratory chemicals	Commercial use as a laboratory chemical	5.5E−01	3.8E−03	1,189	9.1E−04	7.6E−07
Disposal/Disposal/Disposal	General waste handling, treatment, and disposal	12	8.3E−02	1,189	1.9E−02	1.6E−05
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	7.9	5.5E−02	1,189	1.3E−02	1.1E−05
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (remediation)	29	0.21	1,189	4.8E−02	4.1E−05
Distribution in Commerce/Distribution in commerce/Distribution in commerce	Distribution in commerce	N/A [d]				

[a] 1,1-Dichloroethane concentration represents the highest modeled benthic pore water concentration via PSC modeling.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (crayfish), incidental ingestion of sediment, and ingestion of water.
[d] Distribution in commerce does not result in surface water releases (Table 3-6).

Table_Apx L-7. RQs for Screening Level Trophic Transfer of 1,1-Dichloroethane from Air Deposition in Insectivorous Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	Earthworm Concentration (mg/kg) [a]	TRV (mg/kg-bw/day) [b]	Short-Tailed shrew (*Blarina brevicauda*)	
				1,1-Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Manufacture/Domestic manufacturing/Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	0.38	1,189	0.25	2.1E−04

[a] Estimated 1,1-dichloroethane concentration in representative soil invertebrate, earthworm, assumed equal to aggregated highest calculated soil and soil pore water concentration via air deposition to soil 10 m from releasing facilities of TRI-reported fugitive emissions.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (earthworm), incidental ingestion of soil, and ingestion of water.

Table_Apx L-8. RQs for Screening Level Trophic Transfer of 1,1-Dichloroethane from Air Deposition in Herbivorous Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	Plant Concentration (mg/kg) [a]	TRV (mg/kg-bw/day) [b]	Meadow Vole (*Microtus pennsylvanicus*)	
				1,1-Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Manufacture/ Domestic manufacturing/ Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	0.15	1,189	8.2E−02	6.9E−05

[a] Estimated 1,1-dichloroethane concentration in representative terrestrial plant *Trifolium* sp., assumed equal to the highest calculated soil pore water concentration via air deposition to soil 10 m from releasing facilities of TRI-reported fugitive emissions.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (*Trifolium* sp.), incidental ingestion of soil, and ingestion of water.

Table_Apx L-9. RQs for Screening Level Trophic Transfer of 1,1-Dichloroethane from Biosolid Land Application in Insectivorous Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	Soil Type	Earthworm Concentration (mg/kg) [a]	TRV (mg/kg-bw/day) [b]	Short-Tailed shrew (*Blarina brevicauda*)	
					1,1-Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	Tilled agricultural	4.8E−02	1,189	3.1E−02	2.6E−05
		Pastureland	9.5E−02	1,189	6.3E−02	5.3E−05

[a] Estimated 1,1-dichloroethane concentration in representative soil invertebrate, earthworm, assumed equal to aggregated highest calculated soil and soil pore water concentration via biosolids land application.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (earthworm), incidental ingestion of soil, and ingestion of water.

Table_Apx L-10. RQs for Screening Level Trophic Transfer of 1,1-Dichloroethane from Biosolid Land Application in Herbivorous Terrestrial Ecosystems Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	Soil Type	Plant Concentration (mg/kg) [a]	TRV (mg/kg-bw/day) [b]	Meadow Vole (*Microtus pennsylvanicus*)	
					1,1-Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Disposal/Disposal/Disposal	Waste handling, treatment, and disposal (POTW)	Tilled agricultural	1.9E−02	1,189	1.0E−02	8.7E−06
		Pastureland	3.7E−02	1,189	2.1E−02	1.7E−05

[a] Estimated 1,1-dichloroethane concentration in representative terrestrial plant *Trifolium* sp., assumed equal to the highest calculated soil pore water concentration via biosolids land application.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (*Trifolium* sp.), incidental ingestion of soil, and ingestion of water.

Table_Apx L-11. Risk Quotient (RQ) Based on Potential Trophic Transfer of 1,1-Dichloroethane from Fish to American Mink (*Mustela vison*) as a Model Aquatic Predator Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	SWC [a] (µg/L)	Fish Concentration (mg/kg)	TRV (mg/kg-bw/day) [b]	American Mink (*Mustela vison*)	
					1,1- Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Manufacture/Domestic manufacturing/ Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	85	0.59	1,189	0.14	1.2E−04

[a] 1,1-dichloroethane concentration represents the highest modeled surface water concentration via PSC modeling.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (fish), incidental ingestion of sediment, and ingestion of water.

Table_Apx L-12. RQ Based on Potential Trophic Transfer of 1,1-Dichloroethane from Crayfish to American Mink (*Mustela vison*) as a Model Aquatic Predator Using EPA's Wildlife Risk Model for Eco-SSLs

COU (Life Cycle Stage/Category/Subcategory)	OES	Benthic Pore Water [a] (µg/L)	Crayfish Concentration (mg/kg)	TRV (mg/kg-bw/day) [b]	American Mink (*Mustela vison*)	
					1,1-Dichloroethane Dietary Exposure (mg/kg/day) [c]	RQ
Manufacture/Domestic manufacturing/Domestic manufacturing	Manufacturing of 1,1-dichloroethane as an isolated intermediate	78	0.55	1,189	0.13	1.1E−04

[a] 1,1-dichloroethane concentration represents the highest modeled benthic pore water concentration via PSC modeling.
[b] Mammal 1,1-dichloroethane TRV value calculated using several studies as per (U.S. EPA, 2007).
[c] Dietary exposure to 1,1-dichloroethane includes consumption of biota (crayfish), incidental ingestion of sediment, and ingestion of water.

Appendix M ANALOG SELECTION FOR READ-ACROSS

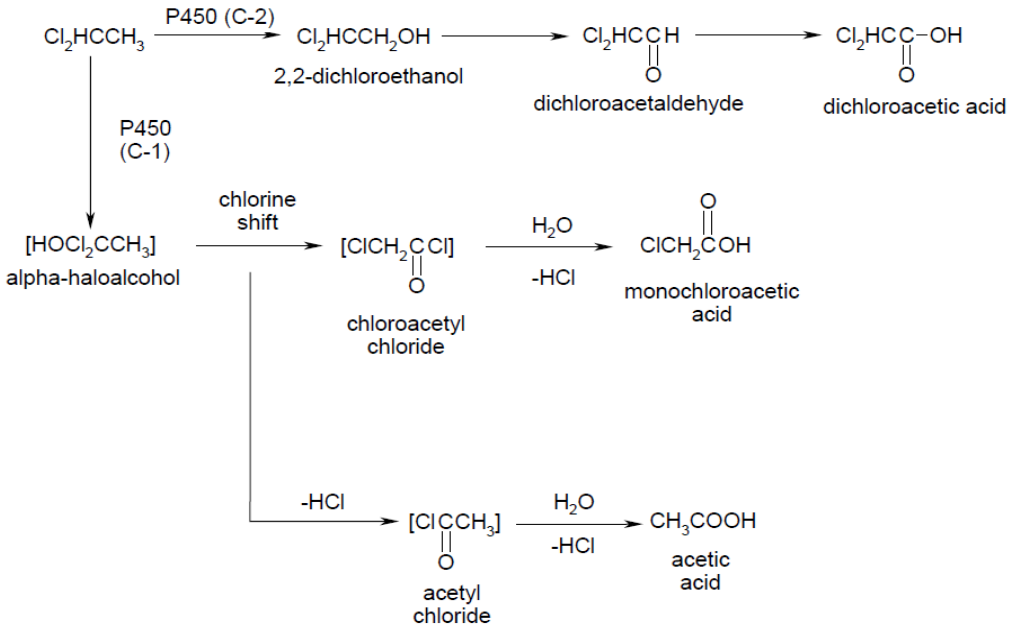
M.1 Analog Selection for Human Health Hazard

EPA identified data gaps for 1,1-dichloroethane for non-cancer PODs for acute, intermediate, and chronic oral, inhalation and dermal routes, and for cancer for oral, inhalation, and dermal routes. Therefore, an analysis of other chlorinated solvents as potential analogs for read-across data was performed following the general principles for read-across as outlined in Lizarraga et al. (2019) and further refinements to the read-across framework presented in a subsequent publication by Lizarraga et al. (2023), taking into consideration structural similarities, physical and chemical properties, metabolism, and toxicological similarities. Overall, the close isomer 1,2-dichloroethane was identified as the best available candidate chemical isomer to fill the identified data gaps for 1,1-dichloroethane.

M.1.1 Metabolic Pathways

In Vitro Metabolism Studies – 1,1-Dichloroethane

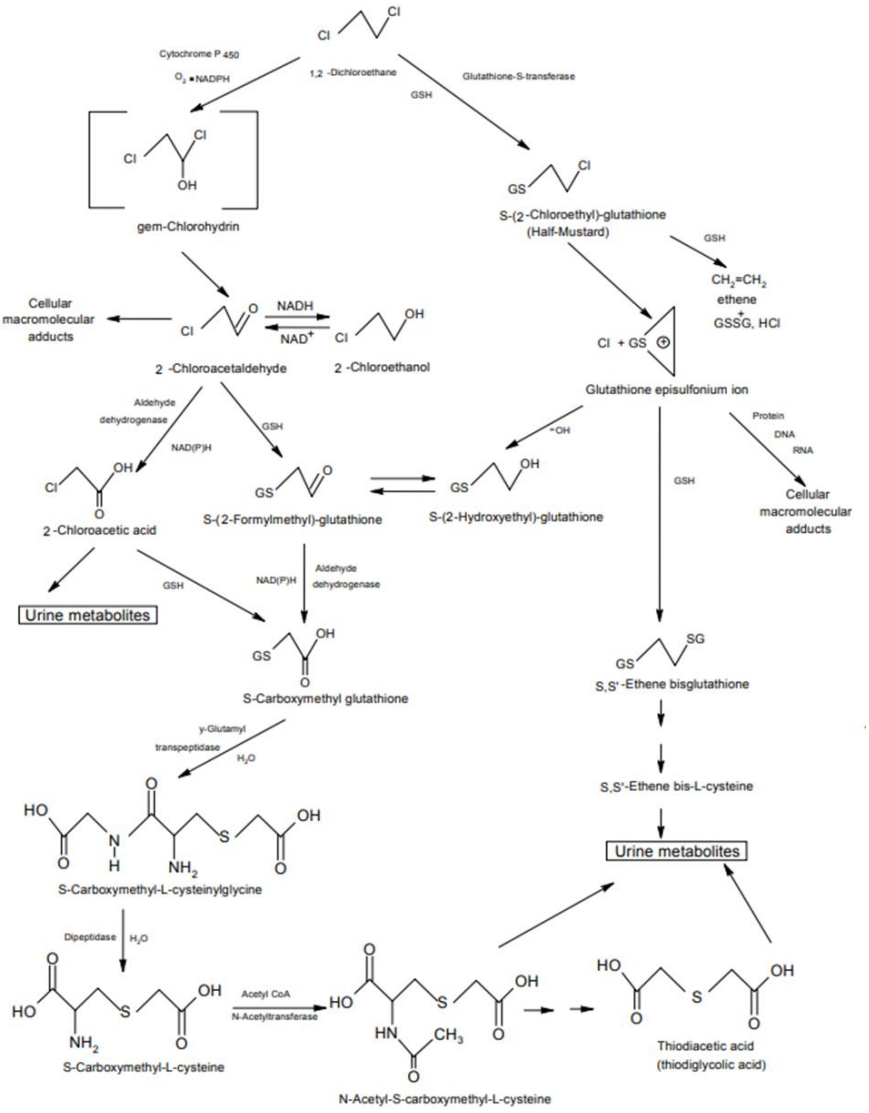
The proposed metabolic pathways as outlined in Figure_Apx M-1 for 1,1-dichloroethane have been elucidated from *in vitro* studies using rat hepatic microsomes (McCall et al., 1983; Sato et al., 1983; Van Dyke and Wineman, 1971). The primary metabolic pathway involves oxidation of the C-1 carbon by cytochrome P450 (CYP) to give an unstable alpha-haloalcohol followed by dechlorination to produce acetyl chloride and acetic acid, which is the major metabolite. The alpha-haloalcohol may also undergo a chlorine shift to yield chloroacetyl chloride and monochloroacetic acid, although this reaction is not favored. CYP oxidation at the C-2 position results in the formation of 2,2-dichloroethanol, dichloroacetaldehyde, and dichloroacetic acid as minor metabolites. Metabolism of 1,1-dichloroethane was increased by induction with phenobarbital and ethanol, but not β-naphthoflavone (McCall et al., 1983; Sato et al., 1983). Similarly, enzymatic dechlorination was inducible by phenobarbital, but not 3-methylcholanthrene (Van Dyke and Wineman, 1971).



Figure_Apx M-1. Proposed Metabolic Scheme for 1,1-Dichloroethane (McCall et al., 1983)

In Vivo and In Vitro Metabolism Studies – 1,2-Dichloroethane

No human studies on the metabolism of 1,2-dichloroethane were located. The proposed metabolic pathways as outlined in Figure_Apx M-2 outlines the primary metabolic pathways for 1,2-dichloroethane, elucidated from *in vitro* studies and *in vivo* studies in rats and mice, include cytochrome P450 (CYP) oxidation and glutathione (GSH) conjugation (IPCS, 1995). Metabolism by CYP results in an unstable gem-chlorohydrin that releases hydrochloric acid, resulting in the formation of 2-chloroacetaldehyde. 2-Chloroacetaldehyde is oxidized by aldehyde dehydrogenase to form chloroacetic acid or reduced to form 2-chloroethanol, and these metabolites are conjugated with GSH and excreted in the urine. *In vivo*, the oral administration of the aldehyde dehydrogenase inhibitor disulfiram, a human drug, increased the blood levels of 1,2-dichloroethane in rats by 5-fold when administered via inhalation indicating this pathway is important for clearance (Cheever et al., 1990). Metabolism via glutathione-S-transferase results in formation of S-(2-chloroethyl)-glutathione, which rearranges to form a reactive episulfonium ion. The episulfonium ion can form adducts with protein, DNA or RNA or interact further with GSH to produce water soluble metabolites that are excreted in the urine.



Figure_Apx M-2. Proposed Metabolic Scheme for 1,2-Dichloroethane (IPCS, 1995)

M.1.2 Read-Across Utilized in Other Program Offices

Historically, EPA and other federal and state agencies have used 1,2-dichloroethane cancer studies routinely to assess the cancer risk for 1,1-dichloroethane. The IRIS assessment of carcinogenic potential of 1,2-dichloroethane was considered to be "supportive" of 1,1-dichloroethane carcinogenic potential "…Because of similarities in structure and target organs…." A comparison of the cancer slope factors across other program offices for 1,1-dichloroethane is provided below in Table_Apx M-1; those for 1,2-dichloroethane are summarized in Table_Apx M-2.

Table_Apx M-1. 1,1-Dichloroethane Cancer Slope Factors Across EPA Offices/Programs

1,1-Dichloroethane Cancer Slope Factors and Cancer Classifications			
Program	**Oral Slope Factor**	**Inhalation Unit Risk**	**Assess for Cancer**
OPPT RE Continuous Exposure	• 0.062 per mg/kg/day • **Read-across** from mouse 1,2-dichloroethane hepatocellular carcinoma data (NTP, 1978) • **High OPPT SR rating**	• 7.1E−06 (per µg/m^3) • **Read-across** from inhalation rat 1,2-dichloroethane (Nagano et al., 2006) • Combined tumors in females • **High OPPT SR rating**	• Yes
IRIS U.S. EPA (1990)	• Not evaluated	• Not evaluated	• Possible human carcinogen partially based on 1,2-dichloroethane data
OW	• 0.0057 per mg/kg/day • **Same as CAL EPA (OEHHA)** • **Read-across** using oral rat 1,2-dichloroethane data (NTP, 1978) • **Uninformative in OPPT SR**	• Not reported	• Yes
OAR	• Not reported	• 1.6E−06 (per µg/m^3) • **Same as CAL EPA (OEHHA)** • **Read-across** from oral 1,2-dichloroethane	• Yes
OLEM	• 0.0057 per mg/kg/day • **Same as Cal EPA (OEHHA)** • **Read-across** using rat 1,2-dichloroethane • **Uninformative in OPPT SR**	• 1.6E−06 (per µg/m^3) • **Same as Cal EPA (OEHHA)** • **Read-across** from oral 1,2-dichloroethane (NTP, 1978)	• Yes
Cal EPA 1992	• 0.0057 per mg/kg/day • **Read-across** using oral rat 1,2-dichloroethane data (NTP, 1978) • **Uninformative in OPPT SR**	• 1.6E−06 (per µg/m^3) • **Read-across** using oral rat 1,2-dichloroethane data (NTP, 1978) • **Uninformative in OPPT SR**	• Yes

Cal EPA = California EPA; IRIS = EPA Integrated Risk Information System; OAR = EPA Office of Air and Radiation; OEHHA = CAL EPA Office of Environmental Health Hazard Assessment; OLEM = EPA Office of Land and Emergency Management; OPPT = EPA Office of Pollution Prevention and Toxics; OW = EPA Office of Water; SR = systematic review

Table_Apx M-2. 1,2-Dichloroethane Cancer Slope Factors Across EPA Offices/Programs

1,2-Dichloroethane Cancer Slope Factors		
EPA Program	**Oral Slope Factor**	**Inhalation Unit Risk**
OPPT RE Continuous Exposure	• 0.062 per mg/kg/day • Mouse (NTP, 1978) • Hepatocellular carcinoma data • **High OPPT SR rating**	• 7.1E−06 per µg/m^3 • Rat inhalation (Nagano et al., 2006) • Combined tumors in females • **High OPPT SR rating**
IRIS NCEA (1987)	• 0.091 per mg/kg/day • Rat hemangiosarcoma data (using a time to death analysis) (NTP, 1978) • **Rat study rated Uninformative OPPT SR**	• 2.6E−05 per µg/m^3 • Rat oral hemangiosarcoma data (using a time to death analysis) (NTP, 1978) • **Rat study rated Uninformative OPPT SR**
OW	• 0.091 per mg/kg/day **based on (**NCEA, 1987**)** • Rat hemangiosarcoma data (using a time to death analysis) (NTP, 1978) • **Rat study rated Uninformative OPPT SR**	• Not reported
OAR	• Not reported	• 2.6E−5 per µg/m^3 **based on (**NCEA, 1987**)** • Rat oral hemangiosarcoma data (using a time to death analysis) (NTP, 1978) • **Rat study rated Uninformative OPPT SR**
OLEM	• 0.091 per mg/kg/day **based on (**NCEA, 1987**)** • Rat oral hemangiosarcoma data (using a time to death analysis) (NTP, 1978) • **Rat study rated Uninformative OPPT SR**	• 2.6E−05 per µg/m^3 **based on (**NCEA, 1987**)** • Rat oral hemangiosarcoma data (using a time to death analysis) (NTP, 1978) • **Rat study rated Uninformative OPPT SR**
Cal EPA	• 0.072 per mg/kg/day • Rat oral hemangiosarcoma data (using a Weibull model) (NTP, 1978) • **Rat study rated Uninformative OPPT SR**	• 2.1E−05 per µg/m^3 • Derived from oral rat data • **Rat study rated Uninformative OPPT SR**

Cal EPA = California EPA; IRIS = EPA Integrated Risk Information System; OAR = EPA Office of Air and Radiation; OEHHA = CAL EPA Office of Environmental Health Hazard Assessment; OLEM = EPA Office of Land and Emergency Management; OPPT = EPA Office of Pollution Prevention and Toxics; OW = EPA Office of Water; SR = systematic review

Appendix N HUMAN HEALTH HAZARD DETAILS

This appendix provides details on the human health hazard assessment for 1,1-dichloroethane and the identified analog 1,2-dichloroethane. Human health hazard data for 1,2-dichloroethane were used to fill data gaps for 1,1-dichloroethane. Appendix N.1 provides a summary of toxicokinetics for both 1,1-dichloroethane and 1,2-dichloroethane. Appendix N.2 provides a non-cancer dose response assessment for both chemicals while Appendix N.3 describes evidence for genotoxicity and cancer for both chemicals. Appendix N.4 provides the equations used in derivation of non-cancer and cancer PODs for the 1,1-dichloroethane risk assessment. Appendix N.5 describes the non-cancer POD derivation for acute, short/intermediate-term, and chronic durations. Appendices N.6 and N.7 provide evidence integration tables for non-cancer health effects of 1,1-dichloroethane. Appendices N.8 and N.9 provide evidence integration tables for cancer for 1,2-dichloroethane. Lastly, Appendix N.10 provides a cancer dose-response assessment for 1,1-dichloroethane using data for 1,2-dichloroethane as read-across.

N.1 Toxicokinetics

N.1.1 Absorption

N.1.1.1 1,1-Dichloroethane

Oral

Oral absorption of 1,1-dichloroethane was demonstrated by the detection of radiolabel in expired air, excreta, and body carcass following gavage administration of 700 mg/kg-bw/day 1,1-dichloroethane (unlabeled) via gavage 5 days/week for 4 weeks followed by a single dose of 700 mg/kg ^{14}C-1,1-dichloroethane in rats or 1,800 mg/kg-bw/day 1,1-dichloroethane (unlabeled) via gavage 5 days/week for 4 weeks followed by a single dose of 1,800 mg/kg ^{14}C-1,1-dichloroethane in mice (Mitoma et al., 1985). Within 48 hours in rats, 91 percent of the administered dose was eliminated in expired air (86 percent unchanged, 5 percent as CO_2). Less than 1 percent of the radiolabel was detected in urine and feces of rats and 1 percent was detected in carcass. In mice, 95 percent of the administered dose was eliminated in expired air (70% unchanged, 25% as CO_2) within 48 hours. Less than 2 percent of the radiolabel was detected in urine and feces of mice and 2 percent was detected in the carcass (Mitoma et al., 1985).

Inhalation

Previous use of 1,1-dichloroethane as a gaseous anesthetic in humans provides evidence of systemic absorption by the inhalation route (ATSDR, 2015). EPA did not identify any *in vivo* animal data evaluating the absorption of 1,1-dichloroethane by the inhalation route of exposure. The blood:air coefficient for 1,1-dichloroethane (4.94 ± 0.24 in humans and 11.2 ± 0.1 in rats) suggests that pulmonary absorption is likely to occur (Gargas and Andersen, 1989).

Dermal

Qualitative evidence of dermal absorption was provided by a rabbit study that detected halogen ion in exhaled breath following application of 1,1-dichloroethane to shaved abdominal skin (ATSDR, 2015). No *in vivo* data were located on the rate and extent of 1,1-dichloroethane absorption through the skin; however, in an OECD 428 study utilizing human skin the dermal absorption was 0.3 percent.

N.1.1.2 1,2-Dichloroethane

Oral

Oral absorption of 1,2-dichloroethane in humans is suggested by case reports of intentional or accidental ingestion resulting in systemic health effects including death (ATSDR, 2024). Experimental animal

studies indicate that oral absorption is rapid and complete (Reitz et al., 1982; Reitz et al., 1980; Spreafico et al., 1980). In rats given a single gavage dose of 150 mg/kg in corn oil, peak blood concentrations were reached within 15 minutes and approximately 94 percent of the administered dose was absorbed within 48 hours (Reitz et al., 1982; Reitz et al., 1980). Spreafico et al. (1980) also demonstrated rapid oral absorption, with peak blood levels occurring between 30 and 60 minutes in rats given gavage doses of 25, 50, or 100 mg/kg in corn oil. Examination of the peak blood level curves at the different doses shows a linear profile up to 50 mg/kg 1,2-dichloroethane and a decrease in steepness of the curve at 100 mg/kg, suggesting a relative saturation of oral absorption at doses exceeding 100 mg/kg. In rats given a single gavage dose of 100 mg/kg 1,2-dichloroethane in corn oil or water, peak blood concentrations (C_{max}) were approximately 4-fold higher and the time to reach C_{max} was 3-fold faster following administration in water compared to corn oil (Withey et al., 1983). Similar findings regarding the rate of absorption were observed in rats given gavage doses of 43 mg/kg/day in water or 150 mg/kg/day in corn oil (C_{max} values of 15 or 30 minutes, respectively) (Dow Chemical, 2006a).

Inhalation

1,2-Dichloroethane was detected in the breast milk of nursing women exposed to 16 ppm in workplace air (with concurrent dermal exposure) (Urusova, 1953). A fatal case report of exposure to 1,2-dichloroethane in an enclosed space for 30 minutes provides further support for absorption through the lungs (Nouchi et al., 1984). Absorption by inhalation was rapid, with steady-state C_{max} concentrations measured 1 to 3 hours after the onset of exposure to 150 to 250 ppm in rats (Dow Chemical, 2006a; Reitz et al., 1982; Reitz et al., 1980; Spreafico et al., 1980) or 25 to 185 ppm in mice (Zhong et al., 2022). In rats exposed to 150 ppm ^{14}C-1,2-dichloroethane for 6 hours, approximately 93 percent absorption occurred based on recovery of radiolabel in urine and feces and as CO_2 in expired air by 48 hours (Reitz et al., 1982). The blood:air coefficients for 1,2-dichloroethane (19.5 ± 0.7 in humans and 30.4 ± 1.2 in rats) also suggest that pulmonary absorption is likely to occur (Gargas et al., 1989).

Dermal

In vivo animal studies have demonstrated that 1,2-dichloroethane is readily absorbed through the skin (Morgan et al., 1991; Jakobson et al., 1982; Tsuruta, 1975). Application of neat 1,2-dichloroethane to the shaved and abraded skin of rats using covered dermal cells resulted in approximately 50 percent absorption of the applied dose with the peak blood level measured at 24 hours (Morgan et al., 1991). Dermal absorption was faster and more complete for aqueous solutions of 1,2-dichloroethane, with peak blood levels measured within 1 to 2 hours and greater than 99 percent of the applied dose absorbed within the 24-hour exposure period (Morgan et al., 1991). In guinea pigs dermally exposed to neat 1,2-dichloroethane, using a covered dermal cell on clipped intact skin, blood concentrations rose rapidly during the first 30 minutes and continued to increase over a 12-hour period (Jakobson et al., 1982). Tsuruta (1975) estimated a percutaneous absorption rate of 480 nmol/minute/cm^2 for 1,2-dichloroethane through the clipped, intact abdominal skin of mice following a 15-minute exposure using a closed dermal cell.

In Vitro

In vitro studies using skin from humans, pigs, and guinea pigs have reported apparent partition coefficients (K_p), steady-state flux (J_{ss}) values, and lag time estimates (*i.e.*, the time to achieve a steady-state concentration) (see Table_Apx N-1). In human skin, 0.13 to 0.21 percent of the applied dose was absorbed over 24 hours at 63.1 and 7.9 mg per cm^2, respectively, with the maximum flux occurring within 10 minutes of exposure (Gajjar and Kasting, 2014). Evaporation from the skin surface accounted for the majority of applied dose in this study. The K_p and lag time values for 1,2-dichloroethane were similar for human and guinea pig skin (Frasch and Barbero, 2009); however, the dermal permeability rate was lower in pig skin (decreased K_p value; longer lag time) (Schenk et al., 2018). In guinea pig skin,

the flux was lower in saturated aqueous solution compared to the undiluted test substance (Frasch et al., 2007). This result appears to differ from the *in vivo* study using abraded skin of rats, which showed a higher percent absorption for an aqueous solution of 1,2-dichloroethane compared to a neat application (Morgan et al., 1991).

Table_Apx N-1. 1,2-Dichloroethane Partition Coefficients Steady State Estimates

Partition Coefficients (K_p) Steady-State Flux (Jss) Estimates from *In Vitro* Dermal Absorption Studies					
Species	**Test Material(s)**	**K_p (cm/hour)**	**Jss (µg/cm^2-hour)**	**Lag Time (minutes)**	**Reference**
Human	Neat	ND	37–193[a]	ND	Gajjar and Kasting (2014)
Human Guinea pig	Neat Neat	0.259 0.259	ND ND	6 6	Frasch and Barbero (2009)
Pig	Neat	1.9E−03	1,360	30.7	Schenk et al. (2018)
Guinea pig	Neat Aqueous	ND ND	6,280[b] 1,076	ND ND	Frasch et al. (2007)
ND = not derived [a] Range of Jss values for applied doses of 7.9, 15.8, 31.5, or 63.1 mg/cm^2. [b] Also reported a Jss value of 3,842 µg/cm^2-hour from a different laboratory.					

N.1.2 Distribution

N.1.2.1 1,1-Dichloroethane

Oral, Inhalation, and Dermal

Distribution to the CNS is suggested by the previous use of 1,1-dichloroethane as a gaseous anesthetic in humans (ATSDR, 2015). No experimental studies were located regarding distribution following oral, inhalation, or dermal exposure to 1,1-dichloroethane.

Other Routes (Intraperitoneal Injection)

Radiolabeled 1,1-dichloroethane was detected as protein, DNA, and RNA adducts in the liver, kidney, lung, and stomach, 22 hours after a single intraperitoneal injection of 1.2 mg/kg ^{14}C-1,1-dichloroethane in Wistar rats and BALB/c mice suggesting that it reacts with cellular biomolecules (Colacci et al., 1985). No additional tissues were examined in this study.

In Vitro

Tissue:air partition coefficients calculated using a vial equilibration method on tissues obtained from male Fischer 344 rats suggest that 1,1-dichloroethane is likely distributed to highly perfused tissues (*i.e.*, liver, muscle) and will accumulate in fat (Table_Apx N-2) (Gargas and Andersen, 1989).

Table_Apx N-2. 1,1-Dichloroethane Partition Coefficients

Species	**Strain**	**Sex**	**Partition Coefficient**			
			Blood/Air	**Liver/Air**	**Muscle/Air**	**Fat/Air**
Rat	F344	Male	11.2 ± 0.1	10.8 ± 0.5	5.12 ± 0.48	164 ± 4
Source: Gargas and Andersen (1989)						

N.1.2.2 1,2-Dichloroethane

Oral

Distribution was rapid following gavage dosing, with concentrations peaking first in the liver at 6 to 7 minutes, followed by lung at 10 to 20 minutes and adipose tissue at 20 to 60 minutes (Manufacturing Chemists Association, 1979). Tissue levels were dose-dependent and the highest peak tissue concentration at any dose was detected in fat. Similar mean peak tissue levels in liver and lung were seen following 11 daily doses of 50 mg/kg, indicating that bioaccumulation does not occur in these tissues with multiple doses. Bioaccumulation in adipose tissue is suggested by higher peak adipose tissue levels after 11 gavage doses, compared to a single gavage dose (Table_Apx N-3).

Table_Apx N-3. Tissue Levels and Time to Peak Tissue Level in Rats Exposed to 1,2-Dichloroethane by Gavage in Corn Oil

Organ/Peak Concentration/Time to Peak Concentration		Dose (mg/kg)			
		25 (Single)	50 (Single)	50 (11 Oral Doses)	150 Single)
Liver	µg/g	30.02 ± 3.29	55.00 ± 4.12	53.12 ± 3.87	92.10 ± 7.58
	Minutes	6	6	6	7.5
Lung	µg/g	2.92 ± 0.38	7.20 ± 0.39	7.19 ± 0.59	8.31 ± 1.27
	Minutes	10	20	15	20
Adipose	µg/g	110.67 ± 6.98	148.92 ± 20.75	161.69 ± 9.93	259.88 ± 25.03
	Minutes	20	60	40	40
Source: (Manufacturing Chemists Association, 1979)					

In pregnant rats exposed to a single dose of 160 mg/kg ^{14}C-1,2-dichloroethane on gestation day (GD) GD 12, the highest tissue concentrations were found in the liver and intestine after 48 hours (radiolabel was also detected in the stomach, kidney, and ovary) (Payan et al., 1995). Distribution across the placenta was demonstrated by detection of radiolabel in the developing fetus within 1 hour; the maximum concentration was detected 4 hours after exposure (Payan et al., 1995). Administration of 160 mg/kg ^{14}C-1,2-dichloroethane on GD 18 showed a greater degree of accumulation in the developing fetuses and the placenta (Payan et al., 1995).

Inhalation

1,2-dichloroethane was detected in breath (14.3 ppm) and breast milk (0.54–0.64 mg % [per 100 mL]) of nursing mothers 1 hour after leaving an occupational facility with exposure concentrations of 15.6 ppm 1,2-dichloroethane (Urusova, 1953). 1,2-Dichloroethane was readily distributed in rats following a 6-hour inhalation exposure and tissue levels were concentration dependent Spreafico et al. (1980). Peak tissue levels in liver and lung were lower than concentrations in blood, but adipose tissue levels were 8 to 9 times higher than blood levels (Spreafico et al., 1980) (see Table_Apx N-4).

Table_Apx N-4. Tissue Levels and Time to Peak Tissue Level in Rats Exposed by Inhalation to 1,2-Dichloroethane for 6 Hours

Organ/Peak Concentration/ Time to Peak Concentration		Concentration (ppm)	
		50	250
Blood	µg/g	1.37 ± 0.11	31.29 ± 1.19
	Hours	6	6
Liver	µg/g	1.14 ± 0.17	22.49 ± 1.12
	Hours	4	6

Organ/Peak Concentration/ Time to Peak Concentration		Concentration (ppm)	
		50	250
Lung	µg/g	0.42 ± 0.05	14.47 ± 1.12
	Hours	4	3
Adipose	µg/g	11.08 ± 0.77	273.32 ± 12.46
	Hours	4	6
Source: Spreafico et al. (1980)			

A similar study in male rats exposed to 160 ppm 1,2-dichloroethane for 6 hours showed the highest tissue levels of 1,2-dichloroethane in abdominal fat Take et al. (2013). In pregnant rats exposed to 150 to 2,000 ppm 1,2-dichloroethane for 5 hours on GD 17, concentrations of 1,2-dichloroethane in maternal blood and fetal tissue increased linearly with exposure concentration, indicating distribution across the placenta (Withey and Karpinski, 1985).

Dermal

No studies were located regarding distribution following dermal exposure to 1,2-dichloroethane.

In Vitro

Tissue:air partition coefficients calculated using a vial equilibration method and tissues obtained from male Fischer 344 rats suggest that 1,2-dichloroethane is preferentially distributed to highly perfused tissues and will accumulate in fat (see following table) (Dow Chemical, 2006a; Gargas and Andersen, 1989).

Table_Apx N-5. 1,2-Dichloroethane Tissue: Air Partition Coefficients

Partition Coefficient							
Blood/Air	**Liver/Air**	**Muscle/Air**	**Fat/Air**	**Brain/Air**	**Kidney/Air**	**Testis/Air**	**Ovary/Air**
30.4 ± 1.2[a]	35.7 ± 1.6[a]	23.4 ± 1.4[a]	344 ± 5[a]	39.5 ± 2.89[b]	44.89 ± 6.77[b]	31.14 ± 7.98[b]	74.59 ± 9.82[b]
[a] Gargas and Andersen (1989). [b] Dow Chemical (2006a).							

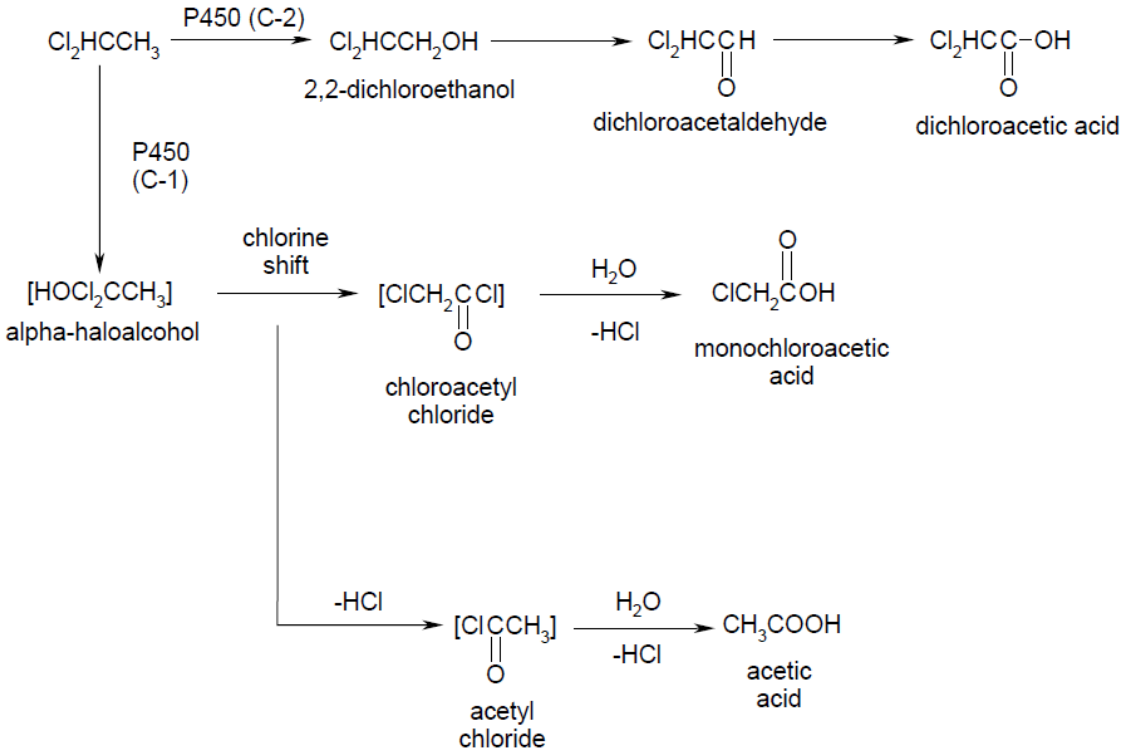
N.1.3 Metabolism

N.1.3.1 1,1-Dichloroethane

In Vitro

The metabolic pathways for 1,1-dichloroethane have been elucidated from *in vitro* studies using rat hepatic microsomes (McCall et al., 1983; Sato et al., 1983; Van Dyke and Wineman, 1971) (see Figure_Apx N-1). The primary metabolic pathway involves oxidation of the C-1 carbon by CYP to give an unstable alpha-haloalcohol followed by dechlorination to produce acetyl chloride and acetic acid, which is the major metabolite. The alpha-haloalcohol may also undergo a chlorine shift to yield chloroacetyl chloride and monochloroacetic acid, although this reaction is not favored. CYP oxidation at the C-2 position results in the formation of 2,2-dichloroethanol, dichloroacetaldehyde, and dichloroacetic acid as minor metabolites. Metabolism of 1,1-dichloroethane was increased by induction with phenobarbital and ethanol, but not β-naphthoflavone (McCall et al., 1983; Sato et al., 1983). Similarly, enzymatic dechlorination was inducible by phenobarbital, but not 3-methylcholanthrene (Van Dyke and Wineman, 1971). 1,1-Dichloroethane generates reactive 2,2-dichloroacetaldehyde during its metabolism and 1,2-dichloroethane generates the DNA crosslinker 2-chloroacetaldehyde during its metabolism. The metabolism of chloroaldehydes by mitochondrial aldehyde dehydrogenase (ALDH) was investigated by Sharpe and Carter with 2,2-dichloroacetaldehyde from 1,1-dichloroethane being

metabolized at a rate 16- to 36-fold slower than 2-chloroacetaldehyde from 1,2-dichloroethane (Sharpe and Carter, 1993). This data suggests that the reactive chloroaldehyde from 1,1-dichloroethane is cleared far slower by mitochondrial ALDH than the reactive chloroaldehyde from 1,2-dichloroethane with relevance to the hazard outcomes.



Figure_Apx N-1. Proposed Metabolic Scheme for 1,1-Dichloroethane (McCall et al., 1983)

Oral

The extent of metabolism was evaluated in Osborne-Mendel rats and B6C3F1 mice administered 700 or 1,800 mg/kg-bw/day 1,1-dichloroethane, respectively, by gavage in corn oil 5 days/week for 4 weeks, followed by a single dose of ^{14}C-1,1-dichloroethane (Mitoma et al., 1985). The total percentages of administered dose found in exhaled CO_2, excreta, and body carcass 48 hours after the administration of the radiolabeled dose were 7.45 percent in rats and 29.3 percent in mice. It is possible that a portion of the radioactivity detected in the urine, feces, and body carcass is present as parent 1,1-dichloroethane and not downstream metabolites.

Inhalation

The metabolic rate constants for 1,1-dichloroethane were estimated for male Fischer 344 rats using a gas uptake method (Gargas et al., 1990) (Table_Apx N-6). The rats were exposed to an initial concentration of 90, 490, 1,100, or 2,175 ppm (360, 1,980, 4,500, or 8,804 mg/m^3) and the disappearance of the gas was studied for about 5 hours. A kinetic model that assumed metabolism occurred exclusively in the liver was used to analyze the data. The metabolism of 1,1-dichloroethane was best described as a saturable process.

Table_Apx N-6. Estimates of Metabolic Parameters for 1,1-Dichloroethane Obtained from Gas Uptake Experiments in Male F344 Rats

V_{maxc}		K_m	
mg/hour*kg	**µmol/hour**	**mg/L**	**µM**
7.5	75.8	0.2	2.02

V_{maxc} = maximum reaction velocity (scaled to 1 kg animal); K_m = concentration at ½ Vmax (Michaelis constant)
Source: Gargas et al. (1990)

Dermal
EPA did not identify *in vivo* animal data that evaluated metabolism of 1,1-dichloroethane by the dermal route of exposure.

N.1.3.2 1,2-Dichloroethane

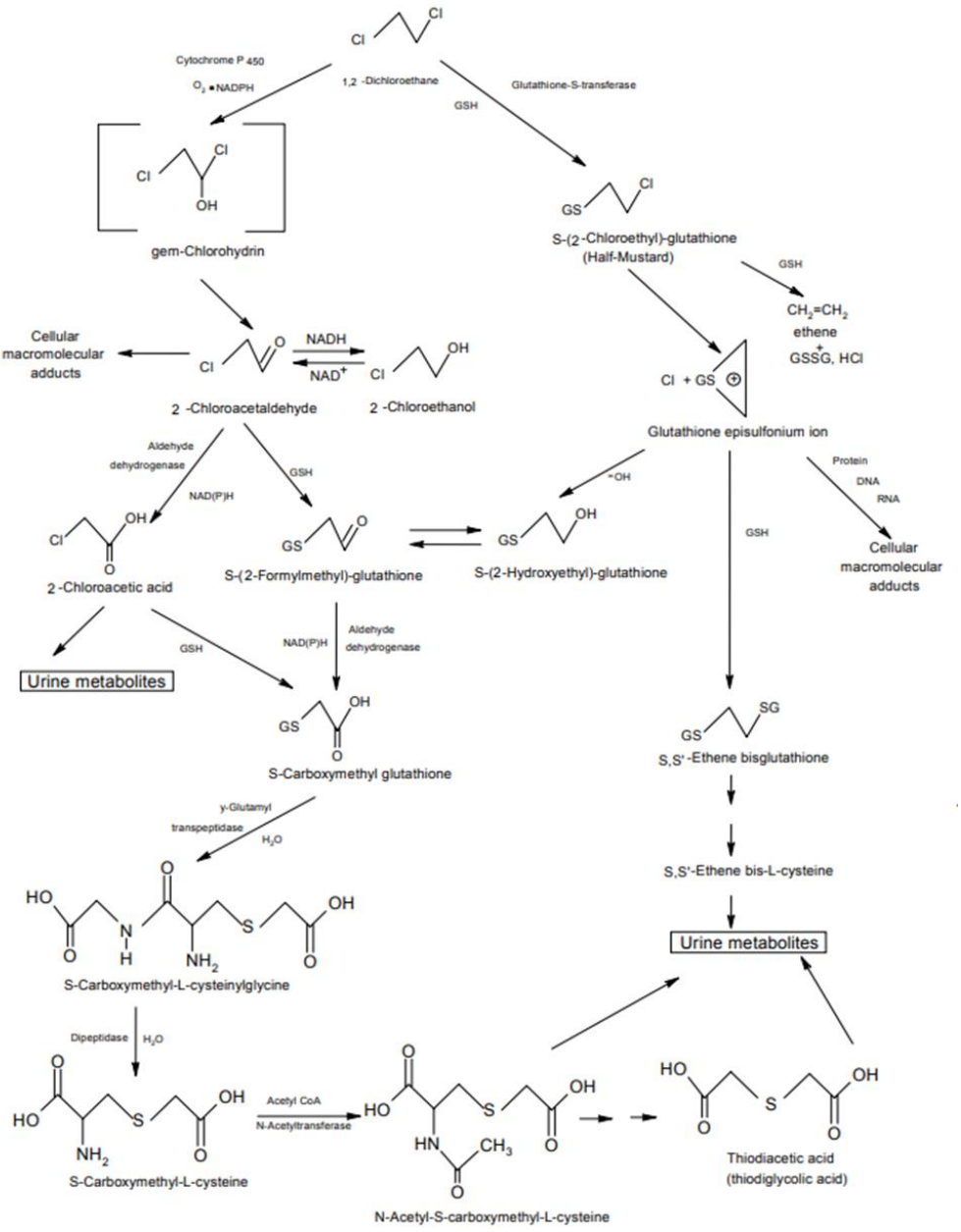
Oral Metabolism
In male rats exposed to a single oral dose of 150 mg/kg [^{14}C]-1,2-dichloroethane, 60 percent of the administered dose was detected as urinary metabolites and 29 percent was released unchanged in expired air, suggesting that metabolic saturation occurred at this dose (Reitz et al., 1982). Although urinary metabolites were not characterized in this study, a decrease in hepatic nonprotein sulfhydryl content suggests that the GSH conjugation pathway was involved.

Inhalation Metabolism
Metabolism was near complete in rats exposed to 150 ppm of [^{14}C]-1,2-dichloroethane for 6 hours, with 84 percent of radiolabel excreted as urinary metabolites and 2 percent released as unchanged compound in expired air (Reitz et al., 1982). Urinary metabolites were not characterized; however, a decrease in the hepatic nonprotein sulfhydryl content suggest involvement of the GSH conjugation pathway. In a rat inhalation study comparing blood concentrations resulting from exposure to 50 or 250 ppm, peak blood levels of 1,2-dichloroethane were 22-fold higher at the higher concentration (Spreafico et al., 1980). Taken together, these results suggest that metabolic saturation occurs at a concentration between 150 and 250 ppm 1,2-dichloroethane, corresponding to blood levels of 5 to 10 µg/mL (Reitz et al., 1982; Spreafico et al., 1980).

Dermal Metabolism
EPA did not identify *in vivo* animal data that evaluated metabolism of 1,2-dichloroethane following exposure by the dermal route.

In Vivo and In Vitro Metabolism Studies
No human studies on the metabolism of 1,2-dichloroethane were located. The primary metabolic pathways for 1,2-dichloroethane, elucidated from *in vitro* studies and *in vivo* studies in rats and mice, include CYP oxidation and GSH conjugation (Figure_Apx N-2) (NTP, 1991). Metabolism by CYP results in an unstable gem-chlorohydrin that releases hydrochloric acid, resulting in the formation of 2-chloroacetaldehyde. 2-Chloroacetaldehyde is oxidized by aldehyde dehydrogenase to form chloroacetic acid or reduced to form 2-chloroethanol, and these metabolites are conjugated with GSH and excreted in the urine (Figure_Apx N-1) (NTP, 1991). Inhibition of aldehyde dehydrogenase by the human drug disulfiram increased the blood levels of 1,2-dichloroethane 5-fold when administered *in vivo* via inhalation supporting that this pathway is important and relevant to people with the aldehyde dehydrogenase mutation having decreased activity for aldehyde clearance (Cheever et al., 1990). Metabolism via glutathione-S-transferase results in formation of S-(2-chloroethyl)-glutathione, which

rearranges to form a reactive episulfonium ion. The episulfonium ion can form adducts with protein, DNA or RNA or interact further with GSH to produce water soluble metabolites that are excreted in the urine (Figure_Apx N-2) (NTP, 1991).



Figure_Apx N-2. Proposed Metabolic Scheme for 1,2-Dichloroethane (IPCS, 1995)

In Vitro Metabolism Studies

In vitro studies using rat and human liver microsomes have demonstrated that oxidative metabolism via CYP2E1 results in the formation of 2-chloroacetaldehyde by dechlorination of an unstable chlorohydrin molecule (Guengerich et al., 1991; Casciola and Ivanetich, 1984; McCall et al., 1983; Guengerich et al., 1980). GSH conjugation of 1,2-dichloroethane was demonstrated in primary rat hepatocytes resulting in the formation of S-(2-hydroxyethyl) glutathione, S-(carboxymethyl) glutathione, and S,S'-(1,2-ethanediyl)bis(glutathione), and GSH depletion was observed (Jean and Reed, 1992). The S-(carboxymethyl) glutathione metabolite likely results from conjugation of 2-chloroacetic acid with GSH (Johnson, 1967). This metabolite can be degraded to form glycine, glutamic acid, and S-

carboxymethylcysteine, which may be oxidized to yield thiodiglycolic acid (see Figure_Apx N-2) (NTP, 1991). Metabolic rate constants were determined using rat liver microsomes and substrate concentrations between 50 µM and 1 mM (V_{max} = 0.24 nmol/minute per mg protein; K_m = 0.14 mM) (Salmon et al., 1981).

N.1.4 Elimination

N.1.4.1 1,1-Dichloroethane

Oral

The elimination pattern in rats exposed to 700 mg/kg-bw/day 1,1-dichloroethane (unlabeled) via gavage 5 days/week for 4 weeks followed by a single dose of ^{14}C-1,1-dichloroethane was as follows: 86 percent eliminated unchanged in expired air, 5 percent eliminated as CO_2, and 0.9 percent in excreta (feces and urine) at 48 hours (Mitoma et al., 1985). The total recovery was 93 percent in rats, with 1.4 percent of the administered dose remaining in the carcass. In mice exposed to 1800 mg/kg-bw/day 1,1-dichloroethane (unlabeled) via gavage 5 days/week for 4 weeks followed by a single dose of ^{14}C-1,1-dichloroethane, 70 percent of the administered dose was eliminated unchanged in expired air, 25 percent was eliminated as CO_2 in expired air, and 1.6 percent was recovered in excreta (feces and urine) at 48 hours (Mitoma et al., 1985). Total recovery in mice was 99 percent, with 2 percent remaining in the carcass.

Oral Metabolism

In male rats exposed to a single oral dose of 150 mg/kg [^{14}C]-1,2-dichloroethane, 60 percent of the administered dose was detected as urinary metabolites and 29 percent was released unchanged in expired air, suggesting that metabolic saturation occurred at this dose (Reitz et al., 1982). Although urinary metabolites were not characterized in this study, a decrease in hepatic nonprotein sulfhydryl content suggests that the GSH conjugation pathway was involved.

Inhalation

No *in vivo* animal data on elimination following exposure to 1,1-dichloroethane by the inhalation route were identified.

Dermal

EPA did not identify *in vivo* animal data that evaluated elimination following exposure to 1,1-dichloroethane by the dermal route.

EPA did not identify any PBPK models for 1,1-dichloroethane.

N.1.4.2 1,2-Dichloroethane

Oral

1,2-dichloroethane was rapidly eliminated following oral exposure, primarily via urinary excretion of water-soluble metabolites and exhalation of unchanged compound or CO_2 (Payan et al., 1993; Mitoma et al., 1985; Reitz et al., 1982). In rats given a single gavage dose of 150 mg/kg [^{14}C]-1,2-dichloroethane, elimination was 96 percent complete within 48 hours, with 60 percent of the radiolabel excreted as urinary metabolites (70% thiodiacetic acid, 26–28% thiodiacetic acid sulfoxide), 29 percent exhaled as unchanged 1,2-dichloroethane, 5 percent exhaled as CO_2, and the remaining 6 percent recovered in feces, carcass, and cage washes (Reitz et al., 1982). The elimination kinetics were described as biphasic with an initial elimination half-life (t½) of 90 minutes, followed by a t½ of approximately 20 to 30 minutes when blood levels were 5 to 10 µg/mL (Reitz et al., 1982).

In rats and mice given gavage doses of 100 and 150 mg/kg [^{14}C]-1,2-dichloroethane, respectively, following pretreatment with unlabeled 1,2-dichloroethane 5 days/week for 4 weeks, recovery of radiolabel in excreta (urine and feces) was 69.5 percent in rats and 81.9 percent in mice after 48 hours (Mitoma et al., 1985). Exhalation of volatile compounds and CO_2 accounted for 11.5 and 8.2 percent, respectively, in rats and 7.7 and 18.2 percent, respectively, in mice. The recovery of radiolabel in the carcass was 7 percent of the administered dose in rats and 2.4 percent of administered dose in mice (Mitoma et al., 1985).

The excretion of thioglycolic acid and other thioether metabolites was measured in rat urine 24 hours after gavage administration of 0.25, 0.5, 2.02, 4.04, or 8.08 mmol/kg (25, 50, 200, 400, or 800 mg/kg) [^{14}C]-1,2-dichloroethane (Payan et al., 1993). The total concentration of urinary metabolites increased linearly with administered doses between 25 and 400 mg/kg; however, the percentage of the administered dose excreted in the urine decreased with increasing dose level, likely due to metabolic saturation (ranging from 63–7.4%) (Payan et al., 1993).

Inhalation

1,2-dichloroethane was detected in expired air of women occupationally exposed to 15.6 ppm by inhalation (Urusova, 1953). Similar findings were noted in women exposed by dermal contact only (Urusova, 1953). In rats exposed via inhalation, elimination occurred by excretion of metabolites in urine and exhalation of unchanged compound or CO_2 (Reitz et al., 1982; Spreafico et al., 1980). Following inhalation of 150 ppm [^{14}C]-1,2-dichloroethane for 6 hours, elimination from the blood was near complete by 48 hours, with 84 percent of the dose detected as urinary metabolites (70% thiodiacetic acid, 26–28% thiodiacetic acid sulfoxide), 2 percent excreted unchanged in feces, and 7 percent exhaled as CO_2 (Reitz et al., 1982). The elimination kinetics of 1,2-dichloroethane in rats were described as monophasic with t½ values of 12.7 and 22 minutes at inhalation concentrations of 25 and 250 ppm 1,2-dichloroethane, respectively (Spreafico et al., 1980). Excretion was dose-dependent, with the percentage exhaled as unchanged 1,2-dichloroethane increased at the highest concentration; elimination from adipose tissue was slower than elimination from blood, liver, or lung (Spreafico et al., 1980).

In mice exposed to 25, 87, or 185 ppm 1,2-dichloroethane for 6 hours, elimination was rapid, with clearance of parent compound from the blood near complete within 1 hour after exposure (Zhong et al., 2022; Liang et al., 2021). In a 28-day study using the same concentrations for 6 hours/day, 5 days/week, 2-chloroacetic acid was detected as the primary metabolite in urine at concentrations of 300, 1,000, and 1,300 µg/L, respectively, supporting that the aldehyde dehydrogenase pathway is important for clearance with relevance to people with this enzyme deficiency (Zhong et al., 2022; Liang et al., 2021).

Dermal

1,2-dichloroethane was detected in expired air of women occupationally exposed by dermal contact only (gas masks were worn to prevent inhalation) (Urusova, 1953).

Physiologically-Based Pharmacokinetic (PBPK) Modeling Approach

Two PBPK models were developed to describe the disposition of 1,2-dichloroethane. The D'Souza (1988; 1987) Model used five compartments (lung, liver, richly perfused tissues, slowly perfused tissues, and fat) and assumed that metabolism occurs only in the liver and lung. Metabolic pathways included a saturable oxidation pathway and GSH conjugation. This PBPK model, which was validated in rats and mice, predicted that inhalation produces less GSH-conjugate metabolites (measured as GSH depletion in the liver) than gavage exposure.

Sweeney et al. (2008) extended and updated the D'Souza et al. (1987; 1988) Model by adding two gastrointestinal compartments—a compartment for the kidney and an additional metabolism pathway for extrahepatic enzymes. Model parameter values that were revised included the oral absorption rate, time delay constant for GSH synthesis following depletion, and GSH levels in liver and lung. Model predictions were compared to experimental rat data for intravenous, oral, and inhalation routes, and the model performed well for single and repeated exposure. Because the model has not been validated in humans, it is unclear whether this model would be useful for extrapolating between rats and humans.

N.2 Non-Cancer Dose-Response Assessment

Appendices N.2.1 and N.2.2 describe dose-response assessment for 1,1-dichloroethane and 1,2-dichloroethane, respectively. Appendices N.2.3, N.2.4, and N.2.5 describe the non-cancer POD derivation for acute, short/intermediate-term, and chronic durations for 1,1-dichloroethane. Appendices N.2.6, N.2.7, and N.2.8 describe the non-cancer POD derivation for acute, short-term/intermediate-term, and chronic durations for 1,2-dichloroethane. Appendix N.4 provides the equations used in derivation of non-cancer and cancer PODs for the 1,1-Dichloroethane Risk Assessment. Finally, Appendix N.5 provides a summary of the non-cancer PODs selected for use in the risk assessment for 1,1-dichloroethane based on read-across from 1,2-dichloroethane, including PODs for both continuous and occupational exposure scenarios.

N.2.1 Non-Cancer Dose-Response Assessment for 1,1-Dichloroethane

EPA evaluated data from studies with adequate quantitative information and sufficient sensitivity as described in Sections 5.2.3.1 and 5.2.7.1. In order to characterize the dose-response relationships of 1,1-dichloroethane. The database for 1,1-dichloroethane toxicity in animals is very limited and many of the available studies were rated Unacceptable/Uninformative for dose response. Table_Apx N-7 shows the studies that were excluded from consideration for dose-response assessment along with the reason for excluding each.

Table_Apx N-7. Studies Not Considered Suitable for PODs for 1,1-Dichloroethane

Reference	Study Rating	Reason Not Suitable for POD
Dow Chemical (1947)	Unacceptable	Rating (based on dermal irritation)
Plaa and Larson (1965)	Unacceptable	Rating
Mellon Institute (1947)	Unacceptable	Rating
Hofmann et al. (1971)	Unacceptable	Rating
Vozovaia (1977)	Unacceptable	Rating
NCI (1978); Rat	Unacceptable	Rating
Weisburger (1977)	Unacceptable	Rating; reports same data as NCI (1978)
Story et al. (1986)	Medium	Reports same data as Milman et al. (1988)
Natsyuk and Chekman (1975)	Low	Tested chemical is uncertain (reported only as dichloroethane)
Natsyuk and Fedurov (1974)	Unacceptable	Rating; tested chemical is uncertain (reported only as dichloroethane)

In addition to the studies above, the EPA mechanistic study by Milman et al. (1988) was excluded from consideration for POD selection. Milman et al. (1988) examined GGT+ (gamma-glutamyl transferase levels) foci in the liver in rats exposed to 1,1-dichloroethane in four separate experiments in a standard tumor initiation/promotion protocol. In the initiation experiments, the rats were exposed once to 1,1-

dichloroethane 1 day after a two-thirds partial hepatectomy, and then were either treated with phenobarbital or vehicle for 7 weeks. 1,1-Dichloroethane did not increase the number of GGT+ foci under either condition, indicating that a single exposure of this chemical is not a tumor initiator. In the promotion experiments, the rats were pretreated (intraperitoneal) with a single dose of diethylnitrosamine or water 1 day after two-thirds partial hepatectomy; 6 days later, the rats were given 1,1-dichloroethane by gavage 5 days/week for 7 weeks. In animals pretreated with diethylnitrosamine, there was a significantly increased number of GGT+ liver foci 2.1-fold higher than the control group, indicating that 1,1-dichloroethane is a tumor promoter. In animals pretreated with water followed by 1,1-dichloroethane, the number of foci was higher than in controls, but the number was not statistically significantly different from control. Other non-cancer endpoints examined in the study were body weight and liver weight; no statistically significant effects were observed in any of the experiments with 1,1-dichloroethane. Milman et al. (1988) was not considered suitable for POD identification for 1,1-dichloroethane because (1) all animals in all experiments were partially hepatectomized prior to treatment, and (2) the only statistically significant effect (increased GGT+ foci) was seen in animals that were pretreated with a single dose of diethylnitrosamine (DEN). However, the group treated with 1,1-dichloroethane alone but without DEN treatment had a robust response 4.3-fold higher than the control group, but due to low animal numbers the results did not have a statistically significant outcome ($p < 0.05$).

Excluding the study by Milman et al. (1988), as well and those provided in Table_Apx N-7, leaves the studies included in Table_Apx N-8 for potential use in POD derivation.

Table_Apx N-8. Summary of Studies Considered for Non-Cancer Dose-Response Assessment of 1,1-Dichloroethane

Reference	Duration Category (Duration)	Species, Strain, and Sex	Study Rating for Non-Cancer Endpoints
Oral			
Dow Chemical (1947)	Acute (once)	Guinea pig	Low
Muralidhara et al. (2001)	Acute (once)	Rat (Sprague-Dawley, male)	Medium
Muralidhara et al. (2001)	Short/intermediate-term (10 days)	Rat (Sprague-Dawley, male)	High
Ghanayem et al. (1986)	Short/intermediate-term (2 weeks)	Rat (F344, male)	Medium
Muralidhara et al. (2001)	Short/intermediate-term (13 weeks)	Rat (Sprague-Dawley, male)	High
Klaunig et al. (1986)	Chronic (52 weeks)	Mouse (B6C3F1, male)	High
NCI (1978)	Chronic (78 weeks)	Mouse (B6C3F1, male and female)	High
Inhalation			
Schwetz et al. (1974)	Short/intermediate-term (10 days)	Rat (Sprague-Dawley, female)	Medium-High
Mellon Institute (1947)	Chronic (26 weeks)	Dog, mongrel	Medium
Hofmann et al. (1971)	Chronic (26 weeks)	Rat, guinea pig, rabbit	Medium

Reference	Duration Category (Duration)	Species, Strain, and Sex	Study Rating for Non-Cancer Endpoints
Dermal			
No data			

No dermal exposure studies received acceptable ratings for 1,1-dichloroethane. Due to the extremely small number of available studies, limited evaluations performed in many studies, and paucity of information available to identify target organs for 1,1-dichloroethane, overall NOAELs and LOAELs were identified for each study, rather than identifying NOAELs and LOAELs by organ/system. Table_Apx N-9 and Table_Apx N-10 summarize the NOAELs and LOAELs identified from the oral and inhalation studies, respectively. Each NOAEL and LOAEL was converted to reflect continuous exposure ($NOAEL_{continuous}$ and $LOAEL_{continuous}$) using Equation_Apx N-4 and Equation_Apx N-5. After adjustment for continuous exposure, each oral NOAEL and LOAEL was converted to a HED using Equation_Apx N-6 and each inhalation NOAEL and LOAEL was converted to a HEC using Equation_Apx N-8. Dose-response considerations for these studies are briefly described below. Benchmark dose (BMD) modeling results are provided in *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Benchmark Dose Modeling* (U.S. EPA, 2025f).

Table_Apx N-9. Summary of Candidate Non-Cancer Oral PODs for 1,1-Dichloroethane

Species (Strain, Sex, n/Group)	Exposure	NOAEL (mg/kg-bw/day)	LOAEL (mg/kg-bw/day)	Effect(s)	Candidate POD (mg/kg-bw/day) (POD type)	Reference	Study Rating for Non-Cancer (Significant Limitations)
Acute							
Guinea pig (strain, sex, and number/group not specified)	Once ("fed")	NOAEL: 300 $NOAEL_{continuous}$: 300 $NOAEL_{HED}$: 81	LOAEL: 1,000 $LOAEL_{continuous}$: 1,000 LOAEL $_{HED}$: 271	100% mortality	81 ($NOAEL_{HED}$)	Dow Chemical (1947)	Low (no control; strain, sex, number/group, method of administration, and duration of follow-up not reported)
Rat (Sprague-Dawley, 8 males/group)	Once (gavage)	NOAEL: 1,000 $NOAEL_{continuous}$: 1,000 $NOAEL_{HED}$: 240	LOAEL: 2,000 $LOAEL_{continuous}$: 2,000 $LOAEL_{HED}$: 480	Sedation	240 ($NOAEL_{HED}$)	Muralidhara et al. (2001)	Medium (evaluated only clinical signs and mortality)
Short/intermediate-term							
Rat (Sprague-Dawley, 24 males/group)	10 days (gavage)	NOAEL: 1,000 $NOAEL_{continuous}$: 1,000 $NOAEL_{HED}$: 240	LOAEL: 2,000 $LOAEL_{continuous}$: 2,000 LOAEL $_{HED}$: 480	≥10% decrease in body weight	1,167 ($BMDL_{10\%}$ for body weight) 280 ($BMDL_{10\%\ HED}$ for body weight)	Muralidhara et al. (2001)	High
Rat (F344, 8 males/group)	2 weeks 5 days/week (gavage)	NOAEL: 700 $NOAEL_{continuous}$: 500 $NOAEL_{HED}$: 120	ND	None	120 ($NOAEL_{HED}$)	Ghanayem et al. (1986)	Medium (evaluated only forestomach histopathology)
Rat (Sprague-Dawley, 15 males/group)	13 weeks, 5 days/week (gavage)	NOAEL: 1,000 $NOAEL_{continuous}$: 714 $NOAEL_{HED}$: 171	LOAEL: 2,000 $LOAEL_{continuous}$: 1,429 $LOAEL_{HED}$: 343	Mortality (1/15 rats); CNS depression; ≥10% decrease in body weight	171 ($NOAEL_{HED}$) 1,248 ($BMDL_{10\%}$ for body weight) 300 ($BMDL_{10\%\ HED}$ for body weight)	Muralidhara et al. (2001)	High
Chronic							
Mouse (B6C3F1, 35 males/group)	52 weeks, 7 days/week (drinking water)	$NOAEL_{continuous}$: 543 $NOAEL_{HED}$: 71	ND	None	71 ($NOAEL_{HED}$)	Klaunig et al. (1986)	High (evaluated only body weight and liver, kidney, and lung weight and histopathology)

Species (Strain, Sex, n/Group)	Exposure	NOAEL (mg/kg-bw/day)	LOAEL (mg/kg-bw/day)	Effect(s)	Candidate POD (mg/kg-bw/day) (POD type)	Reference	Study Rating for Non-Cancer (Significant Limitations)
Mouse (B6C3F1, 50 males and 50 females/group)	15–78 weeks, 5 days/week (gavage)	NOAEL (time-weighted across weeks as reported by NCI): 1,665 (F) $NOAEL_{continuous}$ (adjusted for 5/7 days/week) 1,189 (F) $NOAEL_{HED}$: 155 (F)	LOAEL (time-weighted across weeks as reported by NCI): 3,331 (F) $LOAEL_{continuous}$ (adjusted for 5/7 days/week): 2,379 (F) $LOAEL_{HED}$: 309 (F)	Decreased survival	155 (F) ($NOAEL_{HED}$)	NCI (1978)	High

Table_Apx N-10. Summary of Candidate Non-Cancer Inhalation PODs for 1,1-Dichloroethane

Species (Strain, Sex, n/Group)	Exposure	NOAEL	LOAEL	Effect	Candidate POD (POD Type)	Reference	Study Rating for Non-Cancer (Significant Limitations)
Acute							
No data							
Short/intermediate-term							
Rat (Sprague-Dawley, 20 females/group)	10 days GD 6–15, 7 hours/day	ND	LOAEL: 15,372 mg/m^3 (3,798 ppm) $LOAEL_{continuous}$ = $LOAEL_{HEC}$: 4,485 mg/m^3 (1,108 ppm)	Decreased maternal body weight (9–11% less than controls) on GD 13	4,525 mg/m^3 or 1,118 ppm ($BMCL_{HEC}$)	Schwetz et al. (1974)	High for body weight; medium for other endpoints
Chronic							
Rat (Sprague-Dawley, 5/sex/group), guinea pig (Pirbright-White, 5/sex/group), and rabbit (strain not specified, 2/sex/group)	26 weeks 5 days/week 6 hours/day	NOAEL: 3,036 mg/m^3 (750 ppm) $NOAEL_{continuous}$ = $NOAEL_{HEC}$: 542 mg/m^3 (134 ppm)	ND	No effect on any species	542 mg/m^3 or 134 ppm ($NOAEL_{HEC}$)	(Hofmann et al., 1971)	Medium (histopathology evaluations limited to liver and kidney)
Dog (mongrel, 1 male/group)	6 months, 3.5 days/week, 7 hours/day	ND	LOAEL: 4,319 mg/m^3 (1,067 ppm) $LOAEL_{adj}$ = $LOAEL_{HEC}$: 630 mg/m^3 (156 ppm)	Decreased body weight (magnitude unknown); lung congestion	630 mg/m^3 or 156 ppm ($LOAEL_{HEC}$)	Mellon Institute (1947)	Medium (one dog, body weight reported as percentage of starting weight)

N.2.2 Non-Cancer Dose-Response Assessment for 1,2-Dichloroethane

According to the U.S. EPA (2021c) Draft Systematic Review Protocol, hazard endpoints that receive evidence integration judgments of *demonstrates* and *likely* would generally be considered for dose-response analysis. Endpoints with *suggestive* evidence can be considered on a case-by-case basis. Studies that received high or medium overall quality determinations (or low-quality studies if no other data are available) with adequate quantitative information and sufficient sensitivity can be compared. The only hazard outcome for which evidence *demonstrates* that 1,2-dichloroethane causes the effect was mortality. For neurological/behavioral effects, EPA's evidence integration judgment was *likely*. For nutritional/metabolic, renal/kidney, hepatic/liver, lung/respiratory, immune/hematological, and reproductive effects, the Agency's evidence integration conclusion was that the evidence was *suggestive*. Finally, EPA concluded that the available evidence was *inadequate* to determine whether 1,2-dichloroethane induces developmental effects.

No human studies provided adequate information for POD determination. Animal studies of oral, inhalation, or dermal exposure that received high- or medium-quality determinations for one or more of these health outcomes were considered for dose-response information, with some exceptions. Studies that identified a NOAEL at the highest dose/concentration tested were not considered for dose-response assessment but were considered as part of evidence integration for the relevant health outcomes. In addition, acute lethality studies that did not include untreated or vehicle-treated controls, or other studies that did not present sufficient information to determine a NOAEL or LOAEL were not considered. Finally, only studies in intact, wild-type laboratory animal strains were considered for dose-response assessment. A small number of studies using partially-hepatectomized animals or transgenic models were excluded from consideration, as shown in the tables.

Table_Apx N-11, Table_Apx N-12, and Table_Apx N-13 show the animal studies of oral, inhalation, and dermal exposure (respectively) that were excluded from consideration for dose-response assessment along with the reason for excluding each.

Table_Apx N-11. Oral Studies Not Considered Suitable for PODs for 1,2-Dichloroethane

Duration Category	Reference	HERO ID	Species	Specific Route	Rationale
Acute	Cottalasso et al. (1995)	200280	Rat	Gavage	Not suitable for POD due to dosing uncertainties
Acute	Dow Chemical (2006a)	625286	Rat	Gavage	Freestanding NOAEL[a]
Acute	Kettering Laboratory (1943)	4528351	Rabbit	Gavage	Uninformative
Acute	Kitchin et al. (1993)	6118	Rat	Gavage	Freestanding NOAEL[a]
Acute	Mellon Institute (1948)	5447301	Rat	Gavage	Uninformative
Acute	Mellon Institute (1948)	5447301	Mouse	Gavage	Uninformative
Acute	Mellon Institute (1948)	5447301	Rabbit	Gavage	Uninformative
Acute	Moody et al. (1981)	18954	Rat	Gavage	Not suitable for POD; evaluation limited to liver weight and data not shown
Acute	Munson et al. (1982)	62637	Mouse	Gavage	Low
Acute	Stauffer Chem Co (1973)	6569955	Rat	Gavage	Not suitable for POD; no control group
Acute	Milman et al. (1988)	200479	Rat	Gavage	Study of partially hepatectomized animals

Duration Category	Reference	HERO ID	Species	Specific Route	Rationale
Acute	Zabrodskii et al. (2004)	1048005	Rat	Gavage	Freestanding LOAEL; tested chemical is uncertain (reported only as dichloroethane) but metabolites listed indicative of 1,2-dichloroethane
Short-term	Dow Chemical (2006a)	625286	Rat	Gavage	Freestanding NOAEL[a]
Short-term	NTP (1978)	5441108	Mouse	Gavage	Freestanding NOAEL[a]
Subchronic	Milman et al. (1988)	200479	Rat	Gavage	Mechanistic study for tumor initiation/promotion, study of partially hepatectomized animals
Subchronic	Alumot et al. (1976)	194588	Rat	Diet	Freestanding NOAEL[a] (for 5-week female and 13-week male growth studies); not suitable for POD due to dosing uncertainties (for 5- to 7-week preliminary study)
Subchronic	NTP (1991)	1772371	Rat	Drinking water	Uninformative
Subchronic	NTP (1991)	1772371	Mouse	Drinking water	Uninformative
Subchronic	Munson et al. (1982)	62637	Mouse	Drinking water	Uninformative
Chronic	Alumot et al. (1976)	194588	Rat	Diet	Uninformative
Chronic	Klaunig et al. (1986)	200427	Mouse	Drinking water	Not suitable for POD due to reporting limitations
Chronic	Storer et al. (1995)	200612	Mouse	Gavage	Study of transgenic mice predisposed to cancer
Chronic	NTP (1978)	5441108	Mouse	Gavage	Not suitable for POD due to confounding by tumors
Chronic	NTP (1978)	5441108	Rat	Gavage	Uninformative
Reproduction/ Developmental	Lane et al. (1982)	62609	Mouse	Drinking water	Freestanding NOAEL[a]
Reproduction/ Developmental	WIL Research (2015)	7310776	Rat	Drinking water	Uninformative
Reproduction/ Developmental	Alumot et al. (1976)	194588	Rat	Diet	Uninformative

[a] No effects observed at highest dose tested for all apical health outcomes rated Low or higher.

Table_Apx N-12. Inhalation Studies Not Considered Suitable for PODs for 1,2-Dichloroethane

Duration Category	Reference	HERO ID	Species	Rationale
Acute	Brondeau et al. (1983)	200247	Rat	Not suitable for POD due to limited evaluations
Acute	Dow Chemical (2005)	10699112	Rat	Not suitable for POD determination; no control group
Acute	Dow Chemical (2017)	10699356	Rat	Not suitable for POD determination; no control group

Duration Category	Reference	HERO ID	Species	Rationale
Acute	Sherwood et al. (1987)	200590	Rat	Freestanding NOAEL[a]
Acute	Guo and Niu (2003)	200352	Rat	Uninformative
Acute	Jin et al. (2018b); Jin et al. (2018a)	5431556, 5557200	Mouse	Uninformative
Acute	Mellon Institute (1948)	5447301	Rat	Uninformative
Acute	Mellon Institute (1948)	5447301	Rabbit	Uninformative
Acute	Mellon Institute (1948)	5447301	Mouse	Uninformative
Acute	Spencer et al. (1951)	62617	Rat	Not suitable for POD determination; no control group
Acute	Zhang et al. (2011)	734177	Rat	Uninformative
Short-term	Brondeau et al. (1983)	200247	Rat	Not suitable for POD due to limited evaluations
Short-term	Dow Chemical (2014)	10609985	Rat	Freestanding NOAEL[a]
Short-term	Jin et al. (2018b); Jin et al. (2018a)	5431556, 5557200	Mouse	Uninformative
Short-term	Li et al. (2015b)	4492694	Rat	Uninformative
Short-term	Pang et al. (2018)	4697150	Rat	Uninformative
Short-term	Sherwood et al. (1987)	200590	Rat	Freestanding NOAEL[a]
Short-term	Sherwood et al. (1987)	200590	Mouse	Freestanding NOAEL[a]
Short-term	Spencer et al. (1951)	62617	Rat	Uninformative
Short-term	Spencer et al. (1951)	62617	Guinea pig	Uninformative
Short-term	Sun et al. (2016c)	4451633	Mouse	Uninformative
Short-term	Wang et al. (2013)	1522109	Mouse	Uninformative
Short-term	Wang et al. (2014)	4453007	Mouse	Uninformative
Short-term	Zhang and Jin (2019)	5556105	Mouse	Uninformative
Subchronic	Hofmann et al. (1971)	1937626	Rat	Uninformative
Subchronic	Hofmann et al. (1971)	1937626	Guinea pig	Uninformative
Subchronic	Hofmann et al. (1971)	1937626	Cat	Not suitable for POD due to reporting limitations and small group size[b]
Subchronic	Hofmann et al. (1971)	1937626	Rabbit	Uninformative
Subchronic	Kettering Laboratory (1943)	4528351	Rabbit	Uninformative
Chronic	Cheever et al. (1990)	12097	Rat	Freestanding NOAEL[a]
Chronic	Hofmann et al. (1971)	1937626	Rat	Freestanding NOAEL[a] (17- and 26-week experiments)
Chronic	Hofmann et al. (1971)	1937626	Rabbit	Freestanding NOAEL[a] (17- and 26-week experiments)
Chronic	Hofmann et al. (1971)	1937626	Guinea pig	Freestanding NOAEL[a] (17- and 26-week experiments)

Duration Category	Reference	HERO ID	Species	Rationale
Chronic	Hofmann et al. (1971)	1937626	Cat	Freestanding NOAEL[a] (17-week experiment); Uninformative (26-week experiment)
Chronic	IRFMN (1976)	5447359	Rat	Freestanding NOAEL[a]
Chronic	IRFMN (1987)	94773	Rat	Freestanding NOAEL[a]
Chronic	IRFMN (1987)	94773	Mouse	Freestanding NOAEL[a]
Chronic	IRFMN (1987)	5447260	Rat	Freestanding NOAEL[a]
Chronic	Mellon Institute (1947)	1973131	Rat	Uninformative
Chronic	Mellon Institute (1947)	1973131	Dog	Not suitable for POD due to reporting limitations and small group size[b]
Chronic	Nagano et al. (2006)	200497	Rat	Freestanding NOAEL for non-cancer hazards[a]
Chronic	Nagano et al. (2006)	200497	Mouse	Not suitable for POD due to confounding by tumors
Chronic	Spencer et al. (1951)	62617	Rat	Not suitable for POD due to variable exposure durations and reporting limitations
Chronic	Spencer et al. (1951)	62617	Guinea pig	Not suitable for POD due to variable exposure durations and reporting limitations
Chronic	Spencer et al. (1951)	62617	Rabbit	Not suitable for POD due to variable exposure durations, reporting limitations, and small group size[b]
Chronic	Spencer et al. (1951)	62617	Monkey	Not suitable for POD due to variable exposure durations, reporting limitations, and small group size[b]
Reproduction/ Developmental	Rao et al. (1980)	5453539	Rat	Freestanding NOAEL[a] (one-generation reproduction study)
Reproduction/ Developmental	Zhao et al. (1997)	77864	Rat	Uninformative
Reproduction/ Developmental	Zhao et al. (1989)	200708	Rat	Uninformative
Reproduction/ Developmental	Zhao et al. (1989)	200708	Mouse	Uninformative

[a] No effects observed at highest dose tested for all apical health outcomes rated Low or higher.
[b] Group size of 1–2 per exposure level.

Table_Apx N-13. Dermal Studies Not Considered Suitable for PODs for 1,2-Dichloroethane

Duration Category	Reference	HERO ID	Species	Rationale
Acute	Kronevi et al. (1981)	58151	Guinea pig	Uninformative
Acute	Van Duuren et al. (1979)	94473	Mouse	Uninformative
Acute	Dow Chemical (1956)	725343	Rabbit	Low (no control; LD_{50} study)
Acute	Kettering Laboratory (1943)	4528351	Rabbit	Uninformative
Acute	Dow Chemical (1962)	5447286	Cattle	Low (no sex, strain or n/group reported)
Acute	Mellon Institute (1948)	5447301	Rabbit	Uninformative

Acute	Stauffer Chem Co (1973)	6569955	Rabbit	Negative for skin and eye irritation
Chronic	Van Duuren et al. (1979)	94473	Mouse	Uninformative
Chronic	Suguro et al. (2017)	4451542	Mouse	Cancer study, single dose, dosing only 3 times per week

Table_Apx N-14 shows the studies considered for potential use in POD derivation.

Table_Apx N-14. Summary of Studies Considered for Non-Cancer, Dose-Response Assessment of 1,2-Dichloroethane

Reference	Duration Category (Duration)	Species, Strain, and Sex	Study Rating for Non-Cancer Endpoints
Oral			
Storer et al. (1984)	Acute (once by gavage)	Mouse (B6C3F1, male)	High
Morel et al. (1999)	Acute (once by gavage)	Mouse (Swiss OF1, male)	High
Cottalasso et al. (2002)	Acute (once by gavage)	Rat (Sprague-Dawley, female)	Medium
Salovsky et al. (2002)	Acute (once by gavage)	Rat (Wistar, male)	Medium
Daniel et al. (1994)	Short-term (10 days by daily gavage)	Rat (Sprague-Dawley, male and female)	High
Munson et al. (1982)	Short-term (14 days by daily gavage)	Mouse (CD-1, male)	High
van Esch et al. (1977)	Short-term (2 weeks by gavage 5 days/week)	Rat (Wistar, male)	High
NTP (1978)	Short-term (6 weeks by gavage 5 days/week)	Rat (Osborne-Mendel, male and female)	Medium
Daniel et al. (1994)	Subchronic (90 days by daily gavage)	Rat (Sprague-Dawley, male and female)	High
van Esch et al. (1977)	Subchronic (90 days by gavage 5 days/week)	Rat (Wistar, male and female)	High
NTP (1991)	Subchronic (13 weeks by gavage, 5 days/week)	Rat (F344, males and female)	High
Payan et al. (1995)	Repro/Dev (15 days, GD 6–20 by daily gavage)	Rat (Sprague-Dawley, female)	High
Inhalation			
Francovitch et al. (1986)	Acute (4 hours)	Mouse (CD, male)	Medium
Storer et al. (1984)	Acute (4 hours)	Mouse (B6C3F1, male)	High
Dow Chemical (2006b)	Acute (4 or 8 hours)	Rat (F344/ DUCRL, male and female)	High
Sherwood et al. (1987)	Acute (3 hours)	Mouse (CD-1, female)	High
Zhou et al. (2016)	Acute (1.5 or 4 hours)	Rat (Sprague-Dawley, male)	Medium
Zhang et al. (2010)	Acute (12 hours)	Rat (Sprague-Dawley, male and female)	Medium

Reference	Duration Category (Duration)	Species, Strain, and Sex	Study Rating for Non-Cancer Endpoints
Igwe et al. (1986b)	Short-term (30 days; 5 days/week; 7 hours/day)	Rat (Sprague-Dawley, male)	High
Zhang et al. (2017)	Short-term (1 or 4 weeks; 6 hours/day)	Mouse (Swiss, male)	High
Zeng et al. (2018)	Short-term (28 days; 6 hours/day)	Mouse (Swiss, male)	High
IRFMN (1978)	Chronic (12 months; 5 days/week; 7 hours/day)	Rat (Sprague-Dawley, male and female)	Medium
Rao et al. (1980)	Repro/Dev (10 days; 7 hours/day; GD 6–15)	Rat (Sprague-Dawley, female)	Medium
Rao et al. (1980)	Repro/Dev (13 days; 7 hours/day; GD 6–18)	Rabbit (New Zealand White, female)	Medium
Payan et al. (1995)	Repro/Dev (15 days; 6 hours/day; GD 6–20)	Rat (Sprague-Dawley, female)	High
Dermal			
No data			

No dermal exposure studies of 1,2-dichloroethane were considered suitable for use in determining a POD. Table_Apx N-15 through Table_Apx N-19 summarize the NOAELs and LOAELs identified from the oral (acute and short-term/subchronic) and inhalation (acute, short-term/subchronic, and chronic) studies, respectively. Only the endpoint with the lowest LOAEL for a given study was included in the table (if the lowest LOAEL was for multiple endpoints, all were included in the table). Each NOAEL and LOAEL was converted to reflect continuous exposure ($NOAEL_{continuous}$ and $LOAEL_{continuous}$) using Equation_Apx N-4 and Equation_Apx N-5. After adjustment for continuous exposure, each oral NOAEL and LOAEL was converted to a HED using Equation_Apx N-6 and each inhalation NOAEL and LOAEL was converted to a HEC using Equation_Apx N-7 (for extrarespiratory effects) or Equation_Apx N-8 (for nasal effects).

Table_Apx N-15. Summary of Candidate Acute, Non-Cancer, Oral PODs for 1,2-Dichloroethane

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL (mg/kg-bw)	LOAEL (mg/kg-bw)	Basis for NOAEL/LOAEL	Candidate POD[b] (mg/kg-bw) (POD Type)	Reference	Study Rating for Target Organ/System
Renal/Kidney (*evidence suggests*)	Mouse (B6C3F1, 5 males/group)	Once (gavage)	NOAEL: 200 $NOAEL_{HED}$: 26.0	LOAEL: 300 $LOAEL_{HED}$: 39.0	Significantly increased relative kidney weight (13% higher than controls)	19.9 ($BMDL_{10\%\ HED}$ for kidney weight)	Storer et al. (1984)	High
	Mouse (Swiss OF1, 10 males/group)	Once (gavage)	NOAEL: 1,000 $NOAEL_{HED}$: 130	LOAEL: 1,500 $LOAEL_{HED}$: 195	Increased percentage of damaged proximal tubules	130 ($NOAEL_{HED}$)	Morel et al. (1999)	High
Hepatic/Liver (*evidence suggests*)	Rat (Sprague-Dawley; 10 females/group)	Once (gavage)	ND	LOAEL: 628 $LOAEL_{HED}$: 151	Significantly increased ALT, AST, and LDH (45, 44, and 67% higher than controls, respectively) and liver steatosis	151 ($LOAEL_{HED}$)	Cottalasso et al. (2002)	Medium
Respiratory (*evidence suggests*)	Rat (Wistar, 4–6 males/group)	Once (gavage)	ND	LOAEL: 136 $LOAEL_{HED}$: 32.6	Significantly increased total number of cells in BALF; inflammatory and noninflammatory histological changes in lung (data reported qualitatively)	32.6 ($LOAEL_{HED}$)	Salovsky et al. (2002)	Medium

Table_Apx N-16. Summary of Candidate Short-Term/Intermediate, Non-Cancer, Oral PODs for 1,2-Dichloroethane[a]

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL (mg/kg-bw/day)	LOAEL (mg/kg-bw/day)	Basis for NOAEL/LOAEL	Candidate POD [b] (mg/kg-bw/day) (POD Type)	Reference	Study Rating for Target Organ/System
Mortality (*evidence demonstrates*)	Rat (SPF Wistar, 6 males/group)	2 weeks (gavage, 5 days/week)	NOAEL: 100 $NOAEL_{continuous}$: 71.4 $NOAEL_{HED}$: 7.1	LOAEL: 300 $LOAEL_{continuous}$: 214 $LOAEL_{HED}$: 51.4	Mortality in all animals (6/6 animals by day 5)	17.1 ($NOAEL_{HED}$)	van Esch et al. (1977)	High
Nutritional/ Metabolic (*evidence suggests*)	Rat (Sprague-Dawley; 25–26 females/group)	15 days GD 6–20 (daily gavage)	$NOAEL_{continuous}$: 158 $NOAEL_{HED}$: 37.9	$LOAEL_{continuous}$: 198 $LOAEL_{HED}$: 47.5	Decreased absolute maternal body weight gain [c] on GD 6–21 (reduced ≥30% relative to controls)	10.0 ($BMDL_{10\%\ HED}$ for maternal body weight)	Payan et al. (1995)	High
	Rat (Osborne-Mendel, 5/sex/group)	6 weeks (gavage, 5 days/week)	ND	LOAEL:40 $LOAEL_{continuous}$: 29 $LOAEL_{HED}$: 7.0	Decreased body weights (10%) in females	7.0 ($LOAEL_{HED}$)	NTP (1978)	Medium
Hepatic/Liver (*evidence suggests*)	Rat (Sprague-Dawley; 10/sex/group)	10 days (gavage, daily)	$NOAEL_{continuous}$: 30 $NOAEL_{HED}$: 7.2	$LOAEL_{continuous}$: 100 $LOAEL_{HED}$: 24	Significantly increased relative liver weights (14% relative to controls) and serum cholesterol levels (data not shown) in males	7.2 ($NOAEL_{HED}$)	Daniel et al. (1994)	High
	Rat (Sprague-Dawley; 10/sex/group)	90 days (gavage, daily)	$NOAEL_{continuous}$: 37.5 $NOAEL_{HED}$: 9.00	$LOAEL_{continuous}$: 75 $LOAEL_{HED}$: 18	Significantly increased relative liver weight (20% higher than controls) and serum ALP (data not shown) in males	9.00 ($NOAEL_{HED}$)	Daniel et al. (1994)	High
	Rat (SPF Wistar, 10/sex/group)	90 days (gavage, 5 days/week)	NOAEL: 30 $NOAEL_{continuous}$: 21 $NOAEL_{HED}$: 5.0	LOAEL: 90 $LOAEL_{continuous}$: 64 $LOAEL_{HED}$: 15	Significantly increased relative liver weight (13% higher than controls) in females	5.0 ($NOAEL_{HED}$)	van Esch et al. (1977)	Medium

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL (mg/kg-bw/day)	LOAEL (mg/kg-bw/day)	Basis for NOAEL/LOAEL	Candidate POD [b] (mg/kg-bw/day) (POD Type)	Reference	Study Rating for Target Organ/System
Renal/ Kidney (*evidence suggests*)	Rat (Sprague-Dawley; 10/sex/group)	90 days (gavage, daily)	$NOAEL_{continuous}$: 37.5 $NOAEL_{HED}$: 9.00	$LOAEL_{continuous}$: 75 $LOAEL_{HED}$: 18	Significantly increased relative kidney weights in males and females (18 and 15% higher than controls, respectively)	9.00 ($NOAEL_{HED}$)	Daniel et al. (1994)	High
	Rat (SPF Wistar, 10/sex/group)	90 days (gavage, 5 days/week)	NOAEL: 30 $NOAEL_{continuous}$: 21 $NOAEL_{HED}$: 5.0	LOAEL:90 $LOAEL_{continuous}$: 64 $LOAEL_{HED}$: 15	Significantly increased relative kidney weight (17 and 16% higher than controls in males and females, respectively)	5.0 ($NOAEL_{HED}$)	van Esch et al. (1977)	Medium
	Rat (F344; 10/sex/group)	13 weeks (gavage, 5 days/week)	ND	LOAEL: 30 $LOAEL_{continuous}$: 21 $LOAEL_{HED}$: 5	Significantly increased absolute kidney weights in males (9% higher than controls)	3.4 ($BMDL_{10\%\ HED}$ for absolute kidney weight)	NTP (1991)	High
			NOAEL: 37 $NOAEL_{continuous}$: 26 $NOAEL_{HED}$: 6.2	LOAEL: 75 $LOAEL_{continuous}$: 54 $LOAEL_{HED}$: 13	Increased absolute and relative kidney weights in females (12 and 10% higher than controls, respectively)	6.2 ($NOAEL_{HED}$)		
Immune/ Hematological (*evidence suggests*)	Mouse (CD-1; 10–12 males/group)	14 days (daily gavage)	ND	$LOAEL_{continuous}$: 4.89 $LOAEL_{HED}$: 0.636	Suppression of humoral and cell-mediated immune responses	0.636 ($LOAEL_{HED}$)	Munson et al. (1982)	High

Table_Apx N-17. Summary of Candidate Acute, Non-Cancer, Inhalation PODs for 1,2-Dichloroethane

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL	LOAEL	Basis for NOAEL/LOAEL	Candidate POD[a] (POD Type)	Reference	Study Rating for Target Organ/System
Mortality (*evidence demonstrates*)	Mouse (CD-1, 10–15 males/group)	4 hours	ND	LOAEC: 4,050 mg/m^3 (1,000 ppm) $LOAEC_{continuous}$: $LOAEC_{HEC}$: 675 mg/m^3 (167 ppm)	Dose-related increase in mortality compared with controls (quantitative data not reported)	675 mg/m^3 or 167 ppm ($LOAEL_{HEC}$)	Francovitch et al. (1986)	Medium
Renal/Kidney (*evidence suggests*)	Mouse (B6C3F1, 5 males/group)	4 hours	NOAEC: 639 mg/m^3 (158 ppm) $NOAEC_{continuous}$: $NOAEC_{HEC}$: 107 mg/m^3 (26.3 ppm)	LOAEC: 2,020 mg/m^3 (499 ppm) $LOAEC_{continuous}$: $LOAEC_{HEC}$: 337 mg/m^3 (83.2 ppm)	Significantly increased serum BUN and relative kidney weight (85 and 12% higher than controls, respectively)	207 mg/m^3 or 51.1 ppm ($BMCL_{10\%HEC}$ for relative kidney weight)	Storer et al. (1984)	High
Hepatic/Liver (*evidence suggests*)	Mouse (B6C3F1, 5 males/group)	4 hours	NOAEC: 639 mg/m^3 (158 ppm) $NOAEC_{continuous}$: $NOAEC_{HEC}$: 107 mg/m^3 (26.3 ppm)	LOAEC: 2020 mg/m^3 (499 ppm) $LOAEC_{continuous}$: $LOAEC_{HEC}$: 337 mg/m^3 (83.2 ppm)	Increased serum ALT (2-fold higher than controls [ns]) and SDH (11-fold higher than controls; $p \geq 0.05$)	107 mg/m^3 or 26.3 ppm ($NOAEC_{HEC}$)	Storer et al. (1984)	High
Lung/ Respiratory (*evidence suggests*)	Rat (F344/ DUCRL, 5/sex/group)	4 hours	NOAEC: 212 mg/m^3 (52.4 ppm) $NOAEC_{continuous}$: 35.3 mg/m^3 (8.73 ppm) $NOAEC_{HEC}$: 7.06 mg/m^3 (1.74 ppm)	LOAEC: 794.9 mg/m^3 (196.4 ppm) $LOAEC_{continuous}$: 132.5 mg/m^3 (32.73 ppm) $LOAEC_{HEC}$: 26.50 mg/m^3 (6.547 ppm)	Histological changes to the olfactory mucosa in males and females	1.75 mg/m^3 or 0.432 ppm ($BMCL_{10HEC}$ for degeneration with necrosis in males and females)	Dow Chemical (2006b)	High

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL	LOAEL	Basis for NOAEL/LOAEL	Candidate POD[a] (POD Type)	Reference	Study Rating for Target Organ/System
Lung/ Respiratory (*evidence suggests*)	Rat (F344/ DUCRL, 10/sex/group)	4 hours	ND	LOAEC: 794.9 mg/m^3 (196.4 ppm) $LOAEC_{continuous}$: 132.5 mg/m^3 (32.73 ppm) $LOAEC_{HEC}$: 26.50 mg/m^3 (6.547 ppm)	Histological changes to the olfactory mucosa in males and females	4.636 mg/m^3 or 1.145 ppm ($BMCL_{10HEC}$ for regeneration in males and females)	Dow Chemical (2006b)	High
	Rat (F344/ DUCRL, 5/sex/group)	8 hours	NOAEC 214 mg/m^3 (52.8 ppm) $NOAEC_{continuous}$: 71.3 mg/m^3 (17.6 ppm) $NOAEC_{HEC}$: 14.3 mg/m^3 (3.52 ppm)	LOAEC = 435.1 mg/m^3 (107.5 ppm) $LOAEC_{continuous}$: 145.0 mg/m^3 (35.83 ppm) $LOAEC_{HEC}$: 29.01 mg/m^3 (7.166 ppm)	Histological changes to the olfactory mucosa in males and females	9.78 mg/m^3 or 2.42 ppm ($BMCL_{10HEC}$ for degeneration with necrosis in males and females)	Dow Chemical (2006b)	High
Immune/ Hematological (*evidence suggests*)	Mouse (CD-1, 140 females/ group)	3 hours	NOAEC: 9.3 mg/m^3 (2.3 ppm) $NOAEC_{continuous}$: $NOAEC_{HEC}$: 1.2 mg/m^3 (0.29 ppm)	LOAEC: 22 mg/m^3 (5.4 ppm) $LOAEC_{continuous}$: $LOAEC_{HEC}$: 2.8 mg/m^3 (0.68 ppm)	Mortality following streptococcal challenge	1.2 mg/m^3 or 0.29 ppm ($NOAEC_{HEC}$)	Sherwood et al. (1987)	High (Note: Mice inhaled ≈2E04 aerosolized streptococci 1 hour after exposure. This is unlikely to represent typical immunological challenges in humans).

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL	LOAEL	Basis for NOAEL/LOAEL	Candidate POD[a] (POD Type)	Reference	Study Rating for Target Organ/System
Neurological/ Behavioral (*evidence likely*)	Rat (Sprague-Dawley, 6 males/group)	1.5 hours	ND	LOAEC: 3,950 mg/m^3 (975.9 ppm) $LOAEC_{continuous}$: $LOAEC_{HEC}$: 246.9 mg/m^3 (61.00 ppm)	Changes in brain histopathology	246.9 mg/m^3 or 61.00 ppm ($LOAEC_{HEC}$)	Zhou et al. (2016)	Medium
	Rat (Sprague-Dawley, 12/sex/group)	12 hours	NOAEC: 2,500 mg/m^3 (617.7 ppm) $NOAEL_{continuous}$: $NOAEC_{HEC}$: 1,250 mg/m^3 (308.9 ppm)	LOAEC: 5,000 mg/m^3 (1,240 ppm) $LOAEC_{continuous}$: $LOAEC_{HEC}$: 2,500 mg/m^3 (620 ppm)	Clinical signs of neurotoxicity and changes in brain histology	1,250 mg/m^3 or 308.9 ppm ($NOAEC_{HEC}$)	Zhang et al. (2010)	Medium

LOAEC = lowest-observed-adverse-effect concentration; NOAEC = no-observed-adverse-effect concentration
[a] BMCLs are presented as HECs for comparison with other candidate PODs. BMCL1SD = BMCL for benchmark response of 1 standard deviation change from control mean. BMCL10% = BMCL for benchmark response of 10% relative deviation from control mean. BMCL10 = BMCL for benchmark response of 10% extra risk.

Table_Apx N-18. Summary of Candidate Short-Term/Intermediate, Non-Cancer, Inhalation PODs for 1,2-Dichloroethane

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL	LOAEL	Basis for NOAEL/LOAEL	Candidate POD [a] (POD Type)	Reference	Study Rating for Target Organ/System
Mortality (*evidence demonstrates*)	Rat (Sprague-Dawley, 12 males/group)	30 days 5 days/week 7 hours/day	NOAEC: 619 mg/m^3 (153 ppm) $NOAEC_{continuous}$ = $NOAEC_{HEC}$: 129 mg/m^3 (31.9 ppm)	LOAEC: 1,230 mg/m^3 (304 ppm) $LOAEC_{continuous}$ = $LOAEC_{HEC}$: 256 mg/m^3 (63.3 ppm)	Mortality (1/12 animals)	154 mg/m^3 or 38.0 ppm ($BMCL_{10HEC}$ for mortality)	Igwe et al. (1986b) Igwe et al. (1986c)	High
	Rat (Sprague-Dawley, 16–30 females/group)	10 days 7 hours/day GD 6–15	NOAEC: 405 mg/m^3 (100 ppm) $NOAEC_{continuous}$ = $NOAEC_{HEC}$: 118 mg/m^3 (29.2 ppm)	LOAEC: 1,210 mg/m^3 (300 ppm) $LOAEC_{continuous}$ = $LOAEL_{HEC}$: 353 mg/m^3 (87.5 ppm)	Mortality (10/16 animals)	118 mg/m^3 or 29.2 ppm ($NOAEC_{HEC}$)	Rao et al. (1980)	Medium
	Rat (Sprague-Dawley, 26 females/ group)	15 days 6 hours/day GD 6–20	NOAEC: 1,030 mg/m^3 (254 ppm) $NOAEC_{continuous}$ = $NOAEC_{HEC}$: 258 mg/m^3 (63.5 ppm)	LOAEC: 1,330 mg/m^3 (329 ppm) $LOAEC_{continuous}$ = $LOAEC_{HEC}$: 333 mg/m^3 (82.3 ppm)	Mortality (2/26 dams)	258 mg/m^3 or 63.5 ppm ($NOAEC_{HEC}$)	Payan et al. (1995)	High
	Rabbit (New Zealand White, 19–21 females/ group)	13 days 7 hours/day GD 6–18	ND	LOAEC: 405 mg/m^3 (100 ppm) $LOAEC_{continuous}$ = $LOAEC_{HEC}$: 118 mg/m^3 (29.2 ppm)	Mortality (4/21 animals)	59.4 mg/m^3 or 14.7 ppm ($BMCL_{10HEC}$ for mortality)	Rao et al. (1980)	Medium

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL	LOAEL	Basis for NOAEL/LOAEL	Candidate POD [a] (POD Type)	Reference	Study Rating for Target Organ/System
Hepatic/Liver (*evidence suggests*)	Mouse (Swiss, 10 males/ group)	28 days 6 hours/day	ND	LOAEC: 363.58 mg/m^3 (89.830 ppm) $LOAEC_{continuous}$ = $LOAEC_{HEC}$: 90.895 mg/m^3 (22.457 ppm)	Increased absolute and relative liver weights (≥10% higher than controls)	51.720 mg/m^3 or 12.778 ppm ($BMCL_{10\%HEC}$ for relative liver weight)	Zeng et al. (2018)	High
Reproductive/ Developmental (*evidence suggests*)	Mouse (Swiss, 5–15 males/ group)	4 weeks 6 hours/day	ND	LOAEC: 102.70 mg/m^3 (25.374 ppm) $LOAEC_{continuous}$ = $LOAEC_{HEC}$: 25.675 mg/m^3 (6.3435 ppm)	Changes in sperm parameters (increased total, sperm head, body, and tail abnormalities; decreased sperm concentration; decreased height of seminiferous tubules and height of germinal epithelium)	21.240 mg/m^3 or 5.2500 ppm ($BMCL_{5\%HEC}$ for sperm concentration) 18.815 mg/m^3 or 4.6486 ppm ($BMCL_{1SDHEC}$ for seminiferous tubule height) 8.6304 mg/m^3 or 2.1323 ppm ($BMCL_{1SDHEC}$ for germinal epithelium height)	Zhang et al. (2017)	High

LOAEC = lowest-observed-adverse-effect concentration; NOAEC = no-observed-adverse-effect concentration

[a] BMCLs are presented as HECs for comparison with other candidate PODs. $BMCL_{1SD}$ = BMCL for benchmark response of 1 standard deviation change from control mean. $BMCL_{10\%}$ = BMCL for benchmark response of 10% relative deviation from control mean. $BMCL_{5\%HEC}$ = BMCL for benchmark response of 5% relative deviation from control mean. $BMCL_{10}$ = BMCL for benchmark response of 10% extra risk

.

Table_Apx N-19. Summary of Candidate Chronic, Non-Cancer, Inhalation PODs for 1,2-Dichloroethane

Target Organ/ System	Species (Strain, Sex, n/Group)	Exposure	NOAEL	LOAEL	Basis for NOAEL/LOAEL	Candidate POD [a] (POD Type)	Reference	Study Rating for Target Organ/System
Hepatic/Liver (*evidence suggests*)	Rat (Sprague-Dawley, 8–10/sex/group)	12 months 5 days/week 7 hours/day	NOAEC: 40 mg/m^3 (10 ppm) $NOAEC_{continuous}$ = $NOAEL_{HEC}$: 8.3 mg/m^3 (2.1 ppm)	LOAEC: 200 mg/m^3 (50 ppm) $LOAEC_{continuous}$ = $LOAEL_{HEC}$: 42 mg/m^3 (10 ppm)	Increased ALT (>2-fold higher than controls) and LDH (18% higher than controls) in males	8.3 mg/m^3 or 2.1 ppm ($NOAEC_{HEC}$)	IRFMN (1978)	Medium
			NOAEC: 40 mg/m^3 (10 ppm) $NOAEC_{continuous}$ = $NOAEC_{HEC}$: 8.3 mg/m^3 (2.1 ppm)	LOAEC: 200 mg/m^3 (50 ppm) $LOAEC_{continuous}$ = $LOAEC_{HEC}$: 42 mg/m^3 (10 ppm)	Increased ALT (>2-fold higher than controls) and LDH (25% higher than controls) in females	1.7 mg/m^3 or 0.42 ppm ($BMCL_{1SDHEC}$ for LDH in females)		

LOAEC = lowest-observed-adverse-effect concentration; NOAEC = no-observed-adverse-effect concentration

[a] BMCLs are presented as HECs for comparison with other candidate PODs. BMCL1SD = BMCL for benchmark response of 1 standard deviation change from control mean. BMCL10% = BMCL for benchmark response of 10% relative deviation from control mean. BMCL10 = BMCL for benchmark response of 10% extra risk.

N.2.3 Non-Cancer PODs for Acute Exposures for 1,1-Dichloroethane

Oral

There were two acute-duration oral studies of 1,1-dichloroethane that were rated acceptable: an acute lethality study in guinea pigs by Dow Chemical (1947) and a single-dose lethality study in rats by Muralidhara et al. (2001) (see Table_Apx N-10). The acute lethality study by Dow Chemical (1947) reported no details on the animal strain, sex, age, or condition; number of animals tested; method of administration; or duration of follow-up. The study authors reported only that all guinea pigs survived being fed a dose of 300 mg/kg, while 1,000 mg/kg-bw was lethal for all the animals given this dose. The limitations in the study preclude its use for POD derivation.

Likewise, a single-dose experiment by Muralidhara et al. (2001), with a NOAEL of 1,000 mg/kg-bw and a LOAEL of 2,000 mg/kg-bw was also not considered suitable for POD derivation due to the selection of doses near those exhibiting mortality and the lack of sensitive endpoints other than death. Effects identified included clinical signs of neurotoxicity characterized by the authors as "excitation followed by progressive motor impairment and sedation." The only endpoints evaluated in the experiment were death within the 14 days after dosing and clinical signs. Deaths occurred at doses exceeding 8,000 mg/kg-bw (within 24 hours of dosing) and the LD50 was 8,200 mg/kg-bw. Although the acute-duration oral data are limited, the observation of central nervous system (CNS) effects is consistent with the past use of 1,1-dichloroethane as a human anesthetic (ATSDR, 2015).

Inhalation

No adequate acute-duration (≤ 24 hours) inhalation studies of 1,1-dichloroethane were identified.

Dermal

No adequate acute-duration (≤24 hours) dermal studies of 1,1-dichloroethane were identified.

N.2.4 Non-Cancer PODs for Intermediate-Term Exposures for 1,1-Dichloroethane

Oral

Three short/intermediate-term gavage studies of 1,1-dichloroethane in rats provided sufficient information to identify candidate non-cancer PODs: a 10-day experiment (Muralidhara et al., 2001), a 14-day experiment (Ghanayem et al., 1986), and a 13-week experiment (Muralidhara et al., 2001).

In the 14-day experiment, Ghanayem et al. (1986) identified a NOAEL of 700 mg/kg-bw/day—the only endpoint evaluated in this study was forestomach histopathology. This study was not considered further for the short/intermediate-term oral POD for 1,2-dichloroethane due to the limited evaluations.

In the 10-day experiment by Muralidhara et al. (2001), a NOAEL and LOAEL of 1,000 and 2,000 mg/kg-bw/day, respectively, were identified for decreased body weight. Other endpoints evaluated in this experiment were liver and kidney weights; serum and urinary clinical chemistry markers of liver and kidney effects; and histopathology of the liver, kidney, lung, brain, adrenal, spleen, testis, and epididymis. Dosing was daily, so no adjustment for continuous exposure was necessary. BMD modeling of the data on decreased body weight yielded a $BMDL_{10}$ of 1,167 mg/kg-bw/day. This study was not considered further due to a NOAEL near the limit dose of 1,000 mg/kg-bw/day.

In the 13-week experiment by Muralidhara et al. (2001), evaluations were the same as in the 10-day experiment described above. In this experiment, a NOAEL of 1,000 mg/kg-bw/day and a LOAEL of 2,000 mg/kg-bw/day were identified for mortality (1/15 rats), CNS depression, and decreased body

weight. At the high dose in this study (4,000 mg/kg-bw/day), the rats exhibited protracted narcosis, with 8/15 rats dying between weeks 1 and 11 when the surviving rats in this group were sacrificed. Mortality was not considered to be a suitable endpoint for BMD modeling. Quantitative data on CNS depression were not reported, precluding BMD modeling of this endpoint. BMD modeling of the data on decreased body weight yielded a $BMDL_{10}$ of 1,248 mg/kg-bw/day; however, it is not clear that a POD based on body weight would be adequately protective for mortality and neurotoxicity.

Inhalation

One short/intermediate-term inhalation study provided adequate information to identify a LOAEC. In the inhalation developmental toxicity study of rats by Schwetz et al. (1974), the following maternal endpoints were evaluated: maternal body weight and liver weight, serum ALT, and gross necropsy. Developmental endpoints were also assessed, including gross, skeletal, and visceral anomalies. Effects observed in the study are summarized below:

- Decreased maternal body weight on GD 13 (≈9 and 11% compared with controls at low and high exposure levels, respectively).
- An uncertain effect on the incidence of litters with delayed ossification of the sternebrae at the high exposure level. In this study, each of the two exposure groups had its own control group, and the incidence of this effect differed between the two control groups (61 percent in the control for low exposure and 11 percent in the control for the high exposure). Incidences in low and high exposure groups were 44 and 42 percent, respectively, intermediate between the two control groups.
- Increased relative liver weight (15% compared with controls) 6 days after the end of exposure in nonpregnant rats in the high exposure group. However, no difference in absolute or relative liver weight was seen at the end of the exposure period.

No other short/intermediate-term inhalation studies with a rating of acceptable were located. The data from Schwetz et al. (1974) were not considered adequate for derivation of a short/intermediate-term inhalation POD for the following reasons: (1) the evaluations of maternal endpoints did not include histopathology or effects in organs other than the liver, (2) the disparate findings on delayed ossification in the two control groups mean that a conclusion regarding this endpoint cannot be made with confidence, and (3) there are no supporting studies that evaluated comprehensive endpoints.

Dermal

No adequate short/intermediate-term dermal studies of 1,1-dichloroethane were identified.

N.2.5 Non-Cancer PODs for Chronic Exposures for 1,1-Dichloroethane

Oral

Two chronic-duration oral studies of 1,1-dichloroethane in mice provided sufficient information to identify NOAELs and/or LOAELs: a 52-week drinking water experiment (Klaunig et al., 1986) and a 78-week gavage experiment (NCI, 1978). In the 52-week experiment (Klaunig et al., 1986) (study rating of High for non-cancer endpoints), a freestanding NOAEL of 543 mg/kg-bw/day was identified based on the absence of effects on body weight and liver, kidney, and lung weight and histology. No other endpoints were evaluated. Because this study did not conduct comprehensive toxicological evaluations, it is possible that effects on other organs or systems could have occurred at the NOAEL. Therefore, the freestanding NOAEL from this study was not considered suitable for use as the chronic oral non-cancer POD for 1,1-dichloroethane.

In the 78-week experiment (NCI, 1978) (study rating of High for mice), male and female mice were exposed to increasing doses over time for 78 weeks followed by a 13-week recovery period prior to sacrifice (see Table_Apx N-20).

Table_Apx N-20. Dosing Regimen in (NCI, 1978) Chronic Mouse Study

Group	Dose (mg/kg-bw/day Administered 5 days/week)	Number of Weeks at this Dose	TWA Across 78 Dosing Weeks
Males			
Low dose	900	6	1,442
	1,200	3	
	1,500	69	
	0	13	
High dose	1,800	6	2,885
	2,400	3	
	3,000	69	
	0	13	
Females			
Low dose	900	6	1,665
	1,200	3	
	1,500	11	
	1,800	58	
	0	13	
High dose	1,800	6	3,331
	2,400	3	
	3,000	11	
	3,600	58	
	0	13	

NCI (1978) averaged the doses across the 78 exposure weeks and reported time-weighted average doses of 0, 1,442, or 2,885 mg/kg-bw/day (males) and 0, 1,665, or 3,331 mg/kg-bw/day (females) (these doses were administered 5 days/week). Decreased survival was observed in both males and females in the high dose group, but the findings in males were confounded by reduced survival in untreated control males (beginning around week 35). NCI (1978) did not report cause of death or any explanation for the control male deaths. In females of the high dose group, there was a statistically significant reduction in survival. Based on survival data presented graphically, there were no deaths among female mice exposed for 9 weeks at doses up to 2,400 mg/kg-bw/day. The first high dose female death occurred at around week 15 when the females were receiving 3,000 mg/kg-bw/day, but additional deaths did not occur until around week 30—after the dose had been increased to 3,600 mg/kg-bw/day. Because of the variable dosing regimen, there is significant uncertainty regarding the dose that resulted in decreased survival in females. In addition, the reduced survival of untreated male mice calls into question the reliability of the study findings.

Inhalation

Two chronic-duration inhalation studies of 1,1-dichloroethane were rated acceptable; however, neither provided sufficient information to determine a POD. In the study by Hofmann et al. (1971) (rated Medium), rats, guinea pigs, and rabbits were exposed 6 hours/day, 5 days/week for 13 weeks to

500 ppm followed by 13 weeks at 1,000 ppm 1,1-dichloroethane. Evaluations included clinical signs, body weight, hematology, urinalysis, blood chemistry, and liver function (in rabbits) after 13 weeks, and liver and kidney weight and histopathology at the end of the exposure period (26 weeks). No effects were observed in rats, guinea pigs, or rabbits, so the only exposure level tested is a NOAEC. These data are not sufficient to determine a POD due to the limited evaluations (lack of organ weights and histopathology for organs/systems other than liver and kidney).

The study of dogs by Mellon Institute (1947) received a medium-quality study rating. In this study, a single mongrel dog was exposed to 1,067 ppm 1,1-dichloroethane 7 hours/day, every other day for 6 months. Reporting for this study is very limited, but it appears that there was a significant decrease in the exposed dog's weight compared to the control(s) and marked lung congestion at necropsy. While these results suggest a freestanding LOAEC of 1,067 ppm or 4,319 mg/m^3 (156 ppm or 630 mg/m^3 after adjustment for continuous exposure), the data are not sufficient for use as a POD due to (1) use of a single animal and single exposure concentration; (2) lack of data on the magnitude of body weight change; and (3) failure to identify a NOAEC.

Dermal

No adequate chronic dermal studies of 1,1-dichloroethane were identified.

N.2.6 Non-Cancer PODs for Acute Exposures for 1,2-Dichloroethane

Oral

The acute-duration oral POD for 1,2-dichloroethane was based on increased relative kidney weight in male mice given a single gavage dose of 1,2-dichloroethane (Storer et al., 1984). For this study, a NOAEL of 200 mg/kg-bw/day and a LOAEL of 300 mg/kg-bw/day were identified for kidney weight effects. To obtain a POD, BMD modeling was conducted on the relative kidney weight data using U.S. EPA's Benchmark Dose Software (BMDS; v. 3.3). Table_Apx N-21 shows the relative kidney weights corresponding to each dose. BMD modeling was conducted using a benchmark response (BMR) of 10 percent relative deviation from the control mean (U.S. EPA, 2012a).

Table_Apx N-21. Relative Kidney Weights in Male Mice Exposed to 1,2-Dichloroethane Once by Gavage

Dose (mg/kg-day)	Number of Mice	Mean (g/100 g body weight)	Standard Deviation
0	5	1.50	0.09
200	5	1.58	0.19
300	5	1.69	0.09
400	3	1.75	0.08
500	1[a]	1.82	N/A
600	1[a]	1.61	N/A

Source: Storer et al. (1984)
[a] 4/5 mice died in this group

Following (U.S. EPA, 2012a) guidance, the polynomial 2-degree model with constant variance was selected for these data. The $BMD_{10\%}$ and $BMDL_{10\%}$ values for this model were 270 and 153 mg/kg-bw/day, respectively. The $BMDL_{10\%}$ of 153 mg/kg-bw/day was selected as the POD.

The $BMDL_{10\%}$ of 153 mg/kg-bw/day was converted to a HED of 19.9 mg/kg-bw/day using the DAF of 0.13 for mice utilizing the body weight ¾ scaling method (see Appendix N.4.1.3) and Equation_Apx N-1, as shown below:

Equation_Apx N-1.

$$HED = 153\ mg/kg \times 0.13 = 19.9\ mg/kg$$

The HED of 19.9 mg/kg-bw/day does not need to be adjusted for occupational exposure. The benchmark MOE for this POD is 30 (3× for interspecies extrapolation when a dosimetric adjustment is used to calculate an HED and 10× for human variability).

Inhalation

The acute-duration inhalation POD for 1,2-dichloroethane was based on nasal lesions in rats exposed once by inhalation for 8 hours (Dow Chemical, 2006b). For this study, an NOAEC of 71.3 mg/m^3 and a LOAEC of 145 mg/m^3 were identified for increased incidences of degeneration with necrosis in the olfactory mucosa of the nasal passages in male and female rats. To obtain a POD, BMD modeling was conducted using EPA's BMDS (v. 3.3.2) on the incidence of these nasal lesions in male and female rats (combined). The male and female data were combined for modeling because incidences were similar in both sexes and the combined data set provided increased statistical power relative to the sex-specific data sets. Prior to modeling, the exposure concentrations in the Dow Chemical (2006b) rat 8-hour study were adjusted from the exposure scenario of the original study to continuous (24 hours/day) exposure using Equation_Apx N-5. Table_Apx N-22 shows the nasal lesion incidences corresponding to each exposure concentration. BMD modeling was conducted on the incidences using the continuous equivalent concentrations and the default BMR for quantal data of 10 percent extra risk (U.S. EPA, 2012a).

Table_Apx N-22. Incidence of Nasal Lesions in Male and Female Rats (Combined) Exposed to 1,2-Dichloroethane for 8 Hours

Unadjusted Exposure Concentration (mg/m^3)	Adjusted (Continuous) Exposure Concentration (mg/m^3)	Incidence of Degeneration with Necrosis of the Olfactory Mucosa
0	0	0/10
214	71.3	0/10
435.1	145.0	4/10
630.6	210.2	9/10
Source: Dow Chemical (2006b)		

Following U.S. EPA (2012a) guidance, the multistage 3-degree model was selected for these data. The BMC_{10} and $BMCL_{10}$ for this model were 81.4 and 48.9 mg/m^3, respectively. The $BMCL_{10}$ of 48.9 mg/m^3 was selected as the POD.

U.S. EPA (1994) guidance was used to convert the $BMCL_{10}$ of 48.9 mg/m^3 to a HEC. For nasal lesions, the $RGDR_{ET}$ in rats is used. The $RGDR_{ET}$ of 0.2 was calculated using Equation_Apx N-9 (U.S. EPA, 1994). The $BMCL_{10}$ (48.9 mg/m^3) was multiplied by the $RGDR_{ET}$ (0.2) to calculate the HEC, as shown in the Equation_Apx N-10.

The resulting HEC is 9.78 mg/m^3 for continuous exposure. The continuous HEC of 9.78 mg/m^3 was converted to an equivalent worker HEC using Equation_Apx N-13. The resulting POD for workers is

41.1 mg/m^3. The benchmark MOE for this POD is 30 (3× for interspecies extrapolation when a dosimetric adjustment is used to calculate an HEC and 10× for human variability).

EPA presents all inhalation PODs in equivalents of both mg/m^3 and ppm to avoid confusion and errors. Equation_Apx N-3 was used with the molecular weight of 1,2-dichloroethane (98.96 mg/mmol) to convert the continuous and worker PODs (9.78 and 41.1 mg/m^3, respectively) to 2.42 and 10.2 ppm, respectively.

Dermal

No PODs were identified from acute studies of dermal exposure to 1,2-dichloroethane. Therefore, the acute oral HED of 19.9 mg/kg-bw/day with benchmark MOE of 30 was used for risk assessment of acute dermal exposure for both continuous and worker exposure scenarios. As noted in Appendix N.4.1.4, when extrapolating from oral data that incorporated BW$^{3/4}$ scaling to obtain the oral HED, EPA uses the same HED for the dermal route of exposure. The same uncertainty factors are used in the benchmark MOE for both oral and dermal scenarios.

N.2.7 Non-Cancer PODs for Intermediate Exposures for 1,2-Dichloroethane

Oral

The intermediate duration oral POD for 1,2-dichloroethane was based on relative kidney weight in F344 male rats exposed to 1,2-dichloroethane by gavage for 13 weeks (5 days/week) (NTP, 1991). In this study, a dose-related significant increase in relative kidney weight was observed. Using EPA's BMDS (v. 3.3), BMD modeling was conducted on relative kidney weight data in male rats. The rats in the study by NTP (1991) were exposed 5 days/week, so an adjustment for continuous exposure duration was needed to estimate an equivalent oral dose for animals exposed for 7 days per week. The dose and response data used for the modeling are presented in Table_Apx N-23. Continuous models were used to fit dose-response data.

Table_Apx N-23. Relative Kidney Weight in Male Rats and Associated Doses Selected for Dose Response Modeling for 1,2-Dichloroethane from a 13-Week Oral Exposure Study

Adjusted Dose (mg/kg-day)	Number of Rats	Mean (Organ Weight to Body Weight)	SD (Organ Weight to Body Weight)
0	10	3.9	0.19
21	10	4.1	0.32
43	10	4.5	0.25
86	10	4.9	0.22
Source: NTP (1991)			

Following U.S. EPA (2012a) guidance, the power model was selected for these data. The BMD_{10} and $BMDL_{10}$ for this model were 33 and 27 mg/kg-day, respectively. The $BMDL_{10}$ of 27 mg/kg-day was selected as the POD.

U.S. EPA (1994) guidance was used to convert the $BMDL_{10}$ of 27 mg/kg-day to a HED of 6.5 mg/kg-bw/day using the DAF of 0.24 for rats based on body weight ¾ scaling (see Appendix N.4.1.3) and Equation_Apx N-6.

The continuous HED of 6.5 mg/kg-bw/day was converted to a worker HED of 9.1 mg/kg-bw/day using Equation_Apx N-12. The benchmark MOE for this POD is 30 based on a combination of uncertainty

factors: 3× for interspecies extrapolation when a dosimetric adjustment to derive the HED is used and 10× for human variability.

Inhalation

The short-term/subchronic-duration inhalation POD for 1,2-dichloroethane was based on decreased sperm concentration in male mice exposed to 1,2-dichloroethane by inhalation for 4 weeks (Zhang et al., 2017). In this study, a concentration-related decrease in sperm concentration was observed, reaching statistical significance (relative to controls) at 707.01 mg/m^3. Using EPA's BMDS (v. 3.3.2), BMD modeling was conducted on the sperm concentrations using mouse exposure concentrations. The mice in the study by Zhang et al. (2017) were exposed for 6 hours/day, 7 days/week. Prior to BMD modeling, the exposure concentrations in the Zhang et al. (2017) study were adjusted from the exposure scenario of the original study to equivalent continuous (24 hours/day) exposure concentrations using Equation_Apx N-5. Table_Apx N-24 shows the sperm concentrations corresponding to each exposure concentration. BMD modeling was conducted on these data using a BMR of 5 percent relative deviation from controls.

Table_Apx N-24. Sperm Concentration in Male Mice Exposed to 1,2-Dichloroethane for 4 Weeks

Unadjusted Exposure Concentration (mg/m^3)	Adjusted (Continuous) Exposure Concentration (mg/m^3)	Number of Animals	Mean Sperm Concentration (M/g)	Standard Deviation (M/g)
0.30	0.075	10	4.65	0.52
102.70	25.675	10	4.36	0.40
356.04	89.010	10	3.89	0.47
707.01	176.75	10	3.30	0.57
Source: Zhang et al. (2017)				

Following U.S. EPA (2012a) guidance, the Exponential 3 Model with constant variance was selected for these data. The $BMC_{5\%}$ and $BMCL_{5\%}$ for this model were 26.735 and 21.240 mg/m^3, respectively. The $BMCL_{5\%}$ of 21.240 mg/m^3 was selected as the POD.

U.S. EPA (1994) guidance was used to convert animal inhalation PODs to HECs. For systemic (extrarespiratory) effects, the HEC is calculated by multiplying the animal POD by the ratio of the blood:gas partition coefficients in animals and humans, as shown in Equation_Apx N-8.

A human blood:air partition coefficient of 19.5 ± 0.7 has been reported for 1,2-dichloroethane (Gargas et al., 1989). No blood:air partition coefficient for mice was identified in the literature reviewed. In the absence of a blood:air partition coefficient for mice, the default ratio of 1 is used in the calculation, in accordance with U.S. EPA (1994) guidance. Therefore, the POD of 21.240 mg/m^3 is multiplied by 1 to give the HEC.

The resulting POD is 21.240 mg/m^3 for continuous exposure. The continuous POD of 21.240 mg/m^3 is converted to an equivalent worker POD using Equation_Apx N-14. The resulting POD for workers is 89.208 mg/m^3. The benchmark MOE for this POD is 30 based on a combination of uncertainty factors: 3× for interspecies extrapolation when a dosimetric adjustment is used to calculate an HEC and 10× for human variability.

Dermal

No PODs were identified from short-term or subchronic studies of dermal exposure to 1,2-dichloroethane. Therefore, the short-term/subchronic oral HED of 6.5 mg/kg-bw/day and worker HED

of 9.1 mg/kg-bw/day with benchmark MOE of 30 were used for risk assessment of intermediate dermal exposure. As noted in Appendix N.4.1.4, when extrapolating from oral data that incorporated $BW^{3/4}$ scaling to obtain the oral HED, EPA uses the same HED for the dermal route of exposure. The same uncertainty factors are used in the benchmark MOE for both oral and dermal scenarios.

N.2.8 Non-Cancer PODs for Chronic Exposures for 1,2-Dichloroethane

Oral

No studies of chronic oral exposure in laboratory animals were considered suitable for POD determination (see Table_Apx N-11). Therefore, the intermediate POD was also used for chronic exposure. The intermediate continuous HED was 6.5 mg/kg-bw/day and the worker HED was 9.1 mg/kg-bw/day (see Appendix N.2.7). The benchmark MOE for this POD is 300 based on 3× for interspecies extrapolation when a dosimetric adjustment is used, 10× for human variability, and 10× for extrapolating from a subchronic study duration to a chronic study duration for chronic exposures.

Inhalation.

Only one study of chronic inhalation exposure in laboratory animals (IRFMN, 1978) was considered suitable for POD determination (see Table_Apx N-14). However, the 12-month study by IRFMN (1978) evaluated limited endpoints (serum chemistry changes only) and identified a higher LOAEC than the study of sperm parameters by Zhang et al. (2017) that was used as the basis for the short-term/subchronic POD. Therefore, the POD from Zhang et al. (2017) was also used for chronic exposure. The resulting POD is 21.240 mg/m^3 for continuous exposure. The continuous POD of 21.240 mg/m^3 is converted to an equivalent worker POD using Equation_Apx N-13. Equation_Apx N-3 was used with the molecular weight of 1,2-dichloroethane (98.96 mg/mmol) to convert the continuous and worker short-term/subchronic/chronic PODs (21.240 and 89.208 mg/m^3, respectively) to 5.2478 and 22.041 ppm, respectively. The resulting POD for workers is 89.208 mg/m^3. (see Table_Apx N-28). The benchmark MOE for this POD is 300 based on 3× for interspecies extrapolation when a dosimetric adjustment is used to calculate the HEC, 10× for human variability, and 10× for extrapolation from a 4-week study to chronic exposure duration for chronic exposures.

Dermal

No PODs were identified from chronic-duration studies of dermal exposure to 1,2-dichloroethane (see Table_Apx N-13). Therefore, the oral HEDs of 6.5 mg/kg-bw/day (continuous) and 9.1 mg/kg-bw/day (for workers) with benchmark MOE of 300 were used for risk assessment of chronic-duration dermal exposure. As noted in Appendix N.4.1.3, when extrapolating from oral data that incorporated $BW^{3/4}$ scaling to obtain the oral HED, EPA uses the same HED for the dermal route of exposure. The same uncertainty factors are used in the benchmark MOE for both oral and dermal scenarios.

N.2.9 Other Uncertainty Factors Not Applied in this Assessment

LOAEL-to-NOAEL Uncertainty Factor (UF_L)

A UF_L is applied when adverse effects are identified at the lowest dose/concentration tested and the POD cannot be refined through BMD modeling. A value of 3× or 10× can be applied based on the magnitude of the observed effect and the dose-response curve. The POD chosen to calculate acute, intermediate, and chronic risks is a BMDL and therefore, EPA did not apply this UF.

Database Uncertainty Factor (UF_D)

EPA may consider application of a UF_D on a case-by-case basis when the available quantitative data may insufficiently account for expected adverse effects from chemical exposure. For 1,1-dichloroethane, the Agency is utilizing the most sensitive and well-supported POD for risk estimates. There is

insufficient evidence to indicate that an alternative study would result in a lower POD. Therefore, a UFD is not applied for this assessment.

N.3 Genotoxicity and Cancer

N.3.1 1,1-Dichloroethane

Animal studies provide limited evidence that 1,1-dichloroethane may cause cancer in rodents. Rats and mice exposed via gavage for 78 weeks exhibited a positive dose-related trend in the incidence of liver tumors in male mice and mammary gland tumors and hemangiosarcomas in female rats. Poor survival in both control and treated animals limits the validity of these results. In the Milman et al. (1988) study, 1,1-dichloroethane was positive in the standard *in vivo* rat liver tumor promoter assay Milman et al. (1988). Cancer mode-of-action data for 1,1-dichloroethane are very limited and consist of a small number of genotoxicity Table_Apx N-25 and Table_Apx N-26 show the results of *in vitro* and *in vivo* genotoxicity, respectively, and cell transformation assays of 1,1-dichloroethane.

Table_Apx N-25. *In Vitro* Genotoxicity Tests of 1,1-Dichloroethane

Reference(s)	Test System	Doses and Exposure Conditions	Endpoint	Results	Comment
Simmon et al. (1977)	*Salmonella typhimurium* TA1535, TA1537, TA1538, TA98, TA100	Up to 5 mg/plate or cytotoxic dose	Mutation	Negative	Efforts to mitigate volatility were not reported.
Zeiger et al. (1992)	*S. typhimurium* TA1535, TA1537, TA97, TA98, TA100	Up to 1 mg/plate; capped tubes to prevent evaporation	Mutation	Negative (+/−S9)	
Milman et al. (1988)	*S. typhimurium* TA1535, TA1537, TA98, TA100	Not reported; plates enclosed in 9 L desiccator	Mutation	Positive (+/− S9)	Positive in TA1535 and TA100 with and without S9 from rats and mice of both sexes; positive in TA98 (metabolic activation conditions not reported).
Crebelli et al. (1995) Crebelli et al. (1988)	*Aspergillus nidulans* diploid strain P1	0.2, 0.3, 0.4% (v:v)	Chromosome malsegregation	Equivocal	1,1-Dichloroethane induced significant increase in mitotic segregation (measured as numbers of abnormal colonies) at 0.2% but not at 0.3 or 0.4%.
Matsuoka et al. (1998)	Chinese hamster lung fibroblasts	Up to cytotoxic dose or preparation limit; 6 hours in glass culture bottle with rubber stopper	Chromosomal aberrations	Negative (+/− S9)	
Milman et al. (1988)	B6C3F1 mouse hepatocytes	Not reported	DNA repair	Positive	Assay modified to mitigate volatility. No further details provided.

Reference(s)	Test System	Doses and Exposure Conditions	Endpoint	Results	Comment
Milman et al. (1988) Williams et al. (1989)	Osborne-Mendel rat hepatocytes	Not reported, 18-20 hours	DNA repair	Positive	Lowest positive concentration was 1.3E−02 M. Assay modified to mitigate volatility. No further details provided.
Hatch et al. (1983)	Syrian hamster embryo cells	0, 0.062, 0.125, 0.25, 0.50, 1.0 mL/chamber (vapor) for 20 hours in sealed test system	Cell (viral) transformation	Positive	No cells survived at the highest dose. 1,1-Dichloroethane enhanced transformation of cells by SA7 (simian) adenovirus at doses between 0.062 and 0.5 mL/chamber (1.4- to 2.2-fold).
Arthur D. Little Inc. (1983) Milman et al. (1988) Tu et al. (1985)	BALB/c mouse 3T3 cell line	0, 4, 20, 100, 250 µg/mL for 24 hours in sealed glass incubation chamber	Cell transformation	Negative (−S9)	No metabolic activation. Preliminary cytotoxicity assay showed no effect on survival except at 100 and 250 µg/mL (41–53 and 46–67% survival, respectively).
Colacci et al. (1985)	Calf thymus DNA (cell-free)	2.5 µCi for 90 minutes, with or without microsomes from phenobarbital-induced rat or mouse liver, kidney, lung, stomach	DNA binding	DNA binding observed under all conditions	Significantly higher binding in presence (vs. absence) of liver and lung microsomes from rats or mice. No significant difference with kidney or stomach microsomes of either species. No information provided on methods to mitigate volatilization.

Table_Apx N-26. *In Vivo* Genotoxicity Studies of 1,1-Dichloroethane

Reference	Species	Tissue/Cell Type	Dose, Frequency, and Route	Endpoint	Result
Patlolla et al. (2005)	Male Swiss-Webster mouse	Bone marrow	0, 100, 200, 300, 400, 500 mg/kg (single dose, intraperitoneal)	Chromosomal aberrations and micronuclei 24 hours after dosing	Significant, dose-related increases in percent chromosomal aberrations and percent micronucleated cells at ≥200 mg/kg. Mitotic index was significantly decreased at ≥300 mg/kg.
Taningher et al. (1991)	Male BALB/c mouse	Hepatic nuclei	900 mg/kg (single dose intraperitoneal)	DNA unwinding 4 hours after dosing	No significant effect on percent double-stranded DNA.
Colacci et al. (1985)	Male BALB/c mouse	Liver, kidney, lung, stomach	127 µCi/kg (single dose, intraperitoneal)	DNA binding 22 hours after dosing	Binding highest in liver, followed by stomach, lung, and kidney.
Colacci et al. (1985)	Male Wistar rat	Liver, kidney, lung, stomach	127 µCi/kg (single dose, intraperitoneal)	DNA binding 22 hours after dosing	Binding highest in stomach, followed by liver, lung, and kidney.

In vitro experiments on 1,1-dichloroethane genotoxicity include two bacterial mutagenicity studies, a study of chromosomal aberrations in mammalian cells, studies of DNA repair in mouse and rat, hepatocytes studies of mammalian cell transformation, a test of chromosome malsegregation in fungi, and a study of cell-free DNA binding. *In vitro* genotoxicity testing of 1,1-dichloroethane is hampered by this chemical's volatility, which requires the use of methods to mitigate chemical loss from the test system. 1,1-Dichloroethane was mutagenic both with and without exogenous activation supporting that it directly reacts with DNA without metabolism, in an EPA experiment conducted in a desiccator to mitigate volatilization (Milman et al., 1988); however, negative results were obtained in a preincubation assay using capped tubes to limit volatilization (Zeiger et al., 1992). Another Ames assay yielded negative results, but there was no indication of whether chemical volatility was controlled (Simmon et al., 1977). In mammalian cells tested under conditions controlling for volatility, 1,1-dichloroethane did not increase the frequency of chromosomal aberrations in Chinese hamster lung fibroblasts (Matsuoka et al., 1998) but increased DNA repair in hepatocytes from B6C3F1 mice and Osborne Mendel rats (Williams et al., 1989; Milman et al., 1988).

Assays for cell transformation showed that 1,1-dichloroethane enhanced simian adenovirus transformation of Syrian hamster embryo cells (Hatch et al., 1983) but did not induce morphological transformation of BALB/c mouse 3T3 cells at concentrations associated with approximately 50 percent survival (Milman et al., 1988; Tu et al., 1985; Arthur D. Little Inc., 1983). In tests for chromosome malsegregation in *Aspergillus nidulans* diploid strain P1 (conducted in capped tubes), 1,1-dichloroethane induced a significant increase in mitotic segregation (measured as numbers of abnormal colonies) at a concentration of 0.2 percent (v:v), but not at higher concentrations (0.3 and 0.4%) (Crebelli et al., 1995; Crebelli et al., 1988).

Colacci et al. (1985) evaluated the binding of 1,1-dichloroethane to cell-free calf thymus DNA in the presence or absence of liver, kidney, lung, and stomach microsomes from phenobarbital-pretreated rats and mice. 1,1-Dichloroethane binding to DNA was enhanced when co-cultured with liver and lung microsomes from either rats or mice but not in the presence of kidney or stomach microsomes (Colacci et al., 1985), suggesting that metabolism of 1,1-dichloroethane in the liver and lung results in metabolites capable of binding DNA. Dichloroacetaldehyde is a reactive metabolite. In another experiment by these study authors, addition of glutathione to the incubation system resulted in lower DNA binding (reported to be 26% lower than control without further detail), suggesting that glutathione conjugation is detoxifying for 1,1-dichloroethane. These study authors also measured DNA binding of ^{14}C-1,1-dichloroethane in the liver, kidney, lung, and stomach of male BALB/c mice and Wistar rats 22 hours after an intraperitoneal injection of ^{14}C-1,1-dichloroethane (127 µCi/kg) (Colacci et al., 1985). Table_Apx N-27 shows the results, which indicate the highest binding in the stomach of rats and liver of mice. These results differ from the *in vitro* findings, possibly due to the fact that the animals in the *in vivo* study were not pretreated with phenobarbital to induce liver enzymes.

Table_Apx N-27. Binding of ^{14}C-1,1-Dichloroethane to DNA (pmol/mg) After Intraperitoneal Exposure

Tissue[a]	Rat	Mouse
Stomach	4.78	2.33
Liver	3.10	2.54
Lung	2.24	1.51
Kidney	1.81	0.65

[a] Pooled organs from 4 rats and 12 mice
Source: Colacci et al. (1985)

In another *in vivo* study, 1,1-dichloroethane induced significant, dose-related increases in chromosomal aberrations and micronucleated cells in the bone marrow of male Swiss Webster mice given single intraperitoneal doses of 200 to 500 mg/kg-bw (Patlolla et al., 2005). No increase in DNA unwinding was seen in the livers of mice when sacrificed 4 hours after intraperitoneal injection of 900 mg/kg-bw 1,1-dichloroethane (Taningher et al., 1991).

In summary, MOA information pertaining specifically to tissues susceptible to tumor formation after exposure to 1,1-dichloroethane (*e.g.*, liver, mammary, blood) is limited to studies showing that 1,1-dichloroethane induces (1) DNA repair and binds to DNA in liver cells, and (2) chromosomal aberrations and micronuclei in bone marrow. These data are not sufficient to determine the MOA for any tumor type associated with exposure to 1,1-dichloroethane. Overall, the available data provide limited support for the genotoxicity of 1,1-dichloroethane but no information on alternative modes of carcinogenic action.

N.3.2 1,2-Dichloroethane

1,2-Dichloroethane is considered a "probable human carcinogen" (U.S. EPA, 1987) based on evidence of tumorigenicity in animal studies—including significant increases in tumors of the mammary gland (robust evidence), lung (moderate evidence), liver (slight-to-moderate evidence), circulatory system (slight evidence) and other tissues (indeterminate evidence) in male and/or female rats and/or mice by oral, inhalation, and/or dermal exposure (see Appendix N.11). The occurrence of tumors in multiple tissues and treated groups is suggestive of a genotoxic mode of action, and most data relating to mode of action for 1,2-dichloroethane carcinogenicity are assays for genetic toxicity. Recent comprehensive reviews (ATSDR, 2024; Gwinn et al., 2011) were used to develop an overview of genotoxicity data for 1,2-dichloroethane and the role of metabolism, which is presented below. One metabolite of 1,2-dichloroethane is reactive 2-chloroacetaldehyde, which has been identified as a persistent DNA crosslinker. Potential nongenotoxic modes of action for rat mammary tumors were investigated in one study (Lebaron et al., 2021). Brief discussions of the information (both genotoxic and non-genotoxic mechanisms) that pertain to specific tumor sites associated with 1,2-dichloroethane exposure (mammary gland, lung, liver, and circulatory system) follow the general genotoxicity discussion. Immunotoxicity is a recognized non-genotoxic mechanism for carcinogenesis with several positive immunosuppression findings (Munson et al., 1982).

Genotoxicity Overview

Evidence from *in vivo* studies using multiple animal species and routes of exposure and *in vitro* studies using multiple test systems indicates that 1,2-dichloroethane and/or its metabolites can induce mutations, chromosomal aberrations, DNA damage, and DNA adducts in certain test systems. Alkyl halide

compounds such as 1,2-dichloroethane are recognized as reactive chemicals. The available data show that biotransformation of 1,2-dichloroethane to reactive metabolites via a major CYP450-mediated oxidative pathway and a minor glutathione conjugation pathway contributes to the observed effects. There are species-, sex-, tissue-, and dose-related differences in the interactions between 1,2-dichloroethane and/or its metabolites and DNA.

Evidence that 1,2-dichloroethane induces gene mutation is based largely on *in vitro* studies. Reverse mutation studies in *Salmonella typhimurium* were predominantly positive, especially with metabolic activation (ATSDR, 2024; Gwinn et al., 2011). Mutagenicity was seen more consistently in *Salmonella* strains that detect base-pair substitutions (*e.g.,* TA1535) than those that detect frameshift mutations (*e.g.,* TA97) (ATSDR, 2024; Gwinn et al., 2011). Mutations at the HGPRT locus were increased in Chinese hamster ovary (CHO) cells in the presence of metabolic activation—both when 1,2-dichloroethane was incorporated in media (Tan and Hsie, 1981) and when cells were exposed to 1,2-dichloroethane as a vapor in a closed system (Zamora et al., 1983). There are limited gene mutation data from *in vivo* studies. Oral and inhalation studies assessing various types of mutations in *Drosophila* were generally positive, but many of the studies were limited by lack of methodological details and/or the use of a single exposure level (ATSDR, 2024; Gwinn et al., 2011). A single study of *lacZ* mutations in the liver and testis of Muta^{TM} mice showed no increase in the mutation frequency after exposure to 1,2-dichloroethane by oral or intraperitoneal administration at doses up to 150 or 280 mg/kg-bw, respectively (Hachiya and Motohashi, 2000).

In vivo rodent studies showing clastogenic effects, DNA damage, and DNA adducts in the mammary gland, lung, liver, and circulatory system tissues are discussed in the subsections below on potential mechanisms for carcinogenicity in these tissues. A small number of *in vivo* studies of genotoxicity endpoints in other tissue types showed evidence of DNA damage (Comet assay) in mouse kidney, bladder, and brain (Sasaki et al., 1998); and DNA binding or DNA adducts in mouse and rat stomach, forestomach, and kidney (Watanabe et al., 2007; Hellman and Brandt, 1986; Inskeep et al., 1986; Prodi et al., 1986; Arfellini et al., 1984) after exposure by intraperitoneal injection.

Role of Metabolism

Available data are not sufficient to determine whether metabolism of 1,2-dichloroethane is a necessary first step in its genotoxic action. *In vitro* studies in bacteria have shown that exogenous metabolic activation is either required for, or increases the mutagenic activity of, 1,2-dichloroethane (ATSDR, 2024; Gwinn et al., 2011). In contrast, experiments in human lymphocytes cultured *in vitro* with 1,2-dichloroethane showed increased micronucleus formation in the absence of S9, but not in the presence of S9 fraction (Tafazoli et al., 1998).

Evidence suggests that metabolism of 1,2-dichloroethane, especially via the glutathione pathway, does lead to increased genotoxicity. Crespi et al. (1985) compared the genotoxicity of 1,2-dichloroethane in human cell lines with differing metabolic capacities. Crespi et al. (1985) observed 25-fold higher HGPRT mutation frequencies in AHH-1 compared with TK6 human lymphoblastoid cells. The study authors measured 5-fold greater glutathione-S-transferase activity in the AHH-1 cells than the TK6 cells, suggesting that the glutathione metabolic pathway increased the frequency of mutations induced by 1,2-dichloroethane.

Several studies have inhibited or stimulated enzymes to elucidate the relative importance of the CYP450 and glutathione pathways in 1,2-dichloroethane genotoxicity. In Ames assays, supplementation of the media with glutathione or glutathione-S-transferase increases the mutagenicity of 1,2-dichloroethane (ATSDR, 2024; Gwinn et al., 2011). *Drosophila melanogaster* pretreated with buthionine sulfoximine

(BSO, an inhibitor of glutathione [GSH] synthesis) before inhalation exposure to 1,2-dichloroethane exhibited reduced mutations (measured using somatic mutation and recombination tests [SMARTs]) compared with those that were not pretreated (Romert et al., 1990). Pretreatment of fruit flies with an inducer of glutathione-S-transferase (phenobarbital) significantly increased mutation frequency (Romert et al., 1990). In support of these findings, Chroust et al. (2001) observed increased mutagenicity in transgenic fruit flies expressing human glutathione-S-transferase (A1 subunit), an effect that was mitigated by pretreatment with BSO.

Inhibition of CYP450 metabolism has been shown to potentiate DNA damage and increase DNA binding from exposure to 1,2-dichloroethane. In rats exposed to piperonyl butoxide in addition to 1,2-dichloroethane (via intraperitoneal injection), increased levels of hepatic DNA damage (measured with alkaline DNA unwinding assay) were seen in comparison to the levels in rats treated with 1,2-dichloroethane alone (Storer and Conolly, 1985). Oral dosing of the aldehyde dehydrogenase inhibitor disulfiram increased the blood levels of 1,2-dichloroethane in rats by 5-fold when administered via inhalation and significantly increased the incidences of testes tumors and mammary gland adenocarcinomas which has relevance to humans with this enzyme defect to clear reactive aldehydes Cheever et al. (1990). Similarly, increased DNA binding in the liver, kidney, spleen, and testes was observed in rats exposed to 1,2-dichloroethane by inhalation with concurrent dietary exposure to disulfiram (relative to 1,2-dichloroethane exposure alone) (Igwe et al., 1986a).

Mammary Gland Cancer Mechanisms

Lebaron et al. (2021) conducted *in vivo* experiments to assess potential mechanisms of rodent mammary tumors induced by 1,2-dichloroethane. The study authors exposed female F344 rats by inhalation to 0 or 200 ppm 1,2-dichloroethane for 6 hours/day on at least 28 consecutive days. At sacrifice, blood samples were obtained for assessment of serum prolactin, and mammary tissues were collected for histopathology and assays of epithelial cell proliferation (Ki-67 immunohistochemistry), DNA damage (Comet assay), and levels of glutathione, reduced glutathione, and oxidized glutathione. There was no difference between exposed and control groups for any of these endpoints, nor was there an effect of exposure on 8-oxo-2'-deoxyguanosine (8-OHdG) adduct levels—a marker of oxidative DNA damage. Exposure to 1,2-dichloroethane did, however, induce a significant increase in S-(2-N7-guanylethyl) glutathione DNA adducts, as also found in the liver in this and other studies (see more below). *In vitro* studies have shown these adducts to be mutagenic (Gwinn et al., 2011). Lebaron et al. (2021), however, argue that *in vivo* evidence does not support this conclusion and that these adducts should be considered biomarkers of exposure, rather than mutagenic adducts.

No other data on potential mechanisms were located. The DNA adducts in mammary tissue resulting from 1,2-dichloroethane exposure *in vivo* could plausibly be related to subsequent formation of mammary tumors, although the role of these adducts in carcinogenicity of 1,2-dichloroethane has not been conclusively demonstrated.

Lung Cancer Mechanisms

Studies relevant to carcinogenic mechanisms of 1,2-dichloroethane-induced lung cancers are limited to measurements of DNA damage in the lung of mice exposed by intraperitoneal injection (Sasaki et al., 1998) and quantification of DNA adducts in the lungs of rats and mice also exposed by intraperitoneal injection (Baertsch et al., 1991; Prodi et al., 1988). Increased DNA damage (measured by alkaline single cell gel [SCG] assay and compared with measurement at time 0) was observed in the lungs of mice when measured 3 or 24 hours after dosing with 200 mg/kg 1,2-dichloroethane (Sasaki et al., 1998). DNA binding in the lungs of female rats was observed after 12 hours of inhalation exposure to ^{14}C-1,2-dichloroethane (Baertsch et al., 1991). Prodi et al. (1988) observed higher binding of ^{14}C-1,2-

dichloroethane to DNA in the lungs of mice compared with rats, consistent with the susceptibility of mice, but not rats, to 1,2-dichloroethane-induced lung tumors (Nagano et al., 2006). Experiments on binding of radiolabeled 1,2-dichloroethane to calf thymus DNA in the presence of microsomes and/or or cytosol from mouse and rat lung indicated binding in the presence of lung-derived microsomes (containing CYP450), but not cytosol (containing glutathione-S-transferase) (Prodi et al., 1988).

In an *in vitro* experiment, Matsuoka et al. (1998) observed dose-related increases in chromosomal aberrations in Chinese hamster lung fibroblast (CHL) cells when incubated with 1,2-dichloroethane in the presence of S9. In the absence of S9, the results were judged to be equivocal (Matsuoka et al., 1998).

No other data on potential mechanisms were located. The observed genotoxic effects and DNA binding/adduct formation in lung tissue following 1,2-dichloroethane exposure *in vitro* and *in vivo* could plausibly be related to subsequent formation of lung tumors, although a direct connection between these events and 1,2-dichloroethane-induced lung carcinogenesis has not been conclusively demonstrated.

Liver Cancer Mechanisms

One study evaluated potential mutations in the livers of animals exposed to 1,2-dichloroethane. Hachiya and Motohashi (2000) measured the frequency of hepatic tissue *lacZ* mutations in the Muta™ Mouse model 14 and 28 days after single gavage doses up to 150 mg/kg-bw or after repeated intraperitoneal injections resulting in cumulative doses up to 280 mg/kg-bw. No increase in mutation frequency was observed in the liver in any of the experiments.

When measured 3 and 24 hours after mice were exposed to 1,2-dichloroethane by intraperitoneal injection, an increase in DNA damage in the liver was detected by alkaline SGC assay (when compared to levels seen at time 0) (Sasaki et al., 1998). Significant decreases in the percentage of double-stranded DNA were observed in mice given single intraperitoneal doses of 300 mg/kg (Taningher et al., 1991) or 2 and 3 mmol/kg (200 and 300 mg/kg) (Storer and Conolly, 1983). Storer et al. (1984) assessed route differences in DNA damage in the livers of mice exposed by gavage (100–400 mg/kg), intraperitoneal injection (100–300 mg/kg), and inhalation (4 hours at 150–2,000 ppm). The fraction of double stranded DNA was significantly decreased in a dose-related fashion at all doses (≥100 mg/kg) after gavage administration, at doses greater than or equal to 150 mg/kg after intraperitoneal injection, and at concentrations greater than or equal to 1,000 ppm after inhalation exposure. Although the lower doses producing DNA damage by oral and intraperitoneal exposure did not produce systemic effects in parallel groups of similarly treated mice, all concentrations producing DNA damage by inhalation exposure were lethal to the similarly exposed mice (Storer et al., 1984). In a study comparing alkylation of hepatic DNA in rats and mice exposed to 1,2-dichloroethane by intraperitoneal injection, higher levels of alkylation were observed in mice compared with rats (at least 40-fold higher in the first 30 minutes after dosing) (Banerjee, 1988).

Binding of 1,2-dichloroethane or its metabolites to hepatic DNA of rats and mice exposed *in vivo* has been demonstrated in a number of studies (Lebaron et al., 2021; Watanabe et al., 2007; Baertsch et al., 1991; Prodi et al., 1988; Inskeep et al., 1986). Available data show sex-, species-, and dose-related differences in adduct levels. For example, an early study that compared DNA adduct levels in the livers of male rats and mice exposed to 1,2-dichloroethane by intraperitoneal injection (127 µCi/kg) showed higher binding in mouse compared to rat (Prodi et al., 1988). In contrast, in hepatic tissue from male and female mice and male rats exposed by intraperitoneal administration of a much lower dose of 1,2-dichloroethane (21 µCi/kg, corresponding to 5 mg/kg), the highest levels of adducts were in female mice (57 fmol/mg DNA), followed by male rats (46 fmol/mg DNA) and male mice (29 fmol/mg DNA) (Watanabe et al., 2007). In rats exposed by inhalation (50 ppm) for 2 years and then given a single oral

dose of radiolabeled 1,2-dichloroethane, no exposure-related difference in DNA adduct levels was detected (Cheever et al., 1990). Notably, this exposure level also failed to induce an increase in tumors at any site.

DNA adducts from the glutathione metabolic pathway have been demonstrated to occur in the livers of laboratory rodents exposed *in vivo*. In mice and rats administered 5 mg/kg 1,2-dichloroethane by intraperitoneal injection, the primary adduct was S-(2-N7-guanylethyl) glutathione (Watanabe et al., 2007). Similarly, in rats given 150 mg/kg ^{14}C-1,2-dichloroethane by intraperitoneal injection and sacrificed 8 hours later, prominent adducts in the liver were identified by high-performance liquid chromatography (HPLC) as S-[2-(N7-guanyl)ethyl]glutathione and S-[2-(N7-guanyl)ethyl]cysteinylglycine (Inskeep et al., 1986). Also, after 28 days of inhalation exposure to 200 ppm 1,2-dichloroethane, a significant increase in S-(2-N7-guanylethyl) glutathione DNA adducts was detected in the livers of female rats (Lebaron et al., 2021). As discussed above for mammary tumors, there is some uncertainty as to the toxicological significance of these adducts. Although *in vitro* studies have shown these adducts to be mutagenic (Gwinn et al., 2011), Lebaron et al. (2021) argue that *in vivo* evidence does not support this conclusion and that these adducts should be considered biomarkers of exposure, rather than mutagenic adducts.

One study was located presenting *in vitro* data pertaining to the genotoxicity of 1,2-dichloroethane in the liver. In this study, 1,2-dichloroethane induced DNA repair in both rat and mouse primary hepatocytes (Milman et al., 1988).

No other data on potential mechanisms were located. The observed DNA damage and DNA binding/adduct formation in liver tissue following exposure to 1,2-dichloroethane *in vitro* and *in vivo* could plausibly be related to subsequent formation of liver tumors, although a direct connection between these events and 1,2-dichloroethane-induced liver carcinogenesis has not been conclusively demonstrated.

Circulatory System Cancer Mechanisms

Data pertaining to mechanisms of circulatory system cancers induced by 1,2-dichloroethane consist of genotoxicity studies, including one *in vivo* study in rats (Lone et al., 2016), three *in vivo* studies in mice (Witt et al., 2000; Sasaki et al., 1998; Giri and Que Hee, 1988), and three *in vitro* experiments in human lymphoblastoid cells or lymphocytes (Tafazoli et al., 1998; Doherty et al., 1996; Crespi et al., 1985). Rats exposed by intraperitoneal injection to doses of 80.7, 161.4, or 242.1 mg/kg-bw exhibited statistically significant, dose-related increases in the incidences of chromosomal aberrations and micronuclei in bone marrow, as well as DNA damage (measured by alkaline comet assay) in blood cells (Lone et al., 2016). In mice exposed by intraperitoneal injection, significant increases in sister chromatid exchange frequencies (Giri and Que Hee, 1988) and DNA damage (Sasaki et al., 1998) were observed in bone marrow. However, 90 days of drinking water exposure to 1,2-dichloroethane (up to 8,000 mg/L) did not increase the frequency of micronuclei in mice (Witt et al., 2000). A study of workers exposed to 1,2-dichloroethane and vinyl chloride showed increased sister chromatid exchanges in the blood of those exposed to moderate levels of 1,2-dichloroethane with low levels of vinyl chloride exposure (Cheng et al., 2000).

Several *in vitro* genotoxicity experiments have been conducted in cells of the circulatory system. Increases in mutations (measured using the hypoxanthine-guanine phosphoribosyltransferase [HGPRT] assay) and micronuclei were observed in human lymphoblastoid cells cultured with 1,2-dichloroethane (Doherty et al., 1996; Crespi et al., 1985). Incubation with 1,2-dichloroethane resulted in increased

micronuclei and DNA damage (by Comet assay) in human peripheral lymphocytes in the absence of exogenous metabolic activation (Tafazoli et al., 1998).

No other data on potential mechanisms were located. The observed genotoxic effects of 1,2-dichloroethane in hematopoietic cells and tissues *in vitro* and *in vivo* could plausibly be related to subsequent formation of tumors—although a direct connection between these events and 1,2-dichloroethane-induced circulatory system cancers has not been conclusively demonstrated.

Summary

1,2-dichloroethane is likely to be carcinogenic to humans based on evidence of tumorigenicity in animal studies, including multiple tumor sites in male and/or female rats and/or mice by oral, inhalation, and/or dermal exposure. The occurrence of tumors in multiple tissues and treated groups is suggestive of a genotoxic mode of action, and most data relating to mode of action for 1,2-dichloroethane carcinogenicity are assays for genetic toxicity. Evidence from *in vivo* studies using multiple animal species and routes of exposure and *in vitro* studies using multiple test systems indicates that 1,2-dichloroethane and/or its metabolites can induce mutations, chromosomal aberrations, DNA damage, and DNA binding/adduct formation in certain test systems. The available data also show that biotransformation of 1,2-dichloroethane to reactive metabolites via a major CYP450-mediated oxidative pathway and a minor glutathione conjugation pathway contributes to the observed effects. *In vivo* and *in vitro* data showing genotoxicity and DNA binding/adduct formation in tissues where tumors associated with 1,2-dichloroethane exposure have been observed (mammary gland, lung, liver, and circulatory system) support that these effects could plausibly be related to formation of tumors in these tissues, although a direct connection between these events and 1,2-dichloroethane-induced carcinogenesis has not been conclusively demonstrated. Potential nongenotoxic modes of action were explored only in one study of rat mammary tissue and no supporting results were obtained.

N.4 Equations

Appendix N.4 provides the equations used in derivation of non-cancer and cancer PODs for 1,2-dichloroethane risk assessment. Appendix N.5 describes the non-cancer POD derivation for acute, short/intermediate-term, and chronic durations.

N.4.1 Equations

This section provides equations used in calculating non-cancer PODs, including air concentration conversions (ppm to mg/m^3 and the converse), adjustments for continuous exposure, calculation of human equivalent concentrations (HECs) and human equivalent doses (HEDs), and route-to-route extrapolation calculations. All PODs were initially derived for continuous exposure scenarios (7 days/week, and 24 hours/day for inhalation). See Appendix N.4.1.5 for the calculated continuous exposure PODs as well as PODs converted for use in occupational exposure scenarios (8 hours/day, 5 days/week).

N.4.1.1 Air Concentration Unit Conversion

It is often necessary to convert between ppm and mg/m^3 due to variation in concentration reporting in studies and the default units for different OPPT models. Therefore, EPA presents all PODs in equivalents of both units to avoid confusion and errors. Equation_Apx N-2 presents the conversion of the HEC from ppm to mg/m^3 and Equation_Apx N-3 shows the reverse conversion.

Equation_Apx N-2. Converting ppm to mg/m^3

$$HEC_{continuous}\,(mg/m^3) = HEC_{continuous}\,(ppm) * (molecular\ weight/24.45)$$

Equation_Apx N-3. Converting mg/m^3 to ppm

$$HEC_{continuous}\ (ppm) = HEC_{continuous}\ (mg/m^3\ )\ *\ (24.45/molecular\ weight)$$

For 1,1-dichloroethane, the molecular weight used in the equations is 98.96 mg/mmol.

N.4.1.2 Adjustment for Continuous Exposure

Non-cancer PODs for oral studies are adjusted from the exposure scenario of the original study to continuous exposure following Equation_Apx N-4.

Equation_Apx N-4. Adjusting Non-Cancer Oral POD for Continuous Exposure

$$POD_{continuous}\ =\ POD_{study}\ \times\ (days - week_{study}/days - week_{continous})$$

Where:

$days - week_{continuous}$ = 7 days

Non-cancer PODs for inhalation studies are adjusted from the exposure scenario of the original study to continuous exposure following Equation_Apx N-5.

Equation_Apx N-5. Adjusting Non-Cancer Inhalation POD for Continuous Exposure

$$POD_{continuous}\ =\ POD_{study}\ \times\ (hours - day_{study}/hours - day_{continous})\ \times\ (days - week_{study}/days - week_{continous})$$

Where:

$hours - day_{continous}$ = 24 hours
$days - week_{continous}$ = 7 days

N.4.1.3 Calculation of HEDs and HECs from Animal PODs

Consistent with U.S. EPA (2011b) guidance, oral PODs from animal studies are scaled to HEDs using Equation_Apx N-6.

Equation_Apx N-6. Calculation of Continuous HED from Continuous Animal Oral POD

$$HED_{\text{continous}} = POD_{\text{continous}} \times DAF$$

Where:

$HED_{continous}$ = Human equivalent dose for continuous exposure (mg/kg-day)
$POD_{continous}$ = Oral POD assuming daily doses (mg/kg-day)
DAF = Dosimetric adjustment factor (unitless)

DAFs for scaling oral animal PODs to HEDs are calculated using Equation_Apx N-7.

Equation_Apx N-7. Calculating DAF for Oral HED Calculation

$$DAF = \left(\frac{BW_A}{BW_H}\right)^{\frac{1}{4}}$$

Where:

DAF = Dosimetric adjustment factor (unitless)
BW_A = Body weight of species used in toxicity study (kg)

BW_H = Body weight of adult human (kg)

U.S. EPA (2011b) presents DAFs for extrapolation to humans from several species. However, because those DAFs used a human body weight of 70 kg, EPA has updated the DAFs using a human body weight of 80 kg from the EPA *Exposure Factors Handbook* (U.S. EPA, 2011a). The Agency used the body weights of 0.025 and 0.25 kg for mice and rats, respectively, as presented in U.S. EPA (2011b). The resulting DAFs for mice and rats are 0.13 and 0.24, respectively. For guinea pigs, EPA used a body weight of 0.43 kg, resulting in a DAF of 0.27.

U.S. EPA (1994) guidance was used to convert animal inhalation PODs to HECs. Effects in animals exposed to 1,1-dichloroethane by inhalation consisted of systemic (extrarespiratory) effects. Therefore, consistent with U.S. EPA (1994) guidance, the HEC for extrarespiratory effects is calculated by multiplying the animal POD by the ratio of the blood:gas partition coefficients in animals and humans. Equation_Apx N-8 shows the HEC calculation for extrarespiratory effects.

Equation_Apx N-8. Calculation of HEC from Animal Inhalation POD

$$HEC = POD_{continuous} \times \frac{\left(\frac{HB}{g}\right)_A}{\left(\frac{HB}{g}\right)_H}$$

Where:

$\frac{\left(\frac{HB}{g}\right)_A}{\left(\frac{HB}{g}\right)_H}$ = blood:air partition coefficient for animals (A) to humans (H)

Blood:air coefficients for 1,2-dichloroethane were 19.5 in humans and 30 in rats (Gargas et al., 1989). Blood:air partition coefficients for other species were not located. When the animal blood:air partition coefficient is greater than the human blood:air partition coefficient, the default ratio of 1 is used in the calculation in accordance with U.S. EPA (1994) guidance. Nasal effects were observed in one study of F344 rats exposed by inhalation to 1,2-dichloroethane (Dow Chemical, 2006b). For nasal effects, in accordance with U.S. EPA (1994) guidance, the HEC was calculated using the regional gas dose ratio for extrathoracic effects ($RGDR_{ET}$) using Equation_Apx N-9.

Equation_Apx N-9. Calculating HEC Using Animal Inhalation POD and $RGDR_{ET}$

$$HEC_{\text{continuous}} = POD_{\text{continuous}} \times RGDR_{ET}$$

Where:

$HEC_{continuous}$ = Human equivalent concentration for continuous exposure (mg/m^3)
$POD_{continuous}$ = Animal POD for continuous exposure (mg/m^3)
$RGDR_{ET}$ = Regional gas dose ratio for extrathoracic effects (unitless)

The $RGDR_{ET}$ for nasal effects in F344 rats was calculated as shown in Equation_Apx N-10.

Equation_Apx N-10. Calculating $RGDR_{ET}$ in Rats

$$RGDR_{ET} = \frac{V_{E_a}}{SA_a} \Big/ \frac{V_{E_h}}{SA_h}$$

Where:

$RGDR_{ET}$	=	Regional gas dose ratio for extrathoracic effects (unitless)
V_{E_a}	=	Ventilation rate for male and female F344 rats = 0.211 L/minute (U.S. EPA, 1994)
SA_a	=	Surface area of the extrathoracic region in rats = 15 cm^2 (U.S. EPA, 1994)
V_{E_h}	=	Ventilation rate for humans = 13.8 L/minute (U.S. EPA, 1994)
SA_h	=	Surface area of the extrathoracic region in humans = 200 cm^2 (U.S. EPA, 1994)

The $RGDR_{ET}$ for nasal effects in F344 rats calculated using the equation above is 0.2.

N.4.1.4 Cancer Inhalation Unit Risk

For cancer risk assessment, an Inhalation Unit Risk (IUR) can be converted to a Cancer Slope Factor (CSF) using the exposure parameters described above for non-cancer conversions, as in Equation_Apx N-11.

Equation_Apx N-11. Calculating CSF from IUR

$$CSF = IUR \times \frac{BW_H}{IR_R}$$

Where:

CSF	=	Oral cancer slope factor based on daily exposure (per mg/kg-day)
IUR	=	Inhalation unit risk based on continuous daily exposure (per mg/m^3)
BW_H	=	Body weight of adult humans (kg) = 80
IR_R	=	Inhalation rate for an individual at rest (m^3/day) = 14.7

N.4.1.5 Conversion of Continuous PODs to Worker PODs

All PODs were initially derived for continuous exposure, and then converted to an equivalent POD for occupational exposure for convenience in risk calculations. Equation_Apx N-12 and Equation_Apx N-13 were used to convert from continuous to occupational exposure scenarios for oral and inhalation non-cancer PODs, respectively.

Equation_Apx N-12. Adjusting Non-Cancer Oral POD from Continuous to Occupational Exposure

$$POD_{occupational} = POD_{continuous} \times (7/5\ days/week)$$

Equation_Apx N-13. Adjusting Non-Cancer Inhalation POD from Continuous to Occupational Exposure

$$POD_{occupational} = POD_{continuous} \times (24/8\ hours/day) \times (7/5\ days/week)$$

To adjust a continuous IUR for occupational scenarios, Equation_Apx N-14 was used (days per week adjustment is not required because it is already accounted for in the LADC).

Equation_Apx N-14. Adjusting Continuous IUR For Occupational Scenarios

$$IUR_{occupational} = IUR_{continuous} \times (hours - day_{occupational}/hours - day_{continuous})$$

N.5 Summary of Continuous and Worker Non-Cancer PODs

Each of the continuous non-cancer PODs described in the preceding sections was converted to an equivalent POD for occupational exposure for convenience in risk calculations. Equations used to convert from continuous to occupational exposure scenarios for oral and inhalation exposure, respectively are provided in Appendix N.4. Table_Apx N-28 provides a summary of the non-cancer PODs for both continuous and occupational exposure scenarios for 1,1-dichloroethane using read-across from 1,2-dichloroethane.

Table_Apx N-28. Summary of Non-Cancer PODs for 1,1-Dichloroethane (Read-Across from 1,2-Dichloroethane)

Route	Duration	Continuous POD	Worker POD	Benchmark MOE	Reference
Oral	Acute	19.9 mg/kg-bw/day	19.9 mg/kg-bw/day	30	Storer et al. (1984)
	Intermediate	6.5 mg/kg-bw/day	9.1 mg/kg-bw/day	30	NTP (1991)
	Chronic	6.5 mg/kg-bw/day	9.1 mg/kg-bw/day	300	NTP (1991)
Inhalation	Acute	9.78 mg/m^3	41 mg/m^3	30	Dow Chemical (2006b)
	Short/ Intermediate	21.2 mg/m^3	89 mg/m^3	30	Zhang et al. (2017)
	Chronic	21.2 mg/m^3	89 mg/m^3	300	Zhang et al. (2017)
Dermal (route-to-route extrapolation from oral)	Acute	19.9 mg/kg-bw/day	19.9 mg/kg-bw/day	30	Storer et al. (1984)
	Intermediate	6.5 mg/kg-bw/day	9.1 mg/kg-bw/day	30	NTP (1991)
	Chronic	6.5 mg/kg-bw/day	9.1 mg/kg-bw/day	300	NTP (1991)

N.6 Evidence Integration Tables for Non-Cancer for 1,1-Dichloroethane

Table_Apx N-29. Evidence Integration Table for Reproductive/Developmental Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgment	Inferences Across Evidence Streams and Overall Weight of Scientific (WOSE) Evidence Judgement
Evidence Integration Summary Judgement on Reproductive/Developmental Effects				
Evidence from human studies				*Overall WOSE judgement for reproductive/developmental effects based on integration of information across evidence streams:* Evidence is inadequate to assess whether 1,1-dichloroethane exposure may cause reproductive/ developmental toxicity under relevant exposure circumstances.
• A retrospective case-control study of mother-infant pairs evaluated exposure based on maternal residential proximity to industrial air releases and its association with birth defects (neural tube, oral cleft, and heart defects; limb deficiencies; and anencephaly) (Brender et al., 2014). Study quality: High	Biological gradient/dose-response: • Spina bifida and septal heart defects were associated with maternal residential exposures (any vs. none) to 1,1-dichloroethane. Magnitude and precision: • The study was large and accounted for multiple facilities and their chemical releases, allowing for evaluations of associations between exposure to individual chlorinated solvents and specific birth defects. Quality of the database: • Associations between birth defects and exposure were observed in a high-quality study.	Biological gradient/dose-response: • Analyses based on quartiles of exposure intensity did not show a dose-response relationship with spina bifida or septal heart defects. Magnitude and precision: • Exposure was based on maternal address at delivery and industry releases reported to TRI; changes in address between conception and delivery and failure to account for prevailing wind directions may have contributed to exposure misclassification. • Effect estimates were not adjusted for concurrent exposure to other chemicals.	*Key findings*: Available epidemiological data are limited and inconclusive. *Overall WOSE judgement for reproductive/ developmental toxicity effects based on human evidence:* • Indeterminate	
Evidence from apical endpoints in *in vivo* mammalian animal studies				

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgment	Inferences Across Evidence Streams and Overall Weight of Scientific (WOSE) Evidence Judgement
Oral: • Short-term, subchronic, and chronic gavage studies in male rats and male and female mice examined histology of the testes, epididymis, prostate, mammary gland, ovary, and/or uterus (Muralidhara et al., 2001; NCI, 1978). Study quality: High Inhalation: • A subchronic inhalation toxicity study in male dogs evaluated testis histopathology (Mellon Institute, 1947). Study quality: Medium • An inhalation study that exposed female rats during GD 6–15 evaluated numbers of litters, corpora lutea, implantations, resorptions, and live fetuses; fetal sex, length, and body weights; and gross, soft tissue, and skeletal anomalies (Schwetz et al., 1974). Study quality: Medium Study quality ranked as Uninformative: • Chronic gavage studies in male and female rats [a] examined histology of the testes, epididymis, prostate, mammary gland, ovary, and/or uterus (NCI, 1978). • A subchronic inhalation toxicity study in male rats [b]	Biological gradient/dose-response: • A significantly increased litter incidence of delayed ossification of sternebrae was observed in the offspring of rats exposed via inhalation at the higher of two tested concentrations. • In a study ranked as Uninformative because methodological details were not fully reported, lengthening of the estrus phase was reported in female rats exposed via inhalation for 2–3 months prior to mating, and embryolethality was increased in female rats exposed prior to and throughout gestation (but not in those exposed only prior to gestation).	Consistency: • In the study reporting delayed sternebral ossification associated with exposure, separate control groups used for each exposure level showed significantly different incidences of this outcome. The incidence in the higher exposed group was statistically significant only compared with the concurrent control, which had a much lower incidence than the other control group. Biological plausibility: • Maternal weight gain and food intake were decreased at the same exposure level that resulted in increased incidence of delayed ossification in rat offspring. Magnitude and precision: • Only one concentration was tested in the Uninformative study that identified effects on embryonic mortality. Quality of the database: • The database lacks a 1- or 2-generation reproduction toxicity study of acceptable quality, and only one developmental toxicity study is available. • Data pertaining to effects on estrous cyclicity and preimplantation viability	*Key findings*: Available animal toxicological studies are limited and inconclusive. *Overall WOSE judgement for reproductive/develop-mental effects based on animal evidence:* • Indeterminate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgment	Inferences Across Evidence Streams and Overall Weight of Scientific (WOSE) Evidence Judgement
evaluated testis histopathology (Mellon Institute, 1947). • An inhalation study [c] that exposed female rats during premating, mating, and/or gestation evaluated mating, fertility, fetal development, estrous cyclicity, and histology of the ovaries (Vozovaia, 1977).		are limited to a single study rated Uninformative. • The subchronic inhalation toxicity study in dogs, which did not identify effects on testis histology, used only one mixed-breed animal and lacked methodological details. • Several of the available studies were rated Uninformative based on reporting limitations, high incidences of pathological findings in negative controls, and/or mortality unrelated to exposure.		
Evidence from mechanistic studies – indeterminate (no studies)				

[a] The 78-week study in male and female rats (NCI, 1978) was considered Uninformative owing to high mortality related to pneumonia.
[b] The subchronic inhalation study in male and female rats (Mellon Institute, 1947) was considered Uninformative owing to high incidences of pathological findings in controls and high mortality due to virus or infection.
[c] The reproductive/developmental inhalation study in female rats (Vozovaia, 1977) was considered Uninformative because methodological details regarding exposure (type of inhalation exposure, description of air chamber, number of air changes, etc.) were not reported.

Table_Apx N-30. Evidence Integration Table for Renal Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Renal Effects				
Evidence from human studies (none)			• Indeterminate	*Overall WOSE judgement for renal effects based on integration of information across evidence streams:* Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes renal toxicity under relevant exposure circumstances.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Oral: • Short-term and subchronic gavage studies in male rats evaluated blood urea nitrogen (BUN), urinalysis parameters, kidney weights, and/or gross and microscopic pathology of the kidney (Muralidhara et al., 2001). Study quality: High • A chronic gavage study in male and female mice evaluated gross and microscopic pathology of the kidney and urinary bladder (NCI, 1978). Study quality: High Inhalation: • A subchronic inhalation study in dogs evaluated BUN and kidney histology (Mellon Institute, 1947). Study quality: Medium • Subchronic inhalation studies in male and female rats, guinea pigs, and rabbits evaluated BUN, serum creatinine, urinalysis parameters, kidney weights, and/or kidney histology (Hofmann et al., 1971). Study quality: Medium Study quality ranked as Uninformative: • A chronic gavage study in male and female rats [a] evaluated gross and microscopic pathology of the kidney and urinary bladder (NCI, 1978).	Biological gradient/dose-response: • Absolute kidney weight was significantly decreased at the two highest doses in male rats evaluated after 10 days of gavage exposure. • Urinary excretion of acid phosphatase (ACP) and N-acetylglucosaminidase (NAG) were significantly increased at the three highest doses tested in male rats after 8 weeks of gavage exposure. • In a study ranked as Uninformative, increased BUN and serum creatinine were observed in cats after 26 weeks of exposure via inhalation. Three of four treated cats also showed renal tubular dilatation. • In acute and short-term intraperitoneal studies ranked as Uninformative (due to limited reporting on negative controls and lack of histological examinations in controls, respectively); male mice showed dose-related increases in percentages of animals with "significant" urinary protein and glucose [d] levels; swelling of >50% of the renal proximal tubules was reported in 3/5 mice at the mid-dose. Quality of the database: • Kidney effects were observed in one high-quality study and in two studies ranked as Uninformative.	Biological gradient/dose-response: • Urinary excretion of ACP was significantly decreased at all doses after 12 weeks of gavage exposure in male rats. Urinary NAG in treated rats was not different from the control at this time point. Consistency: • The changes in kidney weights and urinary parameters in the gavage studies did not correspond to adverse histopathology changes in rats, and no renal histopathology changes were seen in mice exposed chronically by gavage or in dogs, rats, guinea pigs, or rabbits exposed subchronically by inhalation. Magnitude and precision: • Changes in BUN and serum creatinine in cats were influenced by values for one cat that was sacrificed after 23 weeks due to poor general condition. In addition, only four cats/group were tested. • In a study ranked as Uninformative due to the lack of histological examinations in controls, a cut-off value was	*Key findings*: Available toxicological studies showed changes in kidney weight, clinical chemistry, urinary excretion, and/or kidney histology. However, many of the studies that observed effects had limitations, and kidney effects were not seen consistently across studies using different species, exposure routes, or study durations. *Overall WOSE judgement for renal effects based on animal evidence:* • Indeterminate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• A subchronic inhalation study in male and female rats [b] evaluated kidney weights and histology (Mellon Institute, 1947). • Subchronic inhalation studies in cats evaluated BUN, serum creatinine, urinalysis parameters, kidney weights, and kidney histology (Hofmann et al., 1971). • Acute and short-term intraperitoneal studies in male mice [c] evaluated urinary glucose and protein and kidney histology (Plaa and Larson, 1965).		used to quantify effects on kidney histology in mice (>50%, or <50% of the proximal tubule area affected) and histological results were only reported for mid-dose animals. Quality of the database: • The subchronic inhalation toxicity study in dogs, which did not identify effects on BUN or kidney histology, used only one mixed-breed animal and lacked methodological details. Biological plausibility: • In the 10-day gavage study in male rats, decreased absolute kidney weights occurred in conjunction with decreased body weight; there were no significant changes in relative kidney weight.		
Evidence from mechanistic studies (none)			• Indeterminate	

[a] The study in male and female rats was ranked as Uninformative owing to high mortality related to pneumonia.
[b] The 6-month study in male and female rats was ranked as Uninformative because negative controls had a high incidence of pathological lesions and there was high mortality related to virus or infection.
[c] The acute and short-term intraperitoneal studies in male mice were ranked as Uninformative because details regarding negative controls were not reported and histology was not performed in controls, respectively.
[d] "Significant" urinary protein and glucose was quantified as 100 and 250 mg%, respectively.

Table_Apx N-31. Evidence Integration Table for Hepatic Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Hepatic Effects				
Evidence from human studies (none)			• Indeterminate	*Overall WOSE judgement for hepatic effects based on integration of information across evidence streams:* Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes hepatic toxicity under relevant exposure circumstances.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Oral: • Short-term and subchronic gavage studies in male rats evaluated serum liver enzymes (ALT, SDH, and OCT), liver weights, and gross and microscopic pathology of the liver (Muralidhara et al., 2001). Study quality: High • A chronic gavage study in male and female mice evaluated gross and microscopic pathology of the liver (NCI, 1978). Study quality: High • Nine-week studies in male rats determined the potential for tumor initiation or promotion based on numbers of GGT-positive foci in the liver (Milman et al., 1988; Story et al., 1986). Study quality: High Inhalation: • A subchronic inhalation study in dogs evaluated liver function (bromsulphthalein excretion and thymol-barbital turbidity) and histology (Mellon Institute, 1947). Study quality: Medium • Subchronic inhalation toxicity studies in male and female rats, guinea pigs, and rabbits evaluated serum ALT and AST and liver function (bromsulphthalein test), weights, and histology (Hofmann et al., 1971). Study quality: Medium • An inhalation study that exposed nonpregnant female rats for 10 days or pregnant rats on GD 6–15 evaluated serum ALT and AST, liver weights, and gross	Biological gradient/dose-response: • Absolute and relative liver weights were significantly decreased in treated male rats after 5 and 10 days of gavage exposure. • Slight changes in hepatocyte histology (mild condensation and changes in cytoplasmic staining consistent with glycogen mobilization) were reported in male rats treated via gavage for 11 weeks. • Exposure resulted in increased numbers of GGT-positive foci in the livers of male rats pretreated with a tumor initiator. • Nonpregnant female rats exposed for 10 days via inhalation showed increased relative liver weight. Quality of the database: • Liver effects were observed in high- and medium-quality studies.	Biological gradient/dose-response: • Changes in hepatocyte histology were observed only at a dose that caused significant mortality (8/15 rats) and in the absence of liver weight or clinical chemistry changes. Consistency: • Changes in liver weight (increased in female rats exposed via inhalation and decreased in male rats treated by gavage) were observed in 10-day toxicity studies but not in longer-duration studies in rats, guinea pigs, rabbits, or cats. • Increased liver weight was observed after a 10-day exposure of nonpregnant rats but there were no liver effects in females exposed to the same concentration during GD 6–15. • Chronic oral exposure of mice did not result in liver pathology. Magnitude and precision: • Only one dose was used in the 9-week tumor initiation and promotion protocols. Quality of the database: • The subchronic inhalation toxicity study in dogs, which did not identify effects on liver functional tests or liver	*Key findings*: Available toxicological studies showed changes in liver weight and/or histology in the absence of relevant clinical chemistry findings. *Overall WOSE judgement for hepatic effects based on animal evidence:* • Slight	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
liver pathology (Schwetz et al., 1974). Study quality: Medium Study quality ranked as Uninformative: • A chronic gavage study in male and female rats [a] evaluated gross and microscopic pathology of the liver (NCI, 1978). • A subchronic inhalation study in male and female rats [b] evaluated icterus index, liver weights, fat content, and histology (Mellon Institute, 1947). • Subchronic inhalation toxicity studies in cats evaluated serum ALT and AST and liver function (bromsulphthalein test), weights, and histology (Hofmann et al., 1971). • An inhalation study [c] that exposed female rats during premating, mating, and/or gestation evaluated liver function (Quick-Pytel test) and/or liver weights (Vozovaia, 1977).		histology, used only one mixed-breed animal and lacked methodological details. • Several of the available studies, which did not identify liver effects, were ranked as Uninformative based on reporting limitations, high incidences of pathological findings in negative controls, and/or mortality unrelated to exposure. Biological plausibility and human relevance: • The toxicological significance of decreased liver weight in the 10-day gavage study in male rats is unclear and may be partly attributable to decreased body weights.		
Evidence from mechanistic studies (none)			• Indeterminate	

[a] The chronic study in male and female rats was ranked as Uninformative owing to high mortality related to pneumonia.
[b] The 6-month study in male and female rats was ranked as Uninformative because negative controls had a high incidence of pathological lesions and there was high mortality related to virus or infection.
[c] The reproductive/developmental inhalation study in female rats was considered Uninformative because methodological details regarding exposure (type of inhalation exposure, description of air chamber, number of air changes per hour, etc.) were not reported.

Table_Apx N-32. Evidence Integration Table for Nutritional/Metabolic Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Nutritional/Metabolic Effects				
Evidence from human studies (none)			• Indeterminate	*Overall WOSE judgement for nutritional/metabolic effects based on integration of information across evidence streams:* Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes body weight decrements under relevant exposure circumstances.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Oral: • Short-term and subchronic gavage studies in male rats evaluated body weight (Muralidhara et al., 2001). Study quality for endpoint: High • Six-week and 2-year gavage studies in male and female mice evaluated body weight (NCI, 1978). Study quality for endpoint: High • A cancer bioassay and a tumor promotion assay in male mice evaluated body weights during a 52-week drinking water exposure (Klaunig et al., 1986). Study quality for endpoint: High • Single dose initiation and 7-week promotion studies (gavage) in partially hepatectomized rats evaluated body weight (Milman et al., 1988). Study quality for endpoint: Medium Inhalation: • An inhalation study that exposed female rats during GD 6–15 evaluated maternal body weights (Schwetz et al., 1974). Study quality for endpoint: High • A 6-month inhalation study in one dog evaluated body weight (Mellon Institute, 1947). Study quality for endpoint: Medium • 26-week inhalation studies in male and female rats, guinea pigs, and rabbits evaluated body weight (Hofmann et al., 1971). Study quality for endpoint: Medium	Biological gradient/dose-response: • In the short-term and subchronic gavage studies in rats, significantly decreased body weights (≥10% relative to controls) were seen at ≥2,000 mg/kg-bw/day. • Maternal body weight was significantly decreased (≥0% relative to controls) at ≥3,798 ppm in rats exposed by inhalation during gestation. • One dog exposed to 1,067 ppm by inhalation for 6 months exhibited lower body weight than the control. Quality of the database: • Decreased body weight was observed in two high quality studies and one medium quality study.	Biological gradient/dose-response and Consistency: • No treatment-related change in body weight was observed in mice exposed to doses up to 2,885–3,331 mg/kg-bw/day by gavage for up to 78 weeks. • No treatment-related change in body weight was observed in rats exposed to doses up to 543 mg/kg-bw/day in drinking water for 52 weeks. • No treatment-related change in body weight was observed in initiation or promotion studies in partially hepatectomized rats exposed by gavage to doses up to 700 mg/kg-bw/day. • No treatment-related change in body weight was observed in male and female rats, guinea pigs, and rabbits exposed to 750 ppm by inhalation for 26 weeks. Magnitude and precision: • The magnitude of the body weight decrease (≈10%) in the gestational exposure study was small and the decrease lacked a dose-response relationship.	*Key findings*: 1,1-dichloroethane induced body weight decrements in rats at high gavage exposures (≥2,000 mg/kg-bw/day) and in one dog exposed by inhalation (1,067 ppm). No body weight effects were seen in mice or in rats at lower exposure levels. *Overall WOSE judgement for nutritional/metabolic effects based on animal evidence:* • Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Study quality ranked as Uninformative for this endpoint: • Six-week and chronic gavage studies in male and female rats [a] evaluated body weight (NCI, 1978). • A 6-month inhalation study in male and female rats [b] evaluated body weight (Mellon Institute, 1947). • A 26-week inhalation study in cats [c] evaluated body weight (Hofmann et al., 1971).		Quality of the database: • No treatment-related effects on body weight were observed in two high quality studies and two medium quality studies.		
Evidence from mechanistic studies (none)			• Indeterminate	

[a] The 6-week gavage study in rats was ranked Uninformative due to inadequate data reporting, and the chronic gavage study in rats was ranked as Uninformative owing to high mortality related to pneumonia.
[b] The 6-month inhalation study in male and female rats was ranked as Uninformative because a significant number of animals died due to apparent lung infections unrelated to exposure.
[c] The 26-week inhalation study in cats was ranked as Uninformative due to an intercurrent "catarrhal" infection that rendered it impossible to differentiate effects of infection from effects of exposure

Table_Apx N-33. Evidence Integration Table for Mortality

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Mortality				
Evidence from human studies (none)			• Indeterminate	*Overall WOSE judgement for mortality based on integration of information across evidence streams:* Evidence indicates that 1,1-dichloroethane exposure is likely to cause death under relevant exposure circumstances.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Oral: • An acute gavage study in guinea pigs evaluated mortality (Dow Chemical, 1947). Study quality for endpoint: Low • Acute, short-term, and subchronic gavage studies in male rats evaluated mortality (Muralidhara et al., 2001). Study quality for endpoint: High • A chronic gavage study in male and female mice evaluated mortality (NCI, 1978). Study quality for endpoint: High • A cancer bioassay and a tumor promotion assay in male mice evaluated mortality during a 52-week drinking water exposure (Klaunig et al., 1986). Study quality for endpoint: High Inhalation: • A 6-month inhalation study in one dog evaluated mortality (Mellon Institute, 1947). Study quality for endpoint: Low • 26-week inhalation studies in male and female rats, guinea pigs, and rabbits evaluated mortality (Hofmann et al., 1971). Study quality for endpoint: Medium Study quality ranked as Uninformative for this endpoint: • Six-week gavage studies in male and female mice and rats [a] evaluated mortality (NCI, 1978). • A chronic gavage study in male and female rats [b] evaluated mortality (NCI, 1978).	Biological gradient/dose-response: • In an acute gavage study, all guinea pigs (sample size not reported) died at 1,000 mg/kg-bw. • In an acute gavage study in rats, deaths occurred at doses ≥8,000 mg/kg-bw within 24 hours of dosing; the LD50 was 8200 mg/kg-bw. • In a short-term gavage study in rats, 3/8 rats died at 8,000 mg/kg-bw/day. • In a subchronic gavage study in rats, 1/15 rats died at 2,000 mg/kg-bw/day and 8/15 died at 4,000 mg/kg-bw/day. • In 6-week gavage studies ranked Uninformative due to the lack of mortality data at doses other than the highest dose, 2/5 female rats died at 3,160 mg/kg-bw/day, and 2/5 male mice and 3/5 female mice died at 5,620 mg/kg-bw/day. • In a chronic gavage study in mice, significantly reduced survival was observed at 2,885–3,331 mg/kg-bw/day. Quality of the database: • Mortalities were reported in high- and low-quality studies.	Biological gradient/dose-response and Consistency: • In the 52-week drinking water study, no effect on survival was observed at doses up to 543 mg/kg-bw/day. • No treatment-related effects on survival were seen in animals exposed by inhalation.	*Key findings*: Mortalities occurred in several species of animal exposed to 1,1-dichloroethane (≥1,000 mg/kg-bw) via gavage in high quality studies. *Overall WOSE judgement for mortality based on animal evidence:* • Robust	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• An inhalation study [c] that exposed female rats during premating, mating, and/or gestation evaluated mortality (Vozovaia, 1977). • A 6-month inhalation study in male and female rats [d] evaluated mortality (Mellon Institute, 1947). • A 26-week inhalation study in cats [e] evaluated mortality (Hofmann et al., 1971). • An acute intraperitoneal study in male mice [f] evaluated mortality (Plaa and Larson, 1965).				
Evidence from mechanistic studies (none)			• Indeterminate	

[a] The 6-week gavage studies in mice and rats were ranked as Uninformative because mortality data were reported only for the high dose group, and statistical analysis was not performed on mortality data.
[b] The chronic gavage study in male and female rats was ranked as Uninformative owing to high mortality related to pneumonia.
[c] The reproductive/developmental inhalation study in female rats was considered Uninformative because methodological details regarding exposure (type of inhalation exposure, description of air chamber, number of air changes per hour, etc.) were not reported
[d] The 6-month inhalation study in male and female rats was ranked as Uninformative because a significant number of animals died due to apparent lung infections unrelated to exposure.
[e] The 26-week inhalation study in cats was ranked as Uninformative due to an intercurrent "catarrhal" infection that rendered it impossible to differentiate effects of infection from effects of exposure.
[f] The acute intraperitoneal study in male mice was ranked as Uninformative because details regarding negative controls were not reported.

Table_Apx N-34. Evidence Integration Table for Neurological Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Neurological Effects				
Evidence from human studies (none)			• Indeterminate	*Overall WOSE judgement for neurological effects based on integration of information across evidence streams:* Evidence suggests, but is not sufficient to conclude, that 1,1-dichloroethane exposure causes neurological effects under relevant exposure circumstances.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Oral: • An acute gavage study in male rats evaluated clinical signs (Muralidhara et al., 2001). Study quality for endpoint: Medium • Short-term and subchronic gavage studies in male rats evaluated clinical signs, brain weight, and brain histopathology (Muralidhara et al., 2001). Study quality for endpoint: Medium Study quality ranked as Uninformative for this endpoint: • A chronic gavage study in male and female rats [a] evaluated clinical signs, brain histopathology, and gross pathology (NCI, 1978).	Biological gradient/dose-response: • Clinical signs of neurotoxicity (excitation followed motor impairment and sedation) were observed in rats given a single gavage dose of ≥2,000 mg/kg-bw. • Central nervous system depression (not further described) was observed in rats exposed to 2,000 mg/kg-bw/day for 13 weeks, and the rats exhibited protracted narcosis at 4,000 mg/kg-bw/day. Biological plausibility: • 1,1-dichloroethane was used as an anesthetic for humans (administered via inhalation) in the past (ATSDR, 2015). Quality of the database: • Clinical signs of central nervous system effects were seen in medium quality studies.	Consistency: • 1,1-dichloroethane exposure did not affect brain weight or histopathology after short-term or subchronic gavage exposure in rats. • 1,1-dichloroethane exposure did not induce clinical signs or changes in brain histopathology in mice exposed by gavage to doses up to 2,885–3,331 mg/kg-bw/day for 78 weeks. Quality of the database: • There are no studies of sensitive neurobehavioral endpoints.	*Key findings*: 1,1-dichloroethane induced central nervous system depression in rats exposed by gavage, and this finding is consistent with its past use as a human anesthetic. *Overall WOSE judgement for neurological effects based on animal evidence:* • Moderate	
Evidence from mechanistic studies (none)			• Indeterminate	

[a] The study in male and female rats was ranked as Uninformative owing to high mortality related to pneumonia.

N.7 Evidence Integration Tables for Non-Cancer for 1,2-Dichloroethane

Table_Apx N-35. 1,2-Dichloroethane Evidence Integration Table for Reproductive/Developmental Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Reproductive/Developmental Effects				
Evidence from human studies				Overall WOSE judgement for reproductive/developmental effects based on integration of information across evidence streams: Evidence indicates that 1,2-dichloroethane likely causes effects on male reproductive structure and/or function under relevant exposure conditions. Evidence is inadequate to determine whether 1,2-dichloroethane may cause effects on the developing organism. There is no evidence that 1,2-dichloroethane causes effects on female reproductive structure and/or function.
• A case-control study examined the association between proximity to point sources of chlorinated solvents and birth defects. Exposure was assessed based on metrics that combined residential distances to industrial sources and annual amounts of chemicals released (using EPA's Toxic Release Inventory), and birth defects were assessed using Texas birth registries. The geocoded address of mothers on day of delivery and the amount of solvent was used in the Emission Weighted Probability model to assign each mother an exposure risk value (Brender et al., 2014). Study quality: High • A retrospective cohort study examined the association between chlorinated solvents in drinking water and birth outcomes in 75 New Jersey towns. Exposure was based on measurements of chlorinated solvents in public water supplies in the maternal town of residence at the time of birth. Birth outcomes and some covariate data were obtained from birth certificates, fetal death certificates, and the NJ Birth Defects Registry (Bove, 1996; Bove et al., 1995). Study quality: Medium	Biological gradient/dose-response: • In women of all ages, any exposure to 1,2-dichloroethane (based on residential proximity to air emissions) was positively associated with neural tube defects OR =1.28 (CI 1.01, 1.62) and in particular spina bifida OR =1.64 (CI 1.24, 2.16). In analyses by intensity of exposure, significant trends were observed for spina bifida and also for septal heart defects. • Exposure to 1,2-dichloroethane in drinking water (detected vs. not detected) was positively associated with major cardiac defects (OR = 2.81, 95% CI 1.11, 6.65). This category of heart defects did not include septal defects, which were evaluated separately. Quality of the database: • Positive associations were found in high and medium quality studies.	Magnitude and precision: • Effect sizes were small and associations weak for all 1,2-dichloroethane outcomes in both studies (ORs ≤ 2.81, lower 95% CI ≤ 1.24). The association between 1,2-dichloroethane in drinking water and major cardiac defects was based on a very small number of cases (6 with detectable 1,2-dichloroethane). • In the Texas study, elective terminations lacked a vital record, so 31% of mothers with neural tube defects were not geocoded. • In both studies, there was the potential for exposure misclassification for mothers that changed residences between the first trimester (period relevant to morphogenesis of birth defects) and delivery, because exposure was based on residence at delivery. Consistency: • No significant associations were observed between 1,2-dichloroethane exposure in public water supplies and	*Key findings:* In high and medium quality studies, associations were observed between 1,2-dichloroethane exposure and various birth defects (neural tube defects including spina bifida and heart defects of different types). However, the effect sizes were small, the associations were weak and in some cases based on very low group sizes, results of the studies were not consistent (neural tube defects/spina bifida in one study but not the other; different types of cardiac defects in the two studies), and both studies were limited in various ways (e.g., incomplete data on neural tube defects, potential exposure misclassification, questionable temporality, co-exposures to other chemicals that were also associated with the same defects). *Overall WOSE judgement for reproductive/ developmental effects based on human evidence:* • Indeterminate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
		neural tube defects, septal heart defects, or total cardiac defects. Biological plausibility and human relevance: • There was limited evidence of temporality (exposure prior to outcome) in either study. • In both studies, subjects had multiple overlapping exposures, and positive associations with spina bifida or neural tube defects, heart defects, and other defects were found for many of the other chemicals considered in the analyses.		
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Effects on male reproductive organs				
• An inhalation study in rats evaluated testis weight and gross and microscopic pathology of the testes after 30 days exposure (Igwe et al., 1986b) Study quality: High • An inhalation study in a single dog evaluated testis histopathology after 6 months exposure (Mellon Institute, 1947) Study quality: Medium • An inhalation study in mice evaluated testis and epididymis weight, sperm parameters and morphology, histology of the testis, seminiferous tubules, and caput epididymis, and plasma and testis hormone levels after 1- or 4-week exposure (Zhang et al., 2017) Study quality: High • An inhalation study in rats and guinea pigs evaluated weight and gross and microscopic pathology of the testes after up to 212 and 246 days of exposure, respectively (Spencer et al., 1951) Study quality: Medium	Biological gradient/dose-response: • In mice exposed by inhalation for one week, decreased sperm concentration and motility, increased sperm abnormalities, and occasional testicular and epididymal histopathology changes) were seen at 700 mg/m3. After 4 weeks, effects seen at ≥ 350 mg/m3 included more pronounced sperm changes, more extensive/severe histological effects, and increases in plasma and testicular testosterone and LH and testicular GnRH. Consistency:	Quality of the database: • No studies of sperm parameters in any species other than mice were available. Consistency: • No testicular histopathology changes were observed in mice exposed by drinking water for subchronic duration. • No testicular histopathology changes were observed in rats, guinea pigs, or a single dog exposed by inhalation for durations between 30 and 246 days. • No testicular histopathology changes were observed in rats exposed by intraperitoneal injection for	*Key findings:* In high-quality studies, mice exposed to 1,2-dichloroethane by inhalation or intraperitoneal injection, but not by drinking water, exhibited effects on testicular pathology and sperm parameters. Most of the data in rats indicated no effect on the testes (or other reproductive organs); however, sperm parameters were not evaluated in rats. *Overall WOSE judgement for male reproductive tract effects based on animal evidence:* • Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• A one-generation reproduction study in rats exposed by inhalation evaluated histopathology of F0 testes after 176 days of exposure (Rao et al., 1980) Study quality: Medium • An inhalation cancer bioassay in rats evaluated gross pathology of the accessory sex organs, testes, and seminal vesicles and histopathology of the prostate and testes after 2 years exposure (Cheever et al., 1990) Study quality: High • Gavage studies in rats evaluated testes weights, gross pathology of the testes, and histopathology (testes, seminal vesicles, prostate, and preputial gland) after 10- or 90-day exposures (Daniel et al., 1994) Study quality: High • A gavage study in rats evaluated testes weights and histopathology of the testes, epididymis, seminal vesicles, and prostate after 13 weeks exposure (NTP, 1991) Study quality: High • A gavage cancer bioassay in mice evaluated comprehensive histopathology after 78 weeks exposure (NTP, 1978) Study quality: High • A drinking water study in mice evaluated testes weights and histopathology of the testes, epididymis, seminal vesicles, and prostate after 13 weeks exposure (NTP, 1991) Study quality: High • A dermal cancer bioassay in transgenic mice susceptible to cancer evaluated testes weights and histopathology of the prostate, seminal vesicle, and epididymis after 26 weeks exposure (Suguro et al., 2017) Study quality: High • An intraperitoneal injection study in mice evaluated histopathology of the testes 8 to 46 days after a 5-day exposure and histopathology and fertility for up to 9 months after a 5-day	• Mice exposed to ≥5 mg/kg/day by daily intraperitoneal injection for 5 days exhibited reduced spermatogenesis, loss of spermatogonia, histopathology changes in the testes, and sterility.	30 days or by gavage for subchronic durations.		

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
exposure plus 45 days recovery for spermatogenesis turnover (Daigle et al., 2009) Study quality: High • An intraperitoneal injection study in rats evaluated testis weight and gross and microscopic pathology of the testes after 30 days exposure (Igwe et al., 1986b) Study quality: Medium				
Effects on female reproductive organs				
• An inhalation study in female rats evaluated serum prolactin levels and morphometry and histopathology of mammary tissue after at least 28 days exposure (Dow Chemical, 2014) Study quality: High • A one-generation reproduction study in female rats exposed by inhalation evaluated histopathology of F0 ovaries and uterus after 176 days of exposure (Rao et al., 1980) Study quality: Medium • An inhalation cancer bioassay in female rats evaluated gross and microscopic pathology of the mammary tissue, ovaries, and uterus after 2 years exposure (Cheever et al., 1990) Study quality: High • Gavage studies in rats evaluated ovary weights, gross pathology of the ovaries, and histopathology (ovaries, uterus, clitoral gland, and mammary gland) after 10- or 90-day exposures (Daniel et al., 1994) Study quality: High • A gavage cancer bioassay in mice evaluated comprehensive histopathology after 78 weeks exposure (NTP, 1978) Study quality: High • A drinking water study in mice and a gavage study in rats evaluated histopathology of the uterus, mammary gland, clitoral gland, and ovaries after 13 weeks exposure (NTP, 1991) Study quality: High		Consistency: • Several high- and medium-quality studies of rats and mice exposed by inhalation, gavage, drinking water, and/or dermal contact reported no treatment-related changes in reproductive organ weights or histopathology.	*Key findings:* Inhalation studies in rats, oral studies in rats and mice, and a dermal study in mice observed no effects of 1,2-dichloroethane on female reproductive organ weights or histopathology. *Overall WOSE judgement for female reproductive tract effects based on animal evidence:* • Moderate evidence of no effect.	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• A dermal cancer bioassay in transgenic mice susceptible to cancer evaluated ovary weights and histopathology of the uterus, mammary gland, and vagina after 26 weeks exposure (Suguro et al., 2017) Study quality: High				
Effects on reproduction or offspring				
• An inhalation study in male and female rats evaluated numbers of live and dead pups; and pup weight, sex, gross pathology, liver and kidney weights, and liver and kidney histopathology after one generation exposure (Rao et al., 1980) Study quality: Medium • Inhalation studies in female rats and rabbits evaluated numbers of corpora lutea; numbers of live, dead, and resorbed fetuses; fetal weight, length, and sex; external and skeletal alterations; and cleft palate after gestational exposure (Rao et al., 1980) Study quality: Medium • Inhalation and gavage studies in female rats evaluated pregnancy outcomes and fetal external, skeletal, and visceral examinations after gestational exposure (Payan et al., 1995) Study quality: High • A drinking water study in male and female mice evaluated fertility and gestation indices, numbers of implantations and resorptions, viability and lactation indices, litter size, pup weight, and teratology after multigenerational exposure (Lane et al., 1982) Study quality: High • An intraperitoneal injection study in male mice evaluated male fertility for up to 9 months after a 5-day exposure plus 45 days recovery for spermatogenesis turnover (Daigle et al., 2009) Study quality: High	Biological gradient/dose-response: • An apparent decrease in necropsy body weight was observed at the high concentration of 150 ppm in a small subset of male F1B weanling rats exposed by inhalation in a one-generation study. • Male mice exposed by daily intraperitoneal injection at ≥ 10 mg/kg-d for 5 days exhibited permanent sterility (defined as sterility for 6 months or longer).	Magnitude and precision: • The apparent body weight decrease in selected male F1B weanlings at 150 ppm was based on only 5 male weanlings per group, was not statistically significantly different from controls, was not seen in female weanlings, and is not supported by the study authors' analysis of the full data set, which showed no effect on neonatal body weight or growth of pups to weaning in either F1A or F1B litters.	*Key findings:* In a high-quality study, sterility was observed in male mice exposed by intraperitoneal injection. Evidence for effects on weanling pup body weight after inhalation exposure is weak and inconsistent. Overall WOSE judgement for developmental effects based on animal evidence: • Slight	
Evidence from mechanistic studies				

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• An in vivo inhalation study in male rats evaluated elemental content in the testes after 30 days exposure (Que et al., 1988). • An in vivo inhalation study in male mice evaluated mRNA expression in the testis and genetic damage in spermatozoa after 1- or 4-week exposure (Zhang et al., 2017) • An in vivo study in mice exposed by intratesticular injection evaluated testicular DNA synthesis (Borzelleca and Carchman, 1982).	Biological gradient/dose-response: • Inhalation exposure to 1,2-dichloroethane did not alter zinc concentration in the testes. Statistically significant changes in other element concentrations included decreased Al, Hg, and S and increased Ca and P at the highest tested concentration (1,840 mg/m^3 or 455 ppm) • Expression consistent with inhibition of CREM/ CREB signaling and the induction of apoptosis was observed in the testis of mice. • Intratesticular injection of 1,2-dichloroethane resulted in a 53% decrease in testicular DNA synthesis in mice at the highest dose tested (250 mg/kg) but not at doses ≤100 mg/kg.	Biological plausibility and human relevance: • The biological relevance of the altered element content in the testes is uncertain. • The human relevance of intratesticular injection exposure is uncertain.	*Key findings:* Evidence for inhibition of CREM/ CREB signaling and apoptosis in testes of male mice exposed to 1,2-dichloroethane in vivo support observed effects on testes pathology, sperm morphology, and fertility in this species. *Overall WOSE judgement for reproductive/ developmental effects based on mechanistic evidence:* • Moderate	

Table_Apx N-36. 1,2-Dichloroethane Evidence Integration Table for Renal Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Renal Effects				
Evidence from human studies			Indeterminate	*Overall WOSE judgement for renal effects based on integration of information across evidence streams:* Evidence indicates that 1,2-dichloroethane likely causes renal effects under relevant exposure circumstances.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Studies evaluating histopathology in conjunction with other renal endpoints: • Acute inhalation studies in male and female rats and male mice evaluated kidney histopathology and weight after a single 4-hour exposure (Dow Chemical, 2006b); Study quality: High. (Francovitch et al., 1986); Study quality: Medium. • A short-term inhalation study in male rats evaluated kidney histopathology and weight and after 30 days of exposure (Igwe et al., 1986b); Study quality: High. • A chronic inhalation study in F0 male and female rats evaluated kidney histopathology and weight after exposure in a reproduction study from pre-breeding through the generation of 2 litters (Rao et al., 1980). Study quality: Medium. • Chronic inhalation studies in male and female rats evaluated kidney histopathology, kidney weight, and/or clinical chemistry after 212 days or 17-weeks of exposure (Hofmann et al., 1971; Spencer et al., 1951); Study quality: Medium. • Chronic inhalation studies in a single dog, guinea pigs, and rabbits evaluated kidney histopathology, kidney weight, and/or clinical chemistry after 6 months, 212 days, or 17 weeks of exposure (Hofmann et al., 1971; Spencer et al., 1951; Mellon Institute, 1947); Study quality: Medium. • Short-term and subchronic gavage studies in male and female rats evaluated kidney	Biological gradient/dose-response: • In acute inhalation studies: ○ Rats exhibited significantly increased incidences of basophilia of the renal tubular epithelium (males) or degeneration/ necrosis (females) in addition to significantly increased absolute and relative kidney weights (≥10%, both sexes) at 8,212 mg/m^3 (2,029 ppm). ○ Male mice exhibited significantly increased kidney weights (>10%) and BUN (86%) at ≥2,020 mg/m^3 (≥499 ppm). ○ In a chronic inhalation study in rats, a statistically significant increase in BUN (≈50%) was reported at 607 mg/m^3 (150 ppm). ○ In acute gavage studies, male mice exhibited significant increases in relative kidney weight (>10%) at ≥300 mg/kg and significantly increased percentage of damaged renal proximal tubules at 1,500 mg/kg.	Biological gradient/dose response: • High-quality short-term and chronic inhalation studies found no treatment-related effects on kidney weight or histopathology in rats exposed up to 647 mg/m^3 (159.7 ppm) or mice exposed up to 368 mg/m^3 (89.8 ppm) • High-quality short-term gavage studies found no treatment-related effects on kidney histopathology, kidney weight, or BUN in rats (both sexes) exposed up to 300 mg/kg-day or on kidney weight or gross pathology in mice (both sexes) exposed up to 49 mg/kg-day. • High-quality subchronic gavage studies in male and female rats found no treatment-related histopathology changes at doses up to 150 mg/kg-day. • A high-quality chronic gavage cancer bioassay in mice found no treatment-related effects on kidney histopathology at doses up to 299 mg/kg-day.	*Key findings*: Several high- and medium-quality studies found associations between 1,2-dichloroethane exposure and increased kidney weights, BUN, and/or renal tubular histopathology in rats (both sexes) and mice following inhalation, oral, dermal, and intraperitoneal injection exposures. *Overall WOSE judgement for renal effects based on animal evidence:* • Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
and bladder histopathology, kidney weight, and/or clinical chemistry, and/or urinary chemistry after 10 or 13 weeks of exposure (Daniel et al., 1994; NTP, 1991); Study quality: High. • A subchronic drinking water study in male and female mice evaluated kidney histopathology, weight of kidney and urinary bladder, and BUN after 13 weeks of exposure (NTP, 1991); Study quality: High. • A dermal cancer bioassay in male and female transgenic mice susceptible to cancer evaluated kidney histopathology and weight after 26 weeks exposure (Suguro et al., 2017); Study quality: High. • A short-term intraperitoneal injection study in male rats evaluated kidney histopathology, kidney weight, and/or clinical chemistry after 30 days of exposure (Igwe et al., 1986b); Study quality: Medium. Studies evaluating histopathology only: • An acute inhalation study in rats, mice, rabbits and guinea pigs evaluated microscopic kidney pathology after 1.5- to 7-hour exposures (Heppel et al., 1945); Study quality: Medium. • Subchronic and chronic inhalation studies in rats, rabbits, guinea pigs, and dogs evaluated kidney histopathology after 13 to 35 weeks of exposure (Heppel et al., 1946); Study quality: Low or Medium. • Inhalation cancer bioassays in male and female rats and mice evaluated histopathology of the kidney and urinary bladder after 2 years exposure (Nagano et	○ In subchronic gavage studies, rats exhibited significantly increased kidney weights (>10%, both sexes) at ≥30 mg/kg-day and increased BUN (20%, males) at 120 mg/kg-day. ○ In a subchronic drinking water study, mice exhibited significantly increased incidences of tubular regeneration (males) at ≥781 mg/kg-day and significantly increased kidney weights (>10%, both sexes) at 244–448 mg/kg-day. ○ In an acute intraperitoneal injection study in male mice, a statistically significant increase in relative kidney weight was observed at ≥400 mg/kg reaching >10% at 500 mg/kg. Consistency: • Renal histopathology changes were also reported in studies that were limited by lack of reporting on control findings. These included: ○ Degeneration of renal tubular epithelium in rats and rabbits after acute inhalation exposure.			

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
al., 2006; Cheever et al., 1990); Study quality: High. • An acute gavage study in male mice evaluated kidney immunohistochemistry after a single exposure (Morel et al., 1999). Study quality: High. • A gavage cancer bioassay in male and female mice evaluated kidney histopathology after 78 weeks of exposure (NTP, 1978); Study quality: High. Studies evaluating kidney weight, gross pathology, and/or clinical chemistry: • An acute inhalation study in mice evaluated kidney weight and BUN levels after a 4-hour exposure (Storer et al., 1984); Study quality: High. • Chronic inhalation studies in male and female rats evaluated serum chemistry and urinalysis parameters after 6, 12, or 18 months of exposure (IRFMN, 1987, 1978, 1976); Study quality: Medium. • An acute gavage study in male mice evaluated kidney weight and BUN after a single exposure (Storer et al., 1984); Study quality: High. • A short-term gavage study in male and female mice evaluated kidney weight and gross pathology after 14 days exposure (Munson et al., 1982); Study quality: High. • Acute intraperitoneal injection studies in male rats and mice evaluated kidney weight and serum chemistry parameters after a single exposure (Storer and Conolly, 1985; Storer et al., 1984; Livesey, 1982); Study quality: High; (Storer and Conolly, 1983); Study quality: Medium. • A short-term intraperitoneal injection study in male mice evaluated kidney gross	○ Increased severity of renal tubular damage in mice after acute inhalation exposure. ○ Moderate fatty degeneration of the kidney in guinea pigs after chronic inhalation exposure. ○ Mild karyomegaly of distal tubules and tubular degeneration in transgenic mice after chronic dermal exposure. Biological plausibility and human relevance: • Metabolism of 1,2-dichloroethane via glutathione-S-transferase is believed to yield a reactive episulfonium ion which can form the potent nephrotoxic conjugate S-(2-chloroethyl)-DL-cysteine.			

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
pathology after 5 days of exposure (NTP, 1978); Study quality: High.				
Evidence from mechanistic studies (none)			• Indeterminate	

Table_Apx N-37. 1,2-Dichloroethane Evidence Integration Table for Hepatic Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Hepatic Effects				
Evidence from human studies				*Overall WOSE judgement for hepatic effects based on integration of information across evidence streams:* Evidence suggests, but is not sufficient to conclude, that 1,2-dichloroethane may cause hepatic effects under relevant exposure conditions.
• A cohort study of 251 male workers from 4 vinyl chloride monomer (VCM) manufacturing plants evaluated associations between exposure to airborne 1,2-dichloroethane (in conjunction with low exposure to VCM) and serum AST, ALT, and GGT. Personal and area air sampling were used to determine VCM and 1,2-dichloroethane exposures and group participants by job category into low 1,2-dichloroethane (job medians of 0.26-0.44 ppm) or moderate 1,2-dichloroethane (job medians of 0.77-1.31 ppm) plus low VCM (job medians of 0.18-0.39 ppm). (Cheng et al., 1999). Study quality: Medium	Biological gradient/dose-response: • Increased odds of abnormal serum AST (>37 IU/L) and ALT (>41 IU/L) were observed when comparing the moderate-1,2-dichloroethane/low-VCM group with the low-1,2-dichloroethane/low-VCM group (OR = 2.2, 95% CI = 1.0–5.4 for abnormal AST; OR = 2.1, 95% CI = 1.1–4.2 for abnormal ALT).	Magnitude/precision: • Exposure concentrations in the low- and moderate-1,2-dichloroethane groups were overlapping. Biological plausibility/human relevance: • All subjects were also exposed to vinyl chloride monomer, a known liver toxicant.	*Key findings*: In a medium-quality study, increased odds of abnormal serum liver enzyme levels were observed among workers with higher exposure to 1,2-dichloroethane, in a cohort with co-exposure to vinyl chloride. *Overall WOSE judgement for hepatic effects based on human evidence:* Indeterminate	
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Studies evaluating histopathology in conjunction with other liver endpoint(s): • Acute inhalation studies in male and female rats and male mice evaluated liver weight and histopathology after	Biological gradient/dose-response: • In an acute inhalation study, rats exhibited minimal histological changes in the liver at 8212.3 mg/m^3 (2,029.0 ppm). Liver weight changes were small (<10%) and inconsistent.	Consistency: • In a high-quality short-term inhalation study in rats, no treatment-related effects on liver weight, serum chemistry or histopathology were	*Key findings*: Several high- and medium-quality studies in rats and mice found associations between 1,2-dichloroethane exposure and increased	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
single 4- and/or 8- hour exposures (Dow Chemical, 2006b); Study quality: High. (Francovitch et al., 1986); Study quality: Medium • A short-term inhalation study in male rats evaluated serum chemistry (ALP, SDH, and 5'NT), liver weight, and histopathology after 30 days exposure (Igwe et al., 1986b, c) Study quality: High • Subchronic and chronic inhalation studies in male and female rats, rabbits, cats, and guinea pigs evaluated serum chemistry (ALT and AST), bromsulphthalein retention, liver weight and/or histopathology after up to 17 weeks exposure (Hofmann et al., 1971) Study quality: Medium. • Chronic inhalation studies in male and female rats and guinea pigs, male monkeys, and a single dog evaluated hepatic lipids/cholesterol, liver function, liver weight, and/or histopathology after 170-248 days exposure (Spencer et al., 1951) Study quality: Medium. (Mellon Institute, 1947) Study quality: Medium. • Chronic inhalation cancer bioassays in male and female rats and mice evaluated liver weight and histopathology after 2 years exposure (Nagano et al., 2006; Cheever et al., 1990) Study quality: High. • A one-generation inhalation reproduction study in rats evaluated parental liver weight and histopathology after up to 176 days	• In an acute inhalation study, male mice exhibited a significant increase in relative liver weight (>10%) at 6071 mg/m^3 (1,500 ppm). Histological observations in the liver included hepatocyte swelling, swollen nuclei, fat accumulation, and occasional small areas of necrosis (incidence and severity were not reported) • In a chronic inhalation cancer bioassay, male (but not female) rats exhibited increased absolute (but not relative) liver weight (>10%) at 204 mg/m^3 (50 ppm) • In a short-term gavage study, male (but not female) rats had significantly increased relative liver weight (>10%) and serum cholesterol at 100 mg/kg-day in the absence of histopathology changes. • In subchronic gavage studies, male and female rats exhibited significantly increased relative liver weights (>10%) at ≥75 mg/kg-day in the absence of biologically significant serum chemistry changes or treatment-related histopathology changes. • In a subchronic drinking water study, male and female mice exhibited significantly increased (>10%) absolute and relative liver weights at ≥2,478 mg/kg-day in the absence of treatment-related histopathology changes. Consistency:	observed in rats at concentrations up to 1840 mg/m^3 (455 ppm). • In high-quality chronic inhalation cancer bioassays in rats and mice, no significant effects on liver weight or histology were observed at concentrations up to 646.4 mg/m^3 (159.7 ppm and 363 mg/m3 (89.8 ppm), respectively.	liver weights, serum enzymes, and/or histopathology changes following inhalation, oral, and intraperitoneal injection exposures. *Overall WOSE judgement for hepatic effects based on animal evidence:* • Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
exposure (Rao et al., 1980) Study quality: Medium. • An acute gavage study in female rats evaluated serum chemistry (ALT, AST, and LDH) and histopathology after a single dose (Cottalasso et al., 2002) Study quality: Medium. • Short-term and subchronic gavage studies in male and female rats evaluated serum chemistry, liver weight, and liver histopathology after 10-day and 13-week exposures (Daniel et al., 1994; NTP, 1991); Study quality: High. • A subchronic drinking water study in male and female mice evaluated liver weight and histopathology after 13 weeks exposure (NTP, 1991) Study quality: High. • A chronic dermal cancer bioassay in male and female transgenic mice evaluated liver weights and histopathology after 26 weeks exposure (Suguro et al., 2017) Study quality: High. Studies evaluating liver histopathology only: • Acute inhalation studies in rats, mice, rabbits, and guinea pigs evaluated gross and microscopic liver pathology after 1.5- to 7-hour exposures (Heppel et al., 1945). Study quality: Medium • Subchronic- and chronic inhalation studies in male and/or female rats, rabbits, guinea pigs, dogs, and cats evaluated liver histopathology after 5 to 35 weeks of exposure (Heppel et	• Hepatic histopathology changes and liver weight increases were also reported in low- and medium-quality studies that were limited by lack of quantitative data reporting and variable exposure regimens. The lesions included: ○ Congestion, fatty degeneration, and/or necrosis in rats, mice, rabbits, and guinea pigs after acute to short-term inhalation exposures that were sometimes lethal. ○ Cloudy swelling, fatty degeneration, necrosis, and/or occasional fat vacuoles in rats and guinea pigs after subchronic to chronic inhalation exposure. ○ Moderate steatosis in rats without biologically significant changes in AST or ALT after a single gavage dose. • In studies that did not evaluate histopathology, findings included: ○ Biologically and/or statistically significant increases in serum SDH and ALT in mice exposed for 4 hours by inhalation. ○ Increased serum ALT, SDH and/or glutamate dehydrogenase in rats after single or repeated inhalation exposures. ○ Increased liver weight in mice exposed by inhalation for 28 days. ○ Increased ALT and AST in rats after single gavage dose. ○ Increased relative liver weight and biologically significant increases in serum SDH and ALT in mice			

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
al., 1946); Study quality: Medium or Low. • A chronic gavage cancer bioassay in male and female mice evaluated liver histopathology after 78 weeks of exposure (NTP, 1978) Study quality: High. Studies evaluating only liver weight, gross pathology and/or clinical chemistry: • An acute inhalation study in male mice evaluated liver weight and serum chemistry (Storer et al., 1984) Study quality: High. • Acute- and short-term inhalation studies in male rats evaluated serum chemistry (Brondeau et al., 1983) Study quality: Medium. • A short-term inhalation study in male mice evaluated liver weight and serum chemistry (Zeng et al., 2018) Study quality: High. • Chronic inhalation studies in male and female rats evaluated serum chemistry (IRFMN, 1987, 1978, 1976) Study quality: Medium. • Acute gavage studies in male and female rats evaluated serum chemistry and/or liver weight (Kitchin et al., 1993); Study quality: High. (Cottalasso et al., 1995) Study quality: Medium. • An acute gavage study in male mice evaluated liver weight and serum chemistry (Storer et al., 1984) Study quality: High. • A short-term gavage study in male and female mice evaluated liver	after a single gavage or intraperitoneal dose.			

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
weight and gross pathology (Munson et al., 1982) Study quality: High. • A subchronic dietary study in rats evaluated serum chemistry (Alumot et al., 1976). Study quality: Medium • Acute, short-term, and subchronic intraperitoneal injection studies in male rats and male mice evaluated liver weight, serum chemistry, and/or gross pathology (Storer and Conolly, 1985; Storer et al., 1984; Livesey, 1982); Study quality: High. (Daigle et al., 2009; Igwe et al., 1986b; Storer and Conolly, 1983) Study quality: Medium.				
Evidence from mechanistic studies				
• An *in vivo* inhalation study in male rats evaluated elemental content in the liver after 30 days exposure (Que et al., 1988). • An *in vivo* inhalation study in male mice evaluated hepatic micro-RNA (miR) expression and gluconeogenesis (Zeng et al., 2018). • *In vivo* genotoxicity tests were conducted in the liver of male mice after single inhalation, oral, and intraperitoneal exposures (Storer et al., 1984). ○ An *in vivo* intraperitoneal injection study in male mice evaluated hepatic enzyme induction (Paolini et al., 1994). ○ A series of studies *in vivo* in rats and *in vitro* in rat hepatocytes evaluated effects on glycolipoprotein metabolism (Cottalasso et al., 2002;	Biological gradient/dose-response: • 1,2-Dichloroethane induced DNA damage after oral and intraperitoneal (but not inhalation) exposure. • 1,2-Dichloroethane induced a dose-related increase in PROD activity (a probe for CYP450 2B1) in mice. Oxidative stress: • Incubation of rat liver slices with 1,2-dichloroethane (up to 10 mM for up to 30 minutes) resulted in dose-and time-dependent increases in MDA production. • Levels of GSH were significantly decreased in rat hepatocytes cultured with 4.4 to 6.5 mM 1,2-dichloroethane for up to 1 hour. • Free radicals were detected in rat hepatocytes cultured with 1,2-dichloroethane under anaerobic (but not aerobic) conditions.	Biological gradient/dose-response: • Rat hepatocytes exposed to 1,2-dichloroethane for 1 hour at 1.2 mM did not show significantly decreased GSH. Consistency: • Rat hepatocytes cultured with 10 mM 1,2-dichloroethane for 2 hours did not show evidence of lipid peroxidation (*i.e.*, increased PCOOH or PEOOH levels).	*Key findings*: Available data on liver toxicity mechanisms are limited and nonspecific. Hepatic enzyme induction was demonstrated in mice exposed by intraperitoneal injection. Limited *in vitro* data indicate that 1,2-dichloroethane may increase oxidative stress or impair glucose and/or lipid metabolism in mice and in rat hepatocytes and liver slices. *Overall WOSE judgement for hepatic effects based on mechanistic evidence:* • Indeterminate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Cottalasso et al., 1995; Cottalasso et al., 1994). ○ *In vitro* studies in rat hepatocytes or rat liver slices evaluated oxidative stress parameters (Cottalasso et al., 1994; Suzuki et al., 1994; Jean and Reed, 1992; Thomas et al., 1989; Tomasi et al., 1984). ○ An *in vitro* study in rat hepatocytes incubated with the cysteine S conjugate of 1,2-dichloroethane, S-(2-chloroethyl)-DL-cysteine (CEC), evaluated cytotoxicity related to oxidative stress (Webb et al., 1987).	• The cysteine S conjugate of 1,2-dichloroethane was cytotoxic and depleted GSH in hepatocytes; co-treatment with antioxidants and GSH precursors mitigated these effects. Effects on gluconeogenesis and glycolipoprotein metabolism*:* • Inhalation exposure increased miR-451a expression and decreased glycerol gluconeogenesis in the liver of exposed mice. • Rats treated with 1,2-dichloroethane via gavage showed impairment of glycoprotein biosynthesis. • 1,2-dichloroethane treatment increased retention and decreased secretion of glycolipoproteins in rat hepatocytes.			

5'-NT = 5'-nucleotidase; ALP = alkaline phosphatase; ALT – alanine aminotransferase; AST = aspartate aminotransferase; F = female; GGT = gamma-glutamyl transferase; GLDH = glutamate dehydrogenase; GSH = glutathione; LDH = lactate dehydrogenase; M = male; MDA = malondialdehyde; ODC = orinithine decarboxylase activity; PCOOH = phosphatidylcholine hydroperoxide; PEOOH = phosphatidylethanolamine hydroperoxide; PROD = pentoxyresorufin dealkylation; SDH = sorbitol dehydrogenase.

[a] Based on a density for 1,2-dichloroethane of 1.25 g/cm^3.

Table_Apx N-38. 1,2-Dichloroethane Evidence Integration Table for Immune/Hematological Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Immune/Hematological Effects				
Evidence from human studies (none)			Indeterminate	*Overall WOSE judgement for immune/hematological effects based on integration of information across evidence streams:* Evidence indicates that 1,2-dichloroethane likely causes immune system suppression under relevant exposure conditions.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Studies of immune function: • An inhalation study evaluated mortality from *Streptococcus zooepidemicus* aerosol challenge in female mice and lymphocyte stimulation, alveolar macrophage inhibition, and pulmonary bactericidal activity against *Klebsiella pneumoniae* in female mice and male rats after exposure once or for 5 (mice) or 12 (rats) days (Sherwood et al., 1987) Study quality: High • An oral gavage study in male mice evaluated hematology (including coagulation), humoral immunity (spleen cell antibody response), cell-mediated immunity (delayed hypersensitivity response), spleen and thymus weight, and gross necropsy after 14 days (Munson et al., 1982) Study quality: High Studies of hematology, organ weights, and histopathology: • Inhalation studies in rats, mice, rabbits, and guinea pigs (sex not specified) evaluated gross pathology and histopathology of the spleen after acute exposures (Heppel et al., 1945). Study quality: Medium • An inhalation study in male rats evaluated spleen weight, gross pathology, and histopathology after 30 days exposure (Igwe et al., 1986b) Study quality: High • Inhalation studies in rats, rabbits, guinea pigs, monkeys, cats and a single dog evaluated hematology (and/or clotting parameters or IgM) and/or spleen histopathology after 5 to 35 weeks of exposure (Heppel et al., 1946)	Biological gradient/dose-response: • Female mice exposed by inhalation for 3 hours exhibited a concentration-related increase in mortality due to *S. zooepidemicus* infection at concentrations ≥22 mg/m^3 (5.4 ppm). Mortality incidences were 1.5 and 2.1-fold higher than controls at 22 and 43.7 mg/m^3, respectively. Female mice also exhibited a small decrease in bactericidal activity against *K. pneumoniae* at 43.7 mg/m^3 (10.8 ppm). • In a gavage study, decreased humoral and cell-mediated immune responses were observed in male mice after 14 days exposure to ≥4.89 mg/kg-day; decreased leukocyte counts were observed at 48.9 mg/kg-day. • In a gavage study in rats, small decreases in erythrocyte count, hemoglobin, and hematocrit were observed in both sexes along with increased platelets (both sexes) and	Consistency: • Male rats exhibited no effects in the *K. pneumoniae* challenge assays after exposures up to 810 mg/m^3 for 5 hours or up to 405 mg/m^3 for 12 days. • In a study rated uninformative due to decreased drinking water intake at the high dose of 189 mg/kg-day, no effect on humoral or cell-mediated immune responses or leukocyte counts were observed in mice exposed to doses of 3, 24, or 189 mg/kg-day via drinking water for 90 days. • No treatment-related changes in hematology were observed in a gavage study of male rats exposed to doses up to 120 mg/kg-day for 13 weeks, or in studies of several species exposed by inhalation for durations from 5 weeks to 2 years. • Multiple studies of several species exposed by inhalation or oral administration for acute, subchronic, or chronic durations showed no effects	*Key findings*: In high-quality inhalation and gavage studies of immune function in mice, an association between 1,2-dichloroethane exposure and immunosuppression was observed; a more limited inhalation study in rats and a longer-term drinking water study in mice rated Uninformative did not show any effects. Evidence from other studies showed only small effects on hematology and no effects on relevant organ weights or histopathology. *Overall WOSE judgement for immune/hematological effects based on animal evidence:* • Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
(IRFMN, 1987, 1978, 1976; Hofmann et al., 1971; Spencer et al., 1951; Mellon Institute, 1947) Study quality: Low to Medium • Inhalation cancer bioassays in male and female rats and mice evaluated hematology and/or comprehensive histopathology after 2 years exposure (Nagano et al., 2006; Cheever et al., 1990) Study quality: High • A drinking water study in male and female mice evaluated comprehensive histopathology after 13 weeks exposure (NTP, 1991) Study quality: High • Gavage studies in male and female rats evaluated hematology, spleen and/or thymus weights, and comprehensive histopathology after 10- and/or 90-day exposures (Daniel et al., 1994; NTP, 1991) Study quality: High • A gavage cancer bioassay in male and female mice evaluated comprehensive histopathology after 78 weeks exposure (NTP, 1978) Study quality: High • A gavage cancer bioassay in male and female transgenic mice susceptible to cancer evaluated hematology and histopathology of the thymus, spleen, lymph nodes, and bone marrow after 40 weeks exposure (Storer et al., 1995) Study quality: Medium • A dermal cancer bioassay in male and female transgenic mice susceptible to cancer evaluated thymus and spleen weights and histopathology of the lymph nodes, thymus, and bone marrow after 26 weeks exposure (Suguro et al., 2017) Study quality: High Studies Rated Uninformative: • An oral study in male mice evaluated hematology, humoral immunity (spleen cell antibody response), cell-mediated immunity (delayed hypersensitivity response), spleen	leukocytes (females only) after 90 days at 150 mg/kg-day. • In a subchronic gavage study, increased incidences of thymus necrosis were observed in male and female rats that died prematurely (≥240 mg/kg-day in males and at 300 mg/kg-day in females).	on relevant organ weights or histopathology. Biological plausibility and human relevance: • In the mouse inhalation study, mice were exposed for 30 minutes to aerosols of streptococcal bacteria (≈2E04 inhaled viable streptococci). The relevance of this immune challenge to typical human bacterial exposures is uncertain.		

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
cell response to mitogens, function of the reticuloendothelial system, spleen and thymus weight, and gross necropsy after 90 days drinking water exposure. (Munson et al., 1982)				
Evidence from mechanistic studies				
• An *in vitro* study investigated phagocytic activity of mouse peritoneal macrophages incubated with 1,2-dichloroethane (Utsumi et al., 1992). • Cell-free and *in vitro* studies investigated 1,2-dichloroethane effects on erythrocyte glutathione-S-transferase (GST) (Ansari et al., 1987) • An inhalation study in rats evaluated elemental content in the spleen after 30 days exposure to 1,2-dichloroethane (Que et al., 1988).	Biological gradient/dose-response: • 1,2-dichloroethane induced dose-related reductions in erythrocyte GST activity in both the cell-free experiment and in human erythrocytes *in vitro*. • 1,2-dichloroethane reduced macrophage phagocytic activity to 76% of control levels at a concentration of 200 mM.		*Key findings*: Limited *in vitro* data showed reductions in macrophage phagocytic activity and erythrocyte GST activity after exposure to 1,2-dichloroethane. *Overall WOSE judgement for immune/hematological effects based on mechanistic evidence:* • Indeterminate	

Table_Apx N-39. 1,2-Dichloroethane Evidence Integration Table for Neurological/Behavioral Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Neurological/Behavioral Effects				
Evidence from human studies				*Overall WOSE judgement for neurological/behavioral effects based on integration of information across evidence streams:* Evidence indicates that 1,2-dichloroethane likely causes neurological/ behavioral effects under relevant exposure circumstances.
• Case reports of human exposure to 1,2-dichloroethane by inhalation or ingestion indicated clinical signs of neurotoxicity (dizziness, tremors, paralysis, coma) as well as histopathology changes in the brain at autopsy (ATSDR, 2024). • Workers exposed to 1,2-dichloroethane for extended periods have developed cerebral edema and toxic encephalopathy (ATSDR, 2024).			*Key findings*: Case reports document clinical signs of neurotoxicity and brain histopathology changes in humans exposed to 1,2-dichloroethane by inhalation or ingestion. *Overall WOSE judgement for neurological/behavioral effects based on human evidence:* • Slight	
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Studies evaluating neurobehavioral endpoints: • An inhalation study in male and female rats evaluated clinical signs, functional observational battery (FOB), grip performance, landing foot splay, rectal temperature, motor activity, brain weight, and gross and microscopic pathology of nervous system tissues after 4 hours exposure (Hotchkiss et al., 2010; Dow Chemical, 2006b) Study quality: High • A range-finding inhalation study in male and female rats evaluated detailed clinical observations (cage-side, hand-held, and open-field; recorded systematically) and gross pathology (tissues not specified) after 4 hours exposure (Dow Chemical, 2005) Study quality: High	Biological gradient/dose-response: • In rats exposed by inhalation once for four hours, neurobehavioral changes including incoordination, palpebral closure, decreased sensory responses, and decreased motor activity were seen at ≥ 7,706 mg/m^3 (1904 ppm) one hour after exposure but not at subsequent times up to 15 days later. • In rats exposed by inhalation for ≥ 1.5 hr to ≥ 4000 mg/m^3 brain edema was seen, and microstructural alterations were detected by diffusion MRI 3 days after exposure. • In rats exposed by inhalation to ≥ 5,000 mg/m^3, increased water content in the cortex was observed	Consistency: • No treatment-related brain weight or histopathology changes were seen in nervous system tissues 15 days after single 4-hour exposure up to 8,212.3 mg/m^3 (2,029.0 ppm). • No histopathology changes were observed in the brain, sciatic nerve, or spinal cord of rats exposed by inhalation for 204 mg/m^3 (50.4 ppm) for 2 years in a cancer bioassay. • No clinical signs of toxicity or histopathology changes in the brain or sciatic nerve were observed in rats exposed by gavage to up to	*Key findings*: Several high- and medium-quality studies using rats exposed to 1,2-dichloroethane by inhalation or gavage or mice exposed by intraperitoneal injection showed the occurrence of neurobehavioral changes, clinical signs of neurotoxicity, and/or changes in brain histopathology. *Overall WOSE judgement for neurological/behavioral effects based on animal evidence:*	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• An intraperitoneal injection study in male mice evaluated righting reflex, bridge test, and operant tests after single exposure (Umezu and Shibata, 2014) Study quality: High Studies evaluating neuropathology: • An inhalation study in male rats evaluated clinical signs and brain MRI and histopathology after 1.5- or 4-hour exposures (Zhou et al., 2016) Study quality: Medium • An inhalation study in male and female rats evaluated clinical signs, histology and electron microscopy, and water content of the brain after 2-, 4-, 6-, or 12-hour exposures (Zhang et al., 2010) Study quality: Medium • An inhalation cancer bioassay in male and female rats evaluated brain, sciatic nerve, and spinal cord gross and/or microscopic pathology after 2 years exposure (Cheever et al., 1990) Study quality: High • A gavage study in male and female rats evaluated clinical signs, brain weight, and gross and/or microscopic pathology of the brain and sciatic nerve after 10- or 90-day exposure (Daniel et al., 1994) Study quality: High • A gavage study in male and female rats evaluated clinical signs, brain weight, and histopathology of the brain, sciatic nerve, and spinal cord after 13 weeks exposure (NTP, 1991) Study quality: High • A drinking water study in male and female mice evaluated clinical signs, brain weight, and histopathology of the	after ≥2-hour exposure and edema and histopathological changes in the brain were observed by light and transmission electron microscopy at the end of ≥ 6-hour exposure. • In animals of several species exposed by inhalation for up to 12 hours, clinical signs including hyperactivity, weakness, sedation, dysphoria, and/or trembling were reported. • In rats exposed by gavage for 13 weeks, clinical signs of neurotoxicity (including tremors and abnormal posture) and necrosis in the cerebellum were observed at ≥240 mg/kg-day. Consistency: • Mice exposed by intraperitoneal injection showed a dose-related decrease in response rate in lever-pressing operant behavior test at ≥ 62.5 mg/kg but no effects on other tests.	300 mg/kg-d for 10 days or 150 mg/kg-d for 90 days. • No histopathology changes were observed in the brain, sciatic nerve, or spinal cord of mice exposed via drinking water for 13 weeks, by gavage for 78 weeks in a cancer bioassay, or in transgenic mice exposed by dermal application for 40 weeks in a cancer bioassay. • Exposure to 1,2-dichloroethane did not alter brain weights of rats exposed by gavage for up to 90 days or in mice exposed by gavage for 14 days or drinking water for 90 days.	• Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
brain, sciatic nerve, and spinal cord after 13 weeks exposure (NTP, 1991) Study quality: High • A gavage cancer bioassay in male and female mice evaluated clinical signs and histopathology of the brain/meninges after 78 weeks exposure (NTP, 1978) Study quality: Medium • A dermal cancer bioassay in male and female transgenic mice evaluated clinical signs, brain weights, and brain, spinal cord, and sciatic nerve histopathology after 26 weeks exposure (Suguro et al., 2017) Study quality: High Studies evaluating clinical signs, brain weight, and/or gross pathology: • Inhalation studies in rats, mice, rabbits, and guinea pigs evaluated clinical signs of neurotoxicity after 1.5- to 7-hour exposures (Heppel et al., 1945) Study quality: Medium • An inhalation study in male and female rats and guinea pigs and male monkeys evaluated clinical signs and/or brain histology after up to 35 weeks exposure (Spencer et al., 1951) Study quality: High • A gavage study in male rats evaluated clinical signs and gross pathology after a single exposure (Stauffer Chem Co, 1973) Study quality: Medium • A gavage study in male and female mice evaluated brain weight and gross pathology after 14-day exposure (Munson et al., 1982) Study quality: High • An intraperitoneal (intraperitoneal) injection study of fertility in male mice evaluated gross pathology of the brain				

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
after 5-day exposure (Daigle et al., 2009) Study quality: Medium				
Evidence from mechanistic studies				
• *In vivo* inhalation studies in mice aimed at identifying mechanisms of brain edema induced by 1,2-dichloroethane evaluated aquaporin and matrix metalloproteinases protein expression or ATP generation and tight junction protein expression after 1-, 2-, or 3-day exposure (Wang et al., 2018a; Wang et al., 2014). • An *in vivo* oral study in rats evaluated neurotransmitter levels in the brain after a single exposure (Kanada et al., 1994). • *In vitro* studies in rat astrocytes exposed to 2-chloroethanol (metabolite of 1,2-dichloroethane) evaluated the roles of mitochondrial function, glutamate metabolism, matrix metalloproteinases, and MAPK cell signaling in cerebral edema induced by 1,2-dichloroethane (Wang et al., 2018b; Wang et al., 2017; Sun et al., 2016a; Sun et al., 2016b).	Biological gradient/dose-response: • Exposure to 1,2-dichloroethane upregulated the mRNA and/or protein expression of aquaporin and a matrix metalloproteinase (MMP9). • Exposure to 1,2-dichloroethane resulted in decreased expression of tight junction proteins (occludin and ZO-1) and mRNA, increased free calcium, decreased ATP content, and decreased ATPase activity in the brains of mice. Consistency: • Exposure to 2-chloroethanol *in vitro* resulted in decreased ATPase activity, mitochondrial function (membrane potential), and glutamate metabolism (expression of enzymes involved in glutamate metabolism) in rat astrocytes. Exposure also upregulated matrix metalloproteinases (MMP2 and MMP9) via increased p38 MAPK signaling. Pretreatment with the antioxidant N-acetyl-l-cysteine mitigated effects on p38 and MMP levels, suggesting a role for oxidative stress.		*Key findings*: 1,2-dichloroethane may downregulate tight junction proteins and energy production and upregulate aquaporin and a matrix metalloproteinase in the brains of exposed mice. *Overall WOSE judgement for neurological/behavioral effects based on mechanistic evidence:* • Slight	

Table_Apx N-40. 1,2-Dichloroethane Evidence Integration Table for Respiratory Tract Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Respiratory Tract Effects				
Evidence from human studies (none)			• Indeterminate	*Overall WOSE judgement for respiratory tract effects based on integration of information across evidence streams:* Evidence suggests, but is not sufficient to conclude, that 1,2-dichloroethane may cause nasal effects under relevant exposure conditions.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Studies examining upper and lower respiratory tract: • An acute inhalation study in male and female rats evaluated BAL, lung weight, and histopathology of the respiratory tract including nasal cavity 24 hours after 4- or 8-hour exposures (Hotchkiss et al., 2010; Dow Chemical, 2006b). Study quality: High • An inhalation cancer bioassay in male and female rats evaluated histopathology of the respiratory tract including nasal cavity after 104 weeks of exposure (Cheever et al., 1990). Study quality: High • Two gavage studies in rats evaluated lung weight and histopathology of the lungs and nasal cavity and turbinates after 10 and 90 days of exposure (Daniel et al., 1994). Study quality: High • A gavage study in male and female rats evaluated histopathology of the respiratory tract including nasal cavity and turbinates, after 13 weeks of exposure (NTP, 1991). Study quality: High • A drinking water study in male and female mice evaluated histopathology of the respiratory tract including nasal cavity and turbinates, after 13 weeks of exposure (NTP, 1991). Study quality: High • A dermal cancer bioassay in male and female transgenic mice susceptible to cancer evaluated lung weight and histopathology of the nasal cavity, trachea, and lungs after 26 weeks of	Biological gradient/dose-response: • In a high-quality study, dose-related increased incidences and/or severity of degeneration/ necrosis of the nasal olfactory mucosa occurred in male and female rats after inhalation exposures ≥795 mg/m^3 (≥196.4 ppm) for 4 hours or ≥ 435 mg/m^3 (≥107.5 ppm) for 8 hours. Regeneration of the olfactory epithelium was seen in groups sacrificed 15 days after a 4-hour exposure to 795 mg/m^3 (196.4 ppm). • Lung effects including a transient decrease in ALP in BALF and histopathology changes (edema, vacuolar changes, desquamation, atelectasis, macrophage proliferation, and inflammation) were reported in rats after a single gavage dose of 136 mg/kg.	Biological gradient/dose-response: • No treatment-related nasal lesions were observed in cancer bioassays of rats exposed by inhalation up to 654 mg/m^3 (160 ppm) for 2 years. • High-quality studies in rats did not show effects of 1,2-dichloroethane on the lung after gavage exposure up to 150 mg/kg/day for 90 days. Magnitude and precision: • Group sizes were small (5/sex) in the acute inhalation study that observed nasal lesions. Consistency: • High- and medium-quality studies in rats did not show effects of 1,2-dichloroethane on the lung after chronic inhalation exposure up to 810 mg/m3 (200 ppm) for 212 days or up to 654 mg/m^3 (160 ppm) for 2 years. • High-quality studies in mice did not show effects of 1,2-dichloroethane on the lungs after 14 days of gavage exposure up to 49 mg/kg/day or 13 weeks of drinking water	*Key findings*: In a high-quality study, an association between 1,2-dichloroethane inhalation exposure and nasal lesions was observed in rats exposed to concentrations ≥ 435 mg/m^3 (≥107.5 ppm). Although one medium-quality study reported lung lesions in rats after a single gavage dose, high- and medium-quality studies of longer duration and higher doses, as well as a high-quality study of acute inhalation exposure, did not show effects of 1,2-dichloroethane on lower respiratory tract tissues of rats. *Overall WOSE judgement for respiratory effects based on animal evidence:* • Slight to moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
exposure (Suguro et al., 2017). Study quality: High Studies examining only lower respiratory tract: • An inhalation cancer bioassay in male and female rats and mice evaluated lung weight and histopathology after 104 weeks of exposure (Nagano et al., 2006). Study quality: High • An inhalation study in male and female rats and guinea pigs evaluated lung weight and histopathology after ≈170–246 days (Spencer et al., 1951). Study quality: Medium • A gavage study in male rats evaluated BALF, lung weight, and lung histopathology 1 to 30 days after a single dose (Salovsky et al., 2002). Study quality: Medium • A gavage study in mice evaluated lung weight and gross pathology after 14 days of exposure (Munson et al., 1982). Study quality: High • A gavage study in male and female mice evaluated the lungs, bronchi, and trachea for histopathology after 78 weeks of exposure (NTP, 1978). Study quality: High • An intraperitoneal injection study in male rats evaluated lung weight and histopathology (Igwe et al., 1986b). Study quality: Medium • An intratracheal injection lethality study in rats (sex NS) evaluated gross pathology of the lungs at death or 3 days after a single dose (Dow Chemical, 1989). Study quality: Medium		exposure up to 4,926 mg/kg/day. • A medium-quality study in guinea pigs did not show effects of 1,2-dichloroethane on the lungs after exposure up to 1,620 mg/m^3 (400 ppm) for 246 days. • BAL parameters, lung weight, and lung histopathology were not affected in rats exposed by inhalation up to 8,212.26 mg/m^3 (2,029.0 ppm) for 4 hours. Quality of the database: • Lung histopathology data in the acute gavage study that reported lung effects were presented qualitatively. Biological plausibility and human relevance: • Lung tumors are associated with chronic inhalation or gavage exposure in mice and with subchronic dermal exposure in susceptible transgenic mice. Increases in lung weight and preneoplastic lesions, such as hyperplasia, in some of these studies are related to tumor development and not indicative of a separate nonneoplastic effect on the lung.		
Evidence from mechanistic studies (none)			• Indeterminate	

Table_Apx N-41. 1,2-Dichloroethane Evidence Integration Table for Nutritional/Metabolic Effects

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Nutritional/Metabolic Effects				
Evidence from human studies (none)			• Indeterminate	*Overall WOSE judgement for nutritional/ metabolic effects based on integration of information across evidence streams:* Evidence suggests that 1,2-dichloroethane may cause nutritional/ metabolic effects under relevant exposure conditions.
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Body weight was evaluated in the following studies: • Acute inhalation studies in male and female rats (Dow Chemical, 2006b); Study quality: High. • Short-term inhalation studies in male mice (Zeng et al., 2018; Zhang et al., 2017); Study quality: High. • A short-term inhalation study in female rats (Dow Chemical, 2014); Study quality: High. • Short-term, subchronic, and chronic inhalation studies in male and/or female rats, mice, rabbits, dogs, guinea pigs, monkeys, and cats (Spencer et al., 1951; Heppel et al., 1946); Study quality: Medium or Low. • A one-generation inhalation reproduction study in rats (Rao et al., 1980); Study quality: Medium. • Chronic inhalation cancer bioassays in male and female rats (Nagano et al., 2006; Cheever et al., 1990); Study quality: High. • An acute oral gavage study in male rats (Moody et al., 1981); Study quality: Medium. • A gavage study in female rats exposed during gestation (Payan et al., 1995); Study quality: High.	Biological gradient/dose-response: Treatment-related adverse [a] effects on body weight occurred in high or medium quality studies of (species, route, exposure level and duration): • Mouse inhalation: ○ ≥707 mg/m^3 (175 ppm), males, 4 weeks • Guinea pig inhalation: ○ 405 mg/m^3 (100 ppm) in females and 809 mg/m^3 (200 ppm) in males, up to 246 d • Rat gavage: ○ ≥40 mg/kg-day, females, 6 weeks ○ 150 mg/kg-day, males, 13 weeks ○ 198 mg/kg-day, maternal weight gain, GD 6–20 • Mouse drinking water: ○ 4,207 mg/kg-day in males and ≥647 mg/kg-day in females, 13 weeks Consistency: • Decreased body weight was observed in male transgenic mice exposed to 200 mg/kg-day by gavage for 40 weeks.	Biological gradient/dose-response: No treatment-related adverse effects on body weight occurred in high or medium quality studies of (species, route, exposure level, and duration): • Rat inhalation: ○ ≤8,212 mg/m^3 (2,029 ppm), males and females, 4 hours ○ 832 mg/m^3 (205 ppm), females, 4 weeks ○ ≤809 mg/m^3 (200 ppm), males and females, up to 212 days ○ ≤648 mg/m^3 (160 ppm), males and females, 2 yrs • Monkey inhalation: ○ 405 mg/m^3 (100 ppm), males, up to 212 days • Rat gavage: ○ 625 mg/kg-day, males, single dose ○ ≤300 mg/kg-day, males, and females, 10 d ○ ≤100 mg/kg-day, males, 2 weeks ○ ≤90 mg/kg-day, males, and females, 13 weeks ○ ≤120 mg/kg-day in males and ≤150 mg/kg-day in females, 13 weeks Consistency: • Body weight was not affected in low quality inhalation studies of female dogs exposed to 1,540 mg/m^3 (380.5	*Key findings*: Decreased body weight was reported in mice and guinea pigs exposed by inhalation and rats and mice exposed orally to 1,2-dichloroethane in high- and medium-quality studies. Several high- and medium-quality studies in a few species via various routes of exposure reported no effect on body weight, sometimes at lower exposure levels and/or shorter exposure durations. *Overall WOSE judgement for nutritional/metabolic effects based on animal evidence:* • Slight	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• A short-term gavage study in male and female mice (Munson et al., 1982); Study quality: High. • Short-term and subchronic gavage studies in male and female rats (Daniel et al., 1994; NTP, 1991; van Esch et al., 1977); Study quality: High. (NTP, 1978); Study quality Medium. • A subchronic drinking water study in male and female mice (NTP, 1991); Study quality: High. • A subchronic dietary study in rats (Alumot et al., 1976); Study quality: Medium. • A multigenerational drinking water study in mice (Lane et al., 1982); Study quality: High. • Chronic gavage and dermal studies in transgenic mice susceptible to cancer (Suguro et al., 2017; Storer et al., 1995); Study quality: High. • Short-term intraperitoneal injection studies in male rats and male mice (Daigle et al., 2009); Study quality: High; (Igwe et al., 1986b); Study quality: Medium.		ppm) for 34–35 weeks or male rabbits exposed to 730 mg/m^3 (180 ppm) for 13–25 weeks. • Body weight was not affected in rats given feed fumigated with 1,2-dichloroethane in a 13-week study with dose uncertainties. • Body weight was not affected in male transgenic mice exposed to dermal doses up to 6,300 mg/kg-day for 26 weeks. • Body weight was not affected after intraperitoneal administration in male rats given 150 mg/kg-day for 30 days or in male mice given 40 mg/kg-day for 5 days.		
Evidence from mechanistic studies (none)			• Indeterminate	

[a] In adult animals, decreases in body weight of at least 10% change from control are considered adverse unless the changes are attributable to food or drinking water intake decreases due to palatability. Statistically significant decreases (relative to controls) in maternal body weight gain during gestation are considered adverse. Effects on body weight of offspring at ages up to sexual maturity are considered developmental effects.

Table_Apx N-42. 1,2-Dichloroethane Evidence Integration Table for Mortality

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Mortality				
Evidence from human studies				*Overall WOSE judgement for mortality effects based on integration of information across evidence streams:* Evidence indicates that 1,2-dichloroethane may cause death under relevant exposure circumstances and lethal levels have been identified in animal studies.
• A retrospective cohort mortality study evaluated all-cause mortality in 7849 white male petrochemical plant workers followed from 1950 to 1983. SMRs were calculated using age-, race-, and calendar year-specific mortality rates of males in the United States (Teta et al., 1991). Study quality: Medium • A retrospective cohort mortality study evaluated all-cause mortality in 251 employees of an herbicide manufacturing facility between 1979 and 1987, followed until 2003. SMRs were calculated using age- and gender-specific mortality rates in the United States. (BASF, 2005). Study quality: Medium		Biological plausibility and human relevance: • Two limited retrospective cohort studies found no increase in mortality of workers with presumed exposure to 1,2-dichloroethane (and other chemicals) relative to the general U.S. population.	*Key findings*: Limited epidemiological data show no increase in mortality among workers with presumed exposure to 1,2-dichloroethane but are insufficient to draw any broader conclusions. *Overall WOSE judgement for mortality effects based on human evidence:* • Indeterminate	
Evidence from apical endpoints in *in vivo* mammalian animal studies				
• Acute-duration inhalation studies evaluated mortality in rats, mice, and guinea pigs (Dow Chemical, 2017, 2006b; Storer et al., 1984; Spencer et al., 1951), Study quality: High.(Zhang et al., 2010; Francovitch et al., 1986; Heppel et al., 1945), Study quality: Medium • Short-term- and subchronic-duration inhalation studies evaluated mortality in rats, guinea pigs, mice, rabbits, dogs, and cats (Dow Chemical, 2014; Payan et al., 1995; Igwe et al., 1986b), Study quality: High. (Rao et al., 1980; Heppel et al., 1946), Study quality: Medium • Chronic-duration inhalation studies evaluated mortality in rats, mice, rabbits, guinea pigs, dogs, monkeys, and cats	Biological gradient/dose-response: Treatment-related deaths [a] or effects on survival occurred in studies of (species, route, exposure, and intended duration): • Rat inhalation: ○ 10,200 mg/m^3 (2,520 ppm), 4 hrs ○ 4,050 mg/m^3 (1,000 ppm), 7 hrs ○ 1,230 mg/m^3 (455 ppm), 30 d ○ ≥730 mg/m^3 (0.73 mg/L), 6 weeks ○ 1,214 mg/m^3 (300 ppm), gestational exposure	Biological gradient/dose-response: No treatment-related[1] deaths/effects on survival were seen in studies of (species, route, exposure, duration): • Rat inhalation: ○ ≤8,212 mg/m^3 (2,029 ppm), 4 hours ○ 5,000 mg/m^3, 2–6 hours ○ 630.6 mg/m^3 (155.8 ppm), 8 hours ○ 10,000 mg/m^3, 12 hours ○ 404 mg/m^3, 17 weeks ○ ≤646.4 mg/m^3 (158 ppm), 2 yrs • Mouse inhalation:	*Key findings*: Treatment-related increases in the incidence of mortality were observed in several animal species exposed to 1,2-dichloroethane via inhalation, oral, or dermal exposure for acute, short-term/intermediate, or chronic durations in multiple studies. *Overall WOSE judgement for mortality effects based on animal evidence:* • Robust	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
(Nagano et al., 2006; Cheever et al., 1990), Study quality: High. (Hofmann et al., 1971; Spencer et al., 1951), Study quality: Medium; (Heppel et al., 1946), Study quality: Low or Medium; (Mellon Institute, 1947), Study quality: Low • Acute-duration gavage studies evaluated mortality in rats and mice (Kitchin et al., 1993; Storer et al., 1984; Moody et al., 1981). Study quality: High; (Stauffer Chem Co, 1973). Study quality: Medium • Short-term- and subchronic-duration gavage studies evaluated mortality in rats (Daniel et al., 1994; NTP, 1991). Study quality: High • Chronic-duration gavage studies evaluated mortality in wild type and transgenic mice (Storer et al., 1995; NTP, 1978). Study quality: High • A subchronic drinking water study evaluated mortality in mice (NTP, 1991). Study quality: High • Chronic-duration drinking water studies evaluated mortality in mice (Klaunig et al., 1986; Lane et al., 1982). Study quality: High • An acute-duration dermal exposure study evaluated mortality in rabbits (Dow Chemical, 1956), Study quality: Medium • A chronic-duration dermal exposure study evaluated mortality in transgenic mice (Suguro et al., 2017), Study quality: High • A single dose intratracheal exposure study evaluated mortality in rats (Dow Chemical, 1989), Study quality: Medium • Single dose intraperitoneal injection studies evaluated mortality mice (Umezu	• Mouse inhalation: ○ ≥4,339 mg/m^3 (1,072 ppm), 4 hours ○ 6,071 mg/m^3 (1,500 ppm), 7 hours • Rabbit inhalation: ○ 12,100 mg/m^3 (3,000 ppm), 7 hours ○ 6,071 mg/m^3 (1,500 ppm), 5 d ○ 1,980 mg/m^3 (490 ppm), 6 weeks ○ 1,540 mg/m^3 (1.54 mg/L), 20 weeks ○ ≥405 mg/m^3 (100 ppm), gestational exposure • Guinea pig inhalation: ○ 6,071 mg/m^3 (1,500 ppm), 7 hr ○ 3,900 mg/m^3 (3.9 mg/L), 4 d ○ 730 mg/m^3 (0.73 mg/L), 25 weeks • Dog inhalation: ○ 3,900 mg/m^3 (3.9 mg/L), 5 weeks • Cat inhalation: ○ 3,900 mg/m^3 (3.9 mg/L), 11 weeks • Rat gavage: ○ ≥1,000 mg/kg, once ○ ≥240 mg/kg-day, 90 d • Mouse gavage: ○ ≥400 mg/kg, once ○ 150 mg/kg-day, 40 weeks (female transgenic) • Mouse drinking water: ○ 4,926 mg/kg-day, 90 d (female)	○ ≤700 mg/m^3, 1 weeks ○ 420 mg/m^3, 4 weeks ○ ≤363 mg/m^3 (89.8 ppm), 2 yrs • Rabbit, guinea pig, and cat inhalation: ○ 404 mg/m^3, 17 weeks • Rat gavage: ○ 625 mg/kg, once ○ 150 mg/kg-day, 90 d ○ 240 mg/kg-day, gestational exposure • Mouse drinking water: ○ 2,710 mg/kg-day, 90 d (male) • Mouse intraperitoneal: ○ 600 mg/kg, once		

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
and Shibata, 2014; Storer et al., 1984), Study quality: High; (Storer and Conolly, 1983), Study quality: Medium; (Crebelli et al., 1999), Study quality: Low	• Rabbit dermal: ○ 2,800 mg/kg (LD50), 24 hours • Rat intratracheal: ○ 120 mg/kg, once • Mouse intraperitoneal: ○ 486 mg/kg (LD50), once			
Evidence from mechanistic studies (none)			• Indeterminate	

[a] Apart from chronic bioassays, most studies did not report statistical significance of mortality incidences. For the purpose of hazard identification, deaths were considered to be related to treatment if they occurred at a higher incidence than in controls, occurred at the highest dose tested or with a relationship to dose, and were not attributed to factors unrelated to treatment (accident or disease). For chronic-duration studies, only statistically significant, treatment-related effects on survival were included.

N.8 Evidence Integration Table for Cancer for 1,1-Dichloroethane

Table_Apx N-43. Evidence Integration Table for Cancer

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary on Cancer				
Evidence from human studies				*Overall WOSE judgement for cancer effects based on integration of information across evidence streams:* Evidence is not adequate to assess whether 1,1-dichloroethane causes cancer in humans under relevant exposure circumstances.
• A prospective study of women from the California Teacher Study Cohort, for which the EPA's National-Scale Air Toxics Assessment (NATA) was used to estimate exposure, evaluated the association between 1,1-dichloroethane exposure and the incidence of invasive breast cancer (Garcia et al., 2015). Study quality: High	Biological gradient/dose-response: • Exposure to 1,1-dichloroethane was associated with estrogen receptor/progesterone receptor-positive (ER+/PR+) tumors and tumors among women who were past or never users of hormone therapy. Magnitude and precision: • The study used quantitative exposure estimates and accounted for covariate information on individual breast cancer risk factors. Quality of the database: • Associations between breast cancer and exposure were observed in a high-quality study.	Biological gradient/dose-response: • The overall risk for invasive breast cancer was not significantly increased in 1,1-dichloroethane-exposed women relative to unexposed controls. • Analyses based on quintiles of exposure did not show a dose-response relationship with ER+/PR+ tumors. Magnitude and precision: • The effect estimates were small (hazard ratios ≤1.35). • Exposure estimates based on modeling of emissions data may have contributed to exposure misclassification; confidence in the exposure assessment was rated "medium" by US EPA. • Concentrations of 1,1-dichloroethane and vinyl chloride were highly correlated in this study and this co-exposure may have confounded the results.	*Key findings*: In a high-quality study, an association between 1,1-dichloroethane exposure in humans and certain breast tumors was observed. This association was seen in the absence of a significant increase in overall risk for invasive breast cancer in 1,1-dichloroethane-exposed women. *Overall WOSE judgement for cancer effects based on human evidence:* • Indeterminate	
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Breast cancer				
• A gavage study in male and female mice examined the mammary gland for neoplasms after 78 weeks of	Biological gradient/dose-response: • In a study ranked as Uninformative due to high mortality related to pneumonia, a significant dose-related trend for increased incidence of	Magnitude and precision: • The incidence of mammary gland tumors in treated female rats was not statistically significantly increased based on pairwise	*Key findings*: Increased breast cancer incidence was observed in female rats in a study ranked as Uninformative.	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
exposure (NCI, 1978). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [a] examined the mammary gland for neoplasms after 78 weeks of exposure (NCI, 1978).	mammary gland adenocarcinomas was observed in female rats using matched vehicle controls (based on analyses of all females and females surviving at least 52 weeks), despite poor survival limiting the ability to detect late-developing tumors.	comparison to pooled or matched vehicle controls or based on a trend test using pooled vehicle controls. [b] Quality of the database: • Increased incidence of mammary tumors was observed only in a study ranked as Uninformative.	*Overall WOSE judgement for breast cancer effects based on animal evidence:* • Indeterminate	
Liver cancer				
• A gavage study in male and female mice examined the liver for neoplasms after 78 weeks of exposure (NCI, 1978). Study quality: High • Nine-week studies in male rats, which were administered 1,1-dichloroethane via gavage, determined the potential for tumor initiation or promotion based on numbers of GGT-positive foci in the liver (Milman et al., 1988; Story et al., 1986). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [d] examined the liver for neoplasms after 78 weeks of exposure (NCI, 1978). • A cancer bioassay and a tumor promotion assay in male mice [e] assessed the incidence of liver adenomas and/or carcinomas after a 52-	Biological gradient/dose-response: • A significant dose-related trend for increased incidence of hepatocellular carcinomas was observed in male mice surviving at least 52 weeks in the 78-week study using pooled vehicle controls, [c] and the pairwise comparison showed a significant increase at the high dose. These effects were observed despite poor survival in high-dose male mice limiting the ability to detect late-developing tumors. • Exposure resulted in increased numbers of GGT-positive foci in the livers of male rats pretreated with a tumor initiator. Quality of the database: • Evidence of increased liver tumor incidence was observed in a high-quality study.	Magnitude and precision: • The incidence of liver tumors in male mice was not statistically significantly increased in pairwise comparison and trend test using matched vehicle controls. • Only one dose was used in the 9-week tumor initiation and promotion protocols. Quality of the database: • Increased incidence of liver tumors was observed in only one study in one sex (males) followed only for 78 weeks.	*Key findings*: In high-quality studies, increased liver tumor incidence was observed in male mice and evidence supporting tumor promotion was observed in male rats. *Overall WOSE judgement for liver cancer effects based on animal evidence:* • Slight	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
week drinking water exposure (Klaunig et al., 1986).				
Endometrial stromal polyps				
• A gavage study in female mice conducted histopathological examination of the uterus after 78 weeks of exposure (NCI, 1978). Study quality: High Study quality ranked as Uninformative: • A gavage study in female rats [f] conducted histopathological examination of the uterus after 78 weeks of exposure (NCI, 1978).	Biological gradient/dose-response: • The incidence of endometrial stromal polyps in female mice showed a significant dose-related trend using either pooled or matched vehicle controls and a significant increase at the high dose in pairwise comparison to the pooled vehicle controls. [g] Quality of the database: • Evidence of increased endometrial stromal polyp incidence was observed in a high-quality study.	Biological gradient/dose-response: • The incidence of endometrial stromal polyps in female mice was not significantly increased in pairwise comparison to matched vehicle controls. Quality of the database: • Increased incidence of endometrial stromal polyps was observed in only one study in mice followed for only 78 weeks. Biological plausibility and human relevance: • The relevance to humans of endometrial stromal polyps in rodents is uncertain due to differences in etiology and hormone sensitivity (Davis, 2012).	*Key findings*: In a high-quality study, increased endometrial stromal polyp incidence was observed in female mice. The relevance of these findings to humans is uncertain due to differences in etiology and hormone sensitivity among rodents and humans. In addition, there is uncertainty within the scientific community whether endometrial stromal polyps should be considered benign tumors or nonneoplastic lesions. *Overall WOSE judgement for uterine cancer effects based on animal evidence:* • Indeterminate	
Circulatory system cancer				
• A gavage study in male and female mice subjected animals to comprehensive histological examinations for neoplasms after 78 weeks of exposure (NCI, 1978). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [h] subjected animals to comprehensive	Biological gradient/dose-response: • In a study ranked as Uninformative due to high mortality related to pneumonia, a significant dose-related trend for increased incidence of hemangiosarcomas was observed in female rats using either pooled or matched vehicle controls, despite poor survival limiting the ability to detect late-developing tumors.	Consistency: • The incidence of hemangiosarcomas was not increased in male rats. Magnitude and precision: • The incidence of hemangiosarcomas in treated female rats was not statistically significantly increased based on pairwise comparison to pooled or matched vehicle controls. Quality of the database:	*Key findings*: Increased incidence of hemangiosarcomas was observed in female rats in a study ranked as Uninformative. *Overall WOSE judgement for circulatory system cancer effects based on animal evidence:* • Indeterminate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
histological examinations for neoplasms after 78 weeks of exposure (NCI, 1978).		• Increased incidence of hemangiosarcomas was observed in a study ranked as Uninformative.		
Evidence from mechanistic studies				
Genotoxicity: • Three *in vitro* experiments evaluated reverse mutation in *Salmonella typhimurium* (Zeiger et al., 1992; Milman et al., 1988; Simmon et al., 1977) • Three *in vitro* experiments evaluated chromosomal aberrations or DNA repair in mammalian cells (Matsuoka et al., 1998; Williams et al., 1989; Milman et al., 1988) • Two *in vitro* experiments evaluated cell transformation (Milman et al., 1988; Tu et al., 1985; Arthur D. Little Inc., 1983; Hatch et al., 1983), one evaluated DNA binding in a cell-free system (Colacci et al., 1985), and one evaluated chromosome malsegregation in fungi (Crebelli et al., 1995; Crebelli et al., 1988). • Four *in vivo* experiments evaluated chromosomal aberrations, micronuclei, DNA binding, or DNA unwinding in rodents (Patlolla et al., 2005; Taningher et al., 1991; Colacci et al., 1985).	Biological gradient/dose-response: • There were significant, dose-related increases in chromosomal aberrations and micronuclei in the bone marrow of treated mice. • 1,1-dichloroethane treatment resulted in dose-related enhancement of Syrian hamster embryo cell transformation by SA7 (simian) adenovirus. Consistency: • Treatment induced DNA repair in cultured hepatocytes from rats and mice. • DNA adducts were induced by treatment *in vivo* and in a cell-free system.	Biological gradient/dose-response: • Increased chromosomal malsegregation in *Aspergillus nidulans* induced by treatment was not strictly concentration-related. Consistency: • 1,1-dichloroethane did not increase the percent double-stranded DNA in hepatic nuclei of mice exposed *in vivo* • Tests of reverse mutations in *S. typhimurium* yielded inconsistent results. • Some tests of reverse mutation in *S. typhimurium* yielded negative results. • No chromosomal aberrations were observed in Chinese hamster lung fibroblasts tested *in vitro*. • Results were negative for cell transformation in BALB/c-3T3 cells Quality of the database: • The available studies did not evaluate mutagenicity in mammalian cells *in vitro* or *in vivo*.	*Key findings*: Available data are limited but suggest that 1,1-dichloroethane may be genotoxic based on evidence of chromosomal abnormalities and micronuclei in mice *in vivo*. Bacterial mutagenicity findings were not consistent. *Overall WOSE judgement for cancer effects based on mechanistic evidence:* • Slight	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement

[a] The study in male and female rats was considered Uninformative due to high mortality related to pneumonia.
[b] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist.
[c] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist.
[d] The study in male and female rats was considered Uninformative due to high mortality related to pneumonia.
[e] The 52-week study in male mice was considered Uninformative because the duration of the study was not adequate to determine tumorigenicity (cancer bioassay) and because the negative control response was too strong, precluding the ability to determine if 1,1-dichloroethane increased tumor incidence (tumor promotion assay).
[f] The study in female rats was considered Uninformative due to high mortality related to pneumonia.
[g] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist.
[h] The study in male and female rats was considered Uninformative due to high mortality related to pneumonia.

N.9 Evidence Integration Table for Cancer for 1,2-Dichloroethane

Table_Apx N-44. 1,1-Dichloroethane Cancer Evidence Integration Table Based on Read-Across from 1,2-Dichloroethane

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Evidence Integration Summary Judgement on Cancer Effects				
Evidence from human studies				*Overall WOSE judgement for cancer effects based on integration of information across evidence streams:* Evidence indicates that 1,2-dichloroethane likely causes cancer under relevant exposure circumstances.
Breast cancer				
• A prospective study of women from the California Teacher Study Cohort, for which the U.S. EPA's National-Scale Air Toxics Assessment (NATA) was used to estimate exposure, evaluated the association between 1,2-dichloroethane exposure and the incidence of invasive breast cancer (Garcia et al., 2015). Study quality: High • A prospective study of women from the Sister Study Cohort, for which the U.S. EPA's NATA was used to estimate exposure, evaluated the association between 1,2-dichloroethane and the incidence of invasive breast cancer and/or ductal carcinoma *in situ* (Niehoff et al., 2019). Study quality: Medium	Biological gradient/dose-response: • The risk for ER+ invasive breast cancer was slightly, but significantly, increased in quintile 4 (but not quintile 5) of exposure relative to quintile 1 in the medium-quality study. Magnitude and precision: • The study used quantitative exposure estimates and accounted for covariate information on individual breast cancer risk factors.	Biological gradient/dose-response: • The overall risk for breast cancer (both studies) and ER- invasive breast cancer (medium-quality study) was not significantly increased in 1,2-dichloroethane-exposed women. • Analyses based on quintiles of exposure did not show an exposure-response relationship between 1,2-dichloroethane exposure and ER+ invasive breast cancer. Magnitude and precision: • The significant effect estimate for ER+ invasive breast cancer was small (hazard ratio = 1.17). • Exposure estimates based on modeling of emissions data and/or at the census tract level may have contributed to exposure misclassification.	*Key findings*: In a medium-quality study, an association between 1,2-dichloroethane exposure and ER+ invasive breast cancer was observed, but it was small and did not show a clear exposure-response relationship. *Overall WOSE judgement for cancer effects based on human evidence:* • Indeterminate	
Circulatory system cancer				
• A nested case-control study of male workers from three Union Carbide facilities, for which job assignment and history of departmental use were taken to estimate exposure (ever/never), evaluated the association	Biological gradient/dose-response: • In the medium-quality study, there was a nonsignificant increase in the OR for nonlymphocytic leukemia (NLL) in 1,2-dichloroethane-	Biological gradient/dose-response: • In the medium-quality study, exposure levels of 1,2-dichloroethane were not provided. Magnitude and precision:	*Key findings*: Significant limitations in the available studies preclude conclusions regarding associations between 1,2-dichloroethane exposure in	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
between 1,2-dichloroethane exposure and the incidence of hematopoietic tissue cancer (Ott et al., 1989; Union Carbide, 1989). Study quality: Medium Study quality ranked as Uninformative: • A retrospective cohort study of male workers [a] from one Union Carbide facility (one of the three evaluated by (Ott et al., 1989; Union Carbide, 1989)), for which exposure (ever/never) was based on the history and/or duration of work in the chlorohydrin unit (which produced 1,2-dichloroethane as a byproduct), evaluated the association between chemical exposure and the risk of mortality due to lymphopoietic cancers (Benson and Teta, 1993).	exposed workers, which was higher in those working more than 5 years. • In a study ranked as Uninformative owing to lack of an appropriate comparison group and lack of 1,2-dichloroethane exposure levels, work in the chlorohydrin unit was significantly associated with mortality from lymphatic and hematopoietic cancers.	• In the medium-quality study, there was potential for confounding because covariates were not considered (race, smoking status, concurrent exposure to other chemicals). • In the medium-quality study, statistical power was limited because cancer case numbers were low (n = 5 for NLL). • In the medium-quality study, statistical methods were not specified and ORs were provided without CIs. Consistency: • In the Uninformative study, analysis was conducted based on work department rather than specific chemicals.	humans and circulatory system cancers. *Overall WOSE judgement for cancer effects based on human evidence:* • Indeterminate	
Pancreatic cancer				
• A case-control study of men and women from 24 states, which estimated intensity and probability of 1,2-dichloroethane exposure (low, medium, high) based on listed occupation and industry (from death certificates) and a job exposure matrix (JEM), evaluated the association between 1,2-Dichloroethane exposure and the risk of pancreatic cancer (Kernan et al., 1999). Study quality: High Study quality ranked as Uninformative: • A retrospective cohort study of male workers [b] from a Union Carbide facility, for which exposure (ever/never) was based on the history and/or duration of work in the	Biological gradient/dose-response: • In the high-quality study, 1,2-dichloroethane exposure was associated with a slight, but borderline significant, increased OR for pancreatic cancer among Black females with low estimated exposure intensity. • In a study ranked as Uninformative owing to lack of an appropriate comparison group and lack of 1,2-dichloroethane exposure levels, work in the chlorohydrin unit was significantly associated with mortality from pancreatic cancer.	Biological gradient/dose-response: • In the high-quality study, the risk for pancreatic cancer in Black females was not increased in groups with medium or high intensity exposure. Consistency: • In the high-quality study, 1,2-dichloroethane exposure was not associated with an increased risk of pancreatic cancer in Black males, White females, or White males. • In the Uninformative study, analysis was conducted based on work department rather than specific chemicals. Magnitude and precision:	*Key findings*: In a high-quality study, a slight, but significant, association between low intensity 1,2-dichloroethane exposure and pancreatic cancer was observed in Black females, but the association did not show an exposure-response relationship, and no association was observed in Black males or White males or females. *Overall WOSE judgement for cancer effects based on human evidence:* • Indeterminate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
chlorohydrin unit (which produced 1,2-dichloroethane as a byproduct), evaluated the association between chemical exposure and the risk of mortality due to pancreatic cancer (Benson and Teta, 1993).		• In the high-quality study, the effect estimate in Black females was small (OR = 1.2, 95% CI 1.0–1.4). • In the high-quality study, there was the potential for exposure misclassification based on the occupation and industry data captured on death certificates.		
Kidney cancer				
• A population-based, case-control study of men and women from the Minnesota Cancer Surveillance System (cases) and the general population of Minnesota or the Health Care Financing administration (controls), for which exposure was estimated based on occupational history and JEMs, evaluated the association between 1,2-dichloroethane exposure and the risk for renal cell carcinoma (Dosemeci et al., 1999). Study quality: Medium	Biological gradient/dose-response: • The risk of renal cell carcinoma was significantly increased in women exposed to all organic solvents combined and all chlorinated aliphatic hydrocarbons combined. Magnitude and precision: • The use of a priori assessment of exposure to solvents (including 1,2-dichloroethane) using JEMs reduced recall bias among men and women and cases and controls.	Biological gradient/dose-response: • No significant increase in the risk of renal cell carcinoma was observed based on exposure to 1,2-dichloroethane among men, women, or all participants. Magnitude and precision: • The number of participants exposed to 1,2-dichloroethane (40 cases and 48 controls) may have been too low to detect effects associated with 1,2-dichloroethane exposure. Quality of the database: • Only one medium-quality study was available to assess risk of renal cancer due to 1,2-dichloroethane exposure.	*Key findings*: In a medium-quality study, no significant association between 1,2-dichloroethane exposure in humans and renal cell carcinoma was observed; however, the number of exposed subjects in the study population was small. *Overall WOSE judgement for cancer effects based on human evidence:* • Indeterminate	
Prostate cancer				
• A retrospective cohort study evaluated cancer incidence in 251 employees of an herbicide manufacturing facility (bentazon unit) between 1979 and 1987, followed until 2003. SMRs were calculated using age-, gender-, and race-specific cancer incidence rates in South	Biological gradient/dose-response: • A statistically significant association was observed between employment in the bentazon unit and prostate cancer incidence (SIR = 2.2, 95% CI = 1.1–3.9)	Magnitude and precision: • The study did not directly assess the association between exposure to 1,2-dichloroethane and prostate cancer. Other chemicals were also used in the bentazon unit.	*Key findings*: In a medium-quality study, an association between work in bentazon production and prostate cancer was observed; however, the association with 1,2-	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Louisiana. (BASF, 2005). Study quality: Medium			dichloroethane was not directly assessed. *Overall WOSE judgement for cancer effects based on human evidence:* Indeterminate	
Evidence from apical endpoints in *in vivo* mammalian animal studies				
Breast cancer				
• A gavage study in male and female mice examined the mammary gland for neoplasms after 78 weeks of exposure (NTP, 1978). Study quality: High • Two inhalation studies in male and female rats (Nagano et al., 2006; Cheever et al., 1990) and one inhalation study in male and female mice (Nagano et al., 2006) examined the mammary gland for neoplasms after 104 weeks of exposure. Study quality: High • A dermal study in male and female transgenic mice susceptible to cancer examined the mammary gland for neoplasms after 26 weeks of exposure (Suguro et al., 2017). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [d] examined the mammary gland for neoplasms after 78 weeks of exposure (NTP, 1978). • An inhalation study in male and female rats and mice [e] examined the mammary gland for neoplasms at natural death after 78 weeks of exposure (Maltoni et al., 1980).	Biological gradient/dose-response: • A significant dose-related trend for increased incidence of mammary gland adenocarcinomas was observed in female mice in the 78-week gavage study using pooled vehicle controls [c] ; pairwise comparisons showed significant increases at both doses. • Significant dose-related trends for increased mammary gland adenomas, fibroadenomas, and/or adenocarcinomas were observed in male and female rats after 104 weeks of inhalation exposure; pairwise comparisons showed significant increases at the highest exposure. • A significant dose-related trend for increased incidence of mammary gland adenocarcinoma was observed in female mice after 104 weeks of inhalation exposure. • In a study ranked as Uninformative due to high mortality from pneumonia, significant dose-related trends for increased mammary gland	Consistency: • The incidence of mammary gland tumors was not increased in a 26-week dermal study in transgenic mice. Magnitude and precision: • Pairwise comparisons were not significant for increased incidence of mammary gland adenocarcinoma in female mice after 104 weeks of inhalation exposure.	*Key findings*: Mammary gland tumors were observed in male and female rats and in female mice exposed to 1,2-dichloroethane orally or via inhalation in high-quality studies. *Overall WOSE judgement for breast cancer effects based on animal evidence*: • Robust	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
	adenocarcinomas or adenocarcinomas and fibroadenomas were observed in female rats in the 78-week study; pairwise comparisons showed a significant increase at the high dose for adenocarcinomas and at both doses for combined tumors. • In a study ranked uninformative due to lack of inhalation exposure details, the incidence of mammary gland fibromas and fibroadenomas was significantly increased in rats after 78 weeks of inhalation exposure. Quality of the database: • Evidence of mammary gland tumors in rats and mice was observed in high-quality studies.			
Liver cancer				
• A gavage study in male and female mice examined the liver for neoplasms after 78 weeks of exposure (NTP, 1978). Study quality: High • Two inhalation studies in male and female rats (Nagano et al., 2006; Cheever et al., 1990) and one inhalation study in male and female mice (Nagano et al., 2006) examined the liver for neoplasms after 104 weeks of exposure. Study quality: High • A dermal exposure study in male and female transgenic mice susceptible to cancer examined the liver for neoplasms after 26 weeks of exposure	Biological gradient/dose-response: • A significant dose-related trend for increased incidence of hepatocellular carcinomas was observed in male (but not female) mice in the 78-week gavage study using pooled and matched vehicle controls [f], and the pairwise comparison to pooled vehicle controls showed a significant increase at the high dose. • A significant dose-related trend for increased incidence of hepatocellular adenomas and adenomas or carcinomas was	Consistency: • The incidence of liver tumors was not increased in transgenic mice following 26 weeks of dermal exposure. Magnitude and precision: • In female mice, incidences of hepatocellular adenomas and adenomas or carcinomas in the 104-week inhalation study were not significantly increased based on pairwise comparisons to controls.	*Key findings*: In high-quality studies, increased liver tumor incidence was observed in male or female mice following exposure via gavage or inhalation, respectively. *Overall WOSE judgement for liver cancer effects based on animal evidence:* • Slight to Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
(Suguro et al., 2017). Study quality: High • Nine-week gavage studies in male rats evaluated the potential for tumor initiation and/or promotion in the liver based on numbers of gamma-glutamyltranspeptidase (GGT)-positive foci (Milman et al., 1988; Story et al., 1986). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [g] examined the liver for neoplasms after 78 weeks of exposure (NTP, 1978). • A cancer bioassay and a tumor promotion assay in male mice [h] assessed the incidence of liver adenomas and/or carcinomas after 52 weeks drinking water exposure (Klaunig et al., 1986). An inhalation study in male and female rats and mice [i] examined the liver for neoplasms at natural death after 78 weeks of exposure (Maltoni et al., 1980). • A dermal exposure study in female mice [j] examined the liver for neoplasms after up to 85 weeks of exposure (Van Duuren et al., 1979).	observed in female (but not male) mice following 104 weeks of inhalation exposure. Quality of the database: • Evidence of increased liver tumor incidence was observed in high-quality studies.			
Lung cancer				
• A gavage study in male and female mice examined the lung for neoplasms after 78 weeks of exposure (NTP, 1978). Study quality: High • Two inhalation studies in male and female rats (Nagano et al., 2006;	Biological gradient/dose-response: • Significant trends and pairwise comparisons for increased incidence of alveolar/bronchiolar adenomas were observed in male and	Magnitude and precision: • Pairwise comparisons did not show a significant increase in the incidence of lung tumors in female mice in the 104-week study.	*Key findings*: In high-quality studies, increased lung tumor incidence was observed in male and/or female mice following gavage,	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Cheever et al., 1990) and one inhalation study in male and female mice (Nagano et al., 2006) examined the lung for neoplasms after 104 weeks of exposure. Study quality: High • A dermal exposure study in male and female transgenic mice examined the lung for neoplasms after 26 weeks of exposure (Suguro et al., 2017). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [k] examined the lung for neoplasms after 78 weeks of exposure (NTP, 1978). • A cancer bioassay and a tumor promotion assay in male mice [l] assessed the incidence of lung adenomas and/or carcinomas after 52 weeks of drinking water exposure (Klaunig et al., 1986). • An inhalation study in male and female rats and mice [m] examined the lungs for neoplasms at natural death after 78 weeks of exposure (Maltoni et al., 1980). • A dermal exposure study in female mice [n] reported neoplasms in the lung (not routinely examined) after up to 82 weeks of exposure (Van Duuren et al., 1979).	female mice in the 78-week gavage study. • Significant trends for increased incidence of bronchiolo-alveolar carcinomas and carcinomas or adenomas were observed in female mice following 104 weeks of inhalation exposure. • Significant increases in the incidence and multiplicity of bronchiolo-alveolar adenomas and adenocarcinomas were observed in both sexes in the dermal study using transgenic mice. Consistency: • In the dermal study ranked as Uninformative due to the use of methods that did not account for the volatility of 1,2-dichloroethane, a significantly increased incidence of benign lung papillomas was observed in female mice. Quality of the database: • Evidence of lung tumors was observed in three high-quality studies.		inhalation, or dermal exposure. *Overall WOSE judgement for lung cancer effects based on animal evidence:* • Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Mesothelioma of the peritoneum				
• A gavage study in male and female mice conducted comprehensive histopathological examination after 78 weeks of exposure (NTP, 1978). Study quality: High • Two inhalation studies in male and female rats (Nagano et al., 2006; Cheever et al., 1990) and one inhalation study in male and female mice (Nagano et al., 2006) conducted comprehensive histopathological examination after 104 weeks of exposure. Study quality: High • A dermal exposure study in male and female transgenic mice susceptible to cancer conducted comprehensive histopathological examination after 26 weeks of exposure (Suguro et al., 2017). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [o] conducted comprehensive histopathological examination after 78 weeks of exposure (NTP, 1978). • An inhalation study in male and female rats and mice [p] conducted comprehensive histopathological examination at natural death after 78 weeks of exposure (Maltoni et al., 1980).	Biological gradient/dose-response: • A significant trend for increased incidence of mesothelioma of the peritoneum was observed in male rats following 104 weeks of inhalation exposure. Quality of the database: • Evidence of mesothelioma of the peritoneum was observed in a high-quality study.	Magnitude and precision: • Pairwise comparisons did not show a significant increase in the incidence of mesothelioma of the peritoneum in male rats in the 104-week inhalation study. Consistency: • There was no significant increase in incidence of mesothelioma of the peritoneum in female rats following 104 weeks of inhalation exposure. • The incidence of mesothelioma of the peritoneum was not increased in transgenic mice following 26 weeks of dermal exposure.	*Key findings*: In a high-quality study, a trend for increased incidence of mesothelioma of the peritoneum was observed in male mice following inhalation exposure; no significant increase was noted in pairwise comparison, and no increase was seen in female mice. *Overall WOSE judgement for mesothelioma of the peritoneum based on animal evidence:* • Indeterminate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
Endometrial stromal polyps				
• A gavage study in female mice conducted histopathological examination of the uterus after 78 weeks of exposure (NTP, 1978). Study quality: High • Two inhalation studies in female rats (Nagano et al., 2006; Cheever et al., 1990) and one inhalation study in female mice (Nagano et al., 2006) conducted histopathological examination of the uterus after 104 weeks of exposure. Study quality: High • A dermal exposure study in female transgenic mice susceptible to cancer conducted histopathological examination of the uterus after 26 weeks of exposure (Suguro et al., 2017). Study quality: High Study quality ranked as Uninformative: • A gavage study in female rats [q] examined the uterus for neoplasms after 78 weeks of exposure (NTP, 1978).	Biological gradient/dose-response: • A significant trend for increased incidence of endometrial stromal polyps or sarcomas was observed in female mice in the 78-week gavage study using pooled vehicle controls [r], and the pairwise comparison showed a significant increase at both doses. • A significant trend for increased incidence of endometrial stromal polyps was observed in female mice following 104 weeks of inhalation exposure. Quality of the database: • Evidence of endometrial stromal polyps in mice was observed in high-quality oral and inhalation studies.	Biological gradient/dose-response: • The incidence of endometrial stromal polyps in female mice was not significantly increased in a 26-week dermal exposure study in transgenic mice. Magnitude and precision: • Pairwise comparisons using matched controls did not show a significant increase in the incidence of stromal polyps or sarcomas, and the incidence of sarcomas (alone) was not significantly increased in female mice in the 78-week gavage study. • Pairwise comparisons did not show a significant increased incidence in stromal polyps in female mice in the 104-week inhalation study. Biological plausibility and human relevance: The relevance to humans of endometrial stromal polyps in mice is uncertain due to differences in etiology and hormone sensitivity (Davis, 2012)	*Key findings*: In high-quality oral and inhalation studies, the incidence of endometrial stromal polyps was increased in female mice. The relevance of these findings to humans is uncertain due to differences in etiology and hormone sensitivity among rodents and humans. In addition, there is uncertainty within the scientific community whether endometrial stromal polyps should be considered benign tumors or nonneoplastic lesions. *Overall WOSE judgement for uterine cancer effects based on animal evidence:* • Indeterminate	
Circulatory system cancer				
• A gavage study in male and female mice subjected animals to comprehensive histological examinations for neoplasms after 78 weeks of exposure (NTP, 1978). Study quality: High	Biological gradient/dose-response: • Significant pairwise increases in the incidence of hemangiosarcoma in the liver were observed in male mice at the two highest exposure	Biological gradient/dose-response: • There was not a significant dose-related trend for increased hemangiosarcomas of the liver in male mice following 104 weeks of inhalation exposure.	*Key findings*: In medium- and high-quality studies, the incidence of circulatory system tumors (e.g., hemangiosarcomas) was increased in mice	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• A gavage study in female transgenic mice susceptible to cancer subjected animals to histological examinations after 40 weeks of exposure (Storer et al., 1995). Study quality: Medium • Two inhalation studies in male and female rats (Nagano et al., 2006; Cheever et al., 1990) and one inhalation study in male and female mice (Nagano et al., 2006) subjected animals to comprehensive histological examinations for neoplasms after 104 weeks of exposure. Study quality: High • A dermal study in transgenic mice susceptible to cancer subjected animals to comprehensive histological examinations for neoplasms after 26 weeks of exposure (Suguro et al., 2017). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [s] subjected animals to comprehensive histological examinations for neoplasms after 78 weeks of exposure (NTP, 1978).	concentrations following 104 weeks of inhalation exposure. • A significantly increased incidence of malignant lymphoma was observed in female transgenic mice in a 40-week gavage study. • In a study ranked as Uninformative due to high mortality from pneumonia, there was a significant trend for increased hemangiosarcomas in male and female rats in a 78-week gavage study using pooled vehicle controls [t], and the pairwise comparison showed a significant increase at both doses. Quality of the database: • Increased incidences of circulatory system cancers were observed in medium- and high-quality studies.	• The incidence of circulatory system cancers was not increased in mice in a 78-week gavage study. There was a significant trend for *decreased* malignant lymphomas of the hematopoietic system in females using matched vehicle controls. • No hemangiomas or hemangiosarcomas were observed in male or female transgenic mice in a 26-week dermal study. Magnitude and precision: • In the 78-week gavage study ranked Uninformative, the trends for increased hemangiosarcomas in male and female rats were not significant using matched controls.	following inhalation and dermal exposure. *Overall WOSE judgement for circulatory system cancer effects based on animal evidence:* • Slight	
• A gavage study in male transgenic mice [u] susceptible to cancer examined the incidence of malignant lymphomas after 40 weeks of exposure (Storer et al., 1995). • An inhalation study in male and female rats and mice [v] examined animals for neoplasms at natural death after 78 weeks of exposure (Maltoni et al., 1980).				
Gastrointestinal tract cancer				

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• A gavage study in male and female mice examined the gastrointestinal tract for neoplasms after 78 weeks of exposure (NTP, 1978). Study quality: High • Two inhalation studies in male and female rats (Nagano et al., 2006; Cheever et al., 1990) and one inhalation study in male and female mice (Nagano et al., 2006) examined the gastrointestinal tract for neoplasms after 104 weeks of exposure. Study quality: High • A dermal exposure study in male and female transgenic mice examined the gastrointestinal tract for neoplasms after 26 weeks of exposure (Suguro et al., 2017). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [x] examined the gastrointestinal tract for neoplasms after 78 weeks of exposure (NTP, 1978). • An inhalation study in male and female rats and mice [y] examined the stomach and intestines for neoplasms at natural death after 78 weeks of exposure (Maltoni et al., 1980). • A dermal exposure study in female mice [z] examined the stomach for neoplasms after up to 85 weeks of exposure (Van Duuren et al., 1979).	Biological gradient/dose-response: • A significant trend for increased incidence of squamous-cell carcinomas in the stomach was observed in female mice in the 78-week gavage study using pooled vehicle controls. • In a study ranked as Uninformative owing to high mortality from pneumonia, a significant trend for increased incidence of squamous-cell carcinomas in the stomach was observed in male rats in the 78-week gavage study using pooled and matched vehicle controls [w]; the pairwise comparisons showed a significant increase at the highest dose.	Biological gradient/dose-response: • The incidence of gastrointestinal tumors (forestomach tumors) was not increased in rats or mice following 104 weeks of inhalation exposure. • The incidence of gastrointestinal tumors was not increased in two dermal studies, including a study in transgenic male and female mice treated for 26 weeks, and an 85-week study in female mice ranked as Uninformative due to the use of methods that did not account for the volatility of 1,2-dichloroethane. Magnitude and precision: • The trend for increased incidence of squamous-cell carcinomas in female mice in the 78-week gavage study was not significant using matched controls, and the pairwise comparisons using pooled and matched controls were not significant.	*Key findings*: In high-quality and Uninformative gavage studies, increased incidences of gastrointestinal tract tumors were observed in female mice and male rats. The effect appears to be route-specific because several high-quality studies did not identify gastrointestinal tumors following inhalation or dermal exposure. *Overall WOSE judgement for gastrointestinal cancer effects based on animal evidence:* • Indeterminate	
Subcutaneous fibromas				
• A gavage study in male and female mice conducted comprehensive histopathological examination after 78 weeks of exposure (NTP, 1978). Study quality: High	Biological gradient/dose-response: • A significant trend for increased incidence subcutaneous fibroma was observed in male and female rats following 104 weeks	Magnitude and precision: • A significant dose-related trend for increased incidence of subcutaneous fibromas was not observed in male rats in the 78-	*Key findings*: In a high-quality study, an increased incidence of subcutaneous fibromas in male and female rats was	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
• Two inhalation studies in male and female rats (Nagano et al., 2006; Cheever et al., 1990) and one inhalation study in male and female mice (Nagano et al., 2006) conducted comprehensive histopathological examination after 104 weeks of exposure. Study quality: High • A dermal exposure study in male and female transgenic mice conducted comprehensive histopathological examination after 26 weeks of exposure (Suguro et al., 2017). Study quality: High Study quality ranked as Uninformative: • A gavage study in male and female rats [aa] conducted comprehensive histopathological examination after 78 weeks of exposure (NTP, 1978). • An inhalation study in male and female rats and mice [bb] conducted comprehensive histopathological examination at natural death after 78 weeks of exposure (Maltoni et al., 1980).	of inhalation exposure; pairwise comparisons showed a significant increase at the high dose in female rats only. • In a study ranked as Uninformative due to high mortality from pneumonia, a significant dose-related trend for increased incidence of subcutaneous fibromas was observed in male rats in the 78-week gavage study using pooled vehicle controls [dd]; pairwise comparisons showed significant increases at both doses. Quality of the database: • Evidence of subcutaneous fibroma was observed in a high-quality study.	week gavage study using matched vehicle controls. Consistency: • The incidence of subcutaneous tumors was not increased in transgenic mice following 26 weeks of dermal exposure.	seen following inhalation exposure. *Overall WOSE judgement for subcutaneous fibromas based on animal evidence:* • Indeterminate	
Evidence from mechanistic studies				
Genotoxicity: [cc] • Two recent authoritative reviews (ATSDR, 2024; Gwinn et al., 2011) were the primary sources used to provide an overview of the database of genotoxicity studies available for 11,2 dichloroethane, including numerous studies of gene mutation in *Salmonella typhimurium*; gene mutation in fruit flies; gene mutation, micronucleus formation, DNA damage, and DNA binding/adduct	Consistency: • In most of the available studies, 1,2 dichloroethane induced mutations in *S. typhimurium* in the presence of metabolic activation. Many of these studies also reported positive results without metabolic activation. • 1,2 dichloroethane induced gene mutations in multiple studies of fruit flies.	Quality of the database: • Alternative modes of action were investigated only for mammary gland tumors and not for other tumor types induced by 1,2-dichloroethane.	*Key findings*: 1,2-dichloroethane has induced mutations, clastogenic effects, DNA damage, and DNA binding/adduct formation *in vitro* and *in vivo*. The preponderance of the substantial database consists of positive results. While these effects could plausibly be related to formation of	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
formation in mammalian cells/tissue isolates *in vitro*; and clastogenicity, DNA damage, and DNA binding/adduct formation in mammals *in vivo*. Other mechanisms: • A 28-day inhalation exposure experiment in female rats evaluated cell proliferation in mammary tissue and serum prolactin levels (Lebaron et al., 2021).	• 1,2 dichloroethane yielded positive results in gene mutation assays in Chinese hamster ovary cells and human lymphoblastoid cells *in vitro*. • 1,2 dichloroethane produced clastogenic effects including micronuclei in human lymphocytes *in vitro* and micronuclei, chromosomal aberrations, and sister chromatid exchanges in rat and mouse bone marrow *in vivo*. • DNA damage was observed in human lymphocytes and rat and mouse hepatocytes exposed to 1,2 dichloroethane *in vitro* and in multiple tissues from rats and mice exposed *in vivo*. • DNA binding/adduct formation after 1,2 dichloroethane exposure was observed in vitro and in multiple tissues from rats and mice *in vivo*. Biological plausibility and human relevance: • Several metabolites of 1,2-dichloroethane, particularly those from the glutathione conjugation pathway, have been shown to bind DNA and induce DNA damage *in vivo*, and to induce mutations in *S. typhimurium in vitro*. Quality of the database: • The genotoxicity database includes numerous *in vitro* and *in vivo* studies evaluating a wide		tumors, a direct connection between these events and 1,2 dichloroethane induced carcinogenesis has not been conclusively demonstrated. Few mechanistic data examining alternative modes of carcinogenic action are available. *Overall WOSE judgement for cancer effects based on mechanistic evidence:* • Moderate	

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
	variety of genotoxic endpoints in multiple test systems.			

[a] The study was ranked as Uninformative because SMRs were calculated based on expected deaths from a reference population matched on sex, but not age, and exposure was assessed based on duration of work in the facility; no information was provided on levels of exposure to 1,2-dichlororethane.
[b] The study was ranked as Uninformative because SMRs were calculated based on expected deaths from a reference population matched on sex and exposure was assessed based on duration of work in the facility; no information was provided on levels of exposure to 1,2-dichloroethane.
[c] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist.
[d] The study in male and female rats was considered Uninformative due to high mortality from pneumonia in all groups (including controls).
[e] Pending evaluation.
[f] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist
[g] The study in male and female rats was considered Uninformative due to high mortality from pneumonia in all groups (including controls).
[h] The study in male mice was considered Uninformative due to inadequate study duration (52-week cancer bioassay) and a high tumor response rate in the initiation-only control group (tumor promotion assay).
[i] This chronic inhalation study was ranked Uninformative due to lack of information on the inhalation exposure methodology.
[j] The study in female mice was considered Uninformative because methods used to conduct the study did not account for volatility of the test substance.
[k] The study in male and female rats was considered Uninformative due to high mortality from pneumonia in all groups (including controls).
[l] The study in male mice was considered Uninformative due to inadequate study duration (52-week cancer bioassay) or a high tumor response rate in the initiation-only control group (tumor promotion assay).
[m] This chronic inhalation study was ranked Uninformative due to lack of information on the inhalation exposure methodology.
[n] The study in female mice was considered Uninformative because methods used to conduct the study did not account for volatility of the test substance.
[o] The study in male and female rats was considered Uninformative due to high mortality from pneumonia in all groups (including controls).
[p] This chronic inhalation study was ranked Uninformative due to lack of information on the inhalation exposure methodology.
[q] The study in female rats was considered Uninformative due to high mortality from pneumonia in all groups (including controls).
[r] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist.
[s] The study in male and female rats was considered Uninformative due to high mortality from pneumonia in all groups (including controls).
[t] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist.
[u] The study in male transgenic mice was considered Uninformative because the duration of the study was potentially inadequate for tumor development and no tumors were observed (the same study in female transgenic mice was considered Informative because tumors were observed).
[v] This chronic inhalation study was ranked Uninformative due to lack of information on the inhalation exposure methodology.
[w] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist.
[x] The study in male and female rats was considered Uninformative due to high mortality from pneumonia in all groups (including controls).
[y] Pending evaluation.
[z] The study in female mice was considered Uninformative due to the use of methods that did not account for the volatility of 1,2-dichloroethane.

Database Summary	Factors that Increase Strength	Factors that Decrease Strength	Summary of Key Findings and Within-Stream Strength of the Evidence Judgement	Inferences Across Evidence Streams and Overall WOSE Judgement
[aa] The study in male and female rats was considered Uninformative due to high mortality from pneumonia in all groups (including controls). [bb] This chronic inhalation study was ranked Uninformative due to lack of information on the inhalation exposure methodology. [cc] Including experiments reviewed by Gwinn et al. (2011) and/or ATSDR (2024) that were not flagged as inconsistent with OECD guidance on genotoxicity testing, as well as the one study published subsequently (Lone et al., 2016). [dd] Pooled controls from several bioassays were used based on data for the same strain, tested by the same laboratory no more than 6 months apart, and diagnosed by the same pathologist.				

N.10 Cancer Dose-Response Assessment (Read-Across from 1,2-Dichloroethane)

The available cancer dose-response data for 1,1-dichloroethane are not adequate for use in deriving cancer PODs. The only available human study was confounded by co-exposure to vinyl chloride (Garcia et al., 2015). Animal studies included a 78-week study in rats and mice exposed by gavage that was limited by premature mortality in both species (due to pneumonia in rats, and with no cause of death identified for mice) (NCI, 1978); a drinking water study in which animals were sacrificed after only 52 weeks (Klaunig et al., 1986); and a 9-week study of GGT+ foci in partially hepatectomized rats (Milman et al., 1988). In the absence of chemical-specific data, as described in Section 5.2.1.3, the cancer risk assessment for 1,1-dichloroethane uses read-across from data for the identified analog 1,2-dichloroethane.

1,2-Dichloroethane IUR for Inhalation Exposures

In 1987, the IRIS program derived an IUR of 2.6×10^{-5} (per µg/m^3) based on route-to-route extrapolation from the oral CSF derived within the report. Additionally, the NTP (1978) study that was used to derive the oral slope factor and the subsequent IUR was identified as "uninformative" by systematic review for dose-response for derivation of the CSF/IUR due to confounding associated with mortality and disease. The inhalation cancer bioassay by Nagano et al. (2006) was not available at the time of the IRIS assessment, thus allowing for the 1,1-dichloroethane risk evaluation to update and derive an IUR based on the inhalation route thus minimizing uncertainties associated with the route-to-route extrapolation. Nagano et al. (2006) treated F344 rats and BDF1 mice at concentrations of 0,10, 40, or 160 ppm or 0, 10, 30, or 90 ppm, respectively, for 6 hours/day 5 days/week for 104 weeks. In the F344 rats, increased incidences of subcutaneous fibromas along with the occurrence of mammary gland adenomas, fibroadenomas, and adenocarcinomas were identified at 160 ppm of 1,2-dichloroethane. Additionally, increased incidences of liver hemangiosarcomas were observed in male mice in the 30 and 90 ppm treatment groups for 1,2-dichloroethane.

IUR estimates based on the tumor data sets in Nagano et al. (2006) were calculated using the following equation: IUR = BMR ÷ HEC, where BMR is the benchmark response and HEC is the human equivalent concentration in µg/m^3.

A BMR of 10 percent extra risk was selected for all datasets. HECs were calculating using the ratio of blood:gas partition coefficients, as shown in Appendix N.1.2. Gargas and Andersen (1989) estimated blood:air partition coefficients for 1,2-dichloroethane of 19.5 and 30.4 in humans and rats, respectively. Because the rat partition coefficient is greater than the human partition coefficient, the default ratio of 1 is used in the calculation in accordance with U.S. EPA (1994) guidance. A blood:air partition coefficient for mice was not available from the literature reviewed; thus, the default ratio of 1 was used to calculate HECs for data in mice.

Details of the BMD modeling are provided in the *Risk Evaluation for 1,1-Dichloroethane – Supplemental Information File: Benchmark Dose Modeling* (U.S. EPA, 2025f) and the BMCL, HEC, and IUR estimate for each dataset is shown in Table_Apx N-45.

Table_Apx N-45. IUR Estimates for Tumor Data from Nagano et al. (2006) Study of 1,2-Dichloroethane Using Linear Low-Dose Extrapolation Approach

Species and Sex	Tumor Type	Selected Model	BMCL10% (ppm)	BMCL10% ($\mu g/m^3$)	HEC ($\mu g/m^3$)	IUR Estimate $(\mu g/m^3)^{-1}$
Male rats	Subcutaneous fibroma	Multistage 1-degree	7	28,332	28,332	3.5E−06
	Mammary gland fibroadenomas	Multistage 1-degree	17	68,807	68,807	1.5E−06
	Mammary gland fibroadenomas and adenomas combined	Multistage 3-degree	15	60,712	60,712	1.6E−06
	Peritoneal mesothelioma	Multistage 3-degree	19	76,901	76,901	1.3E−06
	Combined mammary gland, subcutaneous, and peritoneum tumors	MS Combo	5	20,237	20,237	4.9E−06
Female rats	Subcutaneous fibroma	Multistage 1-degree	17	68,807	68,807	1.5E−06
	Mammary gland adenomas	Multistage 1-degree	9	36,427	36,427	2.7E−06
	Mammary gland fibroadenomas	Multistage 1-degree	8	32,380	32,380	3.1E−06
	Mammary gland fibroadenomas and adenomas combined	Multistage 1-degree	5	20,237	20,237	4.9E−06
	Mammary gland adenocarcinoma	Multistage 3-degree	23	93,091	93,091	1.1E−06
	Mammary gland fibroadenomas adenomas, and adenocarcinomas combined	Multistage 1-degree	4	16,190	16,190	6.2E−06
	Combined mammary gland and subcutaneous tumors	MS Combo	4	16,190	16,190	6.2E−06
Female mice	Bronchiolo-alveolar adenomas	Multistage 3-degree	9	36,427	36,427	2.7E−06
	Bronchiolo-alveolar carcinomas	Multistage 2-degree	14	56,664	56,664	1.8E−06
	Bronchiolo-alveolar adenomas and carcinomas combined	Multistage 2-degree	7	28,332	28,332	3.5E−06
	Mammary gland adenocarcinomas	Multistage 3-degree	10	40,474	40,474	2.5E−06
	Hepatocellular adenomas	Multistage 3-degree	11	44,522	44,522	2.2E−06
	Hepatocellular adenomas and carcinomas combined	Multistage 2-degree	10	40,474	40,474	2.5E−06
	Combined lung, mammary gland, and liver tumors	MS Combo	5	20,237	20,237	4.9E−06

The highest estimated IUR is 6.2×10^{-6} (per $\mu g/m^3$) for combined mammary gland adenomas, fibroadenomas, and adenocarcinomas and subcutaneous fibromas in female rats in the inhalation study

by Nagano et al. (2006). BMD modeling of the combined tumor incidences in female rats was performed as the incidences of the mammary tumors and subcutaneous fibromas showed a significant positive trend with increased concentration and were significantly different from the control group at 160 ppm (combined mammary tumors were also significantly different from controls at 40 ppm). The incidences of mammary tumors in the control group were at incidence rates that did not exceed the maximum tumor incidences when compared to historical controls and thus retained in the modeling.

CSF for Oral Exposures

The IRIS program derived an oral CSF of 9.1×10^{-2} (per mg/kg-bw/day) for 1,2-dichloroethane in 1987 based on the incidence of hemangiosarcomas in male rats in the chronic bioassay by NTP (1978), however, this study did not pass EPA systematic review. The oral CSF for male mice based on hepatocarcinomas of 6.2×10^{-2} (per mg/kg-bw/day) in a reliable study NTP (1978). No oral cancer bioassays of 1,2-dichloroethane have been published since the IRIS assessment. The IRIS CSF was derived using time-to-tumor modeling to account for intercurrent mortality of the rats in the NTP (1978) study. No updates to the time-to-tumor modeling approach have been made since the 1987 assessment. Hemangiosarcomas in male rats were determined to be the most sensitive species, strain, and site, however this study was deemed unacceptable by EPA systematic review. Although CSF does not account for other tumor types induced by 1,2-dichloroethane in the male rat, there is currently no time-to-tumor modeling approach available that accounts for multiple tumor types. Therefore, the oral CSF for 1,2-dichloroethane from the reliable NTP mouse cancer study NTP (1978) was selected for use in assessment of cancer risks associated with exposure to 1,1-dichloroethane. This mouse CSF was used to calculate a drinking water unit risk of 1.8 E-6 per ug/L using a drinking water intake of 2 L/day and body weight of 70 kg.

CSF for Dermal Exposures

There were no identified dermal cancer studies for either 1,1- or 1,2-dichloroethane for quantitative dose-response. The 1,2-dichloroethane dermal study by Suguro et al. (2017) did identify bronchioalveolar adenomas and adenocarcinomas, however, its single dose did not allow calculation of an accurate dermal linear low-dose cancer slope factor (Suguro et al., 2017). A dermal CSF was not derived from 1,1- or 1,2-dichloroethane via route-to-route extrapolation using oral data. Additionally, there are uncertainties associated with extrapolation using 1,2-dichloroethane data from both oral and inhalation dosing for 1,1-dichloroethane dermal route. Use of an oral POD for dermal extrapolation may not be preferred for chemicals known to undergo extensive liver metabolism because the “first-pass effect” that directs intestinally absorbed chemicals to the liver applies to oral ingestion. However, PBPK research also indicates extra-hepatic metabolism for 1,2-dichloroethane. The accuracy of extrapolation of inhalation toxicity data for dermal PODs is dependent on assumptions about inhalation exposure factors such as breathing rate and any associated dosimetric adjustments. However, whole-body inhalation studies may also already be incorporating some level of dermal exposure. Given these uncertainties, in the absence of 1,1-dichloroethane data to support derivation of a dermal CSF from an oral CSF or an inhalation IUR, a dermal CSF was not derived.

N.11 Summary of Continuous and Worker PODs

The continuous IUR was adjusted for occupational scenarios using equations provided in Appendix 1.1.1.1.1N.4.1.5. Table_Apx N-46 provides a summary of the cancer PODs for both continuous and occupational exposure scenarios.

Table_Apx N-46. Summary of Cancer PODs for 1,1-Dichloroethane (Read-Across from 1,2-Dichloroethane)

Route	Continuous POD	Worker POD	Reference
Inhalation	6.0E−06 (per µg/m^3)	2.1E−06 (per µg/m^3)	Nagano et al. (2006)
Oral	6.2E−02 (per mg/kg-bw/day)	Same as continuous	NTP (1978)
Dermal	6.2E−02 (per mg/kg-bw/day)	Same as continuous	Route-to-route extrapolation from oral

N.12 Human Health Hazard Confidence Summary

Table_Apx N-47 presents a summary of confidence for each hazard endpoint and relevant exposure duration based on critical human health hazards considered for the acute, intermediate, chronic, and lifetime exposure scenarios used to calculate risks.

Table_Apx N-47. Confidence Summary for Human Health Hazard Assessment

Hazard Domain	Evidence Integration Conclusion	Selection of Most Critical Endpoint and Study	Relevance to Exposure Scenario	Dose-Response Considerations	PESS Sensitivity	Overall Hazard Confidence
Acute non-cancer						
Oral						
Kidney	+++	+++	+++	++	++	Robust
Inhalation						
Olfactory effects[a]	+++	+++	+++	++	+++	Robust
Intermediate non-cancer						
Oral						
Kidney	+++	+++	+++	++	+++	Robust
Inhalation						
Reproductive[b]	+++	+++	+++	++	+++	Robust
Chronic non-cancer						
Oral						
Kidney	+++	+++	++	++	+++	Robust
Inhalation						
Reproductive[b]	+++	+++	++	++	+++	Robust
Cancer						
Cancer[c]	++	+++	+++	++	+++	Moderate

+ + + Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the hazard estimate.
+ + Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize hazard estimates.
+ Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.
[a] Degeneration with necrosis of olfactory epithelium
[b] Decreased sperm effects
[c] Inhalation based on combined mammary gland adenomas, fibroadenomas, and adenocarcinomas and subcutaneous fibromas

Appendix O OCCUPATIONAL EXPOSURE VALUE DERIVATION

EPA has calculated an 8-hour existing chemical occupational exposure value to summarize the occupational exposure scenario and sensitive health endpoints into a single value. This calculated value may be used to inform risk management efforts for 1,1-dichloroethane under TSCA section 6(a), 15 U.S.C. 2605. EPA calculated the value rounded to 0.044 ppm (0.178 mg/m^3) for inhalation exposures to 1,1-dichloroethane as an 8-hour time-weighted average (TWA) and for consideration in workplace settings (see Appendix O.1) based on the lifetime cancer inhalation unit risk (IUR) for a combined cancer model.

TSCA requires risk evaluations to be conducted without consideration of cost and other non-risk factors, and thus this occupational exposure value represents a risk-only number. In risk management rulemaking for 1,1-dichloroethane following the final risk evaluation, EPA may consider cost and other non-risk factors, such as technological feasibility, the availability of alternatives, and the potential for critical or essential uses. Any existing chemical exposure limit (ECEL) used for occupational safety risk management purposes could differ from the occupational exposure value presented in this appendix based on additional consideration of exposures and non-risk factors consistent with TSCA section 6(c).

This calculated value for 1,1-dichloroethane represents the exposure concentration below which workers and occupational non-users are not expected to exhibit any appreciable risk of adverse toxicological outcomes, accounting for potentially exposed and susceptible populations (PESS). It is derived based on the most sensitive human health effect (*i.e.*, cancer) relative to benchmarks and standard occupational scenario assumptions of 8 hours per day, 5 days per week exposures for a total of 250 days exposure per year and a 40-year working life.

All hazard values used in these calculations are based on non-cancer HECs and associated uncertainty factor derivations and the IUR from this Risk Evaluation for 1,1-Dichloroethane (Section 5.2.6.3).

EPA expects that at the lifetime cancer occupational exposure value of 0.044 ppm (0.178 mg/m^3), a worker or an occupational non-user also would be protected against degeneration with necrosis of the olfactory mucosa and deceases in sperm concentration resulting from acute and intermediate occupational exposures. This calculated lifetime cancer occupational exposure value would protect against excess risk of cancer above the 1×10^{-4} benchmark value resulting from lifetime exposure if ambient exposures are kept below this occupational exposure value. EPA has also separately calculated a short-term occupational exposure value or ceiling limit for 1,1-dichloroethane.

Of the identified occupational monitoring data for 1,1-dichloroethane, there have been measured workplace air concentrations below the calculated exposure value. A summary table of available monitoring methods from the Occupational Safety and Health Administration (OSHA), the National Institute for Occupational Safety and Health (NIOSH), and EPA is included in Appendix O.2. Table_Apx O-1 covers validated methods from governmental agencies and is not intended to be a comprehensive list of available air monitoring methods for 1,1-dichloroethane. The calculated exposure value is above the limit of detection (LOD) and limit of quantification (LOQ) using at least one of the monitoring methods identified.

OSHA has set a permissible exposure limit (PEL; accessed June 11, 2025) as an 8-hour TWA for 1,1-dichloroetane of 100 ppm (Recommended in 1970). As noted on OSHA's website, "OSHA recognizes that many of its permissible exposure limits (PELs) are outdated and inadequate for ensuring protection of worker health. Most of OSHA's PELs were issued shortly after adoption of the Occupational Safety and Health (OSH) Act in 1970 and have not been updated since that time" (Occupational Safety and

Health Administration. Permissible Exposure Limits – Annotated Tables). In addition, OSHA's PEL must undergo both risk assessment and feasibility assessment analyses before selecting a level that will substantially reduce risk under the OSH Act. EPA's calculated exposure value is a lower value and is based on newer information and analysis from this risk evaluation.

Other governmental agencies and independent groups have also set recommended exposure limits established for 1,1-dichloroethane. The American Conference of Governmental Industrial Hygienists (ACGIH) has set a Threshold Limit Value (TLV) at 100 ppm TWA and 100 ppm STEL (confirmed in 1992). This chemical also has a NIOSH Recommended Exposure Limit (REL; accessed June 11, 2025) of 100 ppm TWA (400 mg/m^3) (Recommended in 1992).

NIOSH considers the chloroethanes—ethylene dichloride (1,2-dichloroethane), hexachloroethane, 1,1,2,2-tetrachloroethane, and 1,1,2-trichloroethane—to be potential occupational carcinogens. Additionally, NIOSH recommends that the other five chloroethane compounds (1,1-dichloroethane, ethyl chloride, methyl chloroform, pentachloroethane, and 1,1,1,2-tetrachloroethane) be treated in the workplace with caution because of their structural similarity to the four chloroethanes shown to be carcinogenic in animals.

O.1 Occupational Exposure Value Calculations

This section presents the calculations used to estimate the occupational exposure values using inputs derived in this risk evaluation. Multiple values are presented below for hazard endpoints based on different exposure durations. For 1,1-dichloroethane, the most sensitive occupational exposure value is based on cancer and the resulting 8-hour TWA is rounded to 0.044 ppm. The human health hazard values (HECs, IUR) used in the equations are derived in the risk evaluation for 1,1-dichloroethane.

Lifetime Cancer Occupational Exposure Value

The EV_{cancer} is the concentration at which the extra cancer risk is equivalent to the benchmark cancer risk of 1×10^{-4}:

Equation_Apx O-1.

$$EV_{cancer} = \frac{Benchmark_{Cancer}}{IUR} \times \frac{AT_{IUR}}{ED \times EF \times WY} \times \frac{\mathrm{IR_{resting}}}{\mathrm{IR_{workers}}}$$

$$= \frac{1X10^{-4}}{9.5 \times 10^{-3}\ per\ ppm} \times \frac{24\frac{h}{d} \times \frac{365d}{y} \times 78y}{8\frac{h}{d} \times \frac{250d}{y} \times 40y} \times \frac{0.6125\ \mathrm{m^3}/hr}{1.25\ \mathrm{m^3}/hr}$$

$$= 0.044 \text{ ppm} = 0.179\ \mathrm{mg/m^3}$$

$$\mathrm{EV_{cancer}}\ (mg/m^3) = \frac{EV\ ppm \times MW}{Molar\ Volume} = \frac{0.044\ \text{ppm} \times 98.96\ \frac{g}{mol}}{24.45\ \frac{L}{mol}} = 0.179\ mg/m^3$$

Where:

Molar Volume	=	24.45 L/mol, the volume of a mole of gas at 1 atm and 25 °C
MW	=	Molecular weight of 1,1-dichloroethane (98.96 g/mole)

Chronic Non-Cancer Occupational Exposure Value

The chronic occupational exposure value ($EV_{chronic}$) was calculated as the concentration at which the chronic margin of exposure (MOE) would equal the benchmark MOE for 8-hour chronic occupational exposures with the following equation:

$$EV_{chronic} = \frac{HEC_{chronic,}}{Benchmark\ MOE_{chronic}} \times \frac{AT_{HEC\ chronic}}{ED * EF * WY} \times \frac{IR_{resting}}{IR_{workers}}$$

$$= \frac{22\ ppm}{300} \times \frac{\frac{24h}{d} \times \frac{365d}{y} \times 40\ y \times 0.6125 \frac{m^3}{hr}}{\frac{8h}{d} \times \frac{250d}{y} \times 40\ y \times 1.25 \frac{m^3}{hr}}$$

$$= 0.157\ ppm$$

$$EV_{chronic}\ \left(\frac{mg}{m^3}\right) = \frac{EV\ ppm \times MW}{Molar\ Volume} = \frac{0.157\ ppm \times 98.96 \frac{g}{mol}}{24.45 \frac{L}{mol}} = 0.637\ mg/m^3$$

Intermediate Non-Cancer Occupational Exposure Value

The intermediate occupational exposure value ($EV_{intermediate}$) was calculated as the concentration at which the intermediate MOE would equal the benchmark MOE for intermediate occupational exposure using the following equation:

Equation_Apx O-2.

$$EV_{intermediate} = \frac{HEC_{intermediate}}{Benchmark\ MOE_{intermediate}} \times \frac{AT_{HEC\ intermediate}}{ED \times EF} \times \frac{IR_{resting}}{IR_{workers}}$$

$$= \frac{22\ ppm}{30} \times \frac{\frac{24h}{d} \times 30d}{\frac{8h}{d} \times 22d} \times \frac{0.6125 \frac{m^3}{hr}}{1.25 \frac{m^3}{hr}} = 1.47\ ppm$$

$$EV_{intermediate}\ \left(\frac{mg}{m^3}\right) = \frac{EV\ ppm \times MW}{Molar\ Volume} = \frac{1.47\ ppm \times 98.96 \frac{g}{mol}}{24.45 \frac{L}{mol}} = 5.95\ mg/m^3$$

Acute Non-Cancer Occupational Exposure Value

The acute occupational exposure limit (EV_{acute}) was calculated as the concentration at which the acute MOE would equal the benchmark MOE for acute occupational exposures using the following equation:

Equation_Apx O-3.

$$EV_{acute} = \frac{HEC_{acute}}{Benchmark\ MOE_{acute}} \times \frac{AT_{HEC\ acute}}{ED} \times \frac{IR_{resting}}{IR_{workers}}$$

$$= \frac{10.14\ ppm}{30} \times \frac{24h}{\frac{8h}{d}} * \frac{0.6125\ \frac{m^3}{hr}}{1.25 \frac{m^3}{hr}} = 0.497\ ppm = 2.011\ mg/m^3$$

$$\mathrm{EV_{acute}}\left(\frac{\mathrm{mg}}{\mathrm{m^3}}\right) = \frac{EV\ ppm\ \times\ MW}{Molar\ Volume} = \frac{0.497\ \mathrm{ppm}\ \times\ 98.96\ \frac{g}{mol}}{24.45\ \frac{L}{mol}} =\ 2.011\ mg/m^3$$

Where:

$AT_{HECchronic}$	=	Averaging time for the POD/HEC used for evaluating non-cancer, chronic occupational risk, based on study conditions and/or HEC adjustments (24 hours/day for 365 days/yr) and assuming the number of years matches the high-end working years (WY, 40 years) for a worker
$AT_{HECintermediate}$	=	Averaging time for the POD/HEC used for evaluating non-cancer, intermediate occupational risk, based on study conditions and/or any HEC adjustments (24 hours/day for 30 days)
$AT_{HECacute}$	=	Averaging time for the POD/HEC used for evaluating non-cancer, acute occupational risk, based on study conditions and/or any HEC adjustments (24 hours/day)
AT_{IUR}	=	Averaging time for the cancer IUR, based on study conditions and any adjustments (24 hours/day for 365 days/year) and averaged over a lifetime (78 years)
Benchmark $MOE_{chronic}$	=	Chronic non-cancer benchmark margin of exposure, based on the total uncertainty factor of 300 (Table 5-45)
Benchmark $MOE_{intermediate}$	=	Intermediate non-cancer benchmark margin of exposure, based on the total uncertainty factor of 30 (Table 5-44)
Benchmark MOE_{acute}	=	Acute non-cancer benchmark margin of exposure, based on the total uncertainty factor of 30 (Table 5-43)
$Benchmark_{cancer}$	=	Benchmark for excess lifetime cancer risk
EV_{acute}	=	occupational exposure value based on degeneration with necrosis of the olfactory mucosa
$EV_{intermediate}$	=	Occupational exposure value based on decrease in sperm concentration
$EV_{chronic}$	=	Occupational exposure value based on decrease in sperm concentration
EV_{cancer}	=	Occupational exposure value based on excess cancer risk
ED	=	Exposure duration (8 hours/day)
EF	=	Exposure frequency (250 days/year)
$HEC_{acute,\ intermediate,\ or\ chronic}$	=	Human equivalent concentration for acute, intermediate, or chronic occupational exposure scenarios (Table 5-43, Table 5-44, and Table 5-45)
IUR	=	Inhalation unit risk (per ppm) (Table 5-48)
IR	=	Inhalation rate (default is 1.25 m^3/hr for workers and 0.6125 m^3/hr for the general population at rest)
WY	=	Working years per lifetime at the 95th percentile (40 years)

Unit conversion:

1 ppm = 4.05 mg/m^3 (based on the molecular weight of 98.96 g/mol for 1,1-dichloroethane)

O.2 Summary of Air Sampling Analytical Methods Identified

EPA conducted a search to identify relevant NIOSH, OSHA, and EPA analytical methods used to monitor for the presence of 1,1-dichloroethane in air (see Table_Apx O-1). This table covers validated methods from governmental agencies and is not intended to be a comprehensive list of available air

monitoring methods for 1,1-dichloroethane. The sources used for the search included the following (all access dates June 11, 2025):

1. NIOSH Manual of Analytical Methods (NMAM); 5th Edition
2. NIOSH NMAM 4th Edition
3. OSHA Index of Sampling and Analytical Methods
4. EPA Environmental Test Method and Monitoring Information

Table_Apx O-1. Limit of LOD and LOQ Summary for Air Sampling Analytical Methods Identified

Air Sampling Analytical Methods	Year Published	LOD[a]	LOQ	Notes	Source
NIOSH Method 1003	2003	2.0 µg/ sample	5.1 µg/ sample	The working range is 4 to 250 ppm at 15 L.	NIOSH NMAM, 4th Edition
OSHA Method 07[b]	1979 (last update: 2000)	N/A	N/A	The estimated detection limit is based on the lowest mass per sample injected as a standard.	OSHA Index of Sampling and Analytical Methods

ppm = parts per million; ppb = parts per billion; ppt = parts per trillion
All access dates June 11, 2025.
[a] These sources cover a range of LOD including both below and above the ECEL value.
[b] This method has been withdrawn and is provided for historical record only.

O.3 Summary of 1,1-Dichloroethane Air Sampling from Test Order

In response to a test order, the Vinyl Institute's Consortium submitted a Draft Final Study Plan (DFSP) that was then reviewed by EPA. After addressing EPA's comments, the Consortium submitted a revised DFSP that was approved by EPA in February 2023. The approved DFSP included the use of a modified NIOSH 1003 method capable of detecting below EPA's Occupational Exposure Values. The analytical method recommended in the Test Order, NIOSH 1003, utilizes gas chromatography (GC), flame ionizer detector (FID) technique for analysis of samples. The working range of NIOSH 1003 for 1,1-dichloroethane is 4 to 250 parts per million (ppm) (4,000–250,000 ppb), which is significantly higher than EPA's provisional occupational exposure limit (pOEL) for 1,1-dichloroethane of 250 ppb (0.25 ppm). To allow for a comparison to this value, a validated method of sample analysis using a more sensitive analytical technique, gas chromatography with mass spectroscopy (GC/MS) was developed. The laboratory method validation report is included in Appendix K of the Test Order Inhalation Monitoring Data Package (EPA-HQ-OPPT-2024-0114-0040). The sampling methodology that was used were the Assay Technology 525 TraceAir® II (AT525) activated charcoal passive badges and validation was performed to confirm that this media would result in similar performance as compared to the sorbent tube method recommended in NIOSH 1003.

The 1,1-dichloroethane inhalation monitoring was conducted from July 17 through October 18, 2023. A total of 163 full-shift samples and 81 task length samples across SEGs were collected at 4 facilities from 3e different companies of the Vinyl Institute's Consortium. At the facilities that manufacture 1,1-dichloroethane as an isolated product for use as an intermediate, 63 full-shift samples and 36 task length samples were collected.

In December 2023, the Consortium submitted a final study report with the data requested by the Test Order that was reviewed and accepted by EPA. Of the 63 full-shift samples, 3 were non-detect for a percent non-detect of 4.76 percent. Validation results showed acceptable media and GC/MS method

performance for 1,1-dichloroethane over the concentration range evaluated. The limit of quantification (LOQ) for the modified NIOSH 1003 Method is below EPA's proposed Occupational Exposure Value (see Table_Apx O-2) and was well below the original NIOSH 1003 method as presented above in Table_Apx O-1.

Table_Apx O-2. Overview and Comparison of OEV, LOD, and LOQ Parameters of NIOSH 1003 Modified

Parameter	Value	Unit
Occupational Exposure Value (OEV)	0.044	ppm
Limit of detection (LOD)	3.5	ng/sample
Limit of quantification (LOQ)	13	ng/sample
	0.33	µg/m^3
	3.3E−04	mg/m^3
	8.2E−02	ppb
	8.2E−05	ppm

Appendix P 1,1-DICHLOROETHANE CONDITIONS OF USE

P.1 Additions and Name Changes to Conditions of Use Based on Updated 2020 CDR Reported Data

After the final scope (U.S. EPA, 2020a), EPA received updated submissions under the 2020 CDR reported data. In addition to new submissions received under the 2020 CDR, the reporting name codes change for the 2020 CDR reporting cycle. EPA's review of 2020 CDR reporting did not result in any changes to the COUs in this risk evaluation.

P.2 Changes to Conditions of Use Table

When developing the draft risk evaluation, EPA concluded that an additional subcategory of the conditions of use listed in the final scope (U.S. EPA, 2020a) was needed. EPA added the COU processing – repackaging to account for the repackaging for distribution of 1,1-dichloroethane for use as a laboratory chemical. Table_Apx P-1 summarizes the change to the COU subcategory descriptions.

Table_Apx P-1. Subcategory Editing from the Final Scope Document to the Risk Evaluation

Life Cycle Stage and Category	Original Subcategory in the Final Scope Document	Occurred Change	Revised Subcategory in the 2024 Risk Evaluation
Processing	N/A	Added "Processing: Repackaging" subcategory	Processing: Repackaging

P.3 Descriptions of 1,1-Dichloroethane Conditions of Use

The following descriptions are intended to include examples of uses, so as not to exclude other activities that may also be included in the COUs of the chemical substance. To better describe the COU, EPA considered CDR submissions from the last two CDR cycles for 1,1-dichloroethane (CASRN 75-34-3), and the COU descriptions reflect what the Agency identified as the best fit for that submission. Examples of products, or activities are included in the following descriptions to help describe the COU but are not exhaustive. EPA uses the term "products" in the following descriptions and is generally referring to products as defined by 40 CFR part 751.5.

P.3.1 Manufacturing – Domestic Manufacturing

Domestic manufacture means to manufacture or produce 1,1-dichloroethane within the Unites States. For purposes of the 1,1-dichloroethane risk evaluation, this includes the production of 1,1-dichloroethane and loading and repackaging (but not transport) associated with the manufacturing and/or production of 1,1-dichloroethane. 1,1-Dichloroethane can be manufactured by chlorination of ethane or chloroethane, via thermal chlorination, photochlorination, or oxychlorination. Alternatively, 1,1-dichloroethane can be produced by adding hydrogen chloride to acetylene. This risk evaluation does not include the manufacture of 1,1-dichloroethane as a byproduct during the manufacture of 1,2-dichloroethane (that exposure will be assessed in the risk evaluation for 1,2-dichloroethane). Examples of CDR submissions are provided below:

- In the 2016 CDR cycle, two CDR companies reported domestic manufacturing of 1,1-dichloroethane.
- In the 2020 CDR cycle, two CDR companies reported domestic manufacturing of 1,1-dichloroethane.

P.3.2 Processing – As a Reactant

Processing as a reactant or intermediate is the use of 1,1-dichloroethane as a feedstock in the production of another chemical via a chemical reaction in which 1,1-dichloroethane is consumed to form the product, which is then distributed in commerce.

P.3.2.1 Intermediate in All Other Basic Organic Chemical Manufacture

This COU refers to the use of a chemical substance in chemical reactions for the manufacturing of another chemical substance or product. In this case, 1,1-dichloroethane is used as an intermediate in all other basic organic chemical manufacture. This COU includes the use of 1,1-dichloroethane as an intermediate for the manufacture of chlorinated solvents, mainly 1,1,1-trichloroethane, 1,2-dichloroethane, and vinyl chloride. This COU also includes activities identified by the U.S. Department of Defense. Examples of CDR submissions are provided below:

- In the 2016 CDR cycle, one CDR company reported processing as a reactant as an intermediate in all other basic organic chemical manufacturing of 1,1-dichloroethane.
- In the 2020 CDR cycle, one CDR company reported processing as a reactant as an intermediate in all other basic organic chemical manufacturing of 1,1-dichloroethane

P.3.2.2 Intermediate in All Other Chemical Product and Preparation Manufacturing

This COU refers to the use of a chemical substance in chemical reactions for the manufacturing of another chemical substance or product. In this case, 1,1-dichloroethane is used as an intermediate in all other chemical product and preparation manufacturing. This COU includes the use of 1,1-dichloroethane as chlorinated solvent intermediate. This COU also includes activities identified by the U.S. Department of Defense. Examples of CDR submissions are provided below:

- In the 2016 CDR cycle, one CDR company reported processing as a reactant as an intermediate in all other chemical product and preparation manufacturing of 1,1-dichloroethane.
- In the 2020 CDR cycle, one CDR company reported processing as a reactant as an intermediate in all other chemical product and preparation manufacturing of 1,1-dichloroethane.

P.3.3 Processing – Repackaging

Repackaging refers to preparation of 1,1-dichloroethane for distribution into commerce in a different form, state, or quantity than originally received or stored including chemical product and preparation manufacturing, wholesale and retail trade, and laboratory chemicals manufacturing. This COU includes transferring 1,1-dicloroethane from a bulk storage container into smaller containers. This COU would not apply to the relabeling or redistribution of a chemical substance without removing the chemical substance from the original container it was supplied in.

Repackaging 1,1-dichloroethane as a laboratory chemical was not reported in the 2016 or 2020 reporting cycles. However, EPA identified products containing 1,1-dichloroethane sold as a liquid for research purposes only and not intended for use as drugs, food additives, households, or pesticides (Sigma-Aldrich, 2020).

P.3.4 Processing – Recycling

This COU refers to the process of treating generated waste streams (*i.e.*, which would otherwise be disposed of as waste), containing 1,1-dichloroethane that are collected, either on-site or transported to a third-party site, for commercial purpose. Examples of CDR submissions are provided below:

- In the 2016 CDR cycle, one CDR company reported processing recycling of 1,1-dichloroethane and claimed it as confidential business information, the Agency reviewed the claim and secured their waived claim in 2020.
- In the 2020 CDR cycle, one CDR company claimed processing recycling of 1,1-dichloroethane as confidential business information.

P.3.5 Distribution in Commerce

For purposes of assessment in this risk evaluation, distribution in commerce consists of the transportation associated with the moving of 1,1-dichloroethane or 1,1-dichloroethane-containing products between sites manufacturing or processing 1,1-dichloroethane or 1,1-dichloroethane-containing products, or to final use sites, or for final disposal of 1,1-dichloroethane or 1,1-dichloroethane-containing products. More broadly under TSCA, "distribution in commerce" and "distribute in commerce" are defined under TSCA section 3(5).

P.3.6 Commercial Use in Laboratory Chemicals

This COU refers to the use of 1,1-dichloroethane in laboratory chemicals, such as a chemical standard or reference material during analysis. A commenter (EPA-HQ-OPPT-2018-0426-0026) provided descriptions of their use of 1,1- dichloroethane in analytical standard, research, equipment calibration and sample preparation applications, including reference sample for analysis of terrestrial and extraterrestrial material samples.

This use was not reported to EPA in the 2016 or 2020 CDR cycles and is expected to be below the reporting threshold.

P.3.7 Disposal

Each of the COUs of 1,1-dichloroethane may generate waste streams of the chemical.
For purposes of the 1,1-dichloroethane risk evaluation, this COU refers to the 1,1-dichloroethane in a waste stream that is collected from facilities and commercial sites and is unloaded at and treated or disposed at third-party sites. This COU also encompasses 1,1-dichloroethane contained in wastewater discharged by occupational users to a POTW or other, non-POTW for treatment, as well as other wastes. 1,1-Dichloroethane is expected to be released to other environmental media, such as introductions of biosolids to soil or migration to water sources, through waste disposal (*e.g.*, disposal of formulations containing 1,1-dichloroethane or transport containers). Disposal may also include destruction and removal by incineration. Additionally, 1,1-dichloroethane has been identified in EPA's 2016 report, *Hydraulic Fracturing for Oil and Gas: Impacts from the Hydraulic Fracturing Water Cycle on Drinking Water Resources in the United States* (EPA-600-R-16-236Fb), to be a chemical reported to be detected in produced water, which is subsequently disposed. Recycling of 1,1-dichlrooethane and 1,1-dichloroethane-containing products is considered a different COU. Environmental releases from manufacturing and processing sites that treat or dispose onsite waste are assessed in each COU.

Appendix Q CONFIDENCE STATEMENTS

The following tables provide a visualization of the confidence and weight of scientific evidence for the elements of the risk evaluation by OES.

Table_Apx Q-1. Confidence and Weight of Scientific Evidence per OES for 1,1-Dichloroethane Concentration in Media

OES	Media	Confidence for Releases	Measured/ Monitoring Confidence Level	Modeling/ Estimation Confidence Level	Measured/ Modeling Comparison	Overall Confidence
Manufacturing of 1,1-dichloroethane as an isolated intermediate	Ambient air	Moderate to robust	++	+++	++	Robust
	Indoor air	Moderate to robust	+	++	+	Moderate
	Surface water	Moderate to Robust	++	+++	++	Robust
	Land	Moderate to Robust	+	++	N/A	Moderate
Processing as a reactive intermediate	Ambient air	Moderate to Robust	++	+++	++	Robust
	Indoor air	Moderate to robust	+	++	+	Moderate
	Surface water	Moderate to Robust	++	+++	++	Robust
	Land	Moderate to Robust	+	++	N/A	Moderate
Processing – repackaging	Ambient air	Moderate to Robust	++	+++	++	Robust
	Surface water	Moderate to Robust	++	+++	++	Robust
	Land	Moderate to Robust	+	++	N/A	Moderate
Commercial use as a lab chemical	Ambient air	Moderate	–	++	N/A	Moderate
	Surface water	Moderate	–	++	N/A	Moderate
	Land	Moderate	–	++	N/A	Moderate
General waste handling, treatment, and disposal	Ambient air	Moderate to Robust	++	+++	++	Robust
	Indoor air	Moderate to robust	+	++	+	Moderate
	Surface water	Moderate to Robust	++	+++	++	Robust
	Land	Moderate to Robust	+	++	N/A	Moderate
Waste handling, treatment, and disposal (POTW)	Surface water	Moderate to Robust	++	+++	++	Robust
	Land	Moderate to Robust	+	++	N/A	Moderate
Waste handling, treatment, and	Surface water	Moderate to Robust	++	+++	++	Robust

OES	Media	Confidence for Releases	Measured/ Monitoring Confidence Level	Modeling/ Estimation Confidence Level	Measured/ Modeling Comparison	Overall Confidence
disposal (remediation)	Land	Moderate to Robust	+	++	N/A	Moderate

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

Table_Apx Q-2. 1,1-Dichloroethane Evidence Table Summarizing the Overall Confidence Derived from Hazard Thresholds

Types of Evidence	Quality of the Database	Consistency	Strength and Precision	Biological Gradient/Dose-Response	Relevance[a]	Hazard Confidence
Aquatic						
Acute aquatic assessment	+++	+++	+++	+++	++	Robust
Acute benthic assessment	++	++	+++	+++	++	Moderate
Chronic aquatic assessment	++	++	+++	+++	+++	Robust
Chronic benthic assessment	++	++	+++	+++	+	Moderate
Algal assessment	++	++	+++	++	++	Moderate
Terrestrial						
Chronic mammalian assessment	++	++	++	+++	++	Moderate
Avian assessment	NA[b]	NA	NA	NA	NA	Indeterminate[c]
Soil invertebrate assessment	NA[b]	NA	NA	NA	NA	Indeterminate[c]
Terrestrial plant assessment	+	+	++	++	+	Slight

[a] Relevance includes biological, physical/chemical (including use of analogs), and environmental relevance.
+++ Robust confidence suggests thorough understanding of the scientific evidence and uncertainties. The supporting weight of scientific evidence outweighs the uncertainties to the point where it is unlikely that the uncertainties could have a significant effect on the hazard estimate.
++ Moderate confidence suggests some understanding of the scientific evidence and uncertainties. The supporting scientific evidence weighed against the uncertainties is reasonably adequate to characterize hazard estimates.
+ Slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when the assessor is making the best scientific assessment possible in the absence of complete information. There are additional uncertainties that may need to be considered.
[b] NA indicates that a slight, moderate, or robust confidence cannot be assigned due to the lack of reasonably available data.
[c] Indeterminate is noted when a hazard confidence cannot be assigned to an assessment.

Table_Apx Q-3. Evidence Table Summarizing Overall Confidence for Environmental Risk Characterization

Types of Evidence	Exposure	Hazard	Trophic Transfer	Risk Characterization RQ Inputs
Aquatic				
Acute aquatic assessment	+++	+++	N/A	Robust
Acute benthic assessment	+++	++	N/A	Moderate
Chronic aquatic assessment	+++	+++	N/A	Robust
Chronic benthic assessment	+++	++	N/A	Moderate
Algal assessment	+++	++	N/A	Moderate
Terrestrial				
Chronic avian assessment	N/A	N/A	N/A	Indeterminate
Chronic mammalian assessment (air deposition to soil)	++	++	++	Moderate
Chronic mammalian assessment (biosolids to soil)	++	++	++	Moderate
Chronic mammalian assessment (surface water)	+++	++	++	Moderate
Chronic mammalian assessment (benthic pore water)	+++	++	+	Moderate
Soil invertebrate assessment	N/A	N/A	N/A	Indeterminate
Terrestrial plant assessment, air deposition	++	+	N/A	Slight
Terrestrial plant assessment, biosolid deposition	++	+	N/A	Slight

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

Table_Apx Q-4. Weight of Scientific Evidence Conclusions for General Population Exposure Assessments

OES	Route of Exposure	Media	Relevance to Exposure Scenario	Modeling/ Estimation Confidence Level	Measured/ Monitoring Confidence Level[a]	Measured/ Modeling Comparison	WOSE
Manufacturing of 1,1-dichloroethane as an isolated intermediate	Inhalation	Ambient air	+++	+++	++	++	Robust
	Inhalation	Indoor air	++	++	++	+	Moderate
	Oral/ Ingestion	Drinking water	+++	+++	++	++	Robust
	Oral/Fish Ingestion	Surface water	+++	+++	++	++	Robust
	Oral/ Ingestion	Surface water/ swimming	++	++	++	++	Moderate
	Oral/ Ingestion	Soil (biosolids)	++	++	–	N/A	Slight
	Oral/ Ingestion	Land; soil (air deposition)	++	++	–	N/A	Slight
	Dermal	Swimming	++	++	++	+	Moderate
Processing as a reactive intermediate	Inhalation	Ambient air	+++	+++	++	++	Robust
	Inhalation	Indoor air	++	++	++	+	Moderate
	Oral/ Ingestion	Drinking water	+++	+++	++	++	Robust
	Oral/Fish Ingestion	Surface water	+++	+++	++	++	Robust
	Oral/ Ingestion	Surface water/ swimming	++	++	++	++	Moderate
	Oral/ Ingestion	Soil (biosolids)	++	++	–	N/A	Slight
	Oral/ Ingestion	Land; soil (air deposition)	++	++	–	N/A	Slight
	Dermal	Swimming	++	++	++	+	Moderate
Processing – repackaging	Inhalation	Ambient air	+++	+++	++	++	Robust
	Oral/ Ingestion	Drinking water	+++	+++	++	++	Robust
	Oral/ Fish Ingestion	Surface water	+++	+++	++	++	Robust
	Oral/ Ingestion	Surface water/ swimming	++	++	++	++	Moderate
	Oral/ Ingestion	Soil (biosolids)	++	++	–	N/A	Slight
	Oral/ Ingestion	Land; soil (air deposition)	++	++	–	N/A	Slight
	Dermal	Swimming	++	++	++	+	Moderate
	Inhalation	Ambient air	+++	+++	++	++	Robust

OES	Route of Exposure	Media	Relevance to Exposure Scenario	Modeling/ Estimation Confidence Level	Measured/ Monitoring Confidence Level[a]	Measured/ Modeling Comparison	WOSE
Commercial use as a lab chemical	Oral/ Ingestion	Drinking water	+++	+++	++	++	Robust
	Oral/ Fish Ingestion	Surface water	+++	+++	++	++	Robust
	Oral/ Ingestion	Surface water/ swimming	++	++	++	++	Moderate
	Oral/ Ingestion	Soil (biosolids)	++	++	–	N/A	Slight
	Oral/ Ingestion	Land; soil (air deposition)	++	++	–	N/A	Slight
	Dermal	Swimming	++	++	++	+	Moderate
General waste handling, treatment, and disposal	Inhalation	Ambient air	+++	+++	++	++	Robust
	Inhalation	Indoor air	++	++	++	+	Moderate
	Oral/ Ingestion	Drinking water	+++	+++	++	++	Robust
	Oral/ Fish Ingestion	Surface water	+++	+++	++	++	Robust
	Oral/ Ingestion	Surface water/ swimming	++	++	++	++	Moderate
	Oral/ Ingestion	Soil (biosolids)	++	++	–	N/A	Slight
	Oral/ Ingestion	Land; soil (air deposition)	++	++	–	N/A	Slight
	Dermal	Swimming	++	++	++	+	Moderate
General waste handling, treatment and disposal (POTW)	Oral/ Ingestion	Drinking water	+++	+++	++	++	Robust
	Oral/ Fish Ingestion	Surface water	+++	+++	++	++	Robust
	Oral/ Ingestion	Surface water/ swimming	++	++	++	++	Moderate
	Oral/ Ingestion	Soil (biosolids)	++	++	–	N/A	Slight
	Oral/ Ingestion	Land; soil (air deposition)	++	++	–	N/A	Slight
	Dermal	Swimming	++	++	++	+	Moderate
General waste handling, treatment and disposal (Remediation)	Oral/ Ingestion	Drinking water	+++	+++	++	++	Robust
	Oral/ Fish Ingestion	Surface water	+++	+++	++	++	Robust
	Oral/ Ingestion	Surface water/ swimming	++	++	++	++	Moderate

OES	Route of Exposure	Media	Relevance to Exposure Scenario	Modeling/ Estimation Confidence Level	Measured/ Monitoring Confidence Level[a]	Measured/ Modeling Comparison	WOSE
	Oral/ Ingestion	Soil (biosolids)	++	++	–	N/A	Slight
	Oral/ Ingestion	Land; soil (air deposition)	++	++	–	N/A	Slight
	Dermal	Swimming	++	++	++	+	Moderate

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

Table_Apx Q-5. Overall Confidence for Acute, Intermediate, and Chronic Human Health Non-Cancer Risk Characterization for COUs Resulting in Risks[a] [b]

COU			Exposure Route/ Exposed Group	Exposure Confidence	Hazard Confidence	Risk Characterization Confidence
Life Cycle Stage	**Category**	**Subcategory**				
Occupational						
Manufacturing/ Domestic Manufacturing	Domestic manufacturing	Manufacturing	Inhalation/Worker (operator/process technician)	+++	++	+++
			Inhalation/Worker (maintenance technician)	+++	++	+++
			Dermal/Worker	+++	++	+++
Processing/ As a Reactant	Intermediate in all other basic organic chemical manufacturing/intermediate in all other chemical product and preparation manufacturing/recycling	Processing as reactive intermediate	Inhalation/Worker	++	++	+++
			Dermal/Worker	++	++	+++
Processing/ Processing – Repackaging	Processing – repackaging	Processing – repackaging	Inhalation/Worker	+	++	+
			Inhalation/ONU	+	++	+
			Dermal/Worker	++	++	+++
Commercial Use/Laboratory Chemicals	Laboratory chemicals reference material	Commercial use as a laboratory chemical	Dermal/Worker	++	++	+++
Disposal	Disposal	General waste handling, treatment, and disposal	Inhalation/Worker	+	++	+
			Dermal/Worker	++	++	+++
Disposal	Disposal	Waste handling, treatment, and disposal (WWT)	Inhalation/Worker	+	++	+
			Dermal/Worker	++	++	+++

[a] This table identifies COUs that have any non-cancer risk (acute, intermediate, or chronic) and the route associated with the risk.
[b] Intermediate risks were evaluated for workers only and not the general population.

Table_Apx Q-6. Overall Confidence for Lifetime Human Health Cancer Risk Characterization for COUs Resulting in Risks

COUs			Exposure Route/Exposed Group	Exposure Confidence	Hazard Confidence	Risk Characterization Confidence
Life Cycle Stage	**Category**	**Subcategory**				
Occupational						
Manufacturing/ Domestic Manufacturing	Domestic manufacturing	Manufacturing	Inhalation/Worker (operator/process technician)	+++	++	+++
			Inhalation/Worker (maintenance technician)	+++	++	+++
			Dermal/Worker	NE	NE	NE
Processing/ as a Reactant	Intermediate in all other basic organic chemical manufacturing/intermediate in all other chemical product and preparation manufacturing/recycling	Processing as reactive intermediate	Inhalation/Worker	++	++	+++
			Dermal/Worker	NE	NE	NE
Processing/ Processing – Repackaging	Processing – repackaging	Processing – repackaging	Inhalation/Worker	++	++	+++
			Inhalation/ONU	++	++	+++
			Dermal/Worker	NE	NE	NE
Commercial Use/Laboratory Chemicals	Laboratory chemicals reference material	Commercial use as a laboratory chemical	Dermal/Worker	NE	NE	NE
Disposal	Disposal	General waste handling, treatment, and disposal	Inhalation/Worker	++	++	+++
			Dermal/Worker	NE	NE	NE
Disposal	Disposal	Waste handling, treatment, and disposal (WWT)	Inhalation/Worker	++	++	+++
			Dermal/Worker	NE	NE	NE

NE = not estimated
Dermal cancer risk was not estimated as dermal cancer numbers for 1,1-dichloroethane were not derived.