United States
Environmental Protection Agency

December 2025
Office of Chemical Safety and
Pollution Prevention

Response to Public Comments on the Draft Risk Evaluations for Butyl Benzyl Phthalate (BBP); Dibutyl Phthalate (DBP); Dicyclohexyl Phthalate (DCHP); Diethylhexyl Phthalate (DEHP); and Diisobutyl Phthalate (DIBP)

EPA-HQ-OPPT-2018-0501; EPA-HQ-OPPT-2018-0503; EPA-HQ-OPPT-2018-0504; EPA-HQ-OPPT-2018-0433; EPA-HQ-OPPT-2018-0434; EPA-HQ-OPPT-2022-0918; and EPA-HQ-OPPT-2024-0551

Comment Summaries and Responses

December 2025

TABLE OF CONTENTS

LIST OF TABLES

ABBREVIATIONS AND ACRONYMS

7Q10	Lowest 7-day average flow that occurs (on average) once every 10 years
30Q5	Lowest 30-day average flow that occurs (on average) once every 5 years
ADD	Average daily dose
ADC	Average daily concentration
AGD	Anogenital distance
APDR	Acute potential dose rate
BAF	Bioaccumulation factor
BBP	Butyl benzyl phthalate
BCF	Bioconcentration factor
BLS	Bureau of Labor Statistics (U.S.)
CASRN	Chemical Abstracts Service Registry Number
CBI	Confidential business information
CDC	Centers for Disease Control and Prevention (U.S.)
CDR	Chemical Data Reporting
CEHD	Chemical Exposure Health Data
CEM	Consumer Exposure Model
CFR	Code of Federal Regulations
ChV	Chronic value
COC	Concentration of concern
CPSC	Consumer Product Safety Commission (U.S.)
CRA	Cumulative risk assessment
DBP	Dibutyl phthalate
DCHP	Dicyclohexyl phthalate
DEHP	Diethylhexyl phthalate
DIBP	Diisobutyl phthalate
DIDP	Diisodecyl phthalate
DINP	Dicyclohexyl phthalate
DIY	Do-it-yourself
ECJRC	European Commission's Joint Research Centre
EPA	Environmental Protection Agency (U.S.)
ESD	Emission scenario document

EU	European Union
FDA	Food and Drug Administration (U.S.)
GS	Generic scenario
K_{OC}	Soil organic carbon: water partitioning coefficient
K_{OW}	Octanol: water partition coefficient
HEC	Human equivalent concentration
HED	Human equivalent dose
IADD	Intermediate average daily dose
IIOAC	Integrated Indoor-Outdoor Air Calculator (Model)
IR	Ingestion rate
LCD	Life cycle diagram
LOEC	Lowest-observed-effect concentration
LOAEL	Lowest-observed-adverse-effect level
LOD	Limit of detection
Log K_{OC}	Logarithmic organic carbon: water partition coefficient
Log K_{OW}	Logarithmic octanol: water partition coefficient
MBP	Monobutyl phthalate
MBzP	Monobenzyl phthalate
MOA	Mode of action
MOE	Margin of exposure
NAICS	North American Industry Classification System
NHANES	National Health and Nutrition Examination Survey
NICNAS	National Industrial Chemicals Notification and Assessment Scheme
NOEL	No-observed-effect level
NOAEL	No-observed-adverse-effect level
NPDES	National Pollutant Discharge Elimination System
OCSPP	Office of Chemical Safety and Pollution Prevention (EPA)
OECD	Organisation for Economic Co-operation and Development
OEL	Occupational exposure limits
OES	Occupational exposure scenario
OEV	Occupational exposure value
ONU	Occupational non-user
OPPT	Office of Pollution Prevention and Toxics (EPA)
OSHA	Occupational Safety and Health Administration
PBZ	Personal breathing zone
PESS	Potentially exposed or susceptible subpopulations
PND	Postnatal day
PNOR	Particulates not otherwise regulated
POD	Point of departure
PSC	Point Source Calculator tool (for VVWM)
PV	Production volume
PVC	Polyvinyl chloride
RPF	Relative potency factor
RQ	Risk quotient
SACC	Science Advisory Committee on Chemicals
SDS	Safety data sheet
SOC	Standard occupational classification
SpERC	Specific emission release category
SSD	Species sensitivity distribution

SWC	Surface water concentration
TRI	Toxic Release Inventory
TRV	Toxicity reference value
TSCA	Toxic Substances Control Act
TSD	Technical support document
TWA	Time-weighted average
UF	Uncertainty factor
U.S.	United States
VVWM	Variable Volume Water Model
WWTP	Wastewater treatment plant

INTRODUCTION

Between January 2025 and August 2025, U.S. Environmental Protection Agency (EPA or the Agency) released for public comment and peer review draft risk evaluations and technical support documents for five phthalates: butyl benzyl phthalate (BBP), dibutyl phthalate (DBP), dicyclohexyl phthalate (DCHP), diethylhexyl phthlate (DEHP), and diisobutyl phthalate (DIBP).

On January 7, 2025 the EPA published the *Draft Risk Evaluation for DCHP* and multiple technical support documents (TSDs) and supplemental files (collectively called the "draft risk evaluation package"). EPA accepted public comments on the draft DCHP risk evaluation until May 9, 2025. In January 2025, the Agency also released draft three TSDs each for BBP, DBP, DEHP, and DIBP: draft physical chemistry and fate and transport assessment, draft non-cancer human health hazard assessment, and draft environmental hazard assessment. Public comment on these TSDs were accepted until May 6, 2025. On June 5, 2025, EPA published the *Draft Risk Evaluation for DBP*, the *Draft Risk Evaluation for DEHP*, and multiple TSDs and supplemental files for these chemicals, including the *Draft Revised TSD for the Cumulative Risk Analysis of DEHP, DBP, BBP, DIBP, DCHP, and DINP Under the Toxic Substances Control Act (TSCA)*. EPA accepted public comments on the draft risk evaluations of DBP and DEHP until August 4, 2025. On August 6, 2025, EPA published the *Draft Risk Evaluation for BBP*, the *Draft Risk Evaluation for DIBP* and multiple TSDs and supplemental files for these chemicals. EPA accepted public comments on the draft risk evaluations of BBP and DIBP until October 6, 2025.

The Agency also solicited external peer review input from EPA's Science Advisory Committee on Chemicals (SACC). The SACC final report, published on October 6, 2025, presents discussion and specific recommendations from the SACC on the draft risk evaluations of DBP, DCHP, and DEHP, as well as several TSDs for BBP, DBP, DCHP, DIBP, and DEHP. A virtual public meeting/webinar preparatory meeting was held on July 21, 2025 for SACC reviewers and the public ask clarifying questions of the draft charge questions for peer reviewers. A virtual public meeting/webinar was held from August 4, 2025 to August 8, 2025 for the SACC reviewers to comment on and ask questions regarding the risk evaluation packages, which also included opportunity for the public to comment on these charge questions. All documents submitted for peer review were made available for public comment from June 10, 2025 to July 25, 2025 on this peer review docket, in addition to the public comment periods within the TSCA dockets described above.

Prior to the public release of the draft risk evaluations for BBP, DBP, DCHP, DEHP, and DIBP, the EPA released on February 27, 2023 for public comment the *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act* and the *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substance Control Act*. EPA accepted public comments on these documents until May 12, 2023. The Agency also solicited external peer review input from EPA's SACC. The SACC final report, published on July 12, 2023 presents discussion and specific recommendations from the SACC on these documents outlining proposed approaches to the cumulative assessment of the phthalates under TSCA. A virtual public meeting/webinar was held on May 8, 2023 to May 11, 2023 for SACC reviewers and the public to comment on and ask questions regarding the risk evaluation package and clarity of the draft charge questions for peer reviewers. Peer review feedback from the SACC and public comments on these cumulative assessment documents informed the draft risk evaluations of BBP, DBP, DCHP, DEHP, and DIBP.

Materials on the peer review and risk evaluation packages are available at www.regulations.gov in dockets outlined in Table 1.

A total of 163 public comment submissions were received, as listed in ascending order in Table 2. Note that some docket/comment numbers are skipped because not all submissions were posted to the docket (*e.g.*, duplicates, content not related to the docket). Comments on the selection of peer reviewers and on the *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act* are outside the scope of the response. The Agency has reviewed and considered all public comments received in conjunction with recommendations from the SACC final reports. EPA's Office of Pollution Prevention and Toxics (OPPT) appreciates the valuable input provided by the public and SACC peer reviewers. The input resulted in numerous revisions to the risk evaluation document and several TSDs.

This document summarizes the public comments within the 10 themes listed below—and subsections—which generally align with the organization of the risk evaluation. Responses to external peer review recommendations from the August 2025 SACC according to peer review charge questions are available in *Response to Science Advisory Committee on Chemicals (SACC) Recommendations Regarding Peer Review of the Draft Risk Evaluations of DBP, DEHP, and DCHP and the Technical Support Documents for BBP and DIBP: EPA-HQ-OPPT-2024-0551*. A response to public comment on the *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act* and peer review recommendations from the May 2023 SACC may be published separately at a later date.

Additionally, within each theme, comments that cover similar issues are presented and addressed together across the five phthalates (BBP, DBP, DCHP, DEHP, DIBP). Comments that are specific to a phthalate risk evaluation are addressed independently.

1. Overarching Comments on the Draft Risk Evaluation of BBP, DBP, DCHP, DEHP, and DIBP
2. Chemistry, Fate, and Transport
3. Releases and Concentrations in the Environment
4. Human Exposure Assessment
5. Human Health Hazard Assessment
6. Human Health Risk Characterizations
7. Environmental Risk Assessment
8. Unreasonable Risk Determination
9. Systematic Review
10. Other Comments

Table 1. Index of Relevant Dockets for This Response to Comments Document

Docket Number	Docket Name
EPA-HQ-OPPT-2018-0433	Di-ethylhexyl phthalate (DEHP) (1,2-Benzene- dicarboxylic acid, 1,2- bis(2-ethylhexyl) ester; TSCA Review
EPA-HQ-OPPT-2018-0434	Di-isobutyl phthalate (DIBP) (1,2-Benzene- dicarboxylic acid, 1,2-bis-(2-methylpropyl) ester); TSCA Review
EPA-HQ-OPPT-2018-0501	Butyl benzyl phthalate (BBP) 1,2-Benzene- dicarboxylic acid, 1-butyl 2(phenylmethyl) ester; TSCA Review
EPA-HQ-OPPT-2018-0503	Dibutyl phthalate (DBP) (1,2-Benzene- dicarboxylic acid, 1,2-dibutyl ester); TSCA Review
EPA-HQ-OPPT-2018-0504	Dicyclohexyl phthalate (DCHP); TSCA Review
EPA-HQ-OPPT-2022-0918	Toxic Substances Control Act (TSCA) Science Advisory Committee on Chemicals (SACC) - Notice of Public Meeting and Request for Nominations for a Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer Requested Phthalate
EPA-HQ-OPPT-2024-0551	Science Advisory Committee on Chemicals (SACC) - Peer Review of the Draft Risk Evaluations of Dibutyl phthalate (DBP), Di(2-ethylhexyl) phthalate (DEHP), and Dicyclohexyl phthalate (DCHP), and the Technical Support Documents for Butylbenzyl phthalate (BBP) and Diisobutyl phthalate (DIBP)

Table 2. Index of Public Comment Submissions on the BBP, DBP, DCHP, DEHP, and DIBP Draft Risk Evaluations

Abbreviation	Submission Number	Comment Name
DEHP-0081	EPA-HQ-OPPT-2018-0433-0081	A. Griswold
DEHP-0082	EPA-HQ-OPPT-2018-0433-0082	Karen Fortier
DEHP-0083	EPA-HQ-OPPT-2018-0433-0083	Robin Austin
DEHP-0084	EPA-HQ-OPPT-2018-0433-0084	Heather Hankins
DEHP-0085	EPA-HQ-OPPT-2018-0433-0085	Anonymous public comment
DEHP-0086	EPA-HQ-OPPT-2018-0433-0086	Environmental Defense Fund (EDF)
DEHP-0119	EPA-HQ-OPPT-2018-0433-0119	Anonymous public comment
DEHP-0120	EPA-HQ-OPPT-2018-0433-0120	B&C Consortia Management, L.L.C. (BCCM)
DEHP-0121	EPA-HQ-OPPT-2018-0433-0121	Earthjustice et al.
DEHP-0123	EPA-HQ-OPPT-2018-0433-0123	Earthjustice et al.
DEHP-0124	EPA-HQ-OPPT-2018-0433-0124	Earthjustice et al. (Part 1 of 7)
DEHP-0125	EPA-HQ-OPPT-2018-0433-0125	Earthjustice et al. (Part 2 of 7)
DEHP-0126	EPA-HQ-OPPT-2018-0433-0126	Earthjustice et al. (Part 3 of 7)
DEHP-0127	EPA-HQ-OPPT-2018-0433-0127	Earthjustice et al. (Part 4 of 7)
DEHP-0128	EPA-HQ-OPPT-2018-0433-0128	Earthjustice et al. (Part 5 of 7)
DEHP-0129	EPA-HQ-OPPT-2018-0433-0129	Earthjustice et al. (Part 6 of 7)
DEHP-0130	EPA-HQ-OPPT-2018-0433-0130	Earthjustice et al. (Part 7 of 7)
DEHP-0133	EPA-HQ-OPPT-2018-0433-0133	AIHA
DEHP-0134	EPA-HQ-OPPT-2018-0433-0134	AIHA
DEHP-0135	EPA-HQ-OPPT-2018-0433-0135	Di-ethylhexyl Phthalate Consortium
DEHP-0136	EPA-HQ-OPPT-2018-0433-0136	B&C Consortia Management, L.L.C.
DEHP-0137	EPA-HQ-OPPT-2018-0433-0137	Eastman Chemical Company
DEHP-0138	EPA-HQ-OPPT-2018-0433-0138	University of California, San Francisco Program on Reproductive Health and the Environment
DEHP-0139	EPA-HQ-OPPT-2018-0433-0139	Vinyl Institute
DEHP-0140	EPA-HQ-OPPT-2018-0433-0140	Anonymous public comment
DEHP-0141	EPA-HQ-OPPT-2018-0433-0141	Recycled Materials Association (ReMA)
DEHP-0142	EPA-HQ-OPPT-2018-0433-0142	China WTO/TBT National Notification & Enquiry Center, People's Republic of China
DIBP-0088	EPA-HQ-OPPT-2018-0434-0088	Jeff Jones
DIBP-0089	EPA-HQ-OPPT-2018-0434-0089	Anonymous public comment

Abbreviation	Submission Number	Comment Name
DIBP-0090	EPA-HQ-OPPT-2018-0434-0090	Environmental Defense Fund (EDF)
DIBP-0123	EPA-HQ-OPPT-2018-0434-0123	B&C® Consortia Management, L.L.C. (BCCM)
DIBP-0124	EPA-HQ-OPPT-2018-0434-0124	Justin Mills
DIBP-0125	EPA-HQ-OPPT-2018-0434-0125	American Chemistry Council (ACC)
DIBP-0126	EPA-HQ-OPPT-2018-0434-0126	Anonymous public comment
DIBP-0127	EPA-HQ-OPPT-2018-0434-0127	Anonymous public comment
DIBP-0128	EPA-HQ-OPPT-2018-0434-0128	Vinyl Institute (VI)
DIBP-0129	EPA-HQ-OPPT-2018-0434-0129	Aerospace Industries Association (AIA)
DIBP-0130	EPA-HQ-OPPT-2018-0434-0130	Earthjustice et al.
DIBP-0131	EPA-HQ-OPPT-2018-0434-0131	Di-isobutyl Phthalate Consortium
DIBP-0132	EPA-HQ-OPPT-2018-0434-0132	Recycled Materials Association (ReMA)
DIBP-0133	EPA-HQ-OPPT-2018-0434-0133	Earthjustice et al. (Part 01 of 06)
DIBP-0134	EPA-HQ-OPPT-2018-0434-0134	Earthjustice et al. (Part 02 of 06)
DIBP-0135	EPA-HQ-OPPT-2018-0434-0135	Earthjustice et al. (Part 03 of 06)
DIBP-0136	EPA-HQ-OPPT-2018-0434-0136	Earthjustice et al. (Part 04 of 06)
DIBP-0137	EPA-HQ-OPPT-2018-0434-0137	Earthjustice et al. (Part 05 of 06)
DIBP-0138	EPA-HQ-OPPT-2018-0434-0138	Earthjustice et al. (Part 06 of 06)
BBP-0087	EPA-HQ-OPPT-2018-0501-0087	Environmental Defense Fund (EDF)
BBP-0120	EPA-HQ-OPPT-2018-0501-0120	B&C® Consortia Management, L.L.C.
BBP-0121	EPA-HQ-OPPT-2018-0501-0121	American Chemistry Council (ACC)
BBP-0122	EPA-HQ-OPPT-2018-0501-0122	Alliance for Automotive Innovation
BBP-0123	EPA-HQ-OPPT-2018-0501-0123	Butyl Benzyl Phthalate Consortium
BBP-0124	EPA-HQ-OPPT-2018-0501-0124	Earthjustice et al.
BBP-0125	EPA-HQ-OPPT-2018-0501-0125	Earthjustice et al. (Part 01 of 06)
BBP-0126	EPA-HQ-OPPT-2018-0501-0126	Earthjustice et al. (Part 02 of 06)
BBP-0127	EPA-HQ-OPPT-2018-0501-0127	Earthjustice et al. (Part 03 of 06)
BBP-0128	EPA-HQ-OPPT-2018-0501-0128	Earthjustice et al. (Part 04 of 06)
BBP-0129	EPA-HQ-OPPT-2018-0501-0129	Earthjustice et al. (Part 05 of 06)
BBP-0130	EPA-HQ-OPPT-2018-0501-0130	Earthjustice et al. (Part 06 of 06)
DBP-0079	EPA-HQ-OPPT-2018-0503-0079	Environmental Defense Fund (EDF)

Abbreviation	Submission Number	Comment Name
DBP-0113	EPA-HQ-OPPT-2018-0503-0113	B&C Consortia Management, L.L.C. (BCCM)
DBP-0114	EPA-HQ-OPPT-2018-0503-0114	Huntsman
DBP-0115	EPA-HQ-OPPT-2018-0503-0115	Environmental Defense Fund (EDF)
DBP-0116	EPA-HQ-OPPT-2018-0503-0116	Earthjustice et al.
DBP-0118	EPA-HQ-OPPT-2018-0503-0118	Earthjustice et al.
DBP-0119	EPA-HQ-OPPT-2018-0503-0119	Earthjustice et al. (Part 1 of 7)
DBP-0120	EPA-HQ-OPPT-2018-0503-0120	Earthjustice et al. (Part 2 of 7)
DBP-0121	EPA-HQ-OPPT-2018-0503-0121	Earthjustice et al. (Part 3 of 7)
DBP-0122	EPA-HQ-OPPT-2018-0503-0122	Earthjustice et al. (Part 4 of 7)
DBP-0123	EPA-HQ-OPPT-2018-0503-0123	Earthjustice et al. (Part 5 of 7)
DBP-0124	EPA-HQ-OPPT-2018-0503-0124	Earthjustice et al. (Part 6 of 7)
DBP-0125	EPA-HQ-OPPT-2018-0503-0125	Earthjustice et al. (Part 7 of 7)
DBP-0126	EPA-HQ-OPPT-2018-0503-0126	Huntsman International, LLC
DBP-0127	EPA-HQ-OPPT-2018-0503-0127	American Chemistry Council (ACC)
DBP-0128	EPA-HQ-OPPT-2018-0503-0128	AIHA
DBP-0129	EPA-HQ-OPPT-2018-0503-0129	Dibutyl Phthalate Consortium
DBP-0130	EPA-HQ-OPPT-2018-0503-0130	B&C Consortia Management, L.L.C.
DBP-0131	EPA-HQ-OPPT-2018-0503-0131	AIHA
DBP-0132	EPA-HQ-OPPT-2018-0503-0132	B&C® Consortium Management, L.L.C. (BCCM)
DBP-0133	EPA-HQ-OPPT-2018-0503-0133	China WTO/TBT National Notification & Enquiry Center, People's Republic of China
DCHP-0115	EPA-HQ-OPPT-2018-0504-0115	The Adhesive and Sealant Council (ASC)
DCHP-0116	EPA-HQ-OPPT-2018-0504-0116	B&C Consortia Management, L.L.C. (BCCM)
DCHP-0117	EPA-HQ-OPPT-2018-0504-0117	Dane Beito
DCHP-0118	EPA-HQ-OPPT-2018-0504-0118	Anonymous
DCHP-0120	EPA-HQ-OPPT-2018-0504-0120	Curtis Clinton
DCHP-0121	EPA-HQ-OPPT-2018-0504-0121	National Tribal Toxics Council (NTTC)
DCHP-0122	EPA-HQ-OPPT-2018-0504-0122	Environmental Defense Fund (EDF)
DCHP-0123	EPA-HQ-OPPT-2018-0504-0123	SEMI

Abbreviation	Submission Number	Comment Name
DCHP-0124	EPA-HQ-OPPT-2018-0504-0124	The Adhesive and Sealant Council (ASC)
DCHP-0125	EPA-HQ-OPPT-2018-0504-0125	American Chemistry Council (ACC)
DCHP-0126	EPA-HQ-OPPT-2018-0504-0126	B&C Consortia Management, L.L.C. (BCCM)
DCHP-0127	EPA-HQ-OPPT-2018-0504-0127	University of California, San Francisco Program on Reproductive Health and the Environment
DCHP-0128	EPA-HQ-OPPT-2018-0504-0128	Alaska Community Action on Toxics, Center for Environmental Health, Defend Our Health and Earthjustice
DCHP-0129	EPA-HQ-OPPT-2018-0504-0129	Alaska Community Action on Toxics, Center for Environmental Health, Defend Our Health and Earthjustice
DCHP-0130	EPA-HQ-OPPT-2018-0504-0130	Alaska Community Action on Toxics, Center for Environmental Health, Defend Our Health and Earthjustice
DCHP-0131	EPA-HQ-OPPT-2018-0504-0131	Alaska Community Action on Toxics, Center for Environmental Health, Defend Our Health and Earthjustice
DCHP-0132	EPA-HQ-OPPT-2018-0504-0132	Alaska Community Action on Toxics, Center for Environmental Health, Defend Our Health and Earthjustice
DCHP-0133	EPA-HQ-OPPT-2018-0504-0133	Alaska Community Action on Toxics, Center for Environmental Health, Defend Our Health and Earthjustice
DCHP-0134	EPA-HQ-OPPT-2018-0504-0134	Alaska Community Action on Toxics, Center for Environmental Health, Defend Our Health and Earthjustice
DCHP-0135	EPA-HQ-OPPT-2018-0504-0135	Alaska Community Action on Toxics, Center for Environmental Health, Defend Our Health and Earthjustice
SACC23-0018	EPA-HQ-OPPT-2022-0918-0018	B&C® Consortia Management, L.L.C. (BCCM)
SACC23-0019	EPA-HQ-OPPT-2022-0918-0019	North American Metal Council (NAMC)
SACC23-0023	EPA-HQ-OPPT-2022-0918-0023	Dibutyl Phthalate Consortium
SACC23-0024	EPA-HQ-OPPT-2022-0918-0024	Butyl Benzyl Phthalate Consortium
SACC23-0027	EPA-HQ-OPPT-2022-0918-0027	E. Cronan
SACC23-0028	EPA-HQ-OPPT-2022-0918-0028	Lara Adler
SACC23-0029	EPA-HQ-OPPT-2022-0918-0029	Patrick Masseo

Abbreviation	Submission Number	Comment Name
SACC23-0030	EPA-HQ-OPPT-2022-0918-0030	American Chemistry Council (ACC)
SACC23-0031	EPA-HQ-OPPT-2022-0918-0031	SciPinion
SACC23-0032	EPA-HQ-OPPT-2022-0918-0032	Earthjustice et al.
SACC23-0034	EPA-HQ-OPPT-2022-0918-0034	American Water Works Association (AWWA)
SACC23-0035	EPA-HQ-OPPT-2022-0918-0035	Earthjustice et al.
SACC23-0036	EPA-HQ-OPPT-2022-0918-0036	Styrene Information and Research Center (SIRC)
SACC23-0037	EPA-HQ-OPPT-2022-0918-0037	Society of Chemical Manufacturers & Affiliates (SOCMA)
SACC23-0038	EPA-HQ-OPPT-2022-0918-0038	Fragrance Creators Association
SACC23-0039	EPA-HQ-OPPT-2022-0918-0039	NCASI
SACC23-0040	EPA-HQ-OPPT-2022-0918-0040	University of California, San Francisco Program on Reproductive Health and the Environment (UCSF PRHE)
SACC23-0041	EPA-HQ-OPPT-2022-0918-0041	Utility Solid Waste Activities Group (USWAG)
SACC23-0042	EPA-HQ-OPPT-2022-0918-0042	Environmental Protection Network (EPN)
SACC23-0043	EPA-HQ-OPPT-2022-0918-0043	National Tribal Toxics Council (NTTC)
SACC23-0044	EPA-HQ-OPPT-2022-0918-0044	National Tribal Toxics Council (NTTC)
SACC23-0045	EPA-HQ-OPPT-2022-0918-0045	Environmental Protection Network (EPN)
SACC23-0046	EPA-HQ-OPPT-2022-0918-0046	American Chemistry Council (ACC)
SACC23-0047	EPA-HQ-OPPT-2022-0918-0047	B&C Consortia Management, L.L.C.
SACC23-0048	EPA-HQ-OPPT-2022-0918-0048	American Alliance for Innovation (AAI) et al.
SACC23-0049	EPA-HQ-OPPT-2022-0918-0049	American Fuel & Petrochemical Manufacturers (AFPM) and American Petroleum Institute (API)
SACC23-0050	EPA-HQ-OPPT-2022-0918-0050	Project TENDR (Targeting Environmental Neurodevelopmental Risks)
SACC23-0051	EPA-HQ-OPPT-2022-0918-0051	Kathleen Curtis
SACC23-0052	EPA-HQ-OPPT-2022-0918-0052	Environmental Defense Fund (EDF)
SACC23-0058	EPA-HQ-OPPT-2022-0918-0058	Jessica Trowbridge
SACC23-0059	EPA-HQ-OPPT-2022-0918-0059	Liora Fiksel
SACC23-0060	EPA-HQ-OPPT-2022-0918-0060	ExxonMobil Product Solutions Company
SACC23-0061	EPA-HQ-OPPT-2022-0918-0061	Environmental Defense Fund (EDF)

Abbreviation	Submission Number	Comment Name
SACC23-0062	EPA-HQ-OPPT-2022-0918-0062	Amy D. Kyle
SACC23-0063	EPA-HQ-OPPT-2022-0918-0063	Alaska Community Action on Toxics (ACAT)
SACC23-0064	EPA-HQ-OPPT-2022-0918-0064	American Chemistry Council (ACC)
SACC23-0065	EPA-HQ-OPPT-2022-0918-0065	SciPinion, L.L.C.
SACC23-0066	EPA-HQ-OPPT-2022-0918-0066	Swati Rayasam
SACC25-0125	EPA-HQ-OPPT-2024-0551-0125	Anonymous public comment
SACC25-0129	EPA-HQ-OPPT-2024-0551-0129	B&C Consortia Management, L.L.C. (BCCM)
SACC25-0130	EPA-HQ-OPPT-2024-0551-0130	People for the Ethical Treatment of Animals (PETA)
SACC25-0131	EPA-HQ-OPPT-2024-0551-0131	Environmental Defense Fund (EDF)
SACC25-0132	EPA-HQ-OPPT-2024-0551-0132	Earthjustice et al.
SACC25-0135	EPA-HQ-OPPT-2024-0551-0135	B&C Consortia Management, L.L.C. (BCCM)
SACC25-0136	EPA-HQ-OPPT-2024-0551-0136	Dibutyl Phthalate Consortium
SACC25-0137	EPA-HQ-OPPT-2024-0551-0137	American Chemistry Council (ACC)
SACC25-0138	EPA-HQ-OPPT-2024-0551-0138	Di-ethylhexyl Phthalate Consortium
SACC25-0139	EPA-HQ-OPPT-2024-0551-0139	Butyl Benzyl Phthalate Consortium
SACC25-0140	EPA-HQ-OPPT-2024-0551-0140	John Kissel
SACC25-0145	EPA-HQ-OPPT-2024-0551-0145	University of California, San Francisco Program on Reproductive Health and the Environment
SACC25-0148	EPA-HQ-OPPT-2024-0551-0148	Penelope Fenner-Crisp
SACC25-0151	EPA-HQ-OPPT-2024-0551-0151	University of California, San Francisco, Program on Reproductive Health and the Environment (UCSF PRHE)
SACC25-0152	EPA-HQ-OPPT-2024-0551-0152	Huntsman International LLC
SACC25-0153	EPA-HQ-OPPT-2024-0551-0153	B&C® Consortia Management, L.L.C. (BCCM)
SACC25-0154	EPA-HQ-OPPT-2024-0551-0154	PETA
SACC25-0155	EPA-HQ-OPPT-2024-0551-0155	ToxStrategies, LLC
SACC25-0156	EPA-HQ-OPPT-2024-0551-0156	Earthjustice et al. (Part 1 of 7)
SACC25-0157	EPA-HQ-OPPT-2024-0551-0157	Earthjustice et al. (Part 2 of 7)
SACC25-0158	EPA-HQ-OPPT-2024-0551-0158	Earthjustice et al. (Part 3 of 7)
SACC25-0159	EPA-HQ-OPPT-2024-0551-0159	Earthjustice et al. (Part 4 of 7)

Abbreviation	Submission Number	Comment Name
SACC25-0160	EPA-HQ-OPPT-2024-0551-0160	Earthjustice et al. (Part 5 of 7)
SACC25-0161	EPA-HQ-OPPT-2024-0551-0161	Earthjustice et al. (Part 6 of 7)
SACC25-0162	EPA-HQ-OPPT-2024-0551-0162	Earthjustice et al. (Part 7 of 7)
SACC25-0163	EPA-HQ-OPPT-2024-0551-0163	ExxonMobil Biomedical Sciences

1 OVERARCHING COMMENTS ON THE DRAFT RISK EVALUATIONS OF BBP, DBP, DCHP, DEHP, AND DIBP

Comments associated with this topic are summarized in the subsections below.

1.1 Scope of the Draft Risk Evaluations and Cumulative Risk Evaluations

1.1.1 Scope of Individual Draft Risk Evaluations

Summary: A public comment (DIBP-0089) stated that "To avoid fragmentation, EPA should clearly coordinate timelines and risk communication efforts across DIBP, DBP, DEHP, BBP, and DCHP evaluations to produce an integrated public health protection strategy."

EPA Response: In December 2019 EPA brought seven phthalates into TSCA risk evaluation: BBP, DBP, DCHP, DEHP, and DIBP, as high-priority substances; and DIDP and DINP as manufacturer requested risk evaluations. EPA's evaluation and receipt of public comment and peer review advice on these seven phthalates within the same general timeframe supports the type of coordination and communication the commentor recommends. It is also supported by EPA's *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act* (U.S. EPA, 2023) and a *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substances Control Act* (U.S. EPA, 2023). By evaluating each of the phthalates individually as well as viewing them through the lens of cumulative risk, and focusing the individual and cumulative assessments on sensitive populations (*e.g.*, women of reproductive age, male infants, male children), EPA is confident that its assessments are protective of human health and the environment. EPA has attempted to present a complete and holistic picture of hazard, exposure and risk for these phthalates, individually and together. Where EPA has identified unreasonable risk to people or the environment, as well as where it has not, the approach taken for the phthalates positions the EPA to look broadly at how to mitigate any unreasonable risks. It also supports the ability of the Agency, as the TSCA section 6 process proceeds, to communicate both generally as appropriate, and specifically where necessary, on how these phthalates can continue to be used in ways that do not present unreasonable risk to people or the environment.

Summary: A public comment (SACC23-0052) recommended EPA combine consumer, worker, and general population exposures for all receptors, not just fenceline communities, for example, considering cumulative consumer exposures within the cumulative occupational exposure assessment.

EPA Response: EPA recognizes that consumers, workers, and the general population may be exposed to multiple phthalates associated with TSCA COUs. However, EPA did not combine exposure scenarios in its assessment for workers and consumers because the Agency did not find any evidence to support such an aggregate analysis based on the reasonably available information, such as exposure from various exposure scenarios within a given relevant exposure duration, particularly given the quick metabolism and elimination of phthalates. Instead, as stated in the phthalate CRA, EPA used NHANES urinary biomonitoring data, which provides an estimate of aggregate exposure via all routes and pathways that cannot be attributed to specific sources or pathways of exposure, to capture additional exposures experienced by workers and consumers.

Summary: A public comment (SACC23-0052) suggested there is a greater potential for EPA to underestimate of risk as a result of perceived gaps in exposure assessment (limited scope of phthalates, failure to consider other chemicals that act on the same endpoints, limited combining of exposures), rather than an overestimation of risk from "double-counting."

EPA Response: EPA acknowledges the inherent possibility to over or underestimate risk and indicates possible instances throughout the 2025 CRA TSD and the 2023 draft proposal for CRA of phthalates.

1.1.2 Scope of the Cumulative Risk Assessment

Summary: A public comment (SACC23-0030) stated that the need for a CRA should be established as part of problem formulation on a case-by-case basis.

EPA Response: EPA has taken the approach that the need for a CRA should be established on a case-by-case basis in a manner consistent with the best available science and reasonably available information. Although EPA did not issue a formal problem formulation for the phthalate CRA, EPA did issue a proposed approach for CRA of phthalates in 2023, which was subject to public comment and SACC peer-review. Similar to a problem formulation, the CRA proposed approach outlined key issues. These key issued include that the U.S. population is co-exposed to the phthalates currently being evaluated under TSCA, as demonstrated by NHANES urinary biomonitoring data, and that a subset of these phthalates (DEHP, DBP, BBP, DIBP, DCHP, and DINP, but not DIDP) are toxicologically similar and induce effects on the developing male reproductive system consistent with phthalate syndrome and cause these effects in a dose-additive manner.

Summary: Several comments (SACC23-0028, SACC23-0031, SACC23-0032, SACC23-0043, SACC23-0050, SACC23-0065, DCHP-0128, DIBP-0089) expressed general support for EPA's decision to consider the cumulative risk of phthalates.

One of these public comments (DCHP-0128) further stated "TSCA compels EPA's consideration of the cumulative risks arising from co-exposure to DCHP and other chemicals that cause overlapping health effects. Accordingly, we strongly support EPA's development of the Draft CRA and its consideration of cumulative risks in determining whether DCHP presents unreasonable risks to human health and which COUs contribute to the chemical's unreasonable risks."

EPA Response: EPA believes that these comments align with the below rationale stated in its Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act. In light of these and other comments on a number of phthalates and cumulative risk, EPA decided to incorporate cumulative risk into a number of the final phthalate risk evaluations.

"EPA recognizes that for some chemical substances undergoing risk evaluation, the best available science may indicate that the development of a CRA is appropriate to ensure that any risks to human health and the environment are adequately characterized. TSCA also gives the Agency the authority to consider the combined risk from multiple chemical substances when there is an interrelated group of chemicals or mixtures [15 U.S.C. § 2625(c)]. Under TSCA section 26(c), EPA may take "any action authorized" under any provision of TSCA, in accordance with that provision with respect to a category of chemical substances or mixtures of chemical substances. Because individuals are co-exposed to

many chemicals in their daily lives, some of which may have the same health effects, EPA believes that in some cases the best approach to assess risk to human health may be to look at the combined risk to health from exposure to multiple chemicals." (EPA-740-P-23-001, p. 5)

Summary: Several comments (SACC23-0027, SACC23-0029, SACC23-0043, DCHP-0121, DCHP-0128, SACC25-0132) recommended that EPA consider ecological taxa, including plants and animals, in the cumulative risk assessment in order to better protect ecological taxa and human health.

Several comments (SACC23-0043, DCHP-0121) argue that ecological taxa show the effects of the cumulative risks of multiple agents/stressors before humans due to increased bioaccumulation, increased sensitivity, shorter lifespans, and reproduction cycles citing historical examples as reference, biomagnification in ecological taxa can lead to increased human exposure through ingestion of plant or animals, and environmental health is core to the mission of the U.S. EPA.

One of the comments (DCHP-0128) states there is evidence that wildlife are exposed to biologically significant levels of phthalates. Another comment (SACC25-0132) similarly points to evidence of mixed phthalate exposure within the environment.

One of the comments (SACC25-0132) argues there is sufficient evidence of shared endpoints across ecological taxa (endocrine disruption and regulation of antioxidant defense enzymes) to support a cumulative risk evaluation based on common adverse outcomes.

EPA Response: As described in EPA's *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act* (U.S. EPA, 2023), at this time, EPA is focusing its CRA efforts on human health, not on ecological taxa. This is because established Agency cumulative risk guidance documents are available to support human health, but not yet for an ecological CRA. Currently, EPA does not have a peer-reviewed framework for assessing environmental cumulative risks of chemical substances, however, it may develop an approach for conducting CRA under TSCA for ecological taxa in the future. Therefore, EPA is not actively pursuing a CRA of phthalates for terrestrial or aquatic organisms.

EPA believes that its current ecological risk evaluation, which represents the best available science, does not support the development of a cumulative ecological risk evaluation for this group of phthalates with the information that is reasonably available. Unlike in the human health hazard assessment, ecological hazard assessment includes consideration of many plant and animal species with substantially different physiological characteristics, life histories, and environmental exposures. Due to the lack of an established common endpoint for developing a measure of comparative exposure across phthalates for environmental taxa, or a peer-reviewed framework for obtaining such a measure, EPA did not believe that it was reasonable to pursue a cumulative ecological assessment in the timeframe of this risk evaluation that would provide findings consistent with the best available science.

Summary: A public comment (SACC23-0051) stated that EPA's risk evaluation and risk management rules should take into account both cumulative chemical exposures and non-chemical stressors, and characterized this approach as cumulative impact assessment which they consider a superior approach to cumulative risk assessment. The commenter stated that cumulative impact assessment, hazard-based chemical regulation, and requirement of safer alternatives would provide a

superior basis for chemical management.

The commenter also requested that EPA begin consideration of cumulative risk before the formal risk evaluation process begins, and that such consideration should feature outreach to impacted groups, collection of information about chemical co-exposure and non-chemical stressors, and revision of the TSCA to establish minimum requirements for cumulative risk assessment.

EPA Response: EPA recognizes that for some chemical substances undergoing risk evaluation, the best available science may indicate that the development of cumulative risk assessment is appropriate to ensure that any risks to human health are adequately characterized. However, few scientific methods have been developed that allow for a quantitative analysis of cumulative risk from combined exposure to chemical and non-chemical stressors. EPA continues to actively work to strengthen the scientific underpinning for assessing cumulative impacts, including impacts from non-chemical stressors. Until Agency-wide guidance and established methodologies have been developed, EPA does not expect to quantitatively evaluate non-chemical stressors under TSCA. Nevertheless, when information is available indicating that certain people or communities face socioeconomic, health or other circumstances that may increase their susceptibility to adverse health effects from exposure to a particular chemical or category of chemicals under evaluation, EPA considers those factors within the context of TSCA's requirement that EPA consider potentially exposed or susceptible subpopulations. For example, EPA identified a range of factors that may have the potential to increase biological susceptibility to DEHP, DBP, BBP, DIBP, and DCHP, including lifestage, pre-existing diseases, physical activity, diet, stress, and co-exposures to other environmental stressors that contribute to related health outcomes. As applicable, EPA has included qualitative discussion of these factors in Section 5 of the human health hazard TSDs for DEHP, DBP, DIBP, BBP, and DCHP. EPA maintains that its incorporation of the cumulative risk assessment into these 5 phthalate risk evaluations represents the best available science, as required by TSCA section 26(h).

Additionally, when preparing these risk evaluations and supporting Technical Support Documents, EPA obtained and considered reasonably available information, defined as information that EPA possesses, or can reasonably obtain and synthesize for use in risk evaluations. 40 CFR 702.33. EPA utilized all reasonably available information in the draft risk evaluation documents and incorporated additional information provided from the SACC and public comments on the draft risk evaluations into the final risk evaluations. Throughout the risk evaluation process, EPA sought to obtain information from stakeholders that would inform the risk evaluation of all 5 phthalates and these communications with stakeholders and information received are documented throughout the risk evaluation and technical support documents.

Summary: A public comment (DIBP-0089) stated that "EPA should not exclude DIBP use scenarios that fall under CPSC or FDA jurisdiction from the cumulative risk context. Real-world exposures cross regulatory boundaries." Another comment (SACC-0059) expressed the same comment, but in the context of all five non-MRRE phthalates (DEHP, DBP, BBP, DCHP, DIBP). The second commenter asked EPA to assess all non-TSCA and non-attributable phthalate exposures, including in personal care products and cosmetics. The commenter also asked EPA to include multiple phthalates in a single product even if below 2.5% weight per volume (the proposed cutoff in EPA's draft cumulative assessment). As justification, the commenter notes that the Consumer Product Safety Commission (CPSC) found health concerns for children when phthalates are present in children's toys at low concentrations, as CPSC placed a 0.1% by weight limit on eight phthalates in toys. Finally, the commenter states that not addressing inhalation exposures from non-TSCA sources may

underestimate non-TSCA exposures, because they state that fragrances have high levels of phthalates.

EPA Response: As part of the current risk evaluations of DCHP, DEHP, DBP, DIBP, and BBP, EPA evaluated NHANES urinary biomonitoring data using reverse dosimetry. NHANES provides an estimate of aggregate background phthalate exposure for the U.S. population that cannot be attributed to specific sources or pathways of exposure. Based on previous analyses by U.S. CPSC and Health Canada, dietary phthalate exposure is considered a major source of exposure to phthalates that is expected to be captured within NHANES. EPA believes that its NHANES analysis incorporates exposures from food and food additives, as well as other sources like cosmetics. While under TSCA EPA is not required to evaluate exposures resulting from uses covered under FDA such as from medical devices, cosmetics, food, or food additives, when EPA has reasonably available information on these kinds of exposures, it could assess them as part of a background aggregate exposure assessment. This applies, in the cases medical devices, cosmetics, food, or food additives, whether or not the FDA or other agency considers these exposure scenarios in any assessment/regulation it conducts or promulgates. For TSCA decision support purposes, EPA believes it has appropriately accounted for exposures from sources not regulated under TSCA.

Summary: Two public comments (SACC23-0032, SACC23-0052) stated that the cumulative risk assessment of phthalates should consider all phthalates for which there is any expected co-exposure, not just the high priority ones selected for risk evaluation by EPA. Di-n-pentyl phthalate (DPENP) and di-n-hexyl phthalate (DHEXP) were identified by both comments as phthalates with toxicological similarity that were not included in the CRA.

Several public comments (SACC23-0032, SACC23-0052, SACC23-0058, DCHP-0128) advocated for the inclusion of non-phthalate anti-androgenic chemicals in the EPA cumulative risk assessment for phthalates. Two comments (SACC23-0032, SACC23-0052) stated that exposure to anti-androgens across chemicals classes can contribute to "phthalate syndrome" in an additive manner and citing a publication from the National Research Council. Another two public comments (SACC23-0058, DCHP-0128) similarly state that excluding other anti-androgenic chemicals with overlapping adverse health effects and relevant co-exposures in the cumulative risk analysis underestimates real-world risk. The registered pesticides linuron, vinclozolin, and procymidone were identified in one comment (SACC23-0032) as examples of anti-androgenic chemicals that could be included in the phthalate CRA.

One of these public comments (SACC23-0032) acknowledged that DIDP is not associated with phthalate syndrome, but recommended acknowledgement of its potential to contribute to cumulative risk in either the individual or cumulative risk evaluation due to evidence of developmental effects.

EPA Response: EPA has published a *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act* (U.S. EPA, 2023), a *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substances Control Act* (U.S. EPA, 2023), and a *Technical Support Document for Cumulative Risk Analysis of DEHP, DBP, DIBP, BBP, DCHP, and DINP under TSCA (CRA TSD),* which incorporates peer review and public comments on EPA's cumulative risk assessment approach. As described in these documents, EPA proposed and now finalized an approach for cumulative risk assessment of phthalates. DEHP, DBP, DIBP, BBP, DCHP, and DINP are toxicologically similar and induce effects on the developing male reproductive system consistent with a disruption of androgen action and phthalate syndrome. DIDP is not included in the cumulative assessment because it does not

induce effects consistent with a disruption of androgen action or phthalate syndrome. Although NHANES biomonitoring data clearly shows that the U.S. population is co-exposed to DIDP along with other phthalates such as DINP and DEHP, DIDP causes developmental toxicity that is very different compared to other phthalates such as DEHP and DINP. Studies of experimental rodents demonstrate DIDP can cause reduced offspring survival, decreased pup body weight and weight gain, as well as cause skeletal and visceral variations. Unlike other phthalates that EPA is evaluating for cumulative risk, DIDP does not induce effects on the developing male reproductive system consistent with a disruption of androgen action and phthalate syndrome. For these reasons, EPA is not including DIDP in its cumulative risk assessment.

EPA acknowledges that there may be other phthalates and other classes of chemicals that can disrupt androgen action and cause effects on the developing male reproductive system similar to the six phthalates included in the phthalate CRA. This is discussed further in Section 4.3 of *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substances Control Act* (U.S. EPA, 2023). However, for the cumulative risk assessment of phthalates under TSCA, the assessment was limited in scope to consideration of chemicals that went through the TSCA section 6 prioritization process and were designated as high-priority chemicals for risk evaluation (DEHP, DBP, DIBP, BBP, DCHP) and chemicals with manufacturer requests for risk evaluation (DIDP, DINP). Other phthalates not currently undergoing risk evaluation were not considered. Additionally, other types of chemicals such as antiandrogenic pesticides (*e.g.*, linuron, proloraz, vinclozolin) and pharmaceuticals (*e.g.*, finasteride, flutamide, simvastatin) that are regulated by other federal agencies such as the FDA or other offices at EPA such as Office of Pesticide Programs were not considered, since these are outside the regulatory scope of TSCA and have not been prioritized for risk evaluation under TSCA. Neither TSCA nor EPA's TSCA risk evaluation procedural regulations require the Agency to conduct a cumulative risk assessment or to incorporate chemicals into ongoing TSCA risk evaluations that are not themselves undergoing risk evaluation. EPA has nonetheless completed a cumulative risk assessment for many of the phthalates undergoing risk evaluation as it represents the best available science, while also ensuring EPA can complete its phthalate risk evaluations in a timelier manner given the statutory deadlines for completing risk evaluations.

Summary: A public commenter (SACC23-0049) also provided recommendations on the Draft CRA Phthalates document, with the stated intent to help the SACC with its review, as well as to help EPA with subsequent revisions to the Draft CRA Phthalates Document, including:

1. EPA should clarify if it intends for the CRA to depend on the individual risk evaluations or if the individual risk evaluations will depend on the CRA.
2. EPA should provide more information on what constitutes a chemical with a "low hazard potential."
3. Due to the lack of mode of action data presented by EPA for phthalates, it is unclear from the current draft CRA approach whether the Agency will utilize mode of action data (when available) for other cumulative chemical groups to refine risk estimates and lower uncertainty.
4. EPA should define what criteria need to be met to conduct a cumulative exposure evaluation and be transparent in its selection of cumulative exposure approaches for consideration.
5. As a relatively "data rich" chemical set, the high-priority phthalates are not an ideal "test" group for applying EPA's Draft CRA Principles. These are well-studied chemicals that have already been evaluated as a chemical group by the National Academy of Sciences (NRC, 2008;

NASEM, 2017). Very few chemical groups evaluated under TSCA are going to be data rich, so the Agency should develop case study examples of data-poor chemicals it would consider designating as "cumulative chemical groups."

EPA Response: Many of the comments relate to how EPA would use the 2023 draft CRA proposed approach in risk evaluation and decision making. As they relate to use in risk evaluation, those comments have largely been addressed in the risk evaluations and technical support documents that now have been completed and issued by the Agency. How the phthalates cumulative risk work will inform decision making will be addressed, and proposed for public comment, when EPA moves to the risk management phase of the TSCA section 6 process.

Some comments relate to the broader application of cumulative risk and cumulative impacts assessment beyond the phthalate chemicals for which EPA has completed risk evaluation. EPA's application of CRA approaches and principles was specific to the evaluation of these six chemicals, and at this time no inference should be drawn from this effort to future TSCA section 6 activities beyond those that may be taken for these specific chemicals. Likewise, EPA will continue to consider whether, in the future, more-broadly applicable frameworks or guidance documents are needed for cumulative risk assessment.

Summary: The public comment (SACC23-0042, SACC23-0045) provided the following sources in response to EPA's request that the SACC identify any additional notable phthalate CRAs that may inform EPA's proposed approach:

1. ATSDR 2017 Interaction Profile for: Chlorinated Dibenzo-p-Dioxins, Polybrominated Diphenyl Ethers and Phthalates. U.S. Department of Health and Human Services. Public Health Service. Agency for Toxic Substances and Disease Registry. Atlanta, GA. https://www.atsdr.cdc.gov/interactionprofiles/ip14.html
2. Patton, L.E. 2010. CPSC Staff Toxicity Review of 17 Phthalates for Consideration by the Chronic Hazard Advisory Panel – 2010. Consumer Product Safety Commission. Bethesda, MD. https://www.cpsc.gov/s3fs-public/CPSCStaffToxicity17Phthalates.pdf
3. Dewalque L. , Charlier C., Pirard, C. 2014. Estimated daily intake and cumulative risk assessment of phthalate diesters in a Belgian general population. Toxicol Letters Volume 231, Issue 2, 1 December 2014, Pages 161-168. https://www.sciencedirect.com/science/article/abs/pii/S0378427414002781
4. T. Søeborg, H. Frederiksen, A. M. Andersson 2012. Cumulative risk assessment of phthalate exposure of Danish children and adolescents using the hazard index approach. Int. J. Andrology 33(3): 245-252. https://onlinelibrary.wiley.com/doi/10.1111/j.1365-2605.2011.01240.x
5. Wei-Hsiang Chang, Wei-Chun Chou, Alexander Waits, Kai-Wei Liao, Pao-Lin Kuo, Po-Chin Huang 2021 Cumulative risk assessment of phthalates exposure for recurrent pregnancy loss in reproductive-aged women population using multiple hazard indices approaches. Environment International 154 (2021) 106657
6. Christensen KL, Makris SL, Lorber M 2014. Generation of hazard indices for cumulative exposure to phthalates for use in cumulative risk assessment. Regul Toxicol Pharmacol. 2014 Aug; 69(3):380-9. doi: 10.1016/j.yrtph.2014.04.019.

7. Ji H, Wu Z, Chen D, Miao M, Chen H, Shuai W, Liang H, Yuan W. 2023 Individual and joint effects of phthalates exposure on the risk of early miscarriage. J Expo Sci Environ Epidemiol. 2023 Mar 23. doi: 10.1038/s41370-023-00533-1.

EPA Response: EPA reviewed the additional references of examples of phthalate cumulative risk assessments published in the peer-reviewed literature. EPA notes that the majority of provided references provide examples of estimating cumulative risk from exposure to phthalates using the hazard index approach, not the relative potency factor approach, which was used by EPA to estimate cumulative risk from exposure to phthalates under TSCA. Risk determinations under TSCA are not based on a bright line calculation of risk, and the hazard index approach does not imply an absolute determination of risk vs no risk. The MOE approach separates out the uncertainty factors from the hazard value, promoting increased transparency and allowing more case-specific judgement for "close calls."

Summary: One public commenter (SACC23-0050) strongly supports EPA's commitment to evaluate phthalates in a cumulative risk assessment. The commenter agrees with EPA's conclusion that DEHP, BBP, DBP, DIBP, DCHP, and DINP should be grouped together in cumulative risk assessment and considers this decision to be consistent with NRC's recommendations because there are co-exposures in the human population and all of these chemicals contribute to anti-androgenic effects, including phthalate syndrome. EPA appropriately proposes to comprehensively consider cumulative exposures to phthalates, including exposure sources that are not attributable to TSCA conditions of use because non-TSCA exposure sources include dietary exposures, which are the highest contributor to human exposure for some phthalates. Furthermore, this commenter is supportive of EPA's commitment to account for aggregate and cumulative exposure of fenceline communities to one or more phthalates. Although the commenter agreed with EPA's selection of anti-androgenic outcomes as the basis for the CRA, they recommended that EPA adopt a database uncertainty factor to address the uncertainty around neurodevelopmental harm from exposure to phthalates and likened this uncertainty factor to the FQPA requirement for a 10× safety factor to account for potential neurodevelopmental risk to infants and children.

EPA Response: A database uncertainty factor (UF_D) is applied for deficiencies in the toxicological database that might lead to a lower POD. Under TSCA, there is no universal list of hazard data that is required to consider a database sufficient to conduct risk evaluation, nor is there a minimum set of data required to conduct a risk evaluation. The decision to incorporate the database UF in TSCA risk evaluations is determined on a case-by-case basis, and for DEHP, BBP, DBP, DIBP, and DCHP, EPA determined that the database of studies did not have deficiencies that might suggest an underprotective POD . As such, EPA determined that a database uncertainty factor was not necessary. Additionally, by evaluating each of the phthalates individually as well as viewing them through the lens of cumulative risk, and focusing the individual and cumulative assessments on sensitive populations (*e.g.,* women of reproductive age, male infants, male children), EPA is confident that its assessments are protective of human health and the environment.

Summary: The public comment (SACC23-0042, SACC23-0045) recommended that EPA consider other phthalates beyond the high-priority phthalates and manufacturer-requested risk evaluation phthalates in its CRA to determine if they meet the criteria for inclusion in the cumulative chemical group (CCG), and recommended that EPA consult U.S. CPSC (2010). The commenter recommended that, if additional TSCA or non-TSCA chemicals are identified as potential candidates for inclusion in

the CCG, they should be subjected to the toxicologic similarity screen, and if they meet the criteria, they should be carried forward, along with the already-identified seven phthalates, into the exposure assessment.

Additionally, the commenter asserted that EPA's citing Health Canada's conclusion that di-n-octyl phthalates does not induce effects on the developing male reproductive system consistent with phthalate syndrome does not provide adequate justification for excluding this phthalate from the CCG, so expanded discussion on this issue is warranted.

This commenter went on to say that NHANES data indicates that the general population and at least one PESS are exposed to at least 9 phthalate diesters, which is more than EPA's current CCG candidate list; therefore, the commenter recommends that these other chemicals be subjected to the screening for toxicologic similarity and determine whether they meet EPA's co-exposure criteria.

EPA Response: EPA considered five high-priority (DEHP, DBP, DIBP, DCHP, BBP) and two manufacturer requested phthalates (DINP, DIDP) for inclusion in its cumulative chemical group. DIDP was excluded because it is not toxicologically similar to the other phthalates and does not cause effects consistent with a disruption of androgen action or phthalate syndrome. Although there are potentially other phthalates and chemical substances that can disrupt androgen action and cause phthalate syndrome, these chemicals are not actively being evaluated under TSCA and therefore were not further considered for inclusion in the cumulative risk assessment. TSCA requires EPA to incorporate the "reasonably available information" in the Agency's chemical risk evaluations, which EPA defines to include the "information that EPA possesses or can reasonably generate, obtain, and synthesize for use in risk evaluations, considering the deadlines in TSCA section 6(b)(4)(G) for completing such evaluation. 40 CFR 702.33. EPA has incorporated the best available science into the final phthalate risk evaluations, using the reasonably available information that the Agency had and could utilize within the timeframe provided by the statute to conduct the evaluations.

The TSCA Work Plan includes one additional phthalate (*i.e.,* di-n-octyl phthalate or DnOP) that is not currently prioritized for risk evaluation. This phthalate was not considered for inclusion in the phthalate cumulative risk assessment in part because it is not currently being evaluated under TSCA. Further, Health Canada (2015) concluded that DnOP does not induce effects consistent with phthalate syndrome and Health Canada did not include DnOP in its phthalate CRA. U.S. CPSC reached a similar conclusion and also did not include DnOP it their phthalate CRA (U.S. CPSC 2010). EPA reviewed the underlying studies that form the basis of the conclusions by Health Canada and U.S. CPSC and agrees with the conclusions reached by these agencies that DnOP, like DIDP, does not induce effects consistent with phthalate syndrome and is therefore not appropriate for inclusion in the cumulative risk assessment under TSCA.

EPA considered NHANES data on five high-priority (DEHP, DBP, DIBP, DCHP, BBP) and two manufacturer requested phthalates (DINP, DIDP) as part of its phthalate CRA. Other phthalates included in NHANES but not prioritized for risk evaluation under TSCA (*e.g.,* diethyl phthalate [DEP], dimethyl phthalate [DMP]] were not further considered for the reasons discussed above.

Summary: The public comment (SACC23-0042, SACC23-0045) recommended that EPA "do all of the above" when:

- considering the six sub-options for focusing the CRA on the most sensitive effects.

- evaluating phthalates for cumulative risk using both dose addition (based on the 20+ year history of using dose addition as the default in assessing mixtures) and [response addition] because they believe EPA is "obligated to try out all possibilities and then provide robust justification for selection of the approach to be used and exclusion of the other(s)".
- applying both the relative potency factor (RPF) and Hazard Index (HI) approach and compare the outcomes as a step in building its justification for using the RPF approach in the end.
- including fenceline communities and tribal populations as PESS in the conceptual model. The commenter recommended that EPA also propose an approach to determine co-exposure to individuals who may be part of a tribal population.
- Applying both scenario-based and reverse dosimetry approaches for estimating non-attributable and non-TSCA exposure; compare the results; and justify the choice.

EPA Response: In its draft phthalates assessments, EPA incorporated aspects of many of the commentor's recommendations, submitted those drafts to public comment and peer review, and has completed risk evaluations that, with the help of input received, have advanced the use of CRA to inform Agency decision making.

1.1.3 Conditions of Use (COUs)

Disposal

Summary: A comment (SACC23-0044) recommended that landfills (both lined and unlined) and transfer stations be considered similarly to industrial facilities in TSCA risk evaluations to best characterize handling and exposure to fenceline communities.

EPA Response: Unlined landfills were not considered in these assessments. Disposal of phthalate esters under TSCA in unlined landfills would not reflect exposures on a regional or national scale. Some older and decommissioned landfills are unlined but are subject to quarterly groundwater monitoring before closure as a municipal solid waste landfill under Title 40 of the Code of Federal Regulations (CFR) Part 258, Subpart-E Ground Water Monitoring and Corrective Action. Regardless of presence of a landfill liner, the phthalate esters are not likely candidates for infiltration to and/or transport offsite via groundwater due to the fate and transport characteristics of the phthalate esters (low water solubility, aerobic degradation rates and sorption to organic carbon found in landfill environments). Disposal is discussed in Section 3 of Environmental Media, General Population, and Environmental Exposure TSD and in Section 4.1.3 of each risk evaluation. Phthalates were not anticipated to pose a substantial risk of exposure for the general population or the environment through the biosolids or land pathways due to the limited persistence potential in soils receiving biosolids and limited mobility.

Recycling

Summary: Two public comments (DEHP-0141, DIBP-0132) stated that recycling operations identified in the draft BBP, DBP, DIBP, and DEHP risk evaluations are closer to production than to recycling and thus, overestimate occupational exposure. As support for this statement, the commenter states that recycled materials industry handles these phthalates as a component of a solid material and the concentrations of these phthalates are lower than during initial production due to solubilization or

volatilization during the use phase. In addition, the comments states that the processes of producing bales of recycled materials containing BBP, DBP, DEHP, and/or DIBP entail lower occupational exposure than manufacturing processes.

The commenters (DEHP-0141, DIBP-0132) discuss the processes assessed for recycling as outlined in the Environmental Release and Occupational Exposure TSDs for BBP, DBP, DEHP, and DIBP and states that these are closer to initial use in production or manufacturing than recycling. The commenter discusses differences in recycling processes across facilities, points out that the reference cited for this process is from the United Kingdom where production is considered part of recycling, and highlights that how recycling operations are characterized influences occupational exposure and risks. The comments recommend that EPA risk evaluations focus on recycling operations that supply manufacturing operations.

The public comments (DEHP-0141, DIBP-0132) discussed how recycling OES is treated the same as the waste handling, treatment, and disposal OES in the BBP, DBP, and DIBP assessments, which the commenter stated is not in alignment with the process descriptions described above. One of these public comments (DEHP-0141) noted that the DEHP assessment of recycling selected plastics converting as an appropriate surrogate instead, despite similar processes expected across phthalates.

A public comment (DIBP-0132) noted that recycling was not found to contribute to unreasonable risk for BBP or DIBP, but raised concerns that recycling had higher aggregate MOEs for than PVC plastics converting and PVC plastics compounding in the BBP assessment, but lower MOES than those same OES in the DIBP assessment. A public comment (DEHP-0141) noted that recycling was not found to contribute to unreasonable risk in the DBP assessment, even though the assessment states that "the estimated inhalation exposures to dust are likely overestimated" for recycling. In contrast, it is noted by the same public commenter (DEHP-0141) that recycling was determined to contribute to unreasonable risk in the DEHP assessment and the exposure was not considered an overestimate in that case.

The public comment (DEHP-0141) states that the difference in assessments of recycling in these TSCA risk evaluations could be perceived as arbitrary. Both public comments (DEHP-0141, DIBP-0132) advocated for consistency in how recycling operations are characterized across risk evaluations under TSCA and offered to work with EPA to improve characterization of recycling operations.

EPA Response: The EPA utilizes the best available science to inform the risk evaluations of existing chemicals under TSCA, so the assessments of the same OES may differ across chemicals to best utilize the available data. Ultimately, EPA did not identify unreasonable risk associated with recycling in the BBP, DBP, DEHP, or DIBP risk evaluations.

The EPA agrees with the public comments statement that recycling differs from manufacturing and production. However, the public comment described several different types of recycling processes and did not identify a clear alternative to the characterization within the flow diagrams highlighted within the public comments. EPA considers these process descriptions appropriate for these risk evaluations.

As stated in the public comments, the BBP, DBP, and DIBP risk evaluations consistently evaluated recycling and waste handling, treatment, and disposal the same way. Within the DEHP risk evaluation, surrogate data from PVC plastics converting was used to inform the assessment of recycling, which the public commenter noted characterized similarly across the OES. Data on PVC plastics converting submitted during public comment on the draft DEHP risk evaluation from the Vinyl Institute was used

to update the assessment, which resulting in decreased inhalation exposure and MOEs above the benchmark.

EPA welcomes engagement with stakeholders and the public on how to refine its future characterizations of activities evaluated and regulated under TSCA, including recycling operations.

Maleic Anhydride Synthesis

Summary: Three public comments (DBP-0114, DBP-0126, SACC-0152) recommended that EPA consider the use of DBP in synthesizing maleic anhydride to be a separate COU, rather than part of "Industrial Use – Non-incorporative activities", and provided process, release, and occupational exposure data for the recommended new COU. The commenter provided monitoring and industrial hygiene data for a selection of employees and locations/processes at US and non-US facilities, and stated that the samples collected during these monitoring events were below established regulatory limits and EPA's draft occupational inhalation exposure value for DBP (0.6 mg/m^3).

EPA Response: EPA disagrees with the commenters that the use of DBP in synthesizing maleic anhydride represents a condition of use that is substantially different from the definition of Industrial use – Non-incorporative activities. EPA believes the use of DBP in synthesizing maleic anhydride represents a condition of use that is not substantially different from the definition of Industrial use – Non-incorporative activities since occupational exposure and release points are expected to be similar. In both the Draft and the Final Risk Evaluation for DBP, EPA noted specifically that this COU is inclusive of maleic anhydride manufacturing, within the subcategory 'Solvent, including in maleic anhydride manufacturing technology'. EPA states in the DBP Risk Evaluation, Appendix E – Conditions of Use Descriptions, that "EPA understands that DBP is used in the manufacturing of maleic anhydride; however, DBP is not incorporated into the maleic anhydride product". This non-incorporative use as a solvent is described in the commenters' data, and therefore EPA did not change the COU for the synthesis of maleic anhydride.

EPA considered the provided occupational exposure data for use in the Final Risk Evaluation, but ultimately concluded that the surrogate industrial monitoring data available from an EU risk evaluation (ECB, 2008) were still appropriate since the submitted monitoring data were similar (within 1 order of magnitude) to the exposure estimates used in the assessment. Based on evaluating the weight of scientific evidence, EPA did change the occupational inhalation exposure value for DBP for this COU, from 0.6 mg/m^3 in the draft Risk Evaluation to 0.034 mg/m^3 in the Final Risk Evaluation. The highest exposure data point used previously during the draft risk evaluation was removed from the assessment because that number was a general estimated exposure value during phthalate production and was not specific to DBP. EPA finds that this lower inhalation exposure value is more representative of non-manufacturing industrial uses of DBP.

Additional Conditions of Use

Summary: Public commenters (SACC25-0132, SACC25-0145, SACC25-0151) stated that EPA failed to consider exposure to DEHP and DBP from agricultural films. The commenter noted "EPA must quantify the inhalation, oral, and dermal exposures faced by workers and consumers who use different types of agricultural films both indoors and outdoors. EPA must also quantify the levels of phthalates that enter the environment from these uses including their potential to migrate and the

resulting releases and exposure. Additionally, agricultural films can end up in several waste streams and their disposal must be considered by EPA."

EPA Response: The commenter did not provide any exposure or release data for DEHP or DBP relevant to the use of agricultural films. EPA acknowledges that DEHP and DBP are used in PVC for agricultural films, and this use has been evaluated in the risk assessments of DEHP and DBP. Specifically, the COUs titled "lawn and garden care products" and "other articles with routine direct contact during normal use including rubber articles; plastic articles (hard)" for DEHP and DBP, respectively, include agricultural films (see Appendix E of the DEHP and DBP risk evaluations), and these COUs are evaluated under the OES titled "fabrication or use of final products or articles" for both chemicals. Therefore, environmental releases and occupational exposures associated with the use of agricultural films are characterized by this OES in the DEHP and DBP risk evaluations.

Summary: Public commenters (SACC25-0132, SACC25-0145, SACC25-0151) stated that EPA must assess "prior sanctioned" food packaging uses of DEHP for foods of high water content in the evaluation of risk, as these uses, due to the timing of their approvals (*i.e.*, prior to September 6,1958), do not fall under the definition of "food additive" under the Federal Food, Drug, and Cosmetic Act (FFDCA), and are thus not covered by the TSCA exclusion of "food additives."

EPA Response: EPA agrees that DEHP used in food packaging, specifically in high water content foods, is a "prior sanctioned" use that falls outside of the definition of "food additive" under FFDCA section 201 and, therefore, is not covered by the TSCA section 3(2)(B)(vi) exclusion for food additives. However, EPA notes that the commenters did not provide any additional information to support including this use in the DEHP risk evaluation. EPA does not have reasonably available information that it can "synthesize for use" in the DEHP risk evaluation to further assess this use in particular. *See* 40 CFR 702.33. Regardless, EPA evaluated exposure to DEHP using CDC's NHANES urinary biomonitoring data and reverse dosimetry, as described in EPA's general population exposure assessment for DEHP and EPA's CRA TSD. Biomonitoring data from NHANES provides an estimate of aggregate exposure to DEHP for the general U.S. population. Although exposure estimated via NHANES cannot be attributed to specific sources or routes of exposure (*i.e.*, it is non-attributable), exposure assessments conducted by U.S. CPSC and Health Canada have demonstrated that a large component of exposure captured by NHANES comes from dietary sources, which would likely include migration of DEHP from food packaging materials into food ingested by the U.S. population. Notably, all MOEs calculated using mean and 95th percentile aggregate daily intake values calculated from NHANES are above the benchmark of 30, indicating no risk. For example, the highest 95th percentile exposure from NHANES was 6.44 ug/kg-day for male toddlers (3 to less than 6 years old). Using this exposure and the DEHP POD of 2,100 ug/kg-day, EPA calculated an MOE of 326 (see Table 4-13 in the DEHP risk evaluation for all daily intake values calculated for DEHP). As such, while EPA has not specifically assessed this use of DEHP, EPA believes it has adequately assessed it as part of the phthalate cumulative risk assessment.

Summary: Public commenters (SACC25-0132, SACC25-0145, SACC25-0151, DIBP-0133) stated that EPA must consider the shedding of microplastics from PVC products as a pathway in the analysis of exposure to DEHP, DBP, DIBP, and BBP. SACC25-0132, SACC25-0145, SACC25-0151 state "[i]t is both known, and reasonably foreseen, that microplastics will be released during the use and disposal of plastic products containing DEHP and DBP." Similarly for DIBP and BBP, DIBP-0133 stated that "[t]hese particles, which range in size from five millimeters to one nanometer, have been

detected in household dust (Salthammer, 2022), breast milk (https://www.theguardian.com/environment/2022/oct/07/microplastics-human-breast-milk-first-time), and human blood (Leonard et al. 2024). There is also evidence that microplastics leach both BBP and DIBP into the environment (Paluselli et al. 2019), resulting in exposures to wildlife (Liu et al. 2024) and people (https://www.epa.gov/circulareconomy/national-strategy-prevent-plastic-pollution)."

Similarly, in relation to BBP and DIBP, another commenter (DIBP-0130, DIBP-0133) stated that EPA failed to consider exposures from many sources, such as microplastics, and diapers used by infants and toddlers (Lai et al. 2025; Gul et al. 2025).

One of these public comments (SACC25-0132) further stated that inhalation of DEHP and DBP containing microplastics by birds and terrestrial species should be considered, stating that there is evidence of accumulation of DEHP and DBP in wild avian lungs and marine mammals.

EPA Response: EPA reviewed the evidence submitted by the commenters regarding the detection of microplastics in the environment, in household media, and in biological tissues. Within the phthalate risk evaluations, EPA evaluated the unreasonable risk of chemicals under conditions of use (COUs) and therefore either apportioned each exposure to a source with an identified COU, or else quantified non-apportioned exposures to the general population using biomonitoring (*e.g.,* the National Health and Nutrition Survey, NHANES). Based on the reasonably available scientific evidence, including the evidence submitted by the commenters, EPA concluded that it was unable to apportion potential exposures to phthalates from microplastics to a specific COU (*i.e.,* a category of consumer articles or products). However, for each phthalate under assessment, EPA did conduct a survey of non-apportioned exposure through NHANES. EPA considers this estimate of additional phthalate exposure, which is based on nationally-representative biomonitoring data that is not specific to any particular COU, to encompass potential human exposures to the assessed phthalates through exposure to microplastics.

For environmental (*i.e.,* wildlife) exposure to phthalates through microplastics (via inhalation or otherwise), the same source apportionment/COU concerns apply as in the above response regarding human exposures. Additionally, EPA concluded that the weight of the scientific evidence, including the evidence submitted by the commenters, did not support the conclusion that wildlife are exposed to biologically significant concentrations of phthalates through this route. The evidence submitted by the commenters regarding aquatic wildlife exposures, centering on the review article Liu et al. 2024, provides evidence that microplastics can shed from phthalate-containing articles, that desorption of the phthalate from the particles can occur in water, that phthalates occur in the environment, and that some bioaccumulation from environmental media can occur. However, the submitted evidence (and other evidence reviewed by EPA) do not clearly demonstrate a connection between these disparate pathways; while phthalates may occur in environmental media. EPA did not identify clear evidence from either the systematic review of peer reviewed literature conducted for these risk assessments or public comments to suggest that desorption from microplastics (as opposed to, for example, direct discharge of phthalates or phthalate-containing waste products from an industrial or commercial source) represents a significant (or even quantifiable) contribution to the overall environmental phthalate exposure. Finally, EPA conducted a qualitative assessment of trophic transfer for each phthalate under assessment and concluded that no phthalate currently assessed presents a significant risk for trophic transfer including bioaccumulation from the environment and biomagnification among trophic levels, based on the weight of scientific evidence including physical/chemical properties, biomonitoring studies, and metabolic evidence. Please refer to the Trophic Transfer section of each phthalate's *Environmental Hazard Assessment* for further details.

EPA recognizes the health and environmental concerns related to microplastics and welcomes further public engagement on how to best address microplastics issues as they may arise in future TSCA evaluation of existing chemicals.

Summary: A comment (DIBP-0128) asks that EPA remove PVC compounding as a condition of use for DIBP. The comment notes that US CPSC has recognized the limitation of using DIBP as a plasticizer in PVC in a 2011 toxicity review and a 2014 Chronic Hazard Advisory Panel. Further, the comment points to a US CPSC prohibition of DIBP in toys and childcare articles (in levels greater than 0.1%) since 2018, an FDA deauthorization of DIBP (among other plasticizers) in various food contact applications, and voluntary removal of flooring that used DIBP as a plasticizer from major home improvement retailers in the US as evidence that DIBP is not broadly used as a plasticizer in PVC products.

The comment requests that data from the 2018 Danish EPA survey not be used in the risk evaluation. The commenter notes that only two products in that survey were manufactured in the US and the survey tested neither for the presence of DIBP or other phthalates.

Since the commenter ask EPA to remove PVC compounding as a condition of use for DIBP, they also request that the number of sites for each occupational exposure scenarios in Table 3-4 be adjusted to exclude PVC compounding operations.

EPA Response: EPA disagrees with the commenter because data sources confirm the presence of DIBP in consumer products. For the PVC compounding COU EPA provides a description of the rationale for inclusion in Appendix E.4 in the DIBP risk evaluation document and descriptions of the data sources identified via the systematic review process that confirm the presence of DIBP in consumer products. Briefly, this COU refers to the preparation of a product; that is, the incorporation of DIBP into formulation, mixture, or a reaction product that occurs when a chemical substance is added to a product.

EPA uses reported concentration ranges for each product and calculates exposures based on the best available data and product specific use instructions and patterns to provide representative and health protective risk estimates. EPA performed a search of SDS, existing assessments, CDR, and product testing databases to identify product and article examples for each COU. This search is described in Section 2 of the Consumer and Indoor Exposure Assessment Technical Support Document and the Consumer Exposure Analysis spreadsheet in the supplemental files. For children's toys specifically, EPA acknowledge that the CPSIA final rule for DIBP was effective April 25, 2018, and it limits the concentration of DIBP to 0.1 percent in children's toys and childcare articles. However, the High Priority Chemicals Data System (HPCDS) database contained test data for DIBP measurements in 64 toy/game items from 2017 to 2024. EPA determined that these items are likely composed primarily of plastic and rubber components. For example, some of the descriptions provided for toys were dolls, dolls' furniture, action figures, puppets, board games, card games, developmental toys, scientific toys, and soft toys. EPA assessed exposure to DIBP in children's toys under two scenarios. In the first exposure scenario, new toys produced for the U.S. market are assumed to comply with the regulatory limit (0.1%) and were therefore assessed with DIBP weight fraction of 0.001 w/w in low-, medium-, and high-exposure scenarios. In the second scenario, legacy toys are assessed with weight fractions reported in the HPCDS database (WSDE, 2020). Based on the reported data, the weight fractions of

DIBP used in low-, medium-, and high-exposure scenarios were 0.0001 w/w, 0.0003 w/w, and 0.01 w/w.

With regards to the Danish EPA data, EPA strives to obtain and utilize the most relevant data for all conditions of use. In instances where these data from preferred sources were not available, DIBP content in specific products and articles provided in peer-reviewed literature and government reports originating from Canada and the European Union were used. Manufacturing practices and regulations for DIBP in consumer goods are comparable between these regions and the United States, so it is reasonable to assume that similarly formulated products may be available across these regions. For every identified product and article EPA aimed to use more than one source of product/article DIBP concentrations which is the case for all but car mats and footwear which relied on multiple Danish EPA reports from 2009 to 2020.

Based on the evidence identified, multiple identified products/articles, and the reporting years for the identified data, EPA is confident that this PVC compounding use for DIBP is a foreseeable use and while it may be phasing out of use, DIBP is still present in consumer products.

1.2 Organization of the Draft Risk Evaluations

No comments are associated with this topic.

1.3 Peer Review and Public Comment Process

1.3.1 Peer Review Process

Peer Review Process

Summary: A public comment (DCHP-0121) expressed concern that *DCHP Draft Risk Evaluation* did not include sufficient characterization of the potential cancer risk for DCHP. The comment noted that the *Draft cancer human health hazard assessment for Di(2-ethylhexyl) Phthalate (DEHP), Dibutyl Phthalate (DBP), Butyl Benzyl Phthalate (BBP), Diisobutyl Phthalate (DIBP), and Dicyclohexyl Phthalate (DCHP)* is cited in the risk evaluation, but not available for peer review. The comment expressed concern that the conclusions reached may not represent the "best available science" required under 15 U.S.C. § 2625(h).

Another public commenter (DCHP-0128) also raised concerns about the release of the cancer TSD for DCHP after the risk evaluation and stated that "EPA must permit supplemental comment when it makes the cancer assessment publicly available."

EPA Response: The EPA believes that the cancer assessment of DCHP utilized the best available science required under 15 U.S.C. § 2625(h). Although the *Draft cancer human health hazard assessment for Di(2-ethylhexyl) Phthalate (DEHP), Dibutyl Phthalate (DBP), Butyl Benzyl Phthalate (BBP), Diisobutyl Phthalate (DIBP), and Dicyclohexyl Phthalate (DCHP)* was not available for public comment at the time that this comment was received, the draft document underwent peer review and was available for a 45 day public comment period as part of the August 2025 Science Advisory Committee on Chemicals (SACC) Peer Review of High Priority Phthalates (Docket: EPA-HQ-OPPT-2024-0551), where charge question 8 related to the cancer assessment for DCHP. Further, it was used to inform the risk characterization and determination in the *DCHP Risk Evaluation* .

Summary: A public comment (DCHP-0125) contends that a comprehensive review of the RPFs and their use in the CRA cannot be conducted, because EPA has not yet published all assessments and inputs and subjected them to public comment and/or peer review.

EPA Response: EPA made the draft meta-analysis TSD, all associated modeling files, and the CRA TSD publicly available in December 2024. The revised CRA TSD was made publicly available in May 2025. However, it is important to note that the BMD modeling approach and draft RPFs did not change between the December 2024 draft CRA TSD and the May 2025 revised draft CRA TSD. Although the public commenter is correct to note that certain files (*e.g.*, draft systematic review protocols; draft data quality evaluation information for animal toxicology and human epidemiology; and draft risk evaluations) in the risk evaluation packages were not publicly available when the comment was submitted in May 2025, EPA does not consider these files critical for evaluating the BMD modeling approach or the derived draft RPFs.

1.3.2 Public Comment Process

Summary: Two public comments (DCHP-0115, DCHP-0116) requested a 60 day extension of the comment period on the DCHP Draft Risk Evaluation.

EPA Response: To give stakeholders additional time to review materials and prepare comments, EPA granted the public comment period extension through May 9, 2025 (EPA-HQ-OPPT-2018-0504-0119).

Summary: Two public comments (DBP-0113, DEHP-0120) requested a 60 day extension of the comment period on the DBP and DEHP Draft Risk Evaluations, and the cross-phthalate TSD for the Draft Revised Cumulative Risk Assessment. The comments also requested a 120-day comment period for the BBP and DIBP Draft Risk Evaluations, as they had not yet been released when the comments were submitted.

EPA Response: EPA did not extend the referenced comment periods. The comment period of 60 days meets the requirements under TSCA and extensions would have delayed release of the final risk evaluation. The issues presented for SACC review and public comment that were relevant to DBP and DEHP were addressed in other phthalates evaluations and support documents. In addition, granting the request would have jeopardized EPA meeting it's court-ordered deadlines for issuance of the phthalate risk evaluations.

Summary: Four public comments (SACC25-0139, DBP-0113, DEHP-0120, DIBP-0131, BBP-0123) expressed concern that the SACC would not have the ability to comprehensively evaluate the draft risk evaluations for BBP and DIBP and they would not undergo adequate peer review as required by the Federal Advisory Committee Act (FACA) because, as of July 25, 2025, EPA had not issued draft risk evaluations for BBP and DIBP and stated that it intends to use lessons learned through SACC review of the DEHP, DBP, and DCHP risk evaluations and TSDs during August 4-8, 2025 meeting of the SACC to inform the risk evaluations and draft risk determinations of BBP and DIBP.

EPA Response: The science issues addressed in the SACC charge also applied to the draft risk evaluations of BBP and DIBP, which were released to the public and SACC concurrent with the SACC meeting. In addition, chemical specific charge questions for BBP and DIBP were asked regarding the human health hazard (both cancer and non-cancer) and the chemical specific environmental hazard and physical chemistry and fate and transport TSDs for BBP and DIBP were made available for SACC review and public comment in January 2025 and again in May 2025 ahead of the peer review meeting. Feedback from the SACC review of the DBP, DCHP, and DEHP risk evaluations and exposure TSDs were incorporated into the final risk evaluations of BBP and DIBP.

Summary: A public comment (DBP-0131) stated that EPA should incorporate an additional peer review of the Occupational Exposure Values (OEVs) and Existing Chemical Exposure Limits (ECELs), incorporating dialogue with stakeholders, experts, or the regulated community, in addition to the existing public comment process for the draft Risk Evaluations.

EPA Response: EPA believes that its peer review of the risk evaluations and underlying technical support documents, to the extent the charge questions were relevant to the science underpinning the derivation of OEVs, which are part of the risk evaluation, is consistent with OMB and EPA policy and guidance on peer review. As risk management tools, EPA does not believe that ECELs, which are developed during risk management, as a general matter should be subject to peer review unless specific scientific or technical issues arise from their derivation that make them applicable to OMB and EPA peer review policy and guidance.

Summary: A public comment (DIBP-0089) asked, "Will EPA publish all modeling assumptions, raw exposure data, and use profiles for DIBP in a format accessible to non-industry stakeholders during the official 60-day comment period?"

EPA Response: In July 2025, EPA released the draft DIBP risk evaluation for a 60-day public comment period. The draft risk evaluation and all associated technical support documents were posted to the DIBP docket. All exposure and hazard assessment modeling output files were also made publicly available upon request though the EPA Docket Center, Public Reading Room. All model output files can be accessed in person or by writing: Environmental Protection Agency, Docket Center, 1301 Constitution Ave NW, Room 3334, Washington, D.C. 20004. The Docket Center's hours of operations are 8:30 a.m. – 4:30 p.m., Monday – Friday (except Federal holidays). Telephone: 202-566-1744 Email: Docket-CustomerService@epa.gov.

Summary: A public comment (DEHP-0135) stated that releasing a Memorandum with Additional Context on Occupational Exposure Data for DEHP one week prior to the deadline for comment submission without extending the timeline for public review and comment "does not allow stakeholders adequate time to verify the approach and conclusions of EPA's updated exposure estimates in violation of basic principles of due process."

EPA Response: EPA recognizes that the time allowed to review the memo was short. Use of information in the memo, and questions related to this information, may arise during risk management. Notice and comment opportunities will be available during any proposed rulemaking related to DEHP and EPA will welcome comments on this memorandum.

1.4 Legal and Regulatory Issues

Regulation of Imports

Summary: A commenter (DBP-0133, DEHP-0142) requested a transition period of 1–2 years for any implemented regulations, in the context of imports. The comment was filed in reference to the World Trade Organization Agreement on Technical Barriers to Trade and requires a separate response through the WTO comment process.

EPA Response: This comment is out of scope as it relates to a future risk management rulemaking for DBP and DEHP rather than the risk evaluations at issue here. When EPA proposes TSCA risk management rulemakings for these chemical substances it will request public comments at that time.

Tort Law

Summary: A public comment (SACC23-0028) stated, "It is long-past time for the EPA to consider cumulative exposures, which is the reality of how humans and animals are exposed, in all future risk assessments. We know already from NHANES data that numerous chemicals, including phthalates, bisphenols, and PFAS have been measured in 90%+ of the population. These exposures are coming from thousands of different consumer products, from drinking water, and in some cases, from the very air we breathe. One of the primary safeguards we have in the US is tort law, which allows individuals to sue companies for harm. Chemicals like phthalates are so ubiquitous and used in so many different products, who are consumers to sue? Everyone? How can consumers prove that their breast cancer, or whatever other fill in the blank condition is linked to the phthalate exposure from 1 specific product? They cannot, which undermines entirely the protective benefit of tort law to pursue damages from companies. With the mounting evidence that environmental chemicals, including phthalates, are linked to myriad human health conditions, it's essential that we consider the cumulative exposures that the population is getting every single day so that we may begin to regulate these chemicals with the urgency that is needed."

EPA Response: Conducting risk evaluations on seven phthalates, developing a cumulative risk framework for these seven phthalates, and then applying these cumulative risk approaches to these evaluations has been a significant accomplishment for EPA. Under TSCA, the EPA is charged with evaluating chemical substances. While the seven phthalates were brought into TSCA risk evaluation as individual substances, the EPA believes that its approach, supported by the best available science, of pairing each single-chemical risk evaluation with the application of cumulative risk methods aligns with commentor's recognition that people are exposed to multiple phthalates, and sometimes to more than one phthalate at the same time. By evaluating each of the seven chemicals individually as well as viewing them through the lens of cumulative risk, EPA has a more complete and holistic picture of hazard, exposure, and risk for these seven phthalates, both individually and together consistent the risk-based TSCA statutory obligations. The portions of the comment about United States tort law are out of scope for purposes of these phthalate risk evaluations.

1.5 Other Overarching Comments

General Support for the Risk Evaluations

Summary: A public comment (DCHP-0018) stated "I support this measure" on the docket for the *DCHP Draft Risk Evaluation*.

EPA Response: EPA acknowledges the public support.

Other Concerns

Summary: A public comment (BBP-0122) requests that EPA "consistently apply the exposure evaluation requirements of TSCA 6(c)(2)(D) and (E). For manufacturers of complex durable goods, these sections are critical to demonstrate the negligible exposure potential associated with articles and replacement parts as well as key non-dimensional uses. In the absence of these individual determinations, EPA cannot support the requirements of TSCA 6(i)(1), issue appropriate orders, and ensure that preemption applies."

EPA Response: Pursuant to the current TSCA risk evaluation procedural regulations, TSCA section 6(i)(1) applies when EPA makes a finding that a chemical substance undergoing risk evaluation does not present an unreasonable risk of injury to health or the environment. 89 Fed. Reg. 37028 (May 2, 2024). For each of the phthalate risk evaluations, EPA has made a single risk determination, rather than determinations on individual conditions of use or exposure routes or pathways, finding each phthalate presents an unreasonable risk of injury to health or the environment. Regardless, the requirements of TSCA sections 6(c)(2)(D) and (E) apply to risk management rulemakings that take place after a TSCA section 6(b) risk evaluation is complete. As such, they have no bearing on whether EPA's exposure assessment is adequate enough to meet the TSCA science standards, under TSCA section 26, when issuing TSCA risk evaluations. EPA will consider TSCA sections 6(c)(2)(D) and (E) as part of the forthcoming risk management rulemakings and encourages public comment on those forthcoming proposed rulemakings.

2 CHEMISTRY, FATE, AND TRANSPORT

Comments associated with this topic are summarized in the subsections below.

2.1 Physical and Chemical Properties

Summary: A public comment (DEHP-0139) requested that EPA state that DEHP is produced as a liquid only, and not as a solid. The commenter states that while DEHP may be present in particles of other solid materials, DEHP itself does not exist in a solid form. The commenter states that the references EPA cited to justify the evaluation of DEHP solids/pellets in the occupational exposure scenarios are not accurately applied. The commenter notes that the physical chemistry table accurately states DEHP is liquid under standard temperature and pressure.

Similar to DEHP, a public commenter (DIBP-0128) requests that EPA state that DIBP is produced as a liquid only, and not as a solid.

EPA Response: The Occupational Exposure and Environmental Release TSDs for DEHP and DIBP describe the manufacture of these chemicals as liquids. Assessments of DEHP and DIBP as solids pertain to those COUs where DEHP or DIBP have become part of a solid matrix through processes such as compounding. For those COUs, there are assessments of DEHP and DIBP exposure to solids.

Summary: A public comment (SACC25-0132) stated that EPA should provide a more robust justification for the selection of the DEHP water solubility value (0.003 mg/L) given that high-quality estimates spread over 4 orders of magnitude and the selected value is below the federal drinking water standard maximum contaminant level for DEHP (6 µg/L) (40 C.F.R. § 141.61(c)(1)(xxii)).

EPA Response: EPA agrees that the selected water solubility value for DEHP is below the federal drinking water standard maximum contaminant level for DEHP. DEHP is a data rich chemical, with numerous studies measuring the water solubility of DEHP using techniques with varying levels of accuracy and appropriateness. EPA believes that 0.003 mg/L is a valid and appropriate estimate for the true aqueous solubility of DEHP and this value was used in the final risk evaluation for DEHP. As described in the technical support document for DEHP water solubility values within the upper range is likely due to the tendency of phthalate esters to form colloidal suspensions in water which may result in detection of DEHP above aqueous solubility in water samples. The water solubility indicates the maximum amount of DEHP that in aqueous phase (dissolved) in surface water. The MCL is the maximum level allowed of a contaminant in water which is delivered to any user of a public water system and may include both aqueous and colloidal phases. The MCL is based on the maximum concentration of DEHP in drinking water with no known or anticipated adverse health effects that expected to occur.

2.2 Environmental Fate and Transport

Environmental Fate

Summary: A public comment (DCHP-0121) provides a reference indicating that phthalate degradation is influenced by temperature and exposure to light, and indicates that colder, darker environments, such as the Northern waters, may have increased phthalate concentrations, thus increasing potential for exposure.

EPA Response: EPA agrees that temperature and light exposure are some of the factors playing an important role in the biotic and abiotic degradation of phthalates esters. Abiotic and biotic degradation processes are known to be mainly set by temperature, available sunlight, aerobic conditions, presence of metals, and pH within the environment. Changes in these factors could result in lower or higher degradation half-life values within different environmental compartments and processes. As described in the *Physical Chemistry, and Fate, and Transport Assessment for Dicyclohexyl Phthalate (DCHP)*, these factors are expected to influence the hydrolysis (Section 3.2.1), biodegradation (3.2.3), and persistence (Section 3.5) of DCHP. As described in Section 3.5, DCHP is expected to degrade rapidly under most environmental conditions, with delayed biodegradation in low-oxygen media, and it might persist in sediments or soil. However, due to the inherent hydrophobicity (log K_{OW} = 4.82) and sorption potential (log K_{OC} = 4.47), it is not expected to be bioavailable for uptake.

Summary: A public commenter (SACC25-0137) raised concerns that the selections used within the Level III Fugacity Model in Epi Suite™ under-estimates sediment concentration and overestimates water concentrations. The commenter states that the high Kow (>7) of DEHP falls outside the applicability domain of the default Molecular Connectivity Index (MCI) settings and is more appropriately modeled with the EQC method. Comparison calculations are provided.

EPA Response: EPA disagrees that the selections used within the Level III Fugacity Model in Epi Suite™ under-estimated sediment concentrations and overestimated water concentrations. As described in section 5.2 of the *Draft Physical Chemistry, Fate, and Transport Assessment for Diethylhexyl Phthalate (DEHP)*, media specific concentrations were predicted with the EQC method based on the selected Log Koc of 5.42 (mean of 8 reported values within the best available data sources). With this approach, media specific concentrations are estimated based on the best available Koc values. The default MCI settings were not used and thus the limitations described by the public commenter are not applicable to this estimate.

3 RELEASES AND CONCENTRATIONS IN THE ENVIRONMENT

Comments associated with this topic are summarized in the subsections below.

3.1 Environmental Releases

Summary: A public comment (DCHP-0122) stated that EPA inappropriately characterized modeled release estimates as biased towards overestimation in the evaluation of PVC Compounding in the DCHP risk evaluation. The commenter states that the lower levels of DCHP identified in wastewater monitoring by Washington State Department of Ecology should be attributed to off gassing of PVC products and are not reflective of releases from PVC Compounding facilities.

EPA Response: EPA does not believe that it inappropriately characterized modeled release estimates for PVC Compounding in the DCHP risk evaluation. EPA does not have direct release data for DCHP. However, EPA conducted modeling of releases to surface water at the point of release (*i.e.*, in the immediate receiving waterbody receiving the effluent) to assess the expected resulting environmental media concentrations from TSCA COUs using many conservative assumptions including low flow in the receiving waterbody, consideration of no wastewater treatment for swimming and fish ingestion, and exposure to humans and aquatic species directly at the point of discharge. EPA used models and input parameters consistent with the best available science to estimate release estimates.

EPA utilized all reasonably available data to contextualize modeled results, including the monitoring done by Washington State Department of Ecology (WA DOE, 2022). However, EPA did not solely rely on any monitoring studies because surface water concentrations in the monitoring study could not be attributed to specific COUs. The commenter states that the "a workgroup in Washington State addressing phthalates in sediments concluded that the primary pathway to waterbodies in the state's urban water environment was through phthalates off gassing from PVC products." However, that is not the conclusion of the monitoring study itself but a statement in the introduction of the study which includes discussion of other potential pathways of phthalates as well. The monitoring study does not attribute phthalate concentrations to any specific source. EPA also compared modeled values to Keil et al. (2011) which had higher monitored surface water concentrations for DCHP and still found its modeled values to be four orders of magnitude higher than the monitored value for PVC Plastic Compounding.

Summary: A public comment (SACC23-0052) recommended EPA leverage information collection authorities to obtain occupational exposure and environmental release information directly from manufacturers. The comment raised concerns around the limitations of using CBI data submitted through Chemical Data Reporting (CDR) and recommended EPA improve the usefulness of the program.

EPA Response: EPA strives to foster partnerships with stakeholders to obtain data for risk evaluation assessment and any data received from stakeholders will be considered in the future development of risk evaluations.

Summary: A public comment (DEHP-0139) noted that there is only a single US producer of DEHP and asked EPA to correct the list of manufacturers in the RE to reflect this. The commenter also asked that EPA change the number of days of manufacture for facilities that produce DEHP, to base its assumptions on the 180-day manufacture from the DINP Risk Evaluation rather than the 364-day assumption based on Chemical Data Reporting (CDR) data from a company that uses DEHP. The commenter stated that the company from which the 364-day CDR assumption was derived does not produce DEHP. Another commenter (DEHP-0137), who claimed to be the only manufacturer of DEHP in the United States, provided propriety information on the number of operating days.

Another public comment (DIBP-0128) asked that EPA either revise the number of days of manufacture of DIBP or provide an explanation of why the value of 250 operating days was used in the RE. The commenter provided calculation of how long it could take to manufacture DIBP based on the amount reported in the 2020 cycle of CDR.

EPA Response: The identification of domestic manufacturers of DEHP is based on reported programmatic data. All sites referenced in the DEHP occupational exposure TSD indicated manufacturing activity in either in CDR or TRI database. Suggestions by the commenter on "likely uses" or "may have used" for sites not under control of the submitter are not actionable.

Regarding the operating days for manufacturing: Eastman Chemical has provided an estimate for operating days which compares favorably to the suggested number of operating days in the comment. The DEHP occupational exposure TSD and DEHP risk evaluation were updated accordingly to include the 180 days as the lower end of the possible range of operating days for manufacturing. However, there were no data on the operations schedule for DIBP manufacturing. In absence of facility- or industry-specific operation data, EPA assumes a manufacturing schedule of 5 days per week and 50 weeks per year (with 2 weeks down for turnaround) for a total of 250 operation days per year. Though some facilities may operate more or less than the estimated number of days depending on process specifications, the estimate of 250 days per year is reasonable based on the total production volume presented in the 2020 CDR reporting cycle. Nevertheless, submissions of such data are encouraged during the risk evaluation process, and EPA provides stakeholders numerous opportunities to submit their operations data if estimates are inconsistent with their production schedules.

Summary: A public comment (DEHP-0137) provided propriety information related to the fate of DEHP in an industrial wastewater treatment system.

EPA Response: EPA reviewed the information provided by the public commenter and considered it for the finalization of the risk evaluation. However, the submission did not include enough detail to refine the assessment. The fate of DEHP in industrial wastewater treatment systems, including treatment efficiency and post-treatment concentrations, is already reflected in TRI data, which represent releases after onsite treatment and removal for all reporting facilities. To make the comment usable, EPA would need: (i) ranges for DEHP aerobic biodegradation rates and overall removal efficiencies within the treatment system, (ii) ranges for measured DEHP concentrations in biosolids produced by the industrial wastewater treatment system, and (iii) the biosolids' final disposal method(s) (*e.g.,* thermal oxidation) and corresponding treatment/removal efficiencies, provided as ranges.

Summary: A public commenter (SACC25-0132) stated that EPA underestimated surface water concentrations of DEHP and DBP by assuming in their modeling approach that facilities only discharge chemicals via a single release. The commenter contends that EPA must consider all releases from a facility and other facilities sharing a receiving water body.

EPA Response: With respect to multiple nearby facilities discharging to the same receiving waterbody, EPA considered the proximity of multiple releasing facilities within the Cumulative Risk Assessment for phthalates. The Agency concluded that in a small number of cases, releasing facilities across phthalates analyzed are located in close geographic proximity. However, because of the relatively low persistence, low bioaccumulation potential, and low long-range transport, and the protective screening methods applied for assessing exposure to single facility releases, EPA did not pursue aggregation of release and exposure estimates.

Similarly, EPA is not considering multiple releases from a single facility due to the lack of reasonably available information on releases from multiple OES occuring at the same time from a single facility. However, EPA made many conservative assumptions in its assessment of surface water concentration and resulting fish tissue concentrations including direct discharge to a waterbody, no dilution, and immediate uptake by fish in the direct discharge despite evidence of low persistence, low bioaccumulation potential, and low long-range transport of the phthalates. Using these assumptions along with considering reasonably available data including actual releases from reporting facilities for other OES, surface water concentrations from monitoring studies, and the physical chemical properties of the phthalate, EPA determined that its surface water concentrations and fish tissue concentrations were appropriately conservative.

Summary: A public comment (BBP-0125) noted that based on 2020 Chemical Data Reporting (CDR), EPA identified Polymer Additives Inc. located in Bridgeport, NJ, as a BBP manufacturer. The EPA estimated that the site had annual releases of between 91,400 kg (201,500 lb.; central tendency) and 175,000 kg (385,805 lb.; high-end) of BBP per year to the receiving water body as wastewater. However, Polymer Additives had not reported BBP releases since 1993. The facility has permit limits and monitoring requirements under its National Pollution Discharge Elimination System (NPDES) permit for DEHP, DBP, DEP and dimethyl phthalate of 198 lb/year, 51.9 lb/yr, 155 lb/yr and 48.4 lb/yr, respectively. EPA grossly overestimated the level of discharges to surface water of BBP under the manufacturing condition of use and ignored pollution regulations that would control those discharges. The exposure estimate for this condition of use should be revised accordingly.

EPA identified one site in the United States where BBP was reported to have been used in plastic product manufacturing as part of the 2020 Chemical Data Reporting (CDR), Polymer Additives Inc. located in Bridgeport, NJ. This is the same site noted above as a BBP manufacturing facility. EPA used similar unrealistic assumptions in estimating releases of BBP per year to the receiving water body in wastewater. EPA reported modeled releases of BBP between 8,911 kg per year (19,645 lb.; central tendency) and 17,000 kg per year (37,478 lb.; high end). As noted above, BBP TRI releases have not been reported at this site since 1993 and, as a priority pollutant, BBP discharges must be controlled. The environmental exposure estimate for surface water for this condition of use should be revised to reflect realistic circumstances.

EPA Response: The reason that the Polymer Additives Inc. facility has not reported BBP releases to TRI since 1993 is that it was that year that BBP ceased to be a TRI-listed chemical. The amount of DEHP, DBP, DEP, and dimethyl phthalate that the facility discharges via its NPDES permit is

irrelevant as EPA does not have evidence that the facility is manufacturing those chemicals, while it is not required to monitor its discharge of BBP. Without having any specific release information for this facility, EPA used the national aggregate BBP production volume range that was presented in the 2020 CDR as an input into the model used to estimate releases. As the method and rationale has already been documented in the risk evaluation documents, EPA believes that it has used the best available information to assess the releases.

3.2 Concentrations in the Environment

Surface Water Concentrations

Summary: A public comment (DCHP-0128) states that EPA did not provide sufficient justification for not considering cumulative risks to fenceline communities from facilities that release multiple phthalates. The public commenter points to refinements in the DCHP general population exposure assessment as evidence that conservative assumptions were not used in the assessment or sufficiently protective to negate the need for cumulative assessment of environmental releases. Further, the comment states "Even if fenceline community risks from DCHP alone were not unreasonable, that would not affect the need to consider the increased risks to communities who are exposed to multiple phthalates."

EPA Response: EPA disagrees that the refinements in the DCHP general population exposure assessment is evidence that the assessment is not conservative or sufficiently protective enough. DCHP had no reported facility releases so EPA modeled DCHP releases to the environment using EPA standard models. The refinements pointed out by the public commenter for DCHP's general population exposure assessment was for the surface water concentrations, which affects swimming, drinking water, and fish ingestion exposure and included looking at a range of flows (P50, 75, 90) for the modeling as supported by the SACC as an informative, scenario-based risk characterization. Even though EPA looked at a range of possible flows, the assessment still included multiple other conservative assumptions including using high-end modeled releases with no assumptions of dilution and exposure to humans and uptake to fish directly at the outflow of a release. As stated by the SACC, "using P50 of a multi-year low-flow condition is likely to provide a conservative upper-bound, but pairing this with high-end emissions can amplify conservatism to the point of limiting utility." Without facility specific data for DCHP releasing facilities, looking at other flows in addition to P50 was not a refinement on the assessment to get to a more accurate surface water concentration but to get to surface water concentrations that are environmentally possible. In the absence of reported release data, EPA supported its modeling result contextually with environmental monitoring data of surface water concentrations. The lowest modeled surface water concentration used for the assessment was still 5 orders of magnitude higher than the highest U.S. monitored surface water concentration of 3.00E–06 mg/L (Kiel et al. 2011). EPA has robust confidence that the values used for its general population exposure was sufficiently protective.

EPA still considered the possibility of multiple phthalates exposure, including exposure to DCHP and other phthalates, as part of the cumulative risk assessment described in the CRA TSD. As described above in Section **Error! Reference source not found.**, EPA identified DMR, NEI, and TRI data for DEHP, DBP, and BBP, but not for DCHP, DINP, and DIBP. These EPA databases provide information on facilities releasing phthalates to various environmental media and provide latitude and longitude data for releasing facilities. Using the release information, EPA identified 1,461 facilities that report use of a single phthalate, while 461 report use of multiple phthalates (*i.e.*, any combination

of DEHP, DBP, or BBP). Using the available location data, EPA mapped the reporting facilities to look for geographic patterns or hotspots but concluded that individual facilities are broadly dispersed around the United States. Therefore, based on physical chemical properties that make phthalates not likely candidates for long range transport due to the nature of the phthalate esters to both biodegrade, degrade via photolysis, and sorption to organic carbon present in the environment, EPA does not expect exposure to multiple phthalates from releasing facilities.

4 HUMAN EXPOSURE ASSESSMENTS

Comments associated with this topic are summarized in the subsections below.

4.1 Occupational Exposures

Summary: A public comment (DIBP-0089) stated that "EPA must incorporate real-world worker conditions, including inhalation and dermal exposures during manufacturing, handling of plastics, and recycling. Median and high-end occupational scenarios should reflect conditions in underregulated and non-union worksites."

EPA Response: EPA believes that the inhalation and the dermal exposures estimated during manufacturing, handling of plastics, and recycling incorporate real-world worker conditions. EPA has used both monitoring data and modeling to estimate worker occupational exposures. Monitoring data are given the highest priority in EPA's hierarchy of approaches for occupational exposures as they are collected in actual workplace conditions. Model results are either used to help corroborate monitoring data, especially in cases where such data are limited, or to provide exposure estimates where monitoring data are not available. In general, EPA has incorporated all reasonably available monitoring data that received a quality rating above "unacceptable," as determined through systematic review, into the assessment of each COU. However, on a case-by-case basis, EPA may have elected to exclude data where other more representative data were sufficiently available and preferred. The exposure estimates estimated by EPA should reflect potential conditions expected by workers including in underregulated and non-union worksites.

Summary: A public comment (DEHP-0139) noted that EPA cited "Draft Occupational Inhalation Exposure Data for Diethylhexyl Phthalate (DEHP)" as the source for occupational exposure data associated with each OES, but that the document was not available on the DEHP docket. The commenters request that this document be posted along with a notice of data availability.

EPA Response: Supporting data for the Draft Environmental Release and Occupational Exposure Technical Support Document for the DEHP Risk Evaluation were available upon request by emailing Docket-CustomerService@epa.gov and requesting the modeling files described in EPA-HQ-OPPT-2018-0433-0105. Updated files supporting the final DEHP Risk Evaluation will be similarly available.

Summary: A public comment (DEHP-0137) provided propriety information assumptions around the manufacture of DEHP within the risk evaluation and provided a flow diagram for process descriptions and exposures.

EPA Response: This process description information was used to revise the flow diagram from the draft risk evaluation.

Summary: A public commenter (DIBP-0131) notes that confidential occupational exposure monitoring data were submitted via EPA's CDX in September–October 2025, which supports the determination of no unreasonable risk for pre-catalyst manufacturing.

EPA Response: EPA acknowledges the receipt of this data, which was used to inform the assessment. EPA has determined that there are no unreasonable risks posed from occupational exposures to DIBP from use in pre-catalyst manufacturing.

Summary: A public commenter (DIBP-0131) disputes EPA's assumption that spray application occurs for DIBP-containing adhesives and sealants in industrial and commercial settings, stating no evidence has been identified for current spray uses. The comment goes on to derive different PODs (HED = 26.8 mg/kg-bw/day; HEC = 145.6 mg/m3) to calculate margins of exposure above benchmarks for all worker conditions of use except worst-case, high-end, unprotected spray applications of paints/coatings and adhesives/sealants for average workers and females of reproductive age. They contend these scenarios are not ongoing or reasonably foreseen and that a respirator with an assigned protection factor (APF) of 10 would mitigate the modeled risk if such spraying occurred.

EPA Response: EPA disagrees with the statement by the public commenter that spray application of adhesives and sealants does not occur in industrial and commercial settings. Though EPA did not identify a DIBP-specific adhesive or sealant product that is intended for spray application, phthalates are a common component of adhesive and sealant products that may be spray applied. Since DIBP is a reasonable alternative for other phthalates used in adhesive and sealant products, it is foreseeable that DIBP may be used in an adhesive or sealant product that is spray applied in industrial or commercial settings. Therefore, EPA considers the spray application of adhesive and sealant products containing DIBP a foreseeable use. EPA demonstrated that scenarios less extreme than worst case still result in MOE values below the benchmark. Specifically, in the risk characterization of DIBP, EPA calculates the MOE value for a worker exposed to high-end mist levels (*i.e.*, 95th percentile) but at a reduced product concentration from the central tendency (*i.e.*, 5% concentration rather than 60% concentration). The resultant MOE was well below the benchmark MOE of 30. Consequently, EPA is confident that a sprayable product containing DIBP may lead to an unreasonable risk to health of workers. The effects of respiratory protection on worker risk are shown in the risk characterization of DIBP.

Summary: A public commenter (SACC25-0132, SACC25-0145, DIBP-0133) stated that the EPA's consideration of personal protective equipment (PPE) when making a risk determination is inappropriate, as it is a risk management tool and therefore should be "decoupled" from any evaluation of risk. To support this statement, the commenter cites previous SACC guidance and a hierarchy of controls they contend EPA purports to follow. Furthermore, they state that considering the use of PPE underestimates occupational exposures due to practical considerations (*e.g.*, fit, training, and medical challenges) that impact proper and consistent use of PPE by workers. They also contend the impact of PPE is impossible to measure due to the challenge of sampling within a respirator facepiece. A similar public comment (DCHP-0128, DIBP-0130, DIBP-0133) states that the consideration of PPE within the risk evaluation is not consistent with "best available science" and should be limited to use as a risk management tool considering that PPE falls below hazard limitation, chemical substitution, engineering controls, and administrative controls in the NIOSH "Hierarchy of Controls."

Another public commenter (SACC25-0151) stated that EPA improperly assumes the use and effectiveness of PPE and ignores ingestion as an exposure route in occupational settings. DIBP-0133 states that "workers can be exposed to chemicals through ingestion while at work in multiple unintentional ways, including: nail biting, touching their mouth, chewing pens, removing PPE with their mouth, eating on the worksite, and smoking on break. EPA could use the Cherrie et al. (2006) conceptual model to assess inadvertent ingestion."

In contrast, another public comment (DCHP-0124) disagreed with "EPA's conclusion that it cannot assume that workers wear sufficient PPE or that EPA should make determinations of unreasonable risk based on scenarios that do not assume compliance with OSHA standards generally." The comment explains that "This assumption…. does not represent real-world, current workplace safety and health practices" and "results in highly conservative and overly restrictive risk management decisions for DCHP exposures."

EPA Response: EPA recognizes a range of opinions in the public comments regarding consideration of Personal Protective Equipment (PPE) within the evaluation of risk. 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) regarding the use of respiratory protection or other PPE. This allows EPA to make unreasonable risk determinations based on the available information regarding workers. In the risk evaluations of these five phthalates, the risk estimates calculated reflect use with and without PPE, including information on PPE that could be used to reduce exposures. EPA received some information about PPE use and practices from stakeholders during the public comment period for the draft risk evaluation and has incorporated it into the analysis where possible. The information received was from a few companies about PPE practices, but the information could not be generalized to be representative of all facilities associated with each respective COU. Because EPA has limited information regarding use of PPE under the COUs, the risk determinations are based on the risk estimates that do not reflect use of PPE. EPA generally does not evaluate occupational exposures through the oral route since the frequency and significance of this exposure route are dependent on several factors including the p-chem properties of the substance during expected worker activities, workers' awareness of the chemical hazards, the visibility of the chemicals on the hands while working, workplace practices, and personal hygiene that are difficult to predict. Further, inhalation and dermal exposures are encountered on a routine basis in the workplace, and therefore these routes are expected to comprise the majority of occupational exposures. Workers may inadvertently transfer chemicals from their hands to their mouths through various unintentional behaviors, such as nail-biting and eating on the worksite; however, these oral exposures are expected to be much lower than the dermal and inhalation exposures and therefore, the current aggregate analysis is expected to be protective of oral exposures.

Summary: A public commenter (SACC25-0145) asserted that, for certain occupational COUs, there are serious inconsistencies between EPA's risk estimates and EPA's conclusions regarding unreasonable risk, asserting that EPA repeatedly downplayed or disregarded the risks calculated using high-end exposure estimates without adequate scientific justification and instead used only central tendency estimates of DBP exposure and risk for workers in most conditions of use in its unreasonable risk determination. EPA is therefore disregarding unreasonable risks of non-cancer effects that may be faced by workers with exposures that are greater than median exposure levels, leaving 50% of the worker population unaddressed and at risk.

Similarly, another public comment (SACC25-0151) stated that, in the DEHP and DBP draft risk evaluations, EPA continues a "dangerous pattern of failing to consider high-end exposure scenarios in its unreasonable risk determinations". The commenter stated that EPA claim that central tendency was "most representative" lacked scientific justification for dismissing high-end estimates and ignores 50% of the workers in these COUs, and that this is not aligned with the obligations under TSCA to consider PESS and is not consistent with previous risk evaluations.

Another public comment (DCHP-0128) similarly disagreed with the selection of central tendency exposures as "most reflective of worker exposures." The commenter stated that high-end exposures have been used to inform unreasonable risk determinations in previous TSCA risk assessments, which is more appropriate for capturing populations with "greater exposure" as required under TSCA. As an example, the comment calls the selection of central tendency for PVC Plastic Compounding irrational and states that EPA should 1) utilize reasonable assumptions in the initial exposure, rather than discounting exposure based on uncertainty in underlying assumptions and 2) not arbitrarily label central tendency as most reflective of actual exposure in absence of certainty.

EPA Response: EPA believes that its conclusions regarding unreasonable risk for occupational COUs are consistent with its risk estimates. In order to illustrate the full range of possible worker exposures based on the reasonably available data, EPA estimated occupational exposures representative of central tendency and high-end exposure conditions. Central tendency estimates represent routine exposure scenarios and are most relevant for intermediate or chronic exposure estimates since these exposures may occur regularly at the estimated level. High-end exposure estimates represent extreme or worst-case scenarios and are most relevant for acute exposure estimates since these exposures are unlikely to occur with intermediate or chronic frequency. If there are sufficient monitoring data available, EPA typically calculates the 50th and 95th percentile exposure values from the data distribution to estimate central tendency and high-end exposure levels, respectively. However, if such data do not exist, as was often the case among COUs for phthalate chemicals, EPA relies on various modeling techniques to estimate central tendency and high-end exposure levels that may not directly correspond to 50th and 95th percentile exposure levels for a population. If an exposure is modeled with multiple assumptions that compound conservatism, the resulting high-end estimate serves only as an upper bound that does not necessarily represent a realistic level of exposure and the resulting central tendency overestimates typical levels of worker exposure. In such cases, EPA has examined the underlying assumptions and inputs of the exposure model to determine the most reasonable level of expected exposure based on all available data. EPA does not disregard high-end exposures, but rather, EPA considers how the range of exposure estimates was developed for each COU individually and determines the applicability of the high-end or central tendency exposure estimates.

Summary: A public commenter (SACC25-0132) stated that the assumptions of a central tendency value of 31-year working lifetime underestimates exposures to phthalates and provided evidence that a 40 year lifetime would be appropriate.

Another commenter (DIBP-0130, DIBP-0133) raised concern that EPA understated the length of time that workers are exposed to BBP and DIBP by relying on the assumption of a 31-year working lifetime and as such, dermal exposures to workers were underestimated.
EPA Response: EPA disagrees that the assumption of a central tendency value of 31-year working lifetime underestimates inhalation and dermal exposures to phthalates. EPA evaluated available data from the U.S. Census Bureau (2016) Survey of Income and Program Participation (SIPP) and U.S. Bureau of Labor Statistics Current Population Survey (CPS) data, which resulted in a 50th percentile

value of 31 years and a 95th percentile value of 40 years. The 50th and 95th percentile values are used for central tendency and high-end exposure estimates, respectively. Both sources of information (SIPP and CPS data) used to estimate worker lifetime years underwent evaluation based on the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances* (U.S. EPA, 2021) and was incorporated into the risk evaluation consistent with the best available science. Further, the quantitative values used for number of working years did not affect exposure calculations for phthalates since chronic cancer exposures were not assessed. Chronic non-cancer exposures for inhalation and dermal routes are estimated by the average daily dose (ADD), and the term for "working years per lifetime" occurs in both the numerator and denominator of the inhalation and dermal equations for ADD. Therefore, the claim that exposures were understated based on the number of working years determined by EPA is not valid.

Summary: A public commenter (DCHP-0128) stated that EPA did not sufficiently quantify risks to workers involved in DCHP distribution. Feedback from the DIDP SACC recommended long daily exposure periods for workers in distribution facilities and stated EPA did not provide evidence that distribution workers would have lower exposure than those involved in manufacture or import. In addition, the public commenter noted that high-end occupational exposures for DCHP for manufacturing and import did result in MOEs below benchmark.

A similar public comment (SACC25-0132, DIBP-0133) raised concerns that exposure to workers in warehouses where products containing DEHP, DBP, DIBP and BBP are stored as part of a TSCA requirement to consider distribution in commerce. The comment cited SACC as stating that these workers are "in constant proximity to phthalate-rich products" with long daily exposure periods, then stated that EPA should consider these workers PESS.

EPA Response: EPA disagrees that risks to workers involved in distribution or warehouses were not sufficiently quantified. Worker activities in distribution facilities and warehouses are expected to be similar to worker activities for the Repackaging COU and EPA has estimated inhalation and dermal exposure to workers who work in repackaging facilities. The risk evaluation of the phthalates (DCHP, DEHP, DBP, DIBP, and BBP) includes discussion on whether the central tendency or high-end exposure estimates are more reflective of potential worker exposures. For example, the central tendency values of exposure were found to be most reflective of worker exposures for the Manufacturing and the Import and Repackaging OES for DCHP as explained in section 4.3.2 of the Risk Evaluation for DCHP. The central tendency occupational exposure values were determined to be above the benchmark for DCHP Manufacturing and the Import and Repackaging OES. For distribution in commerce, EPA expects all the products and/or articles to be transported in closed system or otherwise to be transported in a form (*e.g.*, articles containing the chemical) such that there is negligible potential for releases except during an incident. Therefore, no occupational exposures are reasonably expected to occur, and no separate assessment was performed for exposures from distribution in commerce.

Summary: A comment (DIBP-0129) asks EPA to revisit unreasonable risk determinations and exposure assumptions of DIBP uses within the aerospace and defense (A&D) sector. The comment states that within the A&D sector, application of personal protective equipment (PPE) and engineering controls, are standard and reduce exposure to workers within this sector. It points to limited, but specific, uses that the commentor has identified within this sector as justification that uses of DIBP within A&D should be considered separate from industrial and commercial uses of DIBP as an adhesive and sealant.

EPA Response: EPA recognizes the comment regarding consideration of Personal Protective Equipment (PPE) within the aerospace and defense (A&D) sector for the evaluation of risk and unreasonable risk determinations. In the risk evaluations of these five phthalates, the risk estimates calculated reflect use with and without PPE, including information on PPE that could be used to reduce exposures. EPA received some information about PPE use and practices from stakeholders during the public comment period for the draft risk evaluation and has incorporated it into the analysis where possible. The information received was from a few companies about PPE practices, but the information could not be generalized to be representative of all facilities associated with each respective COU. Because EPA has limited information regarding use of PPE under the COUs, the risk determinations are based on the risk estimates that do not reflect use of PPE.

Summary: One commenter (BBP-0123) was unable to identify how 8-hour TWA for vapor exposures was calculated from data contained in ECJRC, 2007 for PVC plastics converting and compounding, because the data contained a mixture of individual data points and summary statistics without standard deviations. The comments did agree with EPA's use of monitoring data from Tarkett Inc. to derive worst-case scenario dust exposures for PVC materials.

EPA Response: When references provided a range for two samples, EPA used the maximum and minimum as distinct measurements. When references provided a range for more than two samples EPA used the mean value as a distinct measurement. When references indicated that a value was based upon one sample, EPA used the value as a distinct measurement. EPA then used all of the EPA identified distinct measurements to create a distribution of exposures and calculated the 95th and 50th percentiles of said distribution.

Scope of Occupational Exposure Assessments

Summary: A public comment (SACC25-0132, DIBP-0133) stated that "EPA violated TSCA by refusing to identify and evaluate risks to fire fighters as a potentially exposed or susceptible subpopulation." The commenter stated that DEHP, DBP, DIBP and BBP are present in a variety of solid articles and that these phthalates readily transfer from burning materials to firefighter protective gear during fire suppression (references provided) leading to dermal exposure, inhalation, and ingestion. The commenter notes that firefighters were identified as PESS in a previous risk assessment (Asbestos Part 2).

EPA Response: EPA reviewed the references provided by the public commenter (SACC25-0132, DIBP-0133), including Barbara and Baxter (2014) and Poutasse et al. (2022). While these studies provide evidence of potential exposure to phthalates through personal protective clothing (Barbara and Baxter, 2014) and through silicone dog tags used as a passive sampling device (Poutasse et al. 2022), these studies do not provide direct evidence of increased phthalate exposure, as would be

demonstrated through urinary biomonitoring. EPA identified one reasonably available urinary biomonitoring study of firefighters conducted by Biomonitoring California. The study, titled the Firefighter Occupational Exposure (FOX) Project evaluated urinary metabolites levels of several phthalates (BBP, DBP, DEHP) in 101 firefighters (99 males, 2 females; mean age 42.8 years) in southern California between 2010 to 2011 (https://biomonitoring.ca.gov/projects/firefighter-occupational-exposures-fox-project). EPA compared the urinary biomonitoring data from the FOX project to 2009-2010 and 2011-2012 NHANES urinary biomonitoring data to determine if firefighters have elevated phthalate exposure. As can be seen from the table below, based on the limited data from the FOX project, firefighters do not appear to have elevated exposure to DEHP, or DBP compared to the general U.S. population (NHANES), while BBP exposure levels appeared slightly elevated compared to NHANES. Overall, no clear patterns in elevated DEHP, DBP, and BBP exposures for firefighters were apparent in the limited data from the FOX project. EPA does not expect there to be a different outcome for DIBP. With no reasonably available information clearly demonstrating increased phthalate exposure, EPA does not believe firefighters should be considered a PESS in any of the above-mentioned phthalate risk evaluations, as EPA has no evidence of increased exposure to this subpopulation.

Urinary Metabolite Measured	**Parent Phthalate**	**Study**	**Sample Size**	**Urinary Concentration (in µg/L)**			
				Geometric Mean (95% CI)	**50th Percentile**	**75th Percentile**	**95th Percentile**
Mono-benzyl phthalate (MBzP)	BBP	FOX (2010-11)[a]	101	8.18 (6.56, 10.2)	7.76	14.9	41.3
		NHANES (2009-10)[b]	1,914	5.61 (4.97, 6.34)	5.95	12.7	39.6
		NHANES (2011-12)[b]	1,705	3.98 (3.60, 4.39)	4.0	9.70	28.0
Mono-(2-ethyl-5-carboxylpentyl) phthalate (MECPP)	DEHP	FOX (2010-11)[a]	101	12.3 (10.1, 14.9)	12.4	23.2	58.3
		NHANES (2009-10)[b]	1,914	19.4 (17.0, 22.0)	18.8	37.7	126
		NHANES (2011-12)[b]	1,705	12.2 (11.3, 18.0)	13.0	23.8	61.8
MnBP	DBP and BBP	FOX (2010-11)[a]	101	10.6 (8.83, 12.8)	10.6	20.2	40.6
		NHANES (2009-10)[b]	1,914	13.5 (12.0, 15.1)	14.5	28.1	68.9
		NHANES (2011-12)[b]	1,705	7.04 (6.04, 8.21)	8.2	18.2	46.4

[a] FOX Project Data from: https://biomonitoring.ca.gov/results/projects/410#Study%20Group

[b] NHANES Urinary biomonitoring data for all adults 20+ years of age from: https://www.cdc.gov/exposurereport/data_tables.html

Summary: A public comment (SACC25-0132, DIBP-0133) recommended that EPA consider occupational exposure to DEHP and DBP for truck drivers, ride share drivers, and others who work in vehicles as these phthalates can volatize from automotive articles within vehicles resulting in high indoor air concentrations. The commenter identified these workers as PESS due to extended exposure times (*e.g.,* truck drivers sleeping in vehicles).

On risk to truck drivers, BBP and DIBP are used in "automotive articles," including car mats and various components of a vehicle's interior, EPA failed to evaluate exposures and risks to truckers, ride share drivers, and others who spend the most time within vehicles and have the greatest exposures from those articles. "EPA calculated risks from "automotive articles" based on the "fabrication of final product from articles" occupational exposure scenario. But that scenario is not specific to vehicle exposures, and it assumes a maximum of 250 days per year of occupational exposure. Long-haul truck drivers are often on the road for at least 300 days per year and may drive ten times more than the average person in the United States. Truck drivers also often sleep within their vehicle cabs, extending their exposure to BBP or DIBP well beyond a standard eight-hour workday. Taxi drivers "usually work 6 days a week," or more than 300 days per year, "for 12-16 hour[s] per day."

EPA Response: EPA agrees that truck drivers, ride share drivers, and others who work in vehicles may have exposures from automotive articles within vehicles. However, these exposures are characterized under the consumer exposure assessment. The commercial uses of automotive articles characterize exposures from the fabrication or installation of the automotive articles, and EPA estimates that workers at fabrication or automotive facilities work 5 days per week, 50 weeks per year.

In the consumer and indoor dust exposure assessment, EPA estimated indoor exposures to DBP, DEHP, BBP, and DIBP in automobile cabins, based on relevant TSCA Conditions of Use (COUs) and exposure routes for drivers and passengers. Although the assessed durations and frequencies of exposure for these scenarios are different than those noted by the commenter for professional drivers, these scenarios were evaluated and summarized in the Weight of Scientific Evidence Summary Per Consumer COU tables within the Consumer and Indoor Dust Exposure technical support documents for each of these phthalates. EPA only identified one source that confirms the use of DEHP, DBP, DIBP, and BBP in car interiors - car mats. EPA recognizes that synthetic leather materials can be used in car interiors, but such confirmation was not captured by the extensive systematic review process used. The TSDs have clarifying statements in the indoor assessment section to explain that the indoor consumer articles exposure scenarios were modeled with stay-at-home parameters (21 hours of inhalation and dust ingestion was used for exposure duration and for chronic exposures 365 days was used as the exposure frequency) that consider use patterns similar to or higher than those in other indoor environments, including vehicles. Therefore, EPA concludes that exposures to similar articles in other indoor environments are included in the indoor assessment as a health protective upper bound scenario. The information is described in Section 2.1 in the DEHP, DBP, DIBP, and BBP consumer exposure assessment TSDs.

Below is a summary for each phthalate:

- DBP: Two scenarios were assessed under the "Other uses; Automotive articles" COU for different use patterns: car mats and synthetic leather seats. Both scenarios were part of the indoor assessment and evaluated for all exposure routes except mouthing. The EPA used a dermal flux approach, assuming that dermal absorption of DBP from solid objects would be limited by its aqueous solubility.
- BBP: One scenario was assessed under the "Other uses; Automotive articles" COU for car

mats. This scenario was part of the indoor assessment and evaluated for all exposure routes except mouthing. For dermal exposure, the EPA used a dermal flux approach, estimated based on *in vitro* dermal absorption of BBP in humans.

- DIBP: Seven scenarios were assessed under the "Other articles with routine direct contact during normal use" COU, including air beds, car mats, in-place wallpaper, wallpaper installation, shower curtains, tire crumb and artificial turf, and various PVC articles with routine contact. Car mats were considered in the indoor assessment for all exposure routes except mouthing. The EPA used a dermal flux approach to assess dermal exposures to DIBP, assuming that dermal absorption from solid objects would be limited by its aqueous solubility.
- DEHP**:** Two indoor scenarios were assessed for the "Other - Automotive articles" COU, involving car mats and tire replacements (only assessed for dermal exposures). The car mat scenario was part of the indoor assessment and was evaluated for all exposure routes except mouthing. The EPA used a dermal flux approach to assess DEHP dermal exposures.

Ingestion Exposure

Summary: Public commenters (DEHP-0138, SACC25-0132, SACC25-0145) noted that EPA considered aggregate exposure to DBP and DEHP by combining worker exposure estimates for the inhalation and dermal routes of exposure. However, EPA ignored ingestion as a route of exposure for workers. These commenters asserted that workers can ingest chemicals through various unintentional behaviors, such as nail-biting and eating on the worksite. They noted EPA must use best available exposure models to quantify these ingestion exposures and provide a more comprehensive risk assessment. One commenter (SACC25-0132) suggested the use of an existing conceptual model (Ng et al. 2012) to assess ingestion exposures.

A similar comment (SACC23-0052, DIBP-0130) raised the same recommendations to include oral exposure in occupational settings within the context of the cumulative risk assessment.

EPA Response: EPA does not believe that the addition of oral exposure to the aggregate exposure will result in a more comprehensive risk assessment. EPA generally does not evaluate occupational exposures through the oral route since the frequency and significance of this exposure route are dependent on several factors including the p-chem properties of the substance during expected worker activities, workers' awareness of the chemical hazards, the visibility of the chemicals on the hands while working, workplace practices, and personal hygiene that are difficult to predict. Further, inhalation and dermal exposures are encountered on a routine basis in the workplace, and therefore these routes are expected to comprise the majority of occupational exposures. Workers may inadvertently transfer chemicals from their hands to their mouths through various unintentional behaviors, such as nail-biting and eating on the worksite; however, these oral exposures are expected to be much lower than the dermal and inhalation exposures and therefore, the current aggregate analysis is expected to be protective of oral exposures.

Inhalation Exposure

Summary: A public comment (DBP-0129) disagreed with EPA's use of National Institute of Occupational Safety and Health (NIOSH) data to derive risk estimates for workers in the Use of laboratory chemicals (liquid) and Use of lubricants and functional fluids Occupational Exposure

Scenarios (OES). The commenter identified several other sources of workers' inhalation exposure data for these OES that they recommended EPA use instead. The commenter stated that an assigned protection factor (APF) of 10 will be sufficient to protect workers exposed to DBP via inhalation if their suggested sources of data are used to update occupational exposures and human health hazard.

A public comment (DEHP-0135) identified COUs where workers' inhalation MOEs were exceeded for DEHP occupational exposures, and conducted a review of the relevance of the exposure data used to derive risk. The commenter also reviewed the assigned protection factor (APF) of the personal protective equipment (PPE) that would be required to protect workers at EPA's draft exposure estimates for DEHP. The commenter agreed with EPA's approach for several of the listed COUs, but disagreed with the approach for others:

- The commenter questioned whether the inhalation exposure value for the Rubber Product Manufacturing COU was representative of actual exposures to workers with modern manufacturing facilities and processes, because "it is not appropriate for EPA to use data from the EU with measured values above 5 mg/m^3 (the current enforceable Permissible Exposure Limit (PEL) under the U.S. Occupational Safety and Health Administration (OSHA)).
- The commenter questioned the use of the rubber manufacturing 8-hour TWA to derive risk from exposure under the Non-Spray Application of Paints, Coatings, Adhesives, and Sealants and Use of Dyes, Pigments, and Fixing Agents COUs, and proposes an alternative derived from OSHA CEHD data. The commenter provided a table of currently available OSHA CEHD data for DEHP for several COUs, and derived 8-hour TWA values using an assumed Limit of Quantification (LOQ) of 0.055 mg/m3 as defined in OSHA 104.
- The commenter "does not agree with EPA's use of the Automotive Refinishing Spray Coating Mist Inhalation Model for the Formulation for the Diffusion Bonding COU. Diffusion bonding uses heat and pressure to join materials and potential occupational exposures are not comparable to application of spray coatings."

Another public comment (DEHP-0139) identified OSHA CEHD data for DEHP, and stated that declining detections for DEHP over time, and especially since 1999, show that facilities have "modernized and improved their workplace environments to minimize exposure to DEHP". The commenter requests that EPA change its assumptions for occupational exposure to DEHP for several COUs based on updated industrial hygiene data which was provided in the comment. The commenter also states that EPA's evaluation of studies that provided the basis for occupational exposures in the PVC Plastic Compounding OES was incorrect. Additionally, the commenter states that EPA's use of monitoring data for ONU exposure is inconsistent for this OES. The commenter states that these issues, plus the use of CEHD and NIOSH Health Hazard Evaluation data collected as a result suspected health hazards or workplace complaints, mean that EPA's analysis is not ideal for characterizing typical exposure and states there is a lack of transparency and replicability in EPA's analysis.

The commenter (DEHP-0139) stated that inhalation dust exposure for plastics compounding and converting should reflect accurate DEHP concentrations, based on information provided by the commenter, because "not all dusts in a DEHP compounding operation or dusts measured in facilities that process flexible PVC compounds into a finished article will contain DEHP". The commenter further states that migration coefficients of DEHP from particles will vary depending on several conditions including the hardness of the PVC material, and that EPA should adjust its modeling to reflect this. The commenter also provides information about the Recycling and Waste Management OES and states that EPA should use more recent data to revise its dust exposure

estimates for DEHP in this OES, because most PVC materials are rigid and do not contain plasticizers and recyclers do not typically commingle rigid and flexible plastics. Further, the commenter states that DEHP use has declined considerably in flexible PVC plastics, and states that EPA has erroneously characterized recycled mixed plastics as being for PVC materials.

EPA Response: EPA believes that using the National Institute of Occupational Safety and Health (NIOSH) monitoring data was appropriate to estimate worker exposures for the Use of laboratory chemicals (liquid) and Use of lubricants and functional fluids Occupational Exposure Scenarios (OES). EPA is required under TSCA to consider all reasonably available information such as NIOSH data. The NIOSH data underwent evaluation based on the Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances (U.S. EPA, 2021) and was incorporated into the risk evaluation consistent with the best available science. EPA considered other sources of workers' inhalation exposure data identified by the commenter but found the NIOSH data to be more appropriate for these occupational exposure scenarios. Although the effect of assigned protection factors (APF) provided by PPE on the MOE values was discussed in the risk evaluation, but EPA did not include PPE assumption in its occupational risk assessment for workers and ONUs. This allows EPA to protect any workers who do not use occupational exposure controls and identify any risk that may exist for these workers.

The OSHA CEHD data suggested by the commenter was not able to be incorporated into EPA's inhalation exposure estimates for workers because the commenter did not provide background information that would allow the Agency to reproduce the industry sector mapping/binning or exposure calculations. Additionally, the data (companies name, industry sector, etc.) that industry provided maps to the plastic converting OES in the RE and not rubber manufacturing. Finally, the public commenter cited one facility which they state provides exposure estimates for Dyes, Pigments, and Fixing Agents. These OSHA CEHD data show 1 blank and 4 non-detects.

EPA acknowledges the commenter's questioning the use of the rubber manufacturing 8-hour TWA to derive risk from exposure under the Non-Spray Application of Paints, Coatings, Adhesives, and Sealants and Use of Dyes, Pigments, and Fixing Agents COUs. To address the uncertainty regarding the relevance of the rubber calendering operation at high temperature (200 °C) as a surrogate for non-spray application of paints, coatings, adhesives, and sealants, EPA compared the dose from non-spray application of application of paints, coatings, adhesives, and sealants for DEHP (based on rubber calendering) to another phthalate (DBP) which had inhalation monitoring data for non-spray application of paints, coatings, adhesives, and sealants from operations at ambient temperatures. The DBP data were comprised of two full-shift PBZ monitoring samples in OSHA's CEHD from two different inspections one from 2011 of a fabric coating mill and one from a janitorial services company (OSHA, 2019) and an additional twelve 8-hour TWA monitoring samples by Rohm and Haas Co. (1990). EPA characterized the data by taking the 50th percentile and the 95th percentile of the combined dataset to represent the central tendency and high end. Importantly, the DEHP inhalation dose to females of reproductive age using rubber calendering data as a surrogate for non-spray applications of paints, coatings, adhesives, and sealants (0.23 mg/kg-day at central tendency and 1.12 mg/kg-day for high-end is very similar to DBP for this same OES (0.21 mg/kg-day at central tendency and 1.02 mg/kg-day for high-end, thereby increasing EPA's confidence in the use of rubber manufacturing inhalation monitoring data as a surrogate for non-spray application of paints, coatings, adhesives, and sealants. However, while EPA has moderate confidence in these data when used to estimate worker exposure to rubber manufacturing, EPA's confidence in these same data to estimate exposure for non-spray application of paints, coatings, adhesives, and sealants is lower, at slight to moderate, given the use of a surrogate OES.

Additionally, the Consortia cites the PNOR respirable dust standard (5 mg/m^3). However, the total dust PNOR standard is 15 mg/m^3, which the exposures for Rubber Manufacturing in the draft risk evaluation are well under.

The letter provides the consortium's independent review of OSHA CEHD data relevant to several OES (Paints/Sealants/Adhesives/Coatings, Rubber Manufacturing, Plastic Compounding, Plastic Converting, etc.). The consortium presented a summary of the CEHD data and not the raw data and calculations which could then be independently verified by EPA. An extended timeframe is needed to allow for an OES-by-OES comparison of the EPAs evaluation of the OSHA CEHD data and the consortium evaluation of OSHA CEHD data. This would require verification of the exposures organized by NAICS and confirmation that the industry sector exposure information extracted from the OSHA CEHD by the consortium matched the industries extracted by EPA, etc. The approach that EPA used for identifying and incorporating OSHA CEHD data is well documented and consistent with past risk evaluations.

Regarding the exposure estimates from Rubber manufacturing OES used as a surrogate for the Diffusion Bonding: EPA makes no claims to the similarities of the industrial processes or applications, simply the relative comparability of the air concentrations for the purposes of this risk evaluation in the absence of OES-specific exposure data. Further, EPA discusses both the strengths and uncertainties with the use of the surrogate data and, despite these uncertainties, considers these data to represent the best available science:

> *EPA used SDSs and product data sheets from identified DEHP-containing products to identify product concentrations, which were then applied to the surrogate mist data to estimate DEHP-specific exposures. The primary limitation is the lack of DEHP-specific monitoring data, with the ESD serving as a surrogate source of monitoring data representing the level of exposure that could be expected at a typical worksite for the given spray application method. The inhalation monitoring data used were specific to the spray application of coating materials, so the estimates may not be representative of exposure during other application methods. Additionally, it is uncertain whether the substrates coated, and products used to generate the surrogate data are representative of those associated with DEHP-containing diffusion bonding formulations.*

Regarding the comment that stated that EPA should revise its dust exposure estimates for DEHP in the "Recycling and Waste Management OES" because: most PVC materials are rigid and do not contain plasticizers; recyclers do not typically commingle rigid and flexible plastics, DEHP use has declined considerably in flexible PVC plastics; and EPA has erroneously characterized recycled mixed plastics as being for PVC materials: EPA asserts that the SDS for Vinoprene 647 (which includes 44% DEHP) is comprised of Acrylonitrile-butadiene copolymer with polyvinyl chloride/ Nitrile polymer blend, which is intended to represent plastic waste and therefore relevant to this OES "Waste handling, treatment, and disposal", not specifically plastic for recycling, which would be covered under the "Recycling" OES. Furthermore, EPA used current industry-specific dust levels based on NAICS codes relevant to this OES, along with the weight fraction in the SDS for Vinoprene 647, considered representative of plastic waste streams that workers may encounter in this OES.

Finally, EPA has evaluated the more recently collected occupational exposure data provided by the submitter and has incorporated this data into the final DEHP risk evaluation. The impacted OES are: plastic compounding, plastic converting, and recycling.

Summary: A public commenter (SACC25-0132, DIBP-0130, DIBP-0133) stated that EPA, when estimating inhalation exposures from monitoring data, was unreasonable in their assumption of zero worker inhalation exposure during the remainder of an 8-hour shift, when the length of the sampling time was 5.5 to less than 8 hours. The commenter states that EPA had no reason to assume that exposure stops when sampling ends, and that doing so is not supported by EPA's own guidelines for Statistical Analysis of Occupational Exposure Data.

EPA Response: EPA agrees that exposure may continue after the sampling time ends in cases when the sampling time was 5.5 hours to less than 8 hours. But, in absence of reasonably available information, EPA assumed exposures were zero for the remainder of the work shift. EPA uses the highest tier data and incorporates it into the exposure estimates in accordance with the EPA Guidelines for Statistical Analysis of Occupational Exposure Data methods for calculating exposure periods less than 8 hours. Higher tier data would provide associated meta-data on worker tasks which would help inform if the sampling duration coincided with a given task in which that exposure would be expected for that duration. In the absence of that information, EPA generally follows the procedure that any sample duration less than 5.5 hours represents a short sampling interval, but that the exposure would be expected to remain consistent throughout an 8-hour work shift; any sample duration 5.5 to 8 hours is assumed to represent the actual duration of exposure; therefore, EPA calculates the 8-hour TWA considering any time outside of the sampling duration to be zero exposure. EPA is currently exploring more health protective approaches and may consider alternative approaches for future assessments for other chemicals.

Occupational Non-Users (ONUs)

Summary: A public comment (DBP-0131) said that EPA may have underestimated exposure to some Occupational Non-Users (ONUs) who have higher peak or average exposures than production workers. The commenter provided the example of workers engaged in the occasional maintenance, repair, or cleaning of machines or containers. The commenter stated that the exposure of these ONUs should be analyzed separately from both production workers and workers with only incidental exposure.

EPA Response: A primary difference between workers and ONUs is that workers may handle the chemical and have direct contact with the chemical, while ONUs are working in the general vicinity of workers but do not handle the chemical and do not have direct contact with the chemical being handled by the workers. Activities such as maintenance, repair and equipment cleaning may lead to direct contact with the material. Therefore, potential exposures from these occupational activities are characterized by worker exposure levels rather than ONU exposure levels.

Summary: A public comment (DBP-0127) stated that "EPA should develop a clear and transparent process for developing OEVs and existing chemical exposure limits (ECELs) under TSCA that incorporates peer review of the approach and the ultimate outcome (value)" and cited two examples (*i.e.,* that solicited "public feedback regarding the technical aspects of their derivation and the benefits and cost associated with their implementation") of a "detailed, transparent, and credible process": 1) the development of the Acute Exposure Guideline Levels (AEGLs), and 2) OSHA's final rule amending its existing standards for occupational exposure to beryllium and beryllium compounds.

EPA Response: EPA appreciates the suggestion for a more transparent process. The chemical occupational exposure value is intended to summarize the occupational exposure scenario and sensitive health endpoints into a single value. 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. By making the OEVs available in the draft risk evaluation, EPA is open to public and peer reviewer comments on the values. Any ECEL used for occupational safety risk management purposes could differ from the occupational exposure value presented in the risk evaluation based on additional consideration of exposures and non-risk factors consistent with TSCA section 6(c).

Summary: A public comment (DCHP-0126) states "EPA has not identified a validated method for detecting DCHP or cumulative exposure to other phthalates in air, and thus it is unclear how EPA will enforce compliance with an OEV for DCHP or for combined phthalates. A validated method must be in place prior to implementation of any regulatory limits." The comment continues "The consortia recognize that there is a validated air monitoring method for DBP, and a partially validated method for DEHP. There are air monitoring methods for DIBP, BBP, and DCHP reported in the peer-reviewed literature. We also provide a link to a peer-reviewed journal publication that describes a validated method for the determination of phthalates in indoor air, designed to detect several phthalates (DIBP, DBP, BBP, and DEHP), with a limit of detection (LOD) at 4 to 45 ng/m^3. Additional literature searches confirm other methods capable of detecting DCHP in the nanogram range from air samples. We urge EPA to develop and validate a method for the remaining phthalates prior to implementation of any regulatory limits on individual or cumulative phthalates."

EPA Response: EPA appreciates the information regarding available monitoring methods and potential limitations for phthalates undergoing TSCA risk evaluation. EPA acknowledges that government-validated analytical methods are available for detecting DBP and DEHP in air (NIOSH Method 5020 for DBP and OSHA Method 104 for DBP and DEHP), and that government-validated methods are not available for DCHP, BBP, or DIBP. With the publication of these final risk evaluations and the identification of unreasonable risk, EPA will be moving forward with risk management by applying one or more of the requirements under TSCA section 6(a) to the extent necessary so that the chemical substance no longer presents an unreasonable risk. During the risk management stage, EPA will consider a variety of relevant factors in choosing one or more of the available TSCA section 6(a) regulatory tools, including the activities that would be involved in each condition of use, the availability of engineering and other controls to reduce exposure, the limitations of respirators, the availability of substitutes, etc.

4.2 Consumer and Indoor Exposures

Summary: A public comment (DIBP-0089) stated that "EPA should evaluate DIBP presence in consumer products beyond plastics—such as personal care items, adhesives, and flooring—particularly in dollar stores or off-market imports where regulatory controls may be weak."

EPA Response: EPA did not evaluate specific uses of DIBP not regulated by TSCA such as personal care items (*i.e.*, cosmetics). However, EPA evaluated exposures from adhesives and flooring. Specifically, EPA assessed adhesives within the Adhesives and Sealants COU and identified four products with various use patterns, see Section 2.1.2 in the Consumer Exposure Assessment Technical Support Document (Consumer TSD). Some of the identified adhesive products are used in amounts that cover large surface areas and hence inhalation and dermal exposures were assessed. Adhesives used in small amounts with small surface area application were assessed for dermal exposures. In addition to flooring adhesives, EPA assessed dermal, inhalation, and ingestion exposures from vinyl flooring, see Section 2.1.1 for description of weight fraction data sources, and Sections 2.2.3 and 2.3.5 for inhalation/ingestion and dermal modeling approaches and parameterization of the models, respectively.

With regards to the dollar stores or off-market imports, see Section 2.1.1 in the Consumer TSD. EPA specifically used data that were reported from two studies conducted by The Ecology Center, a nonprofit, on carpets (Changing Markets Foundation, 2018) and articles purchased from dollar stores (Ecology Center, 2015). Additionally, EPA used data from the High Priority Chemicals Data System (HPCDS) (WSDE, 2020), a database compiling manufacturer reporting requirements from 2017 to 2024 per Washington and Oregon safe children's product regulations.

Additionally, as part of the current risk evaluations of DCHP, DEHP, DBP, DIBP, and BBP, EPA evaluated NHANES urinary biomonitoring data using reverse dosimetry. NHANES provides an estimate of aggregate background phthalate exposure for the U.S. population. Although, EPA cannot attribute exposure measured in NHANES to specific routes or sources of exposure, U.S. CPSC (2014) and Health Canada (2020) have found that the main sources of exposure to phthalates is through diet (non-TSAC use), certain cosmetics (non-TSCA use), indoor air, and indoor dust. To capture highly exposed populations in the analysis, EPA used the 95th percentile cumulative exposure for black non-hispanic women of reproductive age as the basis of its cumulative risk assessment of phthalates (*i.e.*, EPA used the highest exposure estimate [95th percentile] for the highest exposed population (black non-hispanic women of reproductive age)). For TSCA decision support purposes, EPA believes it has appropriately accounted for exposures from sources not regulated under TSCA.

Summary: A public commenter (SACC25-0132, SACC25-0145, SACC25-0151, DIBP-0133) stated that EPA must consider aggregate (combined) exposures to chemicals in plastic parts and products in vehicles for DEHP, DBP, and BBP (including products not assessed such as vehicle upholstery or car parts), as there is a high likelihood of combined exposures from multiple products within vehicle interiors. They stated that off-gassing should be considered in addition to exposure via interior vehicle dust, including the consideration of potentially elevated volatility and emission rates for these chemicals due to the high temperatures possible within vehicles.

EPA Response: EPA is assessing aggregate exposures in the DEHP and DBP risk evaluations but is only pursuing aggregation across exposure routes for each COU and not across COUs or multiple products within a COU due to the lack of reasonably available information on the representativeness

of such results. To include an aggregate assessment across COUs or multiple products within a COU would not be consistent with making a risk determination based on the best available science or be based on the weight of scientific evidence due to these data gaps.

EPA only identified one source that confirms the use of DEHP, DBP, and BBP in car interiors - car mats. EPA recognizes that synthetic leather materials can be used in car interiors, but such confirmation was not captured by the extensive systematic review process used. The TSDs have clarifying statements in the indoor assessment section to explain that the indoor consumer articles exposure scenarios were modeled with stay-at-home parameters that consider use patterns similar to or higher than those in other indoor environments, including vehicles. Therefore, EPA concludes that exposures to similar articles in other indoor environments are included in the residential assessment as a health protective upper bound scenario. The information is described in Section 2.1 in the DEHP, DBP, and BBP consumer exposure assessment TSDs which describe clothing items like coats, raincoats, belts, and accessories.

Summary: A public comment (DCHP-0128) recommends that EPA further consider exposure to DCHP through dust either by evaluating specific conditions of or use or by applying the 95th percentile of dust measured in Dodson et al. 2015 to non-dust exposures. The commenter did not feel that the dust exposures estimated by the Consumer Exposure Model (CEM) showing dust exposures 4 orders of magnitude below the POD for a generic 1 m^2 article with 30% DCHP was sufficient justification to eliminate evaluation of smaller articles or combinations of articles. Additionally, the public commenter recommended evaluating dust exposure for surface areas > 1 m^2 be considered in the consumer assessment for articles such as furniture and flooring.

EPA Response: DCHP dust monitoring data is limited and therefore has limitations in terms of its representativeness of actual dust concentrations in U.S. homes. Of the three identified studies the commenter recommended to use Dodson et al. (2015) to calculate exposure doses and risk. This study reported 49 settled dust samples collected from homes in California during 2006. Samples were collected by slowly dragging a vacuum crevice tool just above the surface of rugs, upholstery, wood floors, windowsills, ceiling fans, and furniture in the primary living areas of the home for about 30 minutes. The low detection frequency reported, 16 percent, indicates that either DCHP is in very low concentrations in indoor dust and thus, not detectable, or the analysis method was limited. This agrees with the use of a generic scenario (items smaller than 1m^2 and 30% DCHP content) to support not assessing small items for dust inhalation. The low number of samples within some of the studies, and few localities, are used to assign a slight confidence in the overall use of these data for risk estimates or representative of the U.S. population. Because the monitoring data was not found to be representative of the U.S. population and was not apportioned to DCHP-containing items and COUs, further analysis or comparison of the monitoring data to modeling data was not conducted.

EPA did not assess furniture and flooring in DCHP consumer exposure because there is no evidence that these articles contain DCHP. See Section 2.1.1 of the consumer exposure TSD for a summary of identified article examples and references.

Summary: A public commenter (SACC25-0132, SACC25-0145, DIBP-0133) stated that the assumption used by EPA of at "at-home time" of 20 or 21 hours when assessing indoor exposure to DEHP, DBP, DIBP, and BBP underestimates exposure. They state that this assumption fails to consider individuals who spend 24 hours at home or individual who spend time in other indoor environments containing phthalates (including those with higher phthalate concentrations than homes).

EPA Response: EPA agrees that using 24-hour exposure duration would result in larger exposure doses. The assumption of the stay-at-home scenario parameter of 20 hours per day spent in the home is applied to all of the high, medium, and low intensity use scenarios, ensuring a protective duration of time spent in the home. EPA agrees that some people may remain indoors for the entirety of the 24-hour period or move to other indoor environments with multiple sources of phthalate containing products and articles. The individual COU assessment provides a range of representative use patterns rather than extreme use patterns.

The TSDs have clarifying statements in the indoor assessment section to explain that the indoor consumer articles exposure scenarios were modeled with stay-at-home parameters that consider use patterns similar to or higher than those in other indoor environments, including vehicles. Therefore, EPA concludes that exposures to similar articles in other indoor environments are included in the residential assessment as a health protective upper bound scenario.

Summary: Public commenters (DEHP-0138, SACC25-0132) asserted that smaller living spaces (*e.g.*, apartments and mobile housing) result in greater inhalation exposures of DBP and DEHP than larger living spaces (*i.e.*, 492 cubic meter home). The commenters stated that "EPA must use a more representative home size, such as 154 cubic meters, to accurately assess risks for individuals in smaller living environments."

Another commenter (DIBP-0130, DIBP-0133) stated that EPA understated exposures for people in vehicles, in indoor environments with higher-than-usual levels of BBP and DIBP; and in apartments, mobile housing, and other smaller homes. EPA underestimated exposures to consumers and workers from BBP in vehicles. BBP is widely used in automotive articles, including in vehicle trim and ornamentation, air bags, electronic controls, radiator grille assemblies, reading light assemblies, and car mats. In the Draft BBP Risk Evaluation, however, EPA only considered consumer exposures from car mats. EPA offered no explanation for its failure to calculate risks from the other sources of BBP within vehicles.

EPA Response: While it is true that residential and indoor spaces volume is variable, the total volume is not the only parameter/input that is used in calculating exposures. When considering a change in volume the air exchange rate and interzone ventilation rate also scale accordingly to the total volume. The consideration of the CEM default whole house volume is used for articles like carpets, vinyl flooring, and wallpaper scenarios in which EPA alters the total surface covered by the article as a percent of the total surface in the selected indoor environment volume. While the volume can be decreased, other parameters would also decrease accordingly and essentially scale down. EPA agrees that some indoor environments and households have larger presence of phthalate-containing products and articles, this variability and the potential sources (TSCA and non-TSCA) was captured and discussed in the indoor assessment, see Section 4 of the Consumer TSD.

Summary: A public commenter (SACC25-0132, DIBP-0133) stated that EPA failed to consider foreseeable dermal exposure to phthalates in articles in the home, and provided a single example of the failure to account for exposures in adults associated with dermal contact with plastic toys during play with children or cleaning.

EPA Response: EPA agrees that adults have direct dermal contact with children's toys but does not agree that EPA failed to account for adults. Infants and toddlers are the populations most at risk when considering exposures from children's toys. Dermal doses for young teens and teenagers were modeled using the same exposure duration than for infants and toddlers, even though the use of those durations may be an overestimation for those populations, and the dose values were smaller for teens than for younger children. Similarly, adults' dermal exposure to children's toys would also result in smaller dose values than for infants. See each of the phthalates Consumer Risk Calculator spreadsheets, the Acute, Chronic, and Aggregate tabs, the children's toys (new and legacy) doses and MOEs which show the pattern described above.

Summary: A public comment (SACC23-0052) expressed concern that EPA relies too heavily on product formulation data from manufacturer websites or safety data sheets to identify the presence and weight fraction of phthalates, stating that it may be biased by manufacturer reporting. Instead, the comment recommends EPA leverage information collection authorities to obtain product formulation information directly from manufacturers.

EPA Response: EPA agrees with the commenter that the consumer assessment relies heavily on manufacturer websites and safety data sheets (SDS) but does not agree that the use of manufacturer SDS is a problem. EPA uses reported concentration ranges for each product and calculates exposures based on the best available data and product specific use instructions and patterns to provide representative and health protective risk estimates. The use of concentrations that may be higher than actual concentrations would result in risk overestimations, and it is the responsibility of the manufacturer to correct the information that is publicly available. EPA relies on the best available science when conducting risk evaluations, EPA integrated all reasonably available product and article specific information into its risk evaluations. EPA strives to obtain and utilize the most relevant data for all conditions of use and urges commenters to submit relevant data during the risk evaluation process. EPA performed a search of SDS, existing assessments, CDR, and product testing databases to identify product and article examples for each COU. This search is described in Section 2 of the Consumer and Indoor Exposure Assessment Technical Support Document and the Consumer Exposure Analysis spreadsheet in the supplemental files. Sources of information and description of product and article use patterns are also available in these files.

Summary: A public comment (DIBP-0133) expressed concern that EPA did not consider the contribution of BBP-containing paints and coatings used on presumptively large and sensitive locations within the home in its Indoor Dust and Exposure Assessment.

EPA Response: As noted in Section 2.1.2 and Table 2-1 of the Consumer and Indoor Dust Exposure technical support document, EPA assessed potential consumer inhalation and dermal exposures to BBP from paints and coatings used in large-scale indoor home repair activities. These exposures are assumed to occur through direct contact and inhalation of emissions from two of the five identified products expected to lead to significant BBP indoor exposures during use. The products include:

- A spray paint with BBP content of 0.1 to 1 percent, used for repairing or renewing surfaces

like bathtubs, sinks, and vanities (Multi-Tech Products Corp, 2015).

- A concrete sealant with BBP content of 0.1 to 1 percent, reported by consumers for use in sealing and refinishing projects such as fireplace stone, concrete countertops, and floors.

As summarized in the Compiled Products tab within the BBP Consumer Exposure Analysis supplemental file, EPA modeled consumer inhalation exposures for an entire indoor residence (492 m^3), assuming a Stay-At-Home activity pattern. The model considered a duration of up to 8 hours and a product usage of up to 16,867 grams (approximately 4 gallons) per event in CEM as a high-end scenario. This assessment of BBP exposures throughout the entire modeled residence includes "large and sensitive locations within the home," referenced by the commenter.

One limitation of this assessment is that EPA did not identify reasonably available monitoring data on the relative contribution of BBP from paints or coatings to residential indoor dust after the paint has dried, chipped, and accumulated in the dust. Exposures to BBP from paints and coatings were not quantified for indoor suspended dust, settled dust, or mouthing pathways but were considered qualitatively.

Adult Toy Exposure

Summary: A public comment (SACC25-0030) recommended *in vitro* studies be used to evaluate safety of adult toys citing recommended methodologies from the International Organization for Standardization (ISO). The commenter highlighted guidance on medical devices from ISO, U.S. Food and Drug Administration, and European Commission that may inform EPA's analysis, while acknowledging that carrying out new and extensive characterization may be excessive for assessing exposure to an adult toy component. Finally, the commenter noted that reconstructed human vaginal epithelium (rHVE) models can be used to evaluate exposure and hazard.

Another comment (SACC25-0154) states that "EPA's request for input on evaluating products that contact mucosal membranes is timely and important." While the comment also points to FDA, ISO, and European Commission for guidance in this space, it also highlights the EpiVaginal model as a potentially useful tool for evaluating the risk of adult toys that are exposed to mucus membranes.

EPA Response: EPA agrees with the commenters and welcomes the shared references and suggested models. As mentioned by the commenters, studies that address mucosal membrane exposures tend to study exposures to medical devices and drug delivery devices and models (*i.e.*, EpiVaginal model) that consider exposure durations that are not representative of adult toy use patterns used in these assessments (15, 30, and 60 minutes). Additionally, due to the wide variability in potential mucosal membrane absorption reported by several studies shared by the commenters, any increases in exposure absorption factor would have a slight confidence. The designation of slight confidence is assigned when the weight of scientific evidence may not be adequate to characterize the scenario, and when there is an absence of complete information and there are additional uncertainties that may need to be considered. For example, it is uncertain if the results of an approach using an absorption factor would fit in the exposure distribution as an upper bound, central tendency, overestimation, or underestimation. All of these factors make the studies inappropriate for use in an extrapolation to absorption of phthalates due to contact with vaginal and anal mucosa.

4.3 General Population Exposures

Summary: Several public commenters (DCHP-0127, DEHP-0138, SACC25-0145, SACC25-0132, DIBP-0133) stated that EPA failed to consider background exposures and/or non-TSCA uses (*e.g.*, from food and food packaging), and recommend that such exposures be included when characterizing risk from TSCA uses.

Commenters (DCHP-0127, DEHP-0138, DIBP-0133) stated "EPA failed to conduct a background exposure assessment, underestimating risk to potentially exposed or susceptible subpopulations." The commenters explained "EPA stated that certain significant pathways of exposure to the general population, including cosmetics, medical devices, food and food packaging materials, were not considered because they constitute 'non-TSCA' uses. EPA's rationale for this decision is that these other pathways of exposure will be assessed and managed by statutes such as the Clean Air Act and the Federal Food, Drug, and Cosmetic Act. However, exposures via these pathways are highly relevant and reasonably foreseeable across the human population and cannot be excluded when evaluating the human health risks posed by…" DBP, DCHP, or DEHP "…as no such regulations are in place nor are they planned." The commenters suggested that EPA must revise the Draft Risk Evaluations for DBP, DCHP, and DEHP so that they address all sources and pathways of exposure, including background exposures.

SACC25-0132, DIBP-0133 similarly state that exposures from food, food additives, drugs, and medical devices (while excluded from the statutory definition of "chemical substance" under TSCA), impact the susceptibility to risk and that their inclusion in aggregate exposure estimates should be considered "best available science", including in the assessment of PESS.

SACC25-0145 and DIBP-0133 similarly state that EPA's approach "underestimates real-world exposures and risks" because aggregate exposure was only considered to a limited extent, and EPA failed to conduct a background exposure assessment, underestimating risk to potentially exposed or susceptible subpopulations.

EPA Response: EPA does not believe that it is underestimating risk to the general population by not incorporating exposures from uses of DEHP, DCHP, DBP, DIBP, and BBP that fall outside the jurisdiction of TSCA. It is correct that EPA has not evaluated specific non-TSCA uses of all 5 phthalates not regulated by TSCA, however, EPA has accounted for background aggregate exposures to all 5 phthalates in the risk evaluations through the use of the CRA. For example, in the general population exposure technical support documents for all 5 phthalates, EPA evaluates urinary biomonitoring data from the CDC's NHANES biomonitoring data set. NHANES provides a relatively recent (data available through 2017 to 2018) and robust source of urinary biomonitoring data that is considered a national, statistically representative sample of the non-institutionalized, U.S. civilian population. Reverse dosimetry modeling does not distinguish between routes or pathways of exposure and does not allow for source apportionment (*i.e.*, exposure from TSCA COUs cannot be isolated). Therefore, NHANES provides an estimate of aggregate background phthalate exposure for the U.S. population that cannot be attributed to specific sources or pathways of exposure. However, based on previous analyses by U.S. CPSC (2014) and Health Canada (2020), dietary phthalate exposure is considered a major source of exposure to phthalate that is expected to be captured within NHANES. MOEs were calculated for exposure estimates derived from NHANES urinary biomonitoring data. Median and 95th percentile MOEs were above the benchmark of 30 for all assessed populations.

Summary: A public comment (DIBP-0089) asked, "Does the technical support document account for fugitive emissions or leaching into groundwater from DIBP-containing materials at landfills, particularly in communities without safe water infrastructure?"

EPA Response: EPA does consider both fugitive emissions and leaching into groundwater from DIBP-containing materials at landfills in its technical support document *Environmental Media, General Population, and Environmental Exposure Diisobutyl Phthalate (DIBP)*. The ambient air assessment detailed in Section 8 of the *Environmental Media, General Population, and Environmental Exposure to DIBP* considers both stack and fugitive emissions. In the absence of any reported releases, EPA estimated releases to air using EPA standard models. For the screening-level assessment, EPA made the conservative assumption to combine the maximum stack and fugitive emissions modeled using the maximum modeled daily release and conservative meteorological data at 100 m from a releasing facility, which resulted in conservative upper-bound estimates of ambient air concentrations that is appropriate for screening-level assessments.

EPA also considered the possibility of leaching into groundwater from DIBP-containing materials at landfills in Section 3.2 the *Environmental Media, General Population, and Environmental Exposure to DIBP*. No studies were identified through systematic review that could provide the concentration of DIBP in refuse or waste in the United States. No TRI data was reported on releases of DIBP into landfills or from recycling facilities. EPA considered the physical and chemical properties of DIBP and determined that DIBP is expected to have a high affinity to particulate (log K_{OC} = 5.5) and organic media (log K_{OW} = 4.34) that would cause significant retardation in groundwater and limit leaching to groundwater. Because of its high hydrophobicity and high affinity for soil sorption, EPA determined it is unlikely that DIBP will migrate from landfills after groundwater infiltration.

Summary: A public commenter (SACC25-0132, SACC25-0145) stated that EPA failed to consider human exposures associated with DEHP and DBP to land via biosolids or landfills. The commenter states that the inclusion of biosolids pathways in the draft DEHP for animal and plant health negates EPA's argument that modeling impacts on human health via these pathways was untenable due to uncertainties in monitoring data. Furthermore, they state that EPA's contention of inadequate monitoring data was inaccurate, providing multiple examples of existing data that supporting evidence exposure to humans via these pathways. Based on this evidence, they state EPA must model exposure via these pathways.

EPA Response: For the risk evaluation for DEHP, EPA had extensive DEHP monitoring data from WWTP sludge survey to act as a surrogate for DEHP concentration in biosolids; whereas, other phthalates were not included in the sludge survey. For DEHP, this estimate of biosolids concentration was used to determine the soil concentration resulting from biosolids application, and the resulting soil concentration was used in a screening level assessment for ecological risk to birds, terrestrial invertebrates, mammals, and plants. Specifically, a screening level assessment for compared hazard values to exposure levels (*e.g.*, from birds and terrestrial mammals eating earthworms at the same concentration as this soil) and determined that these screening level exposure estimates were orders of magnitude lower than the respective hazard values for these species. While it may be possible to conduct similar screening level analysis for human health, it is expected that exposure levels to humans would be considerably lower than the exposure levels to ecological species whose diet is comprised of terrestrial invertebrates assumed to be at the same concentration as the soil. Furthermore, biosolids applications do not constitute a TSCA release that can be attributed to a specific COU or even definitively to TSCA vs non-TSCA (*e.g.*, FDA) uses. Therefore, this approach was only useful as

a screening level assessment to demonstrate that biosolids application is not a pathway of concern, as was determined qualitatively for the other phthalates lacking WWTP sludge monitoring data.

Summary: A public commenter (SACC25-0132, SACC25-0145, DIBP-0133) stated that EPA did not consider down-the-drain releases and exposure pathways for DEHP, DBP, DIBP, and BBP, including releases "to the environment via the cleaning and disposal of adhesives, sealants, paints, lacquers, and coatings." The commenter states that although EPA cited a lack of available data and tools in this decision, EPA had previously considered this pathway in the Risk Evaluation for 1,4 Dioxane using available data or assumptions related to populations contributing to wastewater loadings, consumer product weight fractions of phthalates, and hydrological conditions. Furthermore, the commenter stated that EPA acknowledged that such pathways exist without providing any quantitative analysis or discussing any potential impacts on risk characterization and stated that EPA's assertion that these releases are "not likely [to] lead to environmental concentrations that exceed hazard values for aquatic and terrestrial organisms" is unsupported.

EPA Response: EPA considered down-the-drain releases and exposures in its evaluations. In Section 3.1.4 of the risk evaluation documents for DEHP, DBP, DIBP, BBP, and DCHP, EPA described a qualitative discussion of disposal and subsequent environmental exposure. When conducting individual chemical risk evaluations, EPA considers the specific uses and physical chemical properties of the chemical being assessed to determine the pathways of exposures for humans and the environment. Although EPA has previously quantified the down-the-drain exposures for 1,4-Dioxane, EPA considered the specific physical chemical properties of phthalates, which make them easily removable in wastewater treatment (up to 98% depending on the phthalates) as described in Section 2 of the risk evaluation documents for all phthalates, to determine that a qualitative discussion is appropriate for this risk evaluation. Exposures to the general population can then be assessed using the environmental release values. EPA used the monitoring data in drinking water, surface water, and landfills and biosolids described in the *Environmental Media and General Population Exposure* technical support documents to qualitatively show expected concentrations of phthalates, and chemical fate and transport within each environmental media. Any exposures to the general population are captured by the releases into the environment in the risk evaluation of environmental media and general population.

Summary: A public comment (DCHP-0128) took issue with the screening-level assessment of general population exposure not considering aggregate exposure without explicitly evaluating aggregate exposures of known and reasonably foreseen combinations of DCHP exposure across pathways, COUs, and release facilities.

EPA Response: EPA assessed aggregate exposures in the DBP, DCHP, DEHP, DIBP, and BBP risk evaluations but is only pursuing aggregation across exposure routes for each TSCA COU and not across COUs due to the lack of reasonably available information on the representativeness of such results. To include an aggregate assessment across COUs or multiple products within a COU would not be consistent with making a risk determination based on the best available science or be based on the weight of scientific evidence due to these data gaps.

EPA did not evaluate aggregate risk for the general population from environmental releases. As discussed in section 4.1.3 of the risk evaluations for DBP, DCHP, DEHP, DIBP, and BBP, EPA employed a screening level approach to assess risk from exposure to each phthalate for the general

population from environmental releases. EPA did not evaluate aggregate risk for the general population from environmental releases, because combining exposures from the conservative screening level approach would be inappropriate, without further refinement of each exposure scenario. As described in Section 4.1.3 of the risk evaluations for DBP, DCHP, DEHP, DIBP, and BBP, EPA did assess aggregate exposure to each phthalate for the general U.S. population using NHANES urinary biomonitoring data and reverse dosimetry, which provides context for aggregate exposures in the U.S. non-institutionalized, civilian population. Based on this analysis of NHANES, all aggregate 95th percentile MOEs for DBP, DCHP, DEHP, DIBP, and BBP were above the benchmark of 30 for all assessed populations and age groups.

4.4 Aggregate Human Exposures

Summary: A public comment (SACC23-0050) expressed support for considering aggregate exposures to one or more phthalates to people living in fenceline communities.

EPA Response: EPA acknowledges support from the public commenter. As discussed in EPA's draft proposed approach for CRA of phthalates under TSCA (U.S. EPA, 2023), EPA recognized that the general population, those impacted by facility release of phthalates, could be exposed to multiple phthalates from single facilities that release more than one phthalate or be exposed to multiple phthalates due to living in close proximity to co-located facilities. EPA analyzed the co-location of all the known phthalate-releasing facilities within common watersheds. EPA identified DMR, NEI, and TRI data for DEHP, DBP, and BBP, but not for DCHP, DINP, and DIBP. These EPA databases provide information on facilities releasing phthalates to various environmental media and provide latitude and longitude data for releasing facilities. Using the release information, EPA identified 1,461 facilities that report use of a single phthalate, while 461 report use of multiple phthalates (*i.e.*, any combination of DEHP, DBP, or BBP). Using the available location data, EPA mapped the reporting facilities to look for geographic patterns or hotspots. Individual facilities are broadly dispersed around the United States. EPA also analyzed the locations of the identified facilities by watershed or hydrologic units. A hydrologic unit represents the area of the landscape that drains to a portion of the stream network and is identified by a unique Hydrologic Unit Code (HUC). EPA identified 21 HUC12 watersheds with four or more releasing facilities. The commenter is directed to Section 3.3.2 of the CRA TSD for further discussion of TSCA releasing facilities.

Summary: Several public commenters (DCHP-0127, DEHP-0138, DEHP-0138, SACC25-0145, SACC25-0132, SACC25-0151, DIBP-0130, DIBP-0133) stated that EPA failed to adequately address aggregate exposures.

In relation to DBP, DCHP, and DEHP, three commenters (DCHP-0127, DEHP-0138, DEHP-0138) stated "EPA considered aggregate exposure to only a limited extent." The commenters acknowledge that "In an important improvement, EPA considered aggregate exposure to…" DBP, DCHP, and DEHP "…by combining worker exposure estimates for the inhalation and dermal routes of exposure, and consumer exposure estimates for the inhalation, ingestion, and dermal routes of exposure." However, several of these commenters (DEHP-0138, DEHP-0138, SACC25-0151) noted that EPA ignored ingestion as a route of exposure for workers. These commenters stated "Workers can ingest chemicals through various unintentional behaviors, such as nail-biting and eating on the worksite. The EPA must use best available exposure models to quantify these ingestion exposures and provide a more comprehensive risk assessment." The three commenters continued "EPA should not require

chemical-specific evidence to conduct aggregate exposure assessment. It can reasonably model scenarios in which exposures are combined across products and across worker, consumer, and general population exposures." The commenters additionally stated that "If EPA does not estimate risks from aggregate exposures across COUs and exposure settings in the final…" DBP, DCHP, and DEHP "…risk evaluation[s], the resulting underestimation would then be a consideration that must be incorporated into the unreasonable risk determination."

Similarly, another commenter in relation to BBP and DIBP (DIBP-0130, DIBP-0133) stated that EPA failed to consider aggregate exposures from multiple COUs, ignoring the risks people face from a combination of household and consumer products and the risks workers face in facilities that engage in multiple COUs. In addition, DIBP-0133 raised concern that "characterizing risks to potentially exposed or susceptible subpopulations necessarily requires EPA to consider aggregate exposures and to evaluate risks to groups who experience greater risk because of such exposures. EPA ignored the evidence in the record and the ubiquity of both DIBP and BBP. DIBP is used in many items that are commonly found in the average home, including furniture, flooring, wallpaper, and toys. Similarly, BBP is commonly used in a variety of household industries and consumer products, such as toys, clothing, and arts and crafts materials."

Other commenters (SACC25-0132, SACC25-0151), while supportive of the inclusion of aggregate exposures in general, stated that EPA failed to address multiple known or reasonably foreseen exposures in its aggregate assessment, including combined exposures via multiple COUs, routes, sources, and pathways. The commenter stated the relevance of this weakness to estimating risks to PESS. The commenter also stated that the reliance on limited indoor dust data for estimating aggregate exposures should be avoided by EPA, as these data do not account for all relevant pathways, and may underestimate exposure, especially for PESS. The commenter provided a specific example recommending that EPA aggregate exposures for DEHP and DBP across products using reasonable assumptions and modeling approaches, stating that the ubiquity of the presence of these chemicals in many household products is known (*e.g.*, 5 out of 6 phthalates are used in consumer arts, crafts, and hobby materials). The commenter additionally stated that occupational exposure to multiple COUs was foreseeable, citing TSCA facilities reporting use of DEHP and DBP for multiple COUs, and also recommended occupational exposures should be aggregated with household (consumer) uses. Finally, the commenter stated that not including general population (background) exposures of DEHP and DBP in aggregate exposure assessments was contrary to the goal of estimating risks from combined exposures. Additionally, DIBP-0133, raised concern about using "the limited DIBP and BBP indoor dust monitoring data used in the assessment should not be used to avoid the required calculation of aggregate exposures. The studies do not account for exposures from suspended dust, from direct dermal absorption, from mouthing, or from other relevant exposure pathways. EPA's monitoring data may understate even dust-related exposures to people who have a higher concentration of phthalate-containing goods in their homes. The comparison is solely to "medium" or "central tendency" exposure scenarios. EPA did not compare maximum or higher-end monitoring data with high-end modeling results, even though TSCA requires EPA to make unreasonable risk determinations for groups with "greater exposure" to the chemical substance."

Another commenter (SACC25-0145, SACC25-0151, DIBP-0130), stated that EPA's failure to consider exposures form non-TSCA uses of DBP, DEHP, BBP, and DIBP, including exposures through food and food packaging understates the risk to the general population from the TSCA uses of these chemicals if it does not take into account the background exposures from these and other non-TSCA uses.

For the general population aggregate assessment in DIBP and BBP, DIBP-0133 stated that "even assuming that EPA is correct that no single exposure pathway generates risks of concern for the general population, that would not provide a reasoned basis for ignoring the risks from aggregate exposures across pathways and from multiple facilities and COUs. Additionally, several of EPA's general population risk calculations lie so close to the relevant risk benchmarks that proper consideration of aggregate risks could change EPA's ultimate risk findings. For example, EPA calculated risks of concern for BBP in drinking water and fish consumption that were concerningly high (*i.e.*, below the benchmark margin of exposure ("MOE")), but continually readjusted its assumptions until these findings raised just above the benchmark. Given the proximity to the benchmark cutoffs remaining even after this practice, it is apparent that properly considering factors like aggregate exposures could have relevant impacts on EPA's ultimate risk determination in these areas."

Another public comment (DCHP-0128) stated that "EPA must consider aggregate DCHP exposures from multiple conditions of use, sources, and exposure pathways" and "EPA cannot identify and eliminate unreasonable risks arising from 'any combination of' conditions of use as section 6(a) requires if EPA only assesses the risks posed by individual conditions of use in isolation." In lieu of statistical data to support assumptions of exposure across COUs, the commenter recommends EPA develop exposure scenarios based on reasonable assumptions regarding potential combinations of exposure (*e.g.,* consumers using adhesives and plastic articles containing DCHP or assuming that a worker may experience both occupational and consumer exposures).

EPA Response: EPA assessed aggregate exposures in the DBP, DCHP, DEHP, DIBP, and BBP risk evaluations but is only pursuing aggregation across exposure routes for each TSCA COU and not across COUs or multiple products within a COU due to the lack of reasonably available information on the representativeness of such results. To include an aggregate assessment across COUs or multiple products within a COU would not be consistent with making a risk determination based on the best available science or be based on the weight of scientific evidence due to these data gaps.

Regarding the worker ingestion assessment comments (DEHP-0138, DEHP-0138, and SACC25-0151), EPA generally does not evaluate ingestion as an exposure route in occupational settings. Inhalation and dermal exposures are encountered on a routine basis in the workplace, and therefore these routes are expected to comprise the majority of occupational exposures. However, workers may inadvertently ingest chemicals at the workplace. The frequency and significance of this exposure route are dependent on several factors including the physical and chemical properties of the substance during expected worker activities, workers' awareness of the chemical hazards, the visibility of the chemicals on the hands while working, workplace practices, and personal hygiene that is difficult to predict.

EPA disagrees with DIBP-0130 and DIBP-0133 comment about not assessing exposures across consumer products and articles. This type of aggregate analysis must be based on practical assumptions for use patterns and robust supporting scientific evidence. For example, the indoor dust monitoring data was used in the comparison between indoor dust (specifically settled dust) ingestion modeling and monitoring doses for the same exposure route. The comparison was done using the modeled medium intensity use doses and median indoor dust concentration measurements to allow comparison among similar conditions that may not be captured with the high intensity modeling results and 95th percentiles monitoring data. This information is available in Section 4 of the Consumer and Indoor Exposure Assessment Technical Support Documents for all five phthalates. To compare modeling and monitoring results, EPA must ensure the comparison is done for similar

scenarios. Since monitoring studies data do not provide the relative contributions from consumer products into the reported measurement, EPA used the aggregation of all the indoor modeling scenarios for chronic settled dust ingestion doses only and compared that aggregated result to the monitoring chronic dose. A discussion on the differences between modeling and monitoring data is provided. This indoor analysis is also an aggregation of across products and COUs. However, the comparison between modeling and monitoring data resulted in stark differences pointing to the uncertainties in the aggregation of modeling estimates. While it would be practical to assume households contain products across COUs, the relative contribution of each product/article to indoor dust and subsequent exposures to consumers is far more complex than simply aggregating across COUs.

EPA disagrees that it is underestimating risk to the general population by not incorporating exposures from uses of DEHP, DBP, DCHP, DIBP, and BBP that fall outside the jurisdiction of TSCA. It is correct that EPA has not evaluated specific uses of all 5 phthalates not regulated by TSCA, however, EPA has accounted for background aggregate exposures to the phthalates in the risk evaluation documents as well as background aggregate exposures to the 5 phthalates in the risk evaluation documents and in the CRA. For example, in the *Environmental Media and General Population Exposure for DEHP* technical support document, EPA evaluates urinary biomonitoring data for DEHP from the CDC's NHANES biomonitoring data set. NHANES provides a relatively recent (data available through 2017 to 2018) and robust source of urinary biomonitoring data that is considered a national, statistically representative sample of the non-institutionalized, U.S. civilian population. This same approach was done for DBP, DCHP, DIBP, and BBP. Reverse dosimetry modeling does not distinguish between routes or pathways of exposure and does not allow for source apportionment (*i.e.*, exposure from TSCA COUs cannot be isolated). Therefore, NHANES provides an estimate of aggregate background phthalate exposure for the U.S. population that cannot be attributed to specific sources or pathways of exposure. Based on previous analyses by U.S. CPSC, dietary phthalate exposure is considered a major source of exposure to the 5 phthalates that is expected to be captured within NHANES. MOEs were calculated for exposure estimates derived from NHANES urinary biomonitoring data. Median and 95th percentile MOEs were above the benchmark for all assessed populations.

EPA considers vulnerable populations including women of reproductive age, pregnant women, male infants, male children, and males of reproductive age as PESS as stated in Section 5 of the final non-cancer human health hazard TSD documents. This section describes how susceptibility was addressed in the risk evaluation for these populations through the use of uncertainty factors, which are reflected in the benchmark MOE or captured in the choice of POD, which is based on reproductive and developmental effects. By taking a cumulative approach to phthalates, which considers TSCA uses as well as the broader background of people's exposure to multiple phthalates, EPA is being protective of susceptible subpopulations (*i.e.*, women of reproductive age, pregnant women, male infants, male children) exposures to phthalates. Additionally, EPA discussed possible increased exposures to populations based on specific scenarios and exposure patterns, for example, children under 5 years of age mouthing of electric wires. While mouthing of electric wires is not an intended use, children under 5 exhibit behavioral patterns that may result in increased exposures.

4.5 Cumulative Human Exposures

Summary: A public comment (DCHP-0122) recommends that EPA "strengthen the scenario-based assessments by estimating exposure for potentially exposed or susceptible populations in occupational, consumer and indoor dust exposure scenarios." The comment suggests that EPA could improve the scenario-based assessment by "Including exposures from multiple COUs for workers in facilities that report production of more than one phthalate"; "Evaluating non-TSCA phthalate exposures in scenario-based exposure assessments"; and "Providing sensitivity analyses for phthalate exposure in dust for young children in low-income housing and in daycare."

Another public comment (SACC23-0052) stated that EPA should consider non-TSCA consumer exposures in the same manner as TSCA uses. For example, the comment recommends that non-TSCA inhalation consumer exposures (*e.g.*, cosmetics) be included in the CRA.

EPA Response: As described in Section 4.1.5 of the DCHP, DEHP, DBP, DIBP, and BBP risk evaluations, EPA did not consider aggregate exposure scenarios across COUs because the Agency did not find any evidence to support such an aggregate analysis based on the reasonably available information, such as statistics of populations using certain products represented across COUs, or workers performing tasks across COUs. However, EPA considered combined exposure across all routes of exposure for each individual occupational and consumer COU to calculate aggregate risks.

Under TSCA, EPA is not statutorily required to evaluate non-TSCA exposures. However, as part of the phthalate CRA, EPA evaluated background exposure to DEHP, DBP, DIBP, BBP, and DINP using NHANES urinary biomonitoring data and reverse dosimetry. Although, EPA cannot attribute exposure measured in NHANES to specific routes or sources of exposure, U.S. CPSC (2014) and Health Canada (2020) have found that the main sources of exposure to phthalates is through diet (non-TSAC use), certain cosmetics (non-TSCA use), indoor air, and indoor dust (See Section 3.4 of the CRA TSD for further discussion).

As part of its consumer exposure analyses for DCHP, DEHP, DBP, DIBP, and BBP, EPA considered consumer exposure to phthalates through certain use patterns. Additionally, EPA evaluated cumulative risk from exposure to DCHP, DEHP, DBP, DIBP, BBP, and DINP from exposure to Dust (See section 3.2.3 of the CRA TSD). This analysis demonstrated that exposures through house dust were highest for children 3 to 6 years of age; however, the cumulative MOE for 95th percentile exposures was 880 (benchmark), indicating a contribution of only 3.4 percent to the risk cup.

By taking a cumulative approach to phthalates, which considers TSCA uses as well as the broader background of people's exposure to multiple phthalates, EPA protecting susceptible subpopulations (*i.e.*, women of reproductive age, pregnant women, male infants, male children) from exposure to phthalates.

Summary: A public comment (SACC23-0052) stated that it would be arbitrary and capricious and inconsistent with scientific standards and weigh of scientific evidence requirements under TSCA section 26(h) and (i) to not consider all non-TSCA and non-attributable exposures within the context of the phthalates CRA. The comment applauded EPA for considering non-TSCA and non-attributable exposures in the phthalate CRA, but expressed concern that only "major" non-TSCA sources of phthalate exposure would be included in the CRA. The comment also stated that it was unclear how

EPA defined “major” beyond dietary exposure.

EPA Response: The EPA’s use of NHANES data as part of its cumulative risk approach includes non-TSCA uses and non-attributable exposures. Regarding other recommendations in the comment, EPA’s TSCA risk evaluations are decision-support documents that are required to be conducted within statutorily mandated timeframe and are done with the information reasonably available to the Agency. TSCA cumulative assessments are done for the same reason under the same requirements and constraints, and therefore EPA will make choices about how extensive its approach to cumulative assessment should and can be to meet the requirements of TSCA section 6. To the extent that the aspects of cumulative assessment identified in the comment are aligned with these requirements, EPA may consider some or all of them when deciding how to apply cumulative risk assessment approaches in its evaluation of existing chemicals under TSCA.

Summary: A public comment (SACC23-0052) recommended EPA utilize both reverse dosimetry and scenario-based approaches to evaluate cumulative risks, then select the most protective exposure assessment approach after considering respective uncertainties in order to be most protective of PESS when aggregating exposure.

EPA Response: In the draft 2023 proposed approach document, EPA proposed using a scenario-based exposure assessment to determine non-attributable and non-TSCA source exposure levels to all phthalates and to reconstruct an aggregated daily exposure profile for receptors varied by age (women of reproductive age, male infants, toddlers, and children). The approach proposed was to use similar methods to Health Canada and U.S. CPSC, which determined that diet comprised a large portion of total daily intake for populations of interest. EPA did not identify any other assessments by other regulatory bodies with differing conclusions. In its review of the approach, SACC recommended reviewing literature related to estimates of exposure from diet given the highly diverse U.S. population. EPA conducted a literature search to investigate if there were any large-scale phthalate dietary assessments that would influence a national scale dietary assessment or warrant an update to the previously conducted analyses. However, EPA has concluded that there is limited updated information to substantially change the daily intake estimates previously constructed by the other agencies using scenario-based methods, including for sensitive subpopulations.

Health Canada and U.S. CPSC had both estimated total phthalate daily intake values using reverse dosimetry with human urinary biomonitoring data and scenario-based exposure assessment approaches. Health Canada and U.S. CPSC found that both the reverse dosimetry and scenario-based approaches resulted in daily intake values that were generally similar in magnitude. However, this depended on the recency and quality of data available for use, particularly for data on major exposure pathways like diet. Rather than construct new national estimates of dietary intake, EPA similarly used reverse dosimetry with national human urinary biomonitoring data which provides total intake for total population and subpopulations by demographic category using the most recent publicly available NHANES data to reflect current national exposure to phthalates (see Section 11 of the Environmental Media and General Population Exposure TSDs for DEHP, DBP, DIBP, DCHP, and BBP). National human urinary biomonitoring data is expected to reflect exposure to the major non-TSCA sources of exposure (*e.g.*, diet, personal care products, indoor air, and house dust) identified by U.S. CPSC and Health Canada.

Summary: Public commenters (SACC25-0153, BBP-0120, DIBP-0123) expressed concerns with Option 1 (RPF) for estimation of cumulative risk and referred to Section 3.1 "Occupational Exposure for Workers" in the *Revised Draft TSD for the Cumulative Risk Analysis of the Phthalates under TSCA*, which the commenter reported as stating that EPA:

- does not expect industrial and commercial products containing multiple phthalates to be a significant source of phthalate exposures contributing to cumulative risk under most occupational scenarios; and
- Commercial exposure scenarios and facilities that work with multiple phthalates may run non-overlapping campaigns and therefore workers may not be co-exposed to multiple phthalates.

Therefore, the commenter contended that there is no need for the RPF cumulative phthalate approach for attributable exposures because co-exposures in the workplace are unlikely and recommended that if EPA does pursue a cumulative phthalate exposure scenario, then the common health endpoint of decreased fetal testicular testosterone (FTT) be the basis for the RPF for both attributable and non-attributable exposures. The comment (DIBP-0123) provided the following table with a recalculation of the cumulative MOEs based on the common health endpoint of decreased FTT:

Table 1. Comparison of Calculated Cumulative MOEs

	EPA Option 1	EPA Option 2	Revised* Option 1	Revised* Option 2
DCHP cumulative MOE	29	51	98	96
DEHP cumulative MOE	46	21	77	75

* *See* Appendix A for further information on updated RPFs based on FTT for all phthalates.

This comment (DIBP-0123) also noted that there is a typo in the worked DEHP Recycle OES example for Option 1. Line 1890 on Page 64 states "38.7 and 2.07 ug/kg" but should instead be "46.9 and 2.36 ug/kg."

EPA Response: As discussed in the Section 3.1 of the 2025 CRA TSD, EPA agrees with the public commenters. EPA does not expect industrial and commercial products containing multiple phthalates to be a significant source of phthalate exposures contributing to cumulative risk under most occupational scenarios; and commercial exposure scenarios and facilities that work with multiple phthalates may run non-overlapping campaigns and therefore workers may not be co-exposed to multiple phthalates. To assess cumulative risk for DBP (index chemical), DCHP, BBP, DIBP, and DEHP, EPA used CRA Approach 2. For Approach 2, individual phthalate exposures for consumer and occupational COUs are not scaled by RPFs but use the individual phthalate hazard values and are still combined with non-attributable cumulative exposures estimated using NHANES.

EPA disagrees with how the public commenter calculated cumulative MOEs in the provided table. EPA reviewed the proposed RPFs (Table 5 below, from Appendix A in DIBP-0123) calculated by the public commenter that were used to re-calculate the revised cumulative MOEs. EPA disagrees with the alternative RPFs proposed by the public commenter, as there are two concerns with the RPFs calculated by the public commenter. First, RPFs are calculated based on the ratio of the index chemical (DBP) benchmark dose (BMD) to the BMD of the phthalate of interest, not the ratio of the 95% lower confidence interval associated with the BMD, or BMDL (as was done by the public commenter). EPA notes that there is an error in the public commenters Table 5 below, which indicates that EPA calculated RPFs using $BMDL_{40}$ estimates, when in fact EPA calculated RPFs using BMD_{40} estimates. RPF values account for potency differences among chemicals in a mixture and scale the dose of one chemical to an equitoxic dose of another chemical (*i.e.*, the index chemical). BMDLs do not represent equitoxic doses because the magnitude of the 95% confidence interval varies from

chemical to chemical, which is why BMD estimates are used to calculate RPFs. Second, EPA does not find it appropriate to estimate RPFs using a BMR of 1 control standard deviation because the magnitude of standard deviation associated with the control will vary from phthalate-to-phthalate and therefore does not represent a constant response level in which to calculate equitoxic doses for estimating RPFs. Given the concerns with the proposed RPFs, the subsequent revised cumulative MOEs presented in Table 1 by the public commenters are also of concern to EPA.

Table 5. Proposed RPFs

Chemical	Proposed RPF			EPA's RPF		
	POD for RPF_{FTT}		RPF_{FTT}	$BMDL_{40}$ for FTT		RPF_{40}
DCHP	$BMDL_{1SD}$	19.58	0.51	$BMDL_{40}$	90	1.66
BBP	$BMDL_{1SD}$	61.44	0.16	$BMDL_{40}$	284	0.52
DBP	NOAEL	10	1.00	$BMDL_{40}$	149	1.00
DEHP	$BMDL_{1SD}$	17.96	0.56	$BMDL_{40}$	178	0.84
DIBP	$BMDL_{1SD}$	113.4	0.09	$BMDL_{40}$	279	0.53

EPA has fixed the typo in the DEHP Recycle OES example for Option 1 indicated by the public commenter (DIBP-0123).

Summary: One commentor (SACC23-0030) stated that, while [they] are generally supportive of the use of screening and tiered approaches to risk assessment, including the cumulative risk assessment of mixtures, both clear guidance and advancement to subsequent tiers when refinement is possible for unreasonable risks are necessary.

The commentor believes that screening approaches are not an appropriate method for determining true risk and that a cumulative assessment requires a combination of each substances' aggregated exposures in a realistic manner. Stating: "EPA's established practice for cumulative risk assessment of pesticide active ingredients is to evaluate the common toxic effect associated with concurrent exposure by all relevant pathways and routes of exposure to a group of chemicals that share a common mechanism of toxicity. In the years since the Food Quality Protection Act (FQPA) was enacted, it has proven to be quite challenging for EPA to conduct cumulative risk assessments, even for groups of chemicals having a common mechanism of toxicity."

The commentor goes on to recommend using the Hazard Index approach as the most applicable method for the CRAs but stated that the HI approach is a tier I screening approach and does not estimate true risk. The commentor also stated that tiering can provide helpful "off-ramps" but needs further refinement and advancement to a higher tier when unreasonable risk may be present.

A public commenter (SACC23-0030) believes that the CRA does not meet TSCA's scientific standards, as stated by the EPA in the Draft Principles Introduction.

EPA Response: EPA has utilized a combination of screening-level and more refined approaches as part of its individual chemical assessments of DEHP, DBP, DIBP, BBP, and DCHP and cumulative risk analyses for each phthalate. As discussed throughout the risk evaluations for each phthalate, when EPA calculated aggregate or cumulative MOEs below the benchmark of 30, EPA evaluated the assumptions and model inputs feeding into the exposure scenario to determine if they represented a series of compounding conservative assumptions, which are appropriate for a screening-level assessment, or if the assumptions provide a reasonable/realistic estimate of exposure based on

reasonably available information and the best available science. For exposure scenarios determined to have multiple conservative compounding assumptions, EPA refined the exposure assessment or conducted sensitivity analyses to determine which exposure parameters are most sensitive.

EPA agrees with the public commenter that the HI approach is an applicable method for CRA. However, for phthalates EPA utilized a relative potency factor and cumulative MOE approach, which is consistent with the TSCA risk-based statutory framework.

Although TSCA does not explicitly require EPA to conduct CRAs, TSCA does require that EPA, when conducting TSCA risk evaluations in 3 to 3.5 years [15 U.S.C. § 2605(b)(4)(G)], consider the reasonably available information, consistent with the best available science, and make decisions based on the weight of the scientific evidence. EPA recognizes that for some chemical substances undergoing risk evaluation, the best available science may indicate that the development of a CRA is appropriate to ensure that any risks to human health and the environment are adequately characterized. For 6 of the phthalates undergoing risk evaluation (DEHP, DBP, DIBP, BBP, DCHP, DINP), there is clear evidence of toxicological similarity (*i.e.*, they disrupt fetal testicular testosterone production and cause effects on the developing male reproductive system consistent with phthalate syndrome), and evidence of co-exposure as demonstrated through NHANES urinary biomonitoring data. Further, mixtures toxicology studies also consistently demonstrate that these phthalates act dose-additively to disrupt fetal testis testosterone production and cause phthalate syndrome-related effects. Finally, while as the comment indicates more research can be done in support of CRA, EPA believes that it had adequate information to apply CRA principles and approaches in a useful and appropriate way to support its evaluation of the phthalates. This fit-for-purpose consideration of cumulative risk for phthalates is consistent with the best available science to develop risk evaluations that support EPA decision making under TSCA.

Use of NHANES Dataset

Summary: Two public comments (SACC23-0031, SACC23-0065) SACC on the *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substance Control Act* expressed support for the use of NHANES biomonitoring data to support a Biomonitoring Equivalent (BE) approach in the cumulative assessment.

EPA Response: The EPA acknowledges the reviewers for their support of the use of NHANES data in the cumulative assessment of the phthalates. Ultimately, this data was used to inform the cumulative assessment of BBP, DBP, DCHP, DEHP, and DIBP.

Summary: A public comment (DCHP-0121) points out that NHANES dataset did not include DCHP after 2010, due to a low number of detects. The comment states that DCHP is used as an alternative for DEHP, which has been declining in use since 2011, and further points out that "it would seem logical that DCHP exposures would also increase over time in the NHANES dataset had monitoring continued past 2010, particularly following the then recent DEHP restrictions." The commenter is not aware of another dataset with DCHP data but recommends modeling and establishing monitoring programs in the environment to better estimate DCHP exposure.

EPA Response: As discussed in the DCHP risk evaluation and the 2025 CRA TSD, DCHP urinary metabolites were excluded from NHANES after 2010 due to a low number of detects and low detection levels. EPA did not identify any reasonably available national scale DCHP biomonitoring data after 2010. Although the public commenter states that DCHP is used as an alternative for DEHP and that it would be logical to expect DCHP exposures to increase over time, the commenter does not provide any references to support these statements. Further, as discussed in the 2025 CRA TSD (see Table_Apx D-1), EPA considered national aggregate production volume data from Chemical Data Reporting (CDR). For DCHP, production volume has remained constant, ranging from 500,000 to less than 1,000,000 lbs, from 2012 to 2019.

Summary: A public comment (DCHP-0122) recommends that EPA "consider conducting a subgroup analysis for male children to estimate risk for highly exposed and sensitive population groups" based on the results of the following NHANES analysis:

The commenters performed an NHANES analysis "to investigate any potential differences in exposure in male children by SES. We combined two NHANES cycles, 2015-2016 and 2017-2018, to increase sample size." The commentors state that "Our estimates are slightly higher than those reported by EPA and may be attributable to differences in estimation methods or statistical software, however, the estimates are similar (Table 1)."

The commentors state that "In our subgroup analysis of male children by SES, we found a concerning difference in cumulative risk levels. The risk cup contribution is 29% in male children ages 3-5 who are below the poverty level (95th percentile exposure), compared to 20% in those above the poverty level (Table 2). Since DINP was not available in the 2015-2016 analysis, the risk cup contribution in children below the poverty level could be even larger. The comment concludes that this is "a missed opportunity to demonstrate the intent to protect PESS" and that "the most sensitive children may not be protected with the current exposure estimation approach."

EPA Response: EPA acknowledges that it did not attempt a subgroup analysis for male children due to the small sample size for this age group and sex (N of 267 in 2017 to 2018 NHANES data set). Although it may be possible to combine NHANES data from 2015 to 2016 and 2017 to 2018 to increase the sample size for male children and conduct a subgroup analysis, EPA disagrees with the public commenter that it is appropriate to do so. Temporal trends analysis has clearly demonstrated declining exposure to DEHP, DBP, and BBP for all populations within NHANES. Similarly, for DINP, exposure levels have also been on the decline since 2014. Therefore, by combining 2015-16 and 2017 to 2018 NHANES data, the public commenter has overstated their cumulative exposure and cumulative risk estimate for male children.

Summary: Public comments (SACC25-0137, SACC25-0163) were submitted regarding compounding conservative estimates in the exposure models. Commentors noted that sentinel scenarios were upper-bound estimates useful for screening but are not representative of the distribution of exposures for risk management. Commentors explained how inclusion of both NHANES data and consumer exposure scenarios may double-count common exposures, leading to further overestimation in exposures. The commentors suggested EPA address how these issues have the potential for overestimation and uncertainty in the cumulative exposure estimates.

The public commentor (SACC25-0137) posited that "EPA further overestimates consumer exposure by failing to consider that consumer exposure is likely to be already represented in NHANES data. NHANES biomonitoring data already reflects cumulative exposures in the general population, particularly for frequently used consumer products, as identified in the consumer exposure assessment…" and that consumer exposure may be double-counting risk.

EPA Response: EPA acknowledges that individual exposure scenarios provided estimates of consumer exposure using reasonably available data, but the development of cumulative consumer exposure scenarios that involve combining NHANES measured concentrations with these deterministic exposure estimates across multiple COUs for multiple phthalates without data to support a coherent exposure profile of a consumer may lead to unrealistic cumulative exposure estimates that may yield both large overestimation and underestimation of exposure scenarios, according to the SACC (U.S. EPA, 2025). As part of the phthalate CRA, EPA evaluated background exposure to DEHP, DBP, DIBP, BBP, and DINP using NHANES urinary biomonitoring data and reverse dosimetry. Although, EPA cannot attribute exposure measured in NHANES to specific routes or sources of exposure, U.S. CPSC (2014) and Health Canada (2020) have found that the main sources of exposure to phthalates is through diet (non-TSAC use), certain cosmetics (non-TSCA use), indoor air, and indoor dust (See Section 3.4 of the CRA TSD for further discussion). Therefore, EPA believes that all consumer TSCA COUs may not be represented in the NHANES data and supports adding individual consumer TSCA COUs to the background exposure for assessing cumulative risk. If cumulative risk is identified using upper-bound estimates, EPA refines the exposure assessment by considering additional information as available to refine modeled estimates to represent realistic exposure scenarios.

Cumulative Consumer Exposure

Summary: A public comment (SACC23-0052) stated that EPA did not provide sufficient justification for excluding a product with < 2.5% of DEHP, BBP, and DEHP from the cumulative risk evaluation. The commenter used a Consumer Product Safety Commission (CPSC) ban of phthalates > 0.1% in children's toys as support for evaluating products with low concentrations of phthalates.

EPA Response: As quoted by the commenter, the referenced clay polymer product was not incorporated into the cumulative exposure assessment due to a lack of information regarding the specific relative concentrations (in weight or volume) of each phthalate including DEHP, BBP, and DBP. The reported concentrations were all reported to be less than 2.5 percent for each phthalate with no clear bounds for this range. EPA did not incorporate this product's potential contributions to the cumulative phthalate exposures and risks to reduce uncertainty associated with the lack of clarity associated with the concentrations of phthalates in the clay polymer product.

For children's toys, EPA acknowledged that the CPSIA final rule limits the concentration of certain phthalates to 0.1 percent in children's toys and childcare articles. However, the High Priority Chemicals Data System (HPCDS) database contained test data for each phthalate for a significant number of toys and games from 2017 to 2024. See Section 2 of the TSD for each phthalate for a description of the HPCDS legacy children's toys concentrations. EPA assessed exposure to DEHP, BBP, DIBP, and DBP in children's toys under two scenarios. In the first exposure scenario, new toys produced for the U.S. market are assumed to comply with the regulatory limit (0.1%). In the second scenario, legacy toys are assessed with weight fractions reported in the HPCDS database (WSDE, 2020). The HPCDS database reported above 0.1 percent concentrations for toys tested between 2017

and 2024. Given the identified evidence, EPA assessed exposures to children's toys for concentrations above the regulatory limit.

Summary: A public comment (DCHP-0128) recommended EPA aggregate dust and consumer exposures within the cumulative risk assessment, stating that phthalates are a persistent and continuous source of indoor exposure.

EPA Response: As noted by the commentor, EPA did not combine aggregate dust and consumer exposures in the cumulative risk assessment across all phthalates currently under TSCA risk evaluation. Aggregate dust concentrations of phthalates already likely represent combined sources of phthalate exposure. EPA acknowledges that the phthalates assessed in the CRA can be ubiquitous in the indoor environment. However, the Agency does not have the data necessary to rule out the potential for double counting between the measured phthalate concentrations in dust and the modeled phthalate concentrations from TSCA products which already contributed to the indoor dust. This potential overestimation would add uncertainty and lower the overall confidence in the cumulative risk assessment.

EPA has estimated cumulative exposure and risk from exposure to BBP, DBP, DCHP, DEHP, DIBP, and DINP from ingestion of house dust (see Section 3.2 of the CRA TSD). The highest cumulative phthalate exposure from ingestion of house dust was for children (3–5 years of age) using high-end dust ingestion assumptions and 95th percentile phthalate concentrations in house dust. When comparing these dust intake estimates to cumulative MOEs for NHANES, the percent contribution of NHANES to the risk cup is always much greater than ingestion of settled dust (*i.e.*, the risk cup was 15.5% full for NHANES vs. 3.4% full from ingestion of house dust). This is anticipated as NHANES urinary biomonitoring provides an estimate of aggregate exposure (*i.e.*, exposure via all routes and pathways, including dust ingestion) to each phthalate rather than just through ingestion of phthalates in settled dust.

4.6 Dermal Exposure Assessment

Study Selection

Summary: A public comment (SACC25-0030) supported the selection of *in vitro* human skin models over *in vivo* guinea pig data to estimate dermal absorption of DBP stating it is more biologically relevant and reduces reliance on animal testing.

Similarly, another comment (SACC25-0154) states that "EPA's move to refine dermal exposure estimates for DBP using *in vitro* human skin models marks a significant step toward more biologically relevant science," noting that this method may prove more reliable than other common animal studies due to its accounting for human skin permeability.

EPA Response: After further review of available dermal absorption data of DBP, EPA has identified an *ex vivo* study (Beydon et al. (2010)) that used metabolically active human skin for estimating dermal absorption of DBP. The data represent more biologically relevant estimates of dermal absorption in humans.

Summary: A public comment (DBP-0129) stated that the EPA's original proposed rate of dermal absorption for DBP from the draft Risk Evaluation did not support the best available science. The commenter supported the updated dermal approach as outlined in the Memorandum on Dibutyl Phthalate (DBP) Dermal Absorption Data, published June 2025. The commenter identified an additional study regarding the rate of dermal absorption from human (volunteer) exposures to DBP in a liquid solution (Hopf et al., 2024) and suggested that EPA consider this study as well for estimating dermal exposure.

EPA Response: In the *Draft Risk Evaluation of DBP*, the rate of dermal absorption was estimated using data from *in vivo* guinea pig testing which provided only an upper bound of dermal absorption for humans. However, EPA noted this limitation throughout the *Draft Risk Evaluation of DBP*, and EPA subsequently conducted additional research to identify dermal absorption data that are more relevant for human absorption. Further, additional dermal absorption studies were presented to EPA during the public comment period. Consequently, EPA identified the following studies as potential sources of dermal absorption data: Scott et al. (1987), Hopf et al. (2024), and Beydon et al. (2010). The work of Scott et al. (1987) used non-viable human skin samples and a 50 percent aqueous ethanol solution for the receptor fluid which may overestimate absorption, and the work of Hopf et al. (2024) only measured for metabolites of DBP but did not verify that the previously frozen skin samples were metabolically active which may underestimate absorption. However, the study of Beydon et al. (2010) used metabolically active human skin and measured for both metabolites and parent compound during testing, and the study also used a biologically relevant receptor fluid. Given the limitations of data presented in Scott et al. (1987) and Hopf et al. (2024), and given the strengths and biological relevance of the data presented in Beydon et al. (2010), EPA has selected Beydon et al. (2010) as the best available science for estimating dermal absorption of DBP.

Summary: A public comment (DBP-0131) encouraged EPA to consider several alternative dermal exposure assessment approaches for DBP and DEHP, including the IH SkinPerm model (Tibaldi et al., 2014), along with several submodules of the EPA's Consumer Exposure Model. The commenter "encourage[s] EPA to better characterize those exposure determinants that may dictate which dermal exposure estimation approach is preferred (fit for purpose) for a given chemical and its conditions of use."

EPA Response: For dermal exposure to liquid materials containing DBP and DEHP, EPA identified dermal absorption studies that used metabolically active human skin to estimate the rates of dermal absorption of the chemicals. Such biologically relevant dermal absorption data are preferred to modeling. Dermal exposures to solids containing DEHP were estimated based on *in vivo* absorption data from rats exposed to PVC film, and the reported absorption rate is expected to serve as an upper bound of absorption potential from solid matrices in humans. However, there were no dermal absorption data for exposure to solid materials containing DBP. Therefore, EPA used both CEM (U.S. EPA, 2023) and the *Risk Assessment Guidance for Superfund (RAGS), Volume I: Human Health Evaluation Manual, (Part E: Supplemental Guidance for Dermal Risk Assessment)* (U.S. EPA, 2004) to estimate dermal uptake from a saturated aqueous material. Assuming that DBP must first migrate from a solid matrix to a thin layer of liquid on the surface of the skin, dermal uptake of DBP from a solid material is expected to be less than the dermal uptake of DBP from an aqueous material. Consequently, aqueous absorption modeling serves as an upper bound for dermal uptake from a solid matrix for DBP. EPA is continuing to improve dermal absorption modeling and will consider how and when to use dermal absorption models such as IH Skin Perm in future assessments.

Summary: A public comment (DBP-0131) recommended that EPA work with industry to foster partnerships to obtain high quality information about occupational exposure to phthalates, in reference to EPA's public comment request for input on the possibility and likelihood that a non-volatile chemical with low absorption may be contacted multiple times during a work shift (*i.e.*, the worker is handling the chemical intermittently throughout the work shift) and may exist on the skin surface for a total of 8 hours (or until the material on the skin surface is depleted), including the representativeness of this exposure scenario to the COUs.

The commenter recommends that EPA incorporate the AIHA Principles of Good Practice for Exposure Assessment as baseline recommendations for risk management action on occupational exposures.

EPA Response: EPA strives to foster partnerships with stakeholders to understand the scenarios of exposure and typical workplace practices where the chemicals are being handled. Such questions will be considered and revisited as EPA continues risk evaluation of high-priority substances under TSCA.

EPA will set forth proposed regulations to minimize risks to phthalates in a notice of proposed rulemaking before the issuance of a final rule as required by TSCA. The public will have the opportunity to comment on these notices of proposed rulemaking, including recommendations for risk management actions on occupational exposures.

Summary: A public commenter (SACC25-0138) stated Hopf et al. 2024 represents the best available science for determining maximum dermal absorption of DEHP. The study is an *in vivo* human study that measures urinary elimination of phthalate metabolites after dermal exposure to neat DEHP. Similarly, another public commenter (SACC25-0153) expressed concerns with EPA's metrics used for the proposed rate of dermal absorption for DEHP and DBP and recommended the study by Hopf et al. (2024) as it was conducted using human participants ranging from 18 to 65 years old and evaluated the metabolites of DEHP and DBP excreted in urine, and noted that the use of this *in vivo* human exposure data supports values orders of magnitude lower than what was previously proposed for DEHP and DBP.

EPA Response: EPA reviewed the subsequent dermal absorption study by Hopf et al. (2024) that reports dermal absorption rate of DEHP *in vitro* using human skin and *in vivo* with human subjects and included this study in the section of the human health hazard TSD summarizing the dermal absorption studies and explaining the weight of evidence that led to study and dermal absorption value selection (Note: this study evaluated dermal absorption of only DEHP, and not other phthalates, so was only cited as part of the DEHP assessment). The *in vivo* experiments from Hopf et al. (2024) result in similar levels of estimated dermal uptake (approximately 0.010 µg/cm^2/hour) compared to *in vitro* results (0.025 µg/cm^2/hour) reported in metabolically active skin in the earlier study by Hopf et al. (2024); thereby adding to the weight of evidence supporting the selection of the dermal absorption rate from the *in vitro* studies using metabolically active human skin. EPA considered the *in vitro* data from Hopf et al. (2024) to have higher confidence than the value estimated from the *in vivo* study by Hopf et al. (2024) because the estimation from the *in vivo* study relies exclusively on the excreted DEHP and does not account for any DEHP that was absorbed but not excreted or DEHP that was excreted but was from other sources (*e.g.*, dietary exposure). Further, the *in vitro* experiments in the more recent study by Hopf et al. (2024) likely underestimate dermal absorption of DEHP because the

investigators only measured for metabolites of DEHP but did not verify that the previously frozen skin samples were metabolically active, which is a reasonable explanation for the lower dermal absorption rate (0.0002 µg/cm^2/hour) noted in the more recent *in vitro* study by Hopf et al. (2024) compared to the previous *in vitro* study (0.0013 µg/cm^2/hour) using metabolically active human skin (Hopf et al. 2014).

Summary: A public commentor (SACC25-0132) stated that the dermal absorption data used by EPA in the risk evaluation of DBP did not consider a corrected table of data published in a 1989 erratum associated with the original study (Scott et al. 1987).

EPA Response: Though the 1989 erratum of Scott et al. (1987) was overlooked during systematic review, EPA acknowledges the corrected data associated with the original study. However, after receiving public feedback regarding available dermal absorption studies, EPA identified two additional studies as potential sources of dermal absorption data (Hopf et al. (2024) and Beydon et al. (2010)). The work of Scott et al. (Scott et al. 1987) used non-viable human skin samples and a 50 percent aqueous ethanol solution for the receptor fluid which may overestimate absorption, and the work of Hopf et al. (2024) only measured for metabolites of DBP but did not verify that the previously frozen skin samples were metabolically active which may underestimate absorption. However, the study of Beydon et al. (2010) used metabolically active human skin and measured for both metabolites and parent compound during testing, and the study also used a biologically relevant receptor fluid. Given the limitations of data presented in Scott et al. (1987) and Hopf et al. (2024), and given the strengths and biological relevance of the data presented in Beydon et al. (2010), EPA has selected Beydon et al. (2010) as the best available science for estimating dermal absorption of DBP.

Summary: A public comment (DEHP-0139) requested that EPA revise its assumption of 8 hours of DEHP liquid dermal exposure per shift for workers because employees wash their hands throughout the shift. Further, the commenter stated that DEHP liquid or solids should not be assumed to cover the entire surface of one or both hands during the 8-hour shift, and requested that EPA instead assume that only one-half of the hand surface was exposed.

A similar comment (BBP-0123) questioned EPA's use of continuous 8-hour/day dermal exposure as its worst-case scenario, as it may be unrealistic for workers to have both hands exposed to a liquid for 8 continuous hours without washing hands or using personal protective equipment (PPE).

EPA Response: EPA disagrees with the commenters. Though workers are expected to wash their hands throughout a work shift, it is also expected that workers may handle the material intermittently throughout a work shift with multiple contacts per day. Therefore, it is foreseeable that the material may remain on the skin surface for most or all of a work shift. Further, it should be noted that while the surface area of exposed skin is derived from data for hand surface area, EPA did not assume that only the workers hands may be exposed to the chemical. Nor did EPA assume that the entirety of the hands is exposed for all activities. Rather, the Agency assumed that dermal exposures occur to some portion of the hands plus some portion of other body parts (*e.g.,* arms) such that the total exposed surface area is approximately equal to the surface area of one or two hands for the central tendency and high-end exposure scenario, respectively.

Summary: A public comment (DIBP-0130, DIBP-0133) stated that the DIBP draft risk evaluation dermal approach significantly understated exposures to DIBP by relying on an incorrect dermal absorption rate when calculating DIBP's dermal risks. All of EPA's dermal risk calculations are thus based on an absorption rate of 0.00007 mg/cm^2/hour, taken from a 1987 study by Scott et al. However, Scott et al. published an erratum to that study in 1989 correcting the dermal absorption rate for DBP. The corrected rate is 0.0024 mg/cm2/hour, more than 34 times higher the value originally reported.

EPA Response: EPA did not identify any dermal absorption data specific to the dermal uptake of DIBP in human skin. Because DIBP and DBP are isomers with similar physical chemical properties and similar absorption profiles in rats (Elsisi et al., 1989), human absorption data measured for DBP was used as surrogate to estimate the absorption potential of DIBP in humans. Though the 1989 erratum of Scott et al. (1987) was overlooked during systematic review, EPA acknowledges the corrected data associated with the original study. However, after receiving public feedback regarding available dermal absorption studies, EPA identified two additional studies as potential sources of dermal absorption data (Hopf et al. (2024) and Beydon et al. (2010)). The work of Scott et al. (Scott 1987) used non-viable human skin samples and a 50 percent aqueous ethanol solution for the receptor fluid which may overestimate absorption, and the work of Hopf et al. (2024) only measured for metabolites of DBP but did not verify that the previously frozen skin samples were metabolically active which may underestimate absorption. However, the study of Beydon et al. (2010) used metabolically active human skin and measured for both metabolites and parent compound during testing, and the study also used a biologically relevant receptor fluid. Given the limitations of data presented in Scott et al. (1987) and Hopf et al. (2024), and given the strengths and biological relevance of the data presented in Beydon et al. (2010), EPA has selected Beydon et al. (2010) as the best available surrogate data for estimating dermal absorption of DIBP.

Dermal Exposure in the Context of Charge Questions 13, 15 and 16 (Flux-based Approach, Estimation in the Absence of Data, and Dermal Contact Frequency)

Summary: A public comment (DBP-0127) was submitted on the use of a flux-based approach to estimate dermal exposure to materials with low volatility and low rates of absorption in response to Charge Question 13. The comment expressed appreciation for the EPA's application of a flux-based approach for estimating dermal exposure to phthalates, highlighting its potential benefits compared to fractional absorption methods, and encouraged the EPA to better characterize which dermal estimation approach is "fit for purpose" for specific chemicals and conditions of use (*i.e.,* noting that EPA's Consumer Exposure Model (CEM) "includes a number of approaches for estimating dermal exposure to chemicals in consumer products whether an article or a formulated product"), and to develop a more comprehensive strategy for addressing "potential occupational dermal exposures during manufacturing, processing or use of a chemical".

In response to Charge Question 15, the public commenter (DBP-0127) stated that in the context of estimating dermal absorption in the absence of data, EPA's CEM "includes an approach to estimate Dermal Dose from Skin Contact with an Article (A_DER2)." The public commenter further stated that the CEM dermal approach allows for the "estimation of migration of a chemical within an article to the skin via direct article contact" and contents that the CEM approach "appears to be missing from the approach used in the risk evaluation.". The public comment (DBP-0127) suggested there appears to be sufficient information (*i.e.,* available data or methods) to estimate parameters like fraction absorbed and the solid phase diffusion coefficient and stated that "EPA should attempt to estimate dermal absorption of DBP from an article using the CEM A_DER2."

The public commenter (DBP-0127) also commented on Charge Question 16 regarding dermal contact frequency stating that there are "additional critical parameters, such as dermal loading (and skin loading adherence factors) and transfer coefficients, in estimation of dermal exposures to chemicals in environmental and occupational media" and that "EPA should focus less on 'the possibility and likelihood that a non-volatile chemical with low absorption may be contacted multiple times during a work shift' (*i.e.,* it is possible and likely) and more on what is contact frequency, dermal loading and transfer efficiency of a particular condition of use." The public commenter (DBP-0127) also encouraged EPA to "work with industries and companies where condition of use is relevant" as well as to "foster such partnerships so that it has the necessary information to develop high quality risk evaluations."

EPA Response: Though CEM provides a variety of approaches for estimating dermal exposures among scenarios, EPA is working to create more unified methods of dermal exposure estimation across consumer and occupational scenarios. For instance, CEM A_DER2 (dermal model for articles in CEM label) is available in CEM to estimate dermal dose from skin contact with an article. This model first estimates diffusion of the chemical from a solid matrix to the skin surface. Then, it is assumed within the model that all chemical molecules reaching the skin surface are instantaneously absorbed into the stratum corneum. Finally, CEM A_DER2 calculates the fractional absorption of the chemical in the stratum corneum to estimate overall dermal uptake. EPA determined that the CEM A_DER2 model was not appropriate for dermal absorption of phthalates since aqueous solubility is a limiting factor among phthalate chemicals and this factor is not considered in CEM A_DER2. Therefore, EPA used a bounding approach by assuming that absorption from a solid material will not exceed absorption from a saturated aqueous liquid. This bounding methodology indicated that dermal uptake of phthalates from saturated aqueous materials were not significant in comparison to the PODs and benchmark MOEs. Consequently, EPA did not conduct additional analyses in DBP, DEHP, DCHP, and BBP to model transfer of phthalate chemicals from solid matrices to the skin surface since the only outcome would be reduced dermal uptake in comparison to dermal uptake from the saturated aqueous material. For DIBP children's clothing, EPA did additional refinement to consider migration from the solid clothing material to skin to demonstrate reduced dermal uptake. EPA is committed to continued improvement in dermal exposure estimation and plans to work with stakeholders toward a better understanding of the various parameters that affect dermal exposure such as dermal loading, exposure surface area, exposure frequency, and exposure duration.

Summary: A public comment (SACC25-0140) regarding the estimation of dermal absorption for DBP, DEHP and DCHP stated that "Flux based approaches are generally more scientifically sound than fraction absorbed approaches" and use of flux-based approaches "is a step in the right direction."

EPA Response: In light of this comment, EPA has decided to continue to use the flux based approach in these risk evaluations. Dermal exposure assessment is an active area of development and EPA is committed to continued improvement in assessment methodologies.

Summary: A public commenter (DBP-0127) commended the EPA for its thoughtful approach to human exposure assessment for high-priority phthalates, particularly acknowledging the re-examination and introduction of new dermal exposure assessment methods.

The public commenter (DBP-0127) also emphasized the need for a clear understanding of factors like magnitude, frequency, duration, transfer efficiency, and dermal loading, and encouraged close collaboration with industries to gather high-quality, condition-specific data.

EPA Response: EPA appreciates the feedback as it is helpful in guiding the Agency towards the best available science and methods in the final risk evaluations of phthalate chemicals. EPA is committed to continued improvement in dermal exposure estimation and plans to work with stakeholders toward a better understanding of the various parameters that affect dermal exposure such as dermal loading, exposure surface area, exposure frequency, and exposure duration.

Clarification on finite and infinite does definitions

Summary: A public commenter (SACC25-0140) provided clarification on finite and infinite dose definitions noting that the dose definition (*i.e.,* based on the OECD 2004b guidance) for "finite and infinite dose" as "misleading," and clarified that dose status is dependent on chemical load and driving force, the observed or predicted flux into the skin, and exposure duration.

EPA Response: EPA agrees that "finite" and "infinite" dose definitions from OECD guidance can be misleading, especially when applying such definitions to exposure assessment methodologies. For instance, doses below an "infinite" dose may be flux-limited due to low rates of evaporation and absorption of the chemical. Therefore, a dose defined as "finite" by OECD guidance may mimic absorption behavior of doses defined as "infinite" by OECD guidance. EPA considered this observance in the dermal exposure assessments of phthalate chemicals, and consequently, EPA chose a flux-based approach to estimate dermal uptake from "finite" doses of phthalate chemicals.

Differences for permeability coefficients due to adsorption on solid matric or article

Summary: A public comment (SACC25-0140) identified specific differences for permeability coefficients due to whether or not adsorption occurs on the solid matrix or article. It was noted that "Absorption should vary with concentration below saturation, assuming reduced fugacity due to adsorption to a solid matrix" and that for liquid phthalates, "concentration in solids or articles may or may not matter" – *i.e.,* "If the phthalate is not actually adsorbed in the material but merely exists as free liquid in the interstices of a matrix, then the driving force will be that of the pure liquid. However, the permeability coefficient may be reduced due to increased mass transfer resistance (reduced intimate contact between chemical and skin)."

EPA Response: EPA agrees that flux varies at concentrations below saturation and that adsorption of the chemical within the solid matrix affects absorption concentration. However, EPA utilized a bounding approach to estimate the upper limit of absorption a phthalate chemical from a solid matrix. Specifically, it was assumed that absorption from a solid matrix would be less than absorption from a saturated aqueous material. Because estimation at the bounding level showed low levels of dermal uptake in comparison to PODs and benchmark MOEs, EPA did not apply more advanced

methodologies to account for variations in chemical concentration below the saturation limit.

Availability of empirical data for DBP and DEHP (generally all phthalates)

Summary: A public comment (SACC25-0140) regarding availability of empirical data for DBP in liquid products and DEHP in both liquid products and solid articles noted that empirical evidence exists for direct dermal absorption of DBP from vapor (Weschler et al., 2015) and that dermal absorption from vapor "rivals inhalation exposure." The commenter further stated that "Failure to consider dermal absorption of phthalate vapor is potentially a significant weakness" and that it "depends on the relative magnitudes of inhalation, dermal contact and ingestion exposures." The commenter also noted that while "some related material in the MOE results for occupational exposures" were found, "exposure pathway comparisons for consumer exposures" was not found.

EPA Response: EPA has reviewed the vapor to skin modeling methodology of Weschler et al. (2014), as well as the empirical studies of vapor to skin exposure presented in Weschler et al. (2015) and Morrison et al. (2016). The study of Weschler et al. (2015) measured exposure to participants wearing only shorts, and results showed levels of dermal uptake similar to levels of inhalation exposure. However, Weschler et al. (2015) noted that "[a]lthough these experiments indicate substantial dermal uptake directly from air for both DEP and DnBP, the measured values for the contribution of the dermal pathway directly from air are lower than those predicted in recent studies (Weschler and Nazaroff 2012, 2014)." Therefore, the vapor to skin modeling approach of Weschler et al. (2014), which is based on steady-state dermal uptake from vapors, is shown to overpredict dermal exposure to DBP vapor. Further, Weschler et al. (2015) states that "higher-molecular-weight phthalates such as butyl benzyl phthalate (BBzP), di(2-ethylhexyl) phthalate (DEHP), and di(isononyl) phthalate (DiNP) tend to have low gas-phase concentrations. This results in kinetic constraints on the flux from air to skin; it is too small for dermal uptake from air to be an important pathway for compounds such as DEHP and DiNP." The study of Morrison et al. (2016) investigates the effect of clothing, both clean and contaminated, on dermal uptake of phthalates. Morrison et al. (2016) showed that clean clothes have a significant protective effect on dermal uptake of DBP vapor, while contaminated clothing led to increased levels of dermal exposure. Since it is assumed that workers will wear clean clothing to work, rather than clothing that has been saturated with phthalate chemicals, the results based on the use of clean clothing from Morrison et al. (2016) are most relevant to the dermal exposure assessment. EPA has included discussion of vapor to skin exposures in the risk evaluations of DBP and DIBP.

Few measured values of partition coefficients for other solids in estimating uptake from solid materials

Summary: A public comment (SACC25-0140) regarding the use of aqueous absorption modeling to estimate dermal uptake from solid materials noted that "[u]se of aqueous permeability coefficients to estimate dermal absorption from solids is common and reasonable if partition coefficients are available" and that "[s]oil-water partition coefficients are readily estimable." However, the commenter noted that "[f]ew measured values exist for other solids" and that "[p]hthalate containing products (*e.g.,* floor tile) in which the sorption status of the phthalate is not well characterized are problematic."

EPA Response: EPA agrees that partition coefficients from solids to water will affect dermal uptake of phthalates from solid materials. However, EPA utilized a bounding approach to estimate the upper

limit of absorption a phthalate chemical from a solid material. Specifically, the rate of dermal uptake of a saturated aqueous chemical was used to estimate the potential dermal uptake of a chemical from a solid material, and it is expected that a saturated aqueous chemical will have greater rate of absorption than a chemical within a solid material. Because estimation at the bounding level showed low levels of dermal uptake in comparison to PODs and benchmark MOEs, and bounding level estimates are conservative by definition, EPA did not apply more advanced methodologies to account for variations in solid to water partition coefficients.

Estimation of aqueous permeability coefficients

Summary: A public comment (SACC25-0140) regarding the use of the Consumer Exposure Model (CEM) to approximate the aqueous permeability coefficient and the use of the Superfund Guidance for Dermal Risk Assessment to estimate the dermally absorbed dose noted that "[m]ultiple QSARs for prediction of permeation from water are available" and "give similar results for similar inputs," however "[l]arge discrepancies can be produced depending upon the experimental or estimated value of the Log Kow used." The comment specifically noted that if "CEM is still using Potts-Guy," it is a "reasonable starting place," and that "Log Kow values should be subjected to a sensitivity analysis."

EPA Response: EPA acknowledges that there are several models for estimating aqueous permeability coefficients, and consequently, EPA reviewed the various models to determine the most appropriate for the phthalate chemicals. The Potts-Guy model for estimating aqueous permeability coefficient was developed using a set of chemicals that generally have lower octanol-water partition coefficients and molecular weights than the phthalate chemicals under evaluation. However, the QSAR-based model of ten Berge, described within CEM, was determined to be more suitable for estimating aqueous permeability coefficients of phthalate chemicals given the model compatibility with higher molecular weights and octanol-water partition coefficients.

Use of non-steady state predictor to derive steady state fluxes

Summary: In response to Charge Question 15b, a public commenter (SACC25-0140) stated that the Superfund Guidance for Dermal Risk Assessment remains useful, however, it is not clear whether EPA is using the Superfund guidance correctly. The public commenter further stated: "The text below Figure 2.1 in the Consumer and Indoor Exposure Assessment for DBP implies that the cumulative non-steady predictor (Eq. 3.2 in RAGS E/Eq. 2.1 in the DBP document) result has been run sequentially for up to 24 hours and resulting uptakes are somehow being averaged and declared "steady state" fluxes. Eq. 3.2 in RAGs E produces cumulative absorption (mass per event time) for a specified interval under non-steady conditions For short duration exposures, the non-steady result should be estimated for the actual duration specified and used directly. For longer term exposures, the plausibility of reaching steady state should be evaluated (is Tevent > 2.4·Taulag?) and the steady state solution used if appropriate."

EPA Response: The *Risk Assessment Guidance for Superfund (RAGS), Volume I: Human Health Evaluation Manual, (Part E: Supplemental Guidance for Dermal Risk Assessment)* (U.S. EPA, 2004) provides a set of equations (Eq. 3.2 and Eq. 3.3 in RAGS) for estimation of the dermal uptake of an aqueous chemical for given absorption duration. This set of equations accounts for variations of flux over time. Dividing the estimated dermal uptake quantity by the associated absorption time, the quotient yields the average rate of absorptive flux over the absorption period. Though the rate of

absorption does vary over the absorption period, EPA has calculated the average flux over the absorption period that is specific to each consumer or occupational exposure scenario. As time approaches infinity, Eq. 3.3 in RAGS is driven towards the steady-state solution.

Phthalates as SVOCs

Summary: A public comment (SACC25-0140) stated that "The phthalates of concern here are not 'non-volatile'" and that "they are semi-volatile organic compounds (SVOCs)." The commenter noted that SVOCs "redistribute via vapor deposition…when introduced into indoor environments" and that "[r]eleases via open containers, imperfectly sealed containers, spills, fugitive emissions from production lines, etc. would be expected to produce continuous releases into the workplace, leading to a pseudo steady state condition when balanced against ventilation and cleaning losses." The commenter also stated that "[r]epeat contact would not only be possible but virtually certain" since air handling even to nonproduction areas would have to be "scrupulously segregated."

EPA Response: EPA agrees with the commenter. Phthalate chemicals are semi-volatile organic compounds, and consequently, exposures to phthalate vapors are captured within the phthalate risk evaluations. Further, by considering that dermal absorption may occur for the entirety of a work shift, the occupational dermal exposure estimates of phthalates capture the potential for repeat contacts in the workplace.

Database of vapor pressures of SVOCs above solid products

Summary: In response to Charge Question 15a, a public comment (SACC25-0140) stated: "Use of aqueous permeability coefficients to estimate dermal absorption from solids is common and reasonable if partition coefficients are available (to convert aqueous phase driving force to solid phase driving force). Soil-water partition coefficients are readily estimable. Few measured values exist for other solids. Phthalate containing products (*e.g.*, floor tile) in which the sorption status of the phthalate is not well characterized are problematic. A database (or MSDS requirement) describing vapor pressures of SVOCs above solid products has been needed for a long time.

EPA Response: EPA agrees that further investigation regarding vapor pressures of SVOCs above solid products would be a useful tool in TSCA risk evaluations, and such work will be considered for future development.

Additional references

Summary: A public comment (SACC25-0140) provided the following three additional references cited in the comments but not in the SACC phthalate documents to consider (*i.e.*, in reference to Charge Questions 14, 15 and 16):

1. Weschler, C. J.; Nazaroff, W. W. Semivolatile organic compounds in indoor environments. Atmos. Environ. 2008, 42, 9018−9040
2. Weschler, C. J., & Nazaroff, W. W. (2014). Dermal uptake of organic vapors commonly found

in indoor air. Environmental science & technology, 48(2), 1230-1237.

3. Weschler, C. J., Bekö, G., Koch, H. M., Salthammer, T., Schripp, T., Toftum, J., & Clausen, G. (2015). Transdermal uptake of diethyl phthalate and di (n-butyl) phthalate directly from air: experimental verification. Environmental health perspectives, 123(10), 928-934.

EPA Response: EPA has reviewed the references above regarding dermal uptake to phthalate vapors. Discussion of the relative level of vapor to skin exposure has been included in the risk evaluations of DBP and DIBP. However, vapor to skin exposures are not expected to be as significant for higher-molecular-weight phthalates as noted in the work of Weschler et al. (2015).

Summary: A public commenter (SACC25-0132) stated that the assumptions of a 3-hour exposure shift and an exposure surface area of one full hand when assessing phthalate exposures from protective gloves underestimates exposures, and states that an 8 hour exposure duration and a 2 full hand exposed area would appropriately reflect real world conditions.

EPA Response: EPA believes that the commenter is mixing the occupational scenario and the consumer scenario of the same name even though they (occupational versus consumer) are assessing different subcategories of use. The commenter appears to indicate as much when they discuss fabrication of DEHP-containing final products with the duration of exposure from wearing DEHP-containing (rubber) gloves as PPE.EPA did not assess occupational/commercial risk to DEHP-containing gloves for PPE exposure related scenarios. An occupational/commercial MOE value was evaluated assuming glove use every day (surface area of two hands for 8 hours per day), and the resulting MOEs were greater than 250, which is well above the benchmark of 30.

Exposure to consumers from DEHP-containing gloves was evaluated in the Packaging, paper, plastic, toys, hobby products; Packaging (excluding food packaging) and other articles with routine direct contact during normal use, including rubber articles; plastic articles (hard); plastic articles (soft) COU. EPA identified two sources (Tsumura et al., 2001 and Danish EPA, 2020) for PVC and work gloves available to consumers. These gloves were evaluated under the PVC articles with potential for semi-routine dermal exposure consumer scenario. This consumer exposure scenario was developed for a variety of consumer goods that are (1) not expected to be mouthed, (2) not expected to result in significant inhalation exposure due to their small size and/or outdoor only use, and (3) not expected to result in significant dermal exposures due to short and/or infrequent dermal contact events.

Summary: A public commenter (DIBP-0133) stated that EPA underestimated workers' dermal exposures for DIBP and BBP when assuming the high-end value of 1,070 cm^2 for male workers and 890 cm^2 for female workers. The selected values are not "high-end" estimates, they "are based on the mean two-hand surface area for adults." EPA should use the 95th percentile surface values of 1,310 cm^2 for male workers and 1,060 cm^2 for female workers.

EPA Response: There are no available data on the distribution of exposure surface areas for occupational uses of phthalates. Consequently, EPA considers two distinct cases to estimate potential exposure surface areas in occupational settings: one-hand surface area for central tendency scenarios and two-hand surface area for high-end scenarios. These two cases are based on scenarios outlined by the *EPA/OPPT 1-Hand Dermal Contact Model* and the EPA/OPPT 2-Hand Dermal Contact Model (U.S. EPA, 2015). The EPA/OPPT 2-Hand Dermal Contact Model uses the mean surface area of two hands presented in the EPA Exposure Factors Handbook (U.S. EPA, 2011) (*i.e.*, 1,070 cm^2 for adult

males and 890 cm^2 for adult females), and this model is used to represent high-end exposure activities such as loading/unloading chemicals to and from transport vessels and cleaning residual material from transport containers. Though the surface area of two hands may vary among workers, EPA currently considers the mean value of two adult hands as an appropriate high-end exposure scenario for occupational dermal exposures. Further, it should be noted that while the surface area of exposed skin is derived from data for hand surface area, EPA did not assume that only the worker's hands may be exposed to the chemical. Nor did EPA assume that the entirety of the hands is exposed for all activities. Rather, EPA assumed that dermal exposures occur to some portion of the hands plus some portion of other body parts (*e.g.,* arms) such that the total exposed surface area is approximately equal to the surface area of one or two hands for the central tendency and high-end exposure scenario, respectively.

5 HUMAN HEALTH HAZARD ASSESSMENTS

Comments associated with this topic are summarized in the subsections below.

5.1 Non-Cancer Hazard Assessment

Summary: A public commenter (SACC25-0132) supported EPA's use of animal laboratory data to assess the toxicity of DBP and DEHP, stating "TSCA's mandate to use the "weight of the scientific evidence" when conducting chemical risk evaluations compels the use of laboratory studies of animals that are models to understand health hazards in humans."

EPA Response: EPA acknowledges the support from this public commenter (SACC25-0132) to use animal laboratory data to assess the toxicity of DBP and DEHP, and has retained use of these data in the final risk evaluations of DBP and DEHP.

Developmental and Reproductive Hazards, Point of Departure (POD) Selection

Summary: A public commenter (DCHP-0127) stated that "EPA should use the $BMDL_5$ for DCHP derived from application of the NASEM meta-regression model as the POD in the final DCHP risk evaluation." The comment details 5 reasons they believe "EPA's claim that the DCHP BMDL is too low to be trusted is scientifically inappropriate." These include:

1. "As summarized by EPA, studies of DCHP male reproductive toxicity demonstrate reproductive toxicity at levels very close to the BMD and BMDL."
2. "EPA's disregards the many factors that should give it increased confidence in the DCHP BMDL. Most importantly, it is derived from a meta-regression approach developed and published specifically for anti-androgenic phthalates by the nation's authoritative scientific body, the NASEM."
3. "Using the meta-regression BMD/BMDL to derive RPFs for the phthalates cumulative risk assessment but not for single-chemical phthalates risk evaluations produces inconsistent results."
4. "EPA's use of 10 mg/kg-day as the POD is highly inconsistent with its appropriate and scientifically justified rationale for use of a 5% benchmark response (BMR) for reduced fetal testosterone rather than a higher effect level. The NOAEL of 10 mg/kg-day represents a dose at which a greater than 5% reduction in fetal testosterone occurs; therefore, using this value as the POD contradicts the selection of a 5% BMR.
5. "Using a NOAEL as the POD rather than a benchmark dose (BMD) is not consistent with the best available science, as stated in EPA guidance and reports from the NASEM."

EPA Response: EPA disagrees with the five reasons listed by the public commenter (DCHP-0127); EPA believes that, based on the weight of scientific evidence, its selected POD for DCHP is appropriate for use in the individual assessment of DCHP. EPA considered the $BMDL_5$ of 6 mg/kg-day as the POD, which was derived via meta-analysis of fetal testicular testosterone data from three studies reported in two publications (Furr et al. 2014; Gray et al. 2021). EPA also modeled the three individual fetal testicular testosterone datasets included in the meta-analysis using EPA's BMD Online Software (BMDS Online Version 25.1), and this additional analysis is included in the final non-cancer

human health hazard assessment for DCHP. No viable model fits were obtained for the *ex vivo* fetal testicular testosterone production data from Block 23 rats reported by Furr et al. For the *ex vivo* fetal testicular testosterone production data from Block 33 rats reported by Furr et al., the Exponential 3 model provided the best fit, and supports BMD_5 and $BMDL_5$ estimates of 9.0 and 5.2 mg/kg-day. For the *ex vivo* fetal testicular testosterone production data reported by Gray et al. 2021, the Exponential 3 model provided the best fit, and supports BMD_5 and $BMDL_5$ estimates of 13.7 and 10.0 mg/kg-day. $BMDL_5$ estimates for reduced fetal testicular testosterone range from 5.2 to 10 mg/kg-day. However, a major uncertainty with these $BMDL_5$ estimates is that they are lower than the lowest dose at which there is empirical testosterone data available. Indeed, the lowest dose-to-BMDL ratio ranges from 5.5 to 10 across the three $BMDL_5$ estimates. The lack of data to inform the low-end of the dose-response curve reduces EPA's confidence in using the $BMDL_5$ estimates, and is consistent with EPA's *Benchmark Dose Technical Guidance* (U.S. EPA, 2012), which states (pp. 20), "For some datasets the observations may correspond to response levels far in excess of a selected BMR and extrapolation sufficiently below the observable range may be too uncertain to reliably estimate BMDs/BMDLs for the selected BMR … in such cases, BMD modeling is not recommended and obtaining more data or using the NOAEL/LOAEL approach, while recognizing the inabilities of that approach to resolve the data limitations, may be warranted."

Instead, EPA selected a NOAEL of 10 mg/kg-day based on data from Li et al. (2016) as the basis of its POD for DCHP. As stated in the final non-cancer hazard TSD and final risk evaluation for DCHP, the NOAEL is supported by five studies that support a narrow range of doses between the NOAELs of 10 to 17 mg/kg-day and LOAELs of 20 to 33 mg/kg-day, based on effects on the developing male reproductive system consistent with phthalate syndrome.

EPA recognizes that at the NOAEL of 10 mg/kg-day there is a 10% decrease in testicular testosterone content in postnatal day 1 rats. However, there was a large amount of variability in the testosterone data within control and dose groups, and the change in testosterone at 10 mg/kg-day was not statistically significant (testosterone was not significantly decreased until the next dose from of 100 mg/kg-day). EPA attempted to BMD model testosterone data from Li et al. (2016), and derived BMD_5/$BMDL_5$ estimates of 6.9/1.2 mg/kg-day from the best-fitting Hill model. However, the variability in the testosterone data is reflected in the BMD/BMDL ratio of 5.8, and the fact that the $BMDL_5$ is nearly an order of magnitude below the lowest dose with empirical testosterone data. Similar to the $BMDL_5$ estimates discussed in the paragraph above, the lack of data to inform the low-end of the dose-response curve reduces EPA's confidence in using the $BMDL_5$ estimate.

For these reasons, EPA did not select a $BMDL_5$ for use as the POD for DCHP because the NOAEL of 10 mg/kg-day was more appropriate.

Summary: A public commenter (DCHP-0128) stated that EPA incorrectly characterized the NOAEL and LOAEL values from the Li et al. 2016 study used to inform the DCHP POD and recommends EPA reclassify 10 mg/kg/day as the LOAEL and add an uncertainty factor for the use of a LOAEL. The comment maintains that the study reported clear dose-response for DCHP and phthalate syndrome related effects (*e.g.*, abnormal fetal Leydig cell aggregation) at 10 mg/kg/day. The commenter notes that Health Canada's phthalate assessment concluded that the Li et al. 2016 study did not identify a NOAEL.

In addition, the public comment (DCHP-0128) recommends that EPA consider a Liu et al. 2023 study as evidence that the selected acute POD for DCHP may not be protective of longer durations of

exposures. The referenced study is described as identifying alterations in gene expression in the liver of offspring following 4 weeks of paternal 10 mg/kg/day DCHP exposure.
EPA Response: EPA disagrees with the public commenter that it incorrectly characterized the NOAEL and LOAEL values from the Li et al. (2016) study.

Although this study provides evidence of effects on the developing male reproductive system consistent with phthalate syndrome, EPA did not consider the effects observed at 10 mg/kg-day to be clearly adverse. Statistically significant effects at 10 mg/kg-day are limited to fetal Leydig cell effects, decreased expression of genes and proteins involved in steroidogenesis, and decreased protein expression of INSL3 (all of which are not considered adverse in isolation). Though involved in the MOA of rat phthalate syndrome, the effects are of uncertain toxicological significance by themselves. The remaining effects listed reached statistical significance at higher doses, including decreased testosterone (10% decrease in low-dose group), decreased absolute male AGD (9% decrease in low-dose group), and increased MNGs per seminiferous tubule were also observed at 10 mg/kg-day but did not reach statistical significance until 100 mg/kg-day. Therefore, EPA considers this study to support NOAEL and LOAEL estimates of 10 and 100 mg/kg-day, respectively

EPA acknowledges that the public commenter correctly identified that Health Canada considered Li et al. (2016) to support a LOAEL of 10 mg/kg-day. However, it is also important to note that Health Canada did not apply an additional uncertainty factor for use of a LOAEL.

EPA thanks the public commenter for providing the Liu et al. 2023 reference. EPA reviewed the provided information, however because the study only evaluated a single exposure level (*i.e.*, 10 mg/kg-day) and reported DCHP-related effects on non-apical outcomes, such as sperm small non-coding RNAs including rRNA-derived small RNAs (rsRNAs) and tRNAs-derived small RNAs (tsRNAs), and hepatic transcriptomic changes, and was therefore not suitable for dose-response assessment and not further considered.

Summary: A public comment (SACC25-0139, BBP-0123) asserts that peroxisome proliferator-activated Receptor Alpha (PPAR-α) is an inappropriate MIE, and reported evidence from several studies (Gray et al. 2021; Boberg et al. 2008; Foster et al. 2001; Foster et al. 2005; Parks et al. 2000) which the commenter stated provide evidence of other possible mechanisms of action through which BBP may be acting -- steroid hormone metabolism, inflammatory reactions, and ROS metabolism -- indicating potential alternative MIEs outside of PPARα.

EPA Response: EPA believes the public commenter (SACC25-0139, BBP-0123) has mischaracterized EPA's conclusions pertaining to the MIE associated with reduced fetal testicular testosterone and phthalate syndrome. As discussed in the non-cancer human health hazard TSDs for DEHP, DBP, BBP, DIBP, and DCHP, and as shown in the figure depicting the phthalate syndrome MOA in each TSD (*e.g.*, figure 3-1 in the DEHP non-cancer human health hazard TSD), EPA acknowledges that the MIE associated with phthalate syndrome is unknown and that the MIE is not believed to be androgen receptor or PPARα mediated. Therefore, EPA is in agreement with the public commenter that PPARα is not the MIE associated with the phthalate syndrome MOA.

Summary: A public comment (SACC25-0145) asserted that EPA inappropriately relied on NOAELs and LOAELs instead of relying on results of BMD modeling, contrary to EPA guidance and NASEM's recommendations. Specifically for BBP, the commenter considered the selection of the

NOAEL of 50 mg/kg-day as the POD for characterizing non-cancer risks for BBP inappropriate because PODs for BBP derived from the updated NASEM meta-regression model were available, including the BMD_5 of 31 mg/kg-day and $BMDL_5$ of 17 mg/kg-day with the updated meta-regression linear quadratic model using Metafor version 2.0.0 and a BMD_5 of 22 mg/kg-day and $BMDL_5$ of 17 mg/kg-day with the updated analysis using the linear model from Metafor version 4.6.0. The commenter also noted that EPA excluded lower NOAELs than 50 mg/kg-day from consideration, including the NOAEL of 11 mg/kg-day from Gray (2021), noting that the decreased steroidogenic gene expression a step in the AOP, and the NOAEL of 20 mg/kg-day from the study by Ahmad *et al*.

Similarly, another public comment (SACC25-0139, BBP-0123) suggests that rather than using the NOAEL of 50 mg/kg-day as the POD associated with decreased AGD at 100 mg/kg-day in the study by Tyl et al. (2004), EPA should use decreased fetal testicular testosterone (FTT) as the critical effect because it occurs upstream in the AOP, and would provide a POD of 61.44 mg/kg-day based on the $BMDL_{1SD}$ for FTT (Furr et al., 2014 and Gray et al., 2021). This commenter provided the calculations of the adjusted PODs (HED 14.5 mg/kg-day, HEC 78.9 mg/m^3), and occupational exposure values (OEV) of 3.87 mg/m^3 (acute), 5.27 mg/m^3 (intermediate), and 5.65 mg/m^3 (chronic) for BBP.

EPA Response: EPA disagrees with the public commenter on the dose-response analysis and POD selection for BBP (SACC25-0145). In performing the meta-analysis of *ex vivo* fetal testicular testosterone production, EPA conducted dose-response assessment with updated Metafor Version 4.6.0 and interpreted results according to EPA's Benchmark Dose Technical Guidance. Using best practices in benchmark dose modeling guidance, the meta-analysis did not yield acceptable fits for BBP in the most statistically appropriate model (Linear Quadratic model, based on lowest AIC). Thus, EPA conducted further BMD analysis of individual *ex vivo* fetal testicular testosterone production study datasets. This analysis identified one acceptable model fit from Howdeshell et al. (2008) resulting in a $BMDL_5$ of 81 mg/kg-day.

In revising the BBP human hazard assessment following the 2025 SACC meeting, EPA added additional detail throughout Section 4.2 of the Human Health Hazard TSD describing the consideration of both BMD modeling and NOAEL/LOAEL approaches for studies considered during dose-response analysis to determine the POD for BBP. In revising the dose-response assessment consistent with EPA's Benchmark Dose Technical Guidance, EPA considered multiple studies providing potentially sensitive endpoints for BMD modeling to refine the identified NOAEL/LOAEL values for sensitive studies (*e.g.*, testicular histopathology, sperm parameters, organ weights). Studies that were considered were those that showed a NOAEL or LOAEL at 100 mg/kg-day or below, where a consensus LOAEL of 100 mg/kg-day was noted for BBP effects. For many of the endpoints considered for BMD modeling but not modeled, EPA added discussion on why BMD analysis was not appropriate, including data reporting deficiencies, lack of dose-response data, or data identified as generally not amenable to BMD modeling. BMD analysis notes for each study considered were added to Table 4-1 in the BBP human health hazard assessment TSD. However, of the sensitive studies, testicular histopathology effects of testes softening and seminiferous tubule atrophy were identified in Aso et al. (2005) as sensitive dose-responsive endpoints suitable for BMD analysis. EPA conducted BMD analysis of these endpoints, which resulted in a $BMDL_5$ of 55 mg/kg-day based on increased incidence of seminiferous tubule atrophy in F1 males (see Appendix G of the BBP human health hazard TSD for BMD modeling results). Overall, in considering and conducting further BMD analysis in revisions, BBP effects still fell within a NOAEL/LOAEL of around 50/100 mg/kg-day. Section 4.2.4 in the BBP human health hazard assessment TSD provides a summary of EPA's conclusions on the additional BMD modeling that was conducted between the draft and final assessment.

Also discussed in dose-response assessment, two studies provided effect levels below the NOAEL of 50 mg/kg-day identified in Tyl et al. (2004) based on decreased AGD in F1 and F2 rats. Ahmad et al. (2014) supports a NOAEL of 20 mg/kg-day based on decreased serum testosterone, decreased epididymis and prostate weight, and sperm effects at the LOAEL of 100 mg/kg-day. Gray et al. (2021) supports a NOEL of 11 mg/kg-day based upon fetal testicular gene expression decrease in Insl3 at the LOEL of 33 mg/kg-day. Gray et al. (2021) was noted to show a gene expression change at the low-dose range, but *ex vivo* fetal testicular testosterone was decreased at a LOAEL of 300 mg/kg-day. The low-dose gene expression effect was not considered adverse in isolation of other phthalate-syndrome outcomes (*e.g.*, testosterone production, histopathology, malformations), and the *ex vivo* fetal testicular testosterone from Gray et al. (2021) was included in the BMD analysis of fetal testosterone level meta-analysis and individually (no acceptable model fits were found). Sensitive endpoints reported at the NOAEL in Ahmad et al. (2014) were discussed as well (and considered for BMD analysis), but EPA identified data reporting deficiencies, study limitations, and inconsistencies compared to other studies. Thus, EPA had higher confidence in the next most sensitive NOAEL of 50 mg/kg-day from Tyl et al. (2004). Additionally, the LOAEL of 100 mg/kg-day from Ahmad et al. (2014) was considered with multiple other LOAELs reported in other studies for sensitive phthalate-syndrome effects. Overall, the lowest and most consistent LOAEL for BBP was found to be 100 mg/kg-day across co-critical studies, and the most reliable NOAEL of 50 mg/kg-day was identified by Tyl et al. (2004).

EPA disagrees with the public commenter (SACC25-0139, BBP-0123). As discussed in a similar public comment (SACC25-0145), EPA considered *ex vivo* fetal testicular testosterone data for BBP dose-response assessment through BMD modeling. However, the most appropriate model output for this data (individual dataset by Howdeshell et al. (2008)) resulted in a $BMDL_5$ of 81 mg/kg-day, which was less sensitive than other identified effect levels and was discussed in relation to more sensitive PODs, including the NOAEL of 50 mg/kg-day based on decreased AGD reported in Tyl et al. (2004). In its revisions of the Human Health Hazard TSD for BBP, EPA discussed additional dose-response assessment considerations and added text on the robust and sensitive nature of decreased AGD effects within the phthalate-syndrome MOA, noting decreased AGD is mechanistically linked to reduced fetal testicular testosterone production.

Summary: A public commenter (SACC25-0132) stated that EPA must revise its oral POD for DEHP (4.8 mg/kg-day) to account for toxicological uncertainty. The commenter contends that lower Minimal Risk Levels (MRLs) for DEHP developed by ATSDR (and rejected by EPA) are defensible, and that additionally EPA ignored studies indicating observed adverse effects at doses ranging from 0.03 to 0.05 mg/kg-day. The commentor contends EPA must reconsider PODs selected by ATSDR or include a database deficiency factor of 10. They further contend that retaining the current data sources for oral POD requires benchmark dose analysis.

EPA Response: EPA disagrees with the public commenter (SACC25-0132). EPA reviewed the studies and broader toxicological databases of studies underlying the ATSDR MRLs, including effects on the developing female reproductive system, effects on glucose homeostasis, immune system effects, and neurodevelopmental effects, some of which may have occurred at doses ranging from 0.03 to 0.05 mg/kg-day. As discussed in the non-cancer human health hazard TSD for DEHP, EPA evaluated the weight of scientific evidence for these effects consistent with modified Bradford-Hill criteria, including for dose-response, temporality, strength, consistency, specificity, biological plausibility, and coherence. For these effects, EPA concluded that there was too much scientific uncertainty and inconsistency in observed effects, and therefore they are not scientifically supportable for deriving a point of departure for use in quantitative risk characterization.

EPA disagrees with the public commenter (SACC25-0132) that a database deficiency factor of 10 is needed for DEHP. Under TSCA, there is no universal list of hazard data that is required to consider a database sufficient to conduct risk evaluation, nor is there a minimum set of data required to conduct a risk evaluation. The decision to incorporate the database UF in TSCA risk evaluations is determined on a case-by-case basis, and for DEHP, EPA determined that the database of studies did not identify deficiencies.

EPA disagrees with the public commenter (SACC25-0132) that all oral studies of DEHP considered for dose-response assessment require BMD analysis. As discussed in Section 4 of the non-cancer human health hazard TSD for DEHP, EPA identified 17 studies reporting effects on the developing male reproductive system consistent with disrupted androgen action and phthalate syndrome that support NOAEL and LOAEL values in a narrow dose-range of 1 to 5 mg/kg-day and 10 to 15 mg/kg-day, respectively. For the DEHP, the selected oral POD is based on a NOAEL of 4.8 mg/kg-day for increased incidence of male reproductive tract malformations (TherImmune Research Corporation, 2004; Blystone et al. 2010).

Although BMD analysis of these 17 individual studies is possible, BMD analysis of individual effects within the phthalate syndrome from individual studies would not provide a more robust dose-response assessment than the current integrations of data from 17 studies, which supports the selected POD of 4.8 mg/kg-day based on a NOAEL. Further supporting EPA's selected POD, Blystone et al. conducted BMD modeling on the reproductive tract malformation (RTM) data, which supports BMD_5/$BMDL_5$ estimates of 11.6/7.0 mg/kg-day for the F1 generation, 10.4/2.2 mg/kg-day for the F2 generation, and 8.5/5.6 mg/kg-day for combined F1 and F2 generations. Notably, $BMDL_5$ estimates ranged from 2.2 to 7.0 mg/kg-day and are consistent with the NOAEL of 4.8 mg/kg-day and support its selection as the POD. Discussion of the Blystone et al. BMD analysis of RTM data has been added to Section 4 of the non-cancer human health hazard TSD for DEHP.

Summary: One commenter (BBP-0123) made several comments regarding EPA's draft risk evaluation and occupational exposure limits of BBP. Citing two studies (Li et al., 2023; Jiang et al., 2023), the commenter suggested cellular stress related to reactive oxygen species (ROS) as a potential molecular initiating event (MIE) for BBP. They consequently opposed a transcriptomic point of departure (tPOD) due to the unconfirmed MIE and supported EPA's use of phthalate syndrome effects. They supported fetal testicular testosterone (FTT) as the endpoint for BBP, with the $BMDL_{1SD}$ of 61.44 mg/kg-day as the appropriate POD.

EPA Response: As discussed in the non-cancer human health hazard TSD for BBP (as well as for DEHP, DBP, DIBP, and DCHP) and as shown in the figure depicting the phthalate syndrome MOA in each TSD (*e.g.*, Figure 3-1 in the BBP non-cancer human health hazard TSD), EPA acknowledges that the MIE associated with phthalate syndrome is unknown and that the MIE is not believed to be androgen receptor or PPARα mediated. However, EPA does not agree that there is sufficient evidence to establish ROS processes as an MIE associated with the phthalate syndrome MOA. Further, EPA agrees that apical outcomes of phthalate syndrome (*e.g.*, decreased AGD, reduced fetal testicular testosterone, reproductive tract malformations, etc.) are more appropriate for non-cancer assessment and POD derivation rather than a transcriptomic POD. As discussed throughout the dose-response assessment of the human health hazard TSD for BBP (Section 4), EPA considered *ex vivo* fetal testicular testosterone data for BBP dose-response assessment through BMD modeling. In performing the meta-analysis of *ex vivo* fetal testicular testosterone production, EPA conducted dose-response assessment with updated Metafor Version 4.6.0 and interpreted results according to EPA's Benchmark

Dose Technical Guidance. Using best practices in benchmark dose modeling guidance, the meta-analysis did not yield acceptable fits for BBP in the most statistically appropriate model (Linear Quadratic model, based on lowest AIC). Thus, EPA conducted further BMD analysis of individual *ex vivo* fetal testicular testosterone production study datasets. However, the most appropriate model output for this data (individual dataset by Howdeshell et al. (2008)) resulted in a $BMDL_5$ of 81 mg/kg-day, which was less sensitive than other identified effect levels and was discussed in relation to more sensitive PODs, including the NOAEL of 50 mg/kg-day based on decreased AGD reported in Tyl et al. (2004). In its revisions of the Human Health Hazard TSD for BBP, EPA discussed additional dose-response assessment considerations and added text on the robust and sensitive nature of decreased AGD effects within the phthalate-syndrome MOA, noting decreased AGD is mechanistically linked to reduced fetal testicular testosterone production.

Summary: Several public comments (BBP-0120, BBP-0123, SACC25-0135, SACC25-0153, SACC25-0138, SACC25-0139, DEHP-0135, DIBP-0123 and DCHP-0126) proposed EPA utilize a consistent hierarchy to select PODs across phthalates, prioritizing: (1) Bayesian Benchmark Dose (BBMD)/ToxicR Modeling Average $BMDL_{1SD;}$ then (2) NOAEL, then (3) LOAEL. The comments provided examples of the application of the proposed hierarchy for decision making in DCHP (DCHP-0126, Appendix A of DIBP-0123), DEHP (SACC25-0138, DEHP-0135) with alternative proposed PODs. The comments also apply the proposed PODs to Occupational Exposure Value (OEV) calculations for BBP, DBP, DCHP, DEHP, and DIBP.

A public comment (DCHP-0122) expressed concern regarding inconsistencies across the individual chemical assessments and the CRA in terms of "the selection of the relevant doses. For example, a $BMDL_5$ was selected as the point of departure for the index chemical, DBP and the same approach was done for DEHP. However, in its Draft Risk Evaluation for DCHP, EPA used the NOAEL as the POD, despite presenting $BMDL_5$ and $BMDL_{10}$ values for DCHP in Table 2-4 of the Draft Phthalate CRA. It is unclear what the rationale was for the selection of relevant doses."

A similar comment (DCHP-0126) stated that "EPA's approach in assessing occupational exposure values (OEV), however, was not consistent between individual phthalates. EPA did not provide a justification for using the NOAEL in deriving the OEV instead of the $BMDL_5$ for DCHP. EPA did not include a discussion of the modeling issues that arose or why the $BMDL_5$ was not selected. EPA also did not discuss the impacts of data limitations on the results from the alternate NOAEL approach used by it. EPA should include a discussion in the assessment of the modeling issues that arose and the selection of the NOAEL instead of $BMDL_5$, along with the impacts of any related data limitations on the results from the alternate NOAEL/LOAEL approach."

A similar public comment (SACC25-0138) was submitted specific to the DEHP POD selection, stating that "EPA advanced its DEHP risk evaluation, but has unsuccessfully derived its POD for DEHP using the best available science." Using the proposed framework, the commenter provided BMD_{1SD} analysis of a subset of 4 studies, as well as two individual studies (Saillenfait et al., 2013 and Lin et al., 2008), all of which were included in EPA's BMD analysis.

DCHP-0126 stated that "EPA's Benchmark Dose Technical Guidance (BMD Guidance) recommends a benchmark response (BMR) of 5% extra risk (BMR5) for quantal endpoints in developmental tox studies that are adequately powered to detect such levels of change. EPA's BMD Guidance, however, does not recommend a BMR of 5% relative deviation for continuous endpoints. A better metric for a continuous endpoint, such as fetal testosterone, is a change in the mean equal to one control standard

deviation (SD) from the control mean (BMR_{1SD}). As discussed in EPA's BMD Guidance, developmental toxicity studies with nested study designs provide increased statistical power compared to regular toxicity studies due to the increase in sample size (*i.e.*, use of pups as the observational subject), and as such, a BMR_5 is supported when modeling only quantal data obtained from these studies….As stated above and explained in more detail below, the better metric for the continuous endpoint is BMR_{1SD}.

DCHP-0126 also suggested that for BBP, "rather than using the NOAEL of 50 mg/kg-bw/day for AGD, as proposed by EPA, EPA should use fetal testosterone, which had a NOAEL of 33 mg/kg-bw/day for deriving an OEV based on testosterone production in the study by Furr et al., 2014 and Gray et al., 2021. This approach should be applied to the remaining phthalates to align with guidance and best available science on a consistent endpoint."

A public comment (DBP-0129) stated that they agreed with EPA's POD selection but disagreed with EPA's benchmark dose (BMD) modeling approach, and suggested alternatives including the inclusion of additional data sources and the use of alternative BMD calculation methods.

DIBP-0130 and DIBP-0133 stated that EPA overlooked and understated DIBP and BBP's hazards. The commenter raised concern that EPA's current methodology in calculating MOEs is insufficient. The commenter recommended to use probabilistic dose-response assessments, which have been employed by the World Health Organization's International Programme on Chemical Safety. In doing so, EPA should have relied on the most sensitive study to derive the non-cancer POD for BBP using Sumner et al., 2009. The commenter also noted that EPA should have extended the literature review beyond September 2019. EPA failed to rely on the most sensitive study to derive the POD for BBP.

EPA Response: The EPA disagrees with commenters (BBP-0120, BBP-0123, SACC25-0135, SACC25-0153, SACC25-0138, SACC25-0139, DEHP-0135, DIBP-0123 and DCHP-0126) and DBP-0129) suggesting the adoption of a different approach, including new hierarchy for selecting Points of Departure (PODs). EPA prefers a Benchmark Dose (BMD) approach, where appropriate and supported by data, over a NOAEL/LOAEL approach due to potential uncertainties in the latter, as discussed in EPA's *Benchmark Dose Technical Guidance* (U.S. EPA, 2012; see discussion in Section 1.2 on pp. 6). Although the Bayesian Hierarchical Modeling approach may represent an alternative method to estimate BMD values over frequentist models, this model is not yet available as open-source software for EPA's use in dose-response analysis of non-quantal endpoints.

EPA agreed with commentors (DCHP-0122; DCHP-0126) on the importance of clarifying the rationale for the selection of different relevant doses for the individual phthalate assessments and in the phthalate cumulative risk assessment (CRA). EPA used NOAELs to derive the POD for DEHP and for DCHP in their respective individual assessments because the selected NOAELs were supported by the best available science and the weight of scientific evidence. In some instances, including for DCHP, candidate PODs based on BMD modeling were limited by a lack of testosterone data in the low-end range of the dose-response curve near the 5% response level. This limitation is described in the dose-response analysis section of the non-cancer human health hazard TSD for DCHP.

EPA provides sufficient rationale in the CRA and the individual phthalate assessments selecting relevant doses and believes there is already sufficient justification for differences in values between the individual chemical assessments and the CRA. For example, EPA describes its approach and rationale for the selection of relevant doses for DEHP and DCHP, as well as the other phthalates, in

Appendix B.4 of the CRA, and Appendix E of each individual phthalate assessment ("*Considerations for BMR Selection for Reduced Fetal Testicular Testosterone*"). EPA describes the BMD modeling considerations arose in the Draft Meta-analysis and Benchmark Dose Modeling TSD.

- Because the phthalates (DINP, BBP, DBP, DCHP, DEHP, and DIBP) are toxicologically similar, EPA considered it to be more appropriate to select a single benchmark response (BMR) for decreased fetal testicular testosterone production to provide a consistent basis for dose-response analysis and for deriving PODs relevant to the CRA. Across the key outcomes assessed within the phthalate syndrome mode of action (MOA), decreased fetal testicular testosterone production was determined to be the most robust and sensitive indicator of phthalate syndrome, and EPA considered reasonably available dose-response datasets for different outcomes for each phthalate. The commenter notes exceptions to this in the examples provided for the individual phthalate assessments. In these cases, EPA provides rationale for the selection of one approach over the other. See *Section 5.4.3 Dicyclohexyl phthalate (DCHP)* of the CRA for details.

EPA disagrees with commentors (DCHP-0126; SACC25-0138) that state or suggest the BMR of 1 SD is more appropriate than the selected BMR of 5 percent. EPA disagrees that it's BMD Guidance does not recommend a BMR of 5 percent relative deviation for continuous endpoints in any possible case. As discussed in EPA's *Benchmark Dose Technical Guidance* (U.S. EPA, 2012), a BMR of 5 percent is supported for BMD modeling of most endpoints in developmental and reproductive studies because there is a biological basis for this BMR. EPA provides justification for the selected BMR of 5 percent throughout the final non-cancer human health hazard TSDs, the meta-analysis and BMR Modeling TSD, and the CRA. EPA's *Benchmark Dose Technical Guidance* further states, a BMD_{1SD} might be used, "in the absence of any other idea of what level of response to consider adverse," which is not the case for the critical effect of developmental and reproductive toxicity identified for DEHP.

EPA disagrees with the comment from (DCHP-0126) and (DIBP-0130 and DIBP-0133) that it did not select the most appropriate study to derive a POD and OEV for BBP. EPA identified anti-androgenic effects as the most sensitive and robust non-cancer hazard endpoint identified for the phthalates, including BBP. EPA's weight of scientific evidence conclusions in the non-cancer human health hazard TSD for BBP support a POD of 50 mg/kg-day derived from a NOAEL in Tyl et al. (2004) based on decreased AGD (which was supported by a consensus LOAEL of 100 mg/kg-day across three co-critical studies). EPA notes that the LOAEL for BBP for decreased fetal testicular testosterone production was 100 mg/kg-day identified in the study by Furr et al. (2014) (Block 36 rats; no NOAEL identified). Of note, Furr et al. (2014) did not test for fetal testicular testosterone production at a level below 100 mg/kg-day.

In regard to commenters (DIBP-0130 and DIBP-0133), while EPA agrees that there are numerous ways to characterize risk (EPA notes that statutory authority requires a risk based approach), one of which is the MOE approach, EPA disagrees that it should not use the MOE approach to characterize risk for the phthalates. There will be risk scenarios where one approach may be better than another, and the science of risk characterization is still evolving. EPA used the MOE approach to characterize risk, which EPA believes is the most suitable approach for the risk evaluations of DEHP, BBP, DBP, DCHP, and DIBP given the reasonably available data, the utility of the MOE approach, and the precedent for using the MOE approach in previous TSCA risk evaluations. In general, EPA's approach was also supported by the SACC 2025 peer-review meeting.

Summary: Public commenters (DEHP-0138) stated that, "EPA failed to make appropriate use of benchmark dose modeling to specify a non-cancer point of departure for risk characterization." The commenters asserted that EPA's dose-response assessments for DBP and DEHP were not consistent with the best available science. They suggested that the dose-response assessments should be consistent with EPA's own guidance (namely, the 2012 Benchmark Dose Technical Guidance), and recommendations of NASEM (2017, Application of systematic review methods in an overall strategy for evaluating low-dose toxicity from endocrine active chemicals, p. 158), NAS (2009, Science and Decisions: Advancing Risk Assessment, p. 129.) and SACC [2024 Science Advisory Committee on Chemicals (SACC) Meeting Minutes and Final Report for the "Draft Risk Evaluation for Di-isodecyl Phthalate (DIDP) and Draft Hazard Assessments for Di-isononyl Phthalate (DINP)," p. 92].

A similar comment (SACC25-0145 and SACC25-0151) stated that EPA violated its own guidance and failed to consistently apply benchmark dose modeling to derive non-cancer points of departure for risk characterization for all 4 phthalates, further increasing uncertainty regarding whether the most sensitive endpoints were selected for use in estimating risks. This includes non-male reproductive harm endpoints that EPA considered in its hazard assessments.

EPA Response: EPA acknowledges that within a given study BMD modeling of effects is generally preferred when deriving a POD because a BMD is unaffected by dose-selection, and the model considers factors such as variability and sample size, as discussed in EPA's *Benchmark Dose Technical Guidance* (U.S. EPA, 2012) and noted above in prior comments. For DEHP, however, as described in Section 4.3 in the weight of scientific evidence in the noncancer human health hazard TSD for DEHP, EPA reaffirms that there is robust confidence in the approach and EPAs selected POD derived from a NOAEL in the draft human health hazard TSD for DEHP is appropriate, for the following reasons:

- There are 15 studies comprising 19 publications reporting LOAELs in a narrow dose range of 10 to 15 mg/kg-day based on treatment-related effects on the developing male reproductive system consistent with a disruption of androgen action during the critical window of development.
- The selected POD is based on effects consistent with phthalate syndrome in a high quality three-generation reproductive toxicity study in rats (TherImmune Research Corporation, 2004; Blystone et al., 2010). In fact, Blystone et al. (2010) conducted BMD modeling on reproductive tract malformations in male offspring, and the resulting BMDL5 values (2.2 to 7.0 mg/kg-day) in the F1 and F2 males supports the selection of the NOAEL of 4.8 mg/kg-day as the POD. Furthermore, the co-critical studies by Andrade and Grande et al. (2006a, 2006b, 2006c, 200d), which exposed rats throughout gestation and lactation, established a LOAEL of 15 mg/kg-day and a NOAEL of 5 mg/kg-day, which are similar to the NOAEL and LOAEL in the three-generation reproduction study.
- In addition to the principal and co-critical studies, 13 other studies indicated similar effects on the developing reproductive system in a narrow dose range supporting LOAELs of 10 to 14 mg/kg-day. Eleven of the 13 studies did not test low enough doses to establish a NOAEL. The two remaining studies support NOAELs of 1 and 3 mg/kg-day (Akingbemi et al., 2001; Christiansen et al., 2010). EPA believes that it would be inappropriate to select a lower NOAEL than that which is supported by numerous other studies showing effects on the same suite of endpoints, as this is merely a reflection of dose-selection.

In summary, there are a considerable number of studies supporting an effect of DEHP on the developing male reproductive system in the narrowly refined threshold in which no effects are

occurring at doses as high as 5 mg/kg-day and numerous effects are occurring at doses as low as 10 mg/kg-day across the key events in the AOP in 15 studies. EPA has robust confidence that using the entire body of evidence represents the best available science, compared to BMD modeling of any individual endpoint within an individual study, or selecting the lowest NOAEL in an individual study.

EPA further responds the public comment (DEHP-0138) regarding BMD modeling of DBP in the response directly below.

Summary: A public commenter (SACC25-0145) provided comments for DBP, stating that, although EPA ultimately selected a POD of 9 mg/kg-day based on BMD modeling, it failed to conduct BMD modeling on 11 studies on male reproductive toxicity, instead dismissing those studies because their NOAELs and/or LOAELs fell within range of the selected POD, without running BMD modeling to compare the results to the selected POD. The commenter specifically noted that two studies (Moody et al. 2013 and Lee et al. 2004) rated as medium quality demonstrate effects on sperm at doses of 1 mg/kg-d and 3 mg/kg-d, and recommended EPA conduct BMD modeling on these two studies and other candidate studies and endpoints (not restricted to male reproductive effects), and then use the lowest overall BMDL, or a set of BMDLs (representing different studies, endpoints and organ systems) as the POD for risk characterization of DBP. The commenter asserted that this approach was taken in several previous TSCA risk evaluations, such as those for trichloroethylene,1,4-dioxane, n-methylpyrrolidone, and 1,3-butadiene.

Another public commenter (DEHP-0138) similarly stated that "EPA failed to make appropriate use of benchmark dose modeling to specify a non-cancer point of departure for risk characterization." The commenters asserted that EPA's dose-response assessments for DBP and DEHP were not consistent with the best available science.

Additionally, this public commenter (SACC25-0145) stated that for DBP, EPA dismissed all health-effects studies not related to male reproductive harm from dose-response consideration without adequate scientific justification.

EPA Response: In response to comments, EPA has revised Section 4 (dose-response assessment) of the DBP non-cancer human health hazard assessment to clarify how the NOAEL/LOAEL and BMD approaches were utilized for DBP. For the DBP dose-response assessment, EPA first identified NOAEL and LOAEL values from 11 developmental toxicity studies that were considered for dose-response assessment. Seven of these 11 developmental toxicity studies evaluated effects on the developing male reproductive system in mice and rats exposed to low doses of DBP (*i.e.,* at doses less than 100 mg/kg-day), and data from these 7 studies were further considered for benchmark dose (BMD) analysis (Moody et al., 2013; Lee et al., 2004; Boekelheide et al., 2009; Mylchreest et al., 2000; Mahood et al., 2007; Lehmann et al., 2004; Furr et al., 2014). One of the initial 11 studies did not test doses of DBP below 100 mg/kg-day, but was included as part of EPA's meta-analysis and BMD analysis of fetal testosterone (Martino-Andrade et al., 2008). The remaining 3 of the initial 11 studies were not subjected to BMD analysis because they either evaluated only one dose level (Clewell et al., 2009) or because of data reporting limitation and/or because they were not very sensitive (*i.e.,* evaluated doses of 100 mg/kg-day or higher or did not identify sensitive phthalate-syndrome related effects for modeling) (Wine et al., 1997; Barlow et al., 2004).

In response to comments from the public and SACC, EPA considered conducting BMD modeling of data from the two studies by Moody et al., 2013 and Lee et al., 2004. For Moody et al. (2013), EPA

considered BMD modeling of reduced AGD and increase incidence of partial spermatogenesis (the two most sensitive effects observed in the study). However, AGD was not modeled due to data reporting deficiencies (*i.e.*, reported graphically only and sample size not provided) and due to lack of clear dose-response relationship, while incidence of partial spermatogenesis was high across dose groups (ranging from 50–100%) compared to controls (incidence of 0%). EPA did not attempt to BMD model incidence of partial spermatogenesis data because this type of response is not amenable to BMD modeling because of the lack of data in the low-end range of the curve near the BMR of 5 to 10% (see pp. 20 of EPA's *Benchmark Dose Technical Guidance* (U.S. EPA, 2012)). For Lee et al. (2004), EPA considered BMD modeling of reduced spermatocyte development incidence data, however, the prevalence of this lesion across dose groups ranged from 50 to 100 percent (incidence: 0/8, 4/8, 4/8, 8/8, 8/8 across control and dose groups). EPA did not attempt to BMD model these data because this type of response is not amenable to BMD modeling due to the lack of data in the low-end range of the curve near the BMR of 5 to 10 percent.

In order to facilitate comparison of individual datasets vs. meta-analysis of combined fetal testicular testosterone data, EPA conducted additional BMD modeling of individual fetal testicular testosterone data sets from the 8 studies included in the meta-analysis of fetal testicular testosterone using EPA's BMD Software. As discussed in Section 4.2.2 of the DBP non-cancer human health hazard TSD, BMD analysis of fetal testicular testosterone data from individual studies provides several BMD_5 and $BMDL_5$ estimates (*e.g.*, BMD_5 and $BMDL_5$ values of 22 and 14 mg/kg-day from Kuhl et al. and 24 and 16 mg/kg-day from Martino-Andrade et al.) similar to the BMD_5 and $BMDL_5$ estimates from the updated meta-analysis (*i.e.*, BMD_5/$BMDL_5$ values of 11 and 9 mg/kg-day). EPA also conducted additional BMD modeling of male pup nipple retention data from Mylchreest et al. (2000), which supported BMD/$BMDL_5$ estimates of 33/15 mg/kg-day. Notably, the $BMDL_5$ estimate of 15 mg/kg-day is similar to the selected $BMDL_5$ of 9 mg/kg-day from the meta-analysis of reduced fetal testicular testosterone.

EPA disagrees with the public commenter that health effects other than male reproductive outcomes were inappropriately dismissed. As discussed in Section 1.2 of the DBP non-cancer human health hazard assessment, EPA initially focused its assessment on male reproductive effects because existing assessments of phthalates by other authoritative and regulatory agencies have identified these effects as the most sensitive and scientifically supportable effects for use in risk characterization. EPA also considered newer studies identified with literature searches and updates from 2014 to 2025. These studies provided information pertaining to various hazard outcomes, including: reproduction/ development, neurological, metabolic/nutritional, cardiovascular, and the immune system. Information from studies pertaining to these health outcomes is discussed in the DBP non-cancer human health hazard assessment. Existing assessments have consistently shown that effects on other health outcomes (*i.e.*, female reproduction, neurological, cardiovascular, metabolic) are generally observed at higher dose levels than developmental effects on male reproduction or are not supported by as robust databases of studies. This is further supported by the more recent literature published from 2014 to 2025, as some of the lowest LOAELs were identified for reproductive and developmental effects. Therefore, the Agency focused its non-cancer human health hazard assessment on toxicity to the male reproductive system following developmental exposures. The most sensitive POD representing this hazard outcome thus also represents the most sensitive overall POD across all endpoints. Risk estimates using this value are protective of other hazard endpoints as well.

Summary: A public commenter (SACC25-0145 and SACC25-0151) stated that EPA did not appropriately consider hazard studies for DEHP which provide more sensitive endpoints for selection of the POD for non-cancer human health hazard. Regarding the study by Christensen et al (2010), the commenter asserted that "EPA's rationale for dismissing the NOAEL and claiming that this study aligned with the "consensus" NOAEL of 5 mg/kg-day is scientifically unsupported…EPA cannot simultaneously accept the validity of a NOAEL and dismiss it based solely on its relative position within a dose range". Regarding the study by Hsu et al (2016), the commenter stated that it is inappropriate for EPA to discount the effects on sperm morphology due to the uncertainty regarding plausibility and replicability because these findings were not noted in the 3-generation reproductive toxicity study at much higher doses (Blystone et al., 2010; TherImmune Research Corporation, 2004). Similarly, the commenter stated that EPA dismissed the evidence from studies related to other adverse health outcomes beyond male reproductive toxicity which would provide a more sensitive POD, including effects on female developmental/reproductive endpoints, and effects on nutritional/metabolic endpoints, cardiovascular/kidney toxicity, neurotoxicity, and immunotoxicity; essentially disagreeing with EPA's conclusions about the limitations and uncertainties such as lack of replicability, testing only a single dose, lack of an established AOP, and results that are inconsistent in magnitude, directionality, temporality, and dose-response. Finally, the commenter disagreed with EPA's consensus NOAEL/LOAEL approach in which EPA considered the highest NOAEL at approximately 5 mg/kg-day for the POD (from the principal and co-critical studies) below the lowest LOAEL of 10 mg/kg-day indicated in 11 studies. Instead, the commenter recommended BMD modeling for all studies and endpoints listed in Table 4-3 on the developing male reproductive tract, in addition to other candidate endpoints beyond the male reproductive tract to derive the lowest BMDL or a different BMDL for each hazard.

EPA Response: EPA disagrees with the characterization that it dismissed the evidence from studies related to other adverse health outcomes beyond male reproductive toxicity which would provide a more sensitive POD. In fact, EPA carefully considered studies indicating the very same hazards mentioned by the commenter. As detailed in Section 1.2.3 of the DEHP Human Health Hazard Assessment, EPA provides a detailed explanation that it further considered the 201 studies included in ATSDR's Table 2-2 of LSEs (ATSDR, 2022) to identify studies with sensitive endpoints (LOAEL <20 mg/kg-day) for new information on human health hazards not previously identified in existing assessments— including information that may indicate a more sensitive POD than established by the regulatory bodies prior to the publication of ATSDR in 2022. As described further in the *Systematic Review Protocol for Diethylhexyl Phthalate (DEHP)*, EPA identified 50 animal toxicology studies that provided information pertaining to hazard outcomes associated with exposure to less than or equal to 20 mg/kg-day, including: 25 studies on male and female reproduction/development; 16 studies on metabolic endpoints related to glucose/insulin homeostasis and lipid metabolism; 4 studies on cardiovascular/kidney outcomes, 19 studies indicating effects on the liver, 3 studies reporting neurological effects, three studies indicating effects on the immune system, and one study describing effects on musculoskeletal systems, in addition to 5 studies reporting hazards identified by the inhalation route. Importantly, the evidence supporting each of these hazards was thoroughly evaluated as described in Section 3 of the DEHP Non-cancer Human Health Hazard Assessment, including a detailed evaluation of the individual studies and a discussion of the evidence across studies in evidence synthesis and integration. This weight of evidence analysis was organized around modified Bradford-Hill criteria, including examination of concordance regarding dose-response, temporality, strength, consistency, specificity, and biological plausibility. Notably, EPA considered the evidence supporting an effect of DEHP on the developing male reproductive tract to be the most robust and provide the most sensitive and appropriate POD for use in risk assessment and therefore carried this evidence

forward to dose-response analysis. EPA does not equate this approach to a dismissal of the evidence supporting other hazards.

EPA notes that the "consensus NOAEL" approach is not a novel concept, although it may not always be referred to using that terminology. Risk assessment guidance describes the synthesis and integration of evidence across studies that underlie this approach. Simply put, the approach refines the effect threshold by selecting the highest NOAEL below the lowest LOAEL among a suite of studies with similar endpoints. EPA would also like to point out that this concept is not limited to human health hazard assessment. In determining ecological hazard, some EPA employs a similar approach in deriving wildlife toxicity reference values in its *Guidance for Developing Ecological Soil Screening Levels (Eco-SSLs)*, which selects the lowest LOAEL above either the highest NOAEL, or the geometric mean of the NOAELs, depending on the weight of evidence across the suite of studies with ecologically relevant endpoints (U.S. EPA, 2007).

EPA acknowledges that, within a given study, BMD modeling of effects is generally preferred when deriving a POD because a BMD is unaffected by dose-selection, and the model considers factors such as variability and sample size. However, in consideration of the extensive evidence supporting effects on DEHP on the male reproductive tract, EPA reaffirms that there is robust confidence in the approach and the resulting POD that EPA selected in the draft human health hazard TSD, for the following reasons, detailed further in Section 4.3 on the weight of scientific evidence:

- There are 15 studies comprising 19 publications reporting LOAELs in a narrow dose range of 10 to 15 mg/kg-day based on treatment-related effects on the developing male reproductive system consistent with a disruption of androgen action during the critical window of development. The AOP for effects on the developing male reproductive system from exposure to DEHP and other phthalates is well established.
- The selected POD is based on effects consistent with phthalate syndrome in a high quality three-generation reproductive toxicity study in rats (TherImmune Research Corporation, 2004; Blystone et al. 2010). In fact, Blystone et al. (2010) conducted BMD modeling on reproductive tract malformations in male offspring, and the resulting BMDL5 values (2.2 to 7.0 mg/kg-day) in the F1 and F2 males supports the selection of the NOAEL of 4.8 mg/kg-day as the POD. Furthermore, the medium-quality studies by Andrade and Grande et al. (2006a, 2006b, 2006c, 200d), which exposed rats throughout gestation and lactation, established a LOAEL of 15 mg/kg-day and a NOAEL of 5 mg/kg-day, which are similar to the NOAEL and LOAEL in the three-generation reproduction study (TherImmune Research Corporation, 2004; Blystone et al. 2010). Therefore, consideration of these studies as co-critical studies provides additional strength and confidence in the selected POD, in both the outcomes and the dose at which they occur.
- In addition to the principal and co-critical studies, 13 other studies indicated similar effects on the developing reproductive system in a narrow dose range supporting LOAELs of 10 to 14 mg/kg-day. Eleven of the 13 studies did not test low enough doses to establish a NOAEL. The two remaining studies support NOAELs of 1 and 3 mg/kg-day (Akingbemi et al., 2001; Christiansen et al., 2010). The commenter specifically mentioned the study by Christiansen (2010) and suggested that it be used to establish a NOAEL. EPA affirms that it would be inappropriate to select a lower NOAEL than supported by numerous other studies showing effects on the same suite of endpoints, as this is merely a reflection of dose-selection, and EPA has higher confidence in the POD (NOAEL of 4.8 mg/kg-day) as a robust consensus NOAEL based on a high quality three-generation reproduction study (TherImmune Research

Corporation, 2004; Blystone et al. 2010) co-critical with the studies by Andrade and Grande et al. (2006).

In summary, there are a considerable number of studies supporting an effect of DEHP on the developing male reproductive system in the narrowly refined threshold in which no effects are occurring at doses as high as 5 mg/kg-day and numerous effects are occurring at doses as low as 10 mg/kg-day across the key events in the AOP in 15 studies. EPA has robust confidence that using the entire body of evidence represents the best available science, compared to BMD modeling of any individual endpoint within an individual study, or selecting the lowest NOAEL in an individual study.

Summary: A public commenter (SACC25-0148) agreed with EPA that the weaknesses and uncertainties significantly outweigh the strengths when it comes to being able to reach a definitive conclusion as to whether DEHP exposure is causal in the production of adverse effects on the female reproductive system with sufficient robustness that an identified hazard can also be quantitatively assessed and integrated into a robust risk assessment.

They suggested that EPA include a table for effects on female reproductive tract similar to Table 3-4 on male reproductive tract, to accompany an epidemiology table because the variation in study designs and parameters evaluated makes it difficult for the reader to integrate the findings into a final judgement of adequacy and certainty.

They also noted that there is a large database of studies on the phthalates, but unfortunately very few of these studies, if any, appear to be conducted for regulatory purposes, and it does not appear that the researchers are examining the evidence from other publications when conducting and reporting their results. Further, they suggested that EPA consider contacting the authors of any publications with incompletely reported results to fill data gaps and resolve uncertainties that prevented their use in dose-response and risk characterization.

EPA Response: EPA notes that the commenter agrees with EPA's assessment and conclusions regarding the studies indicating effects on the developing female reproductive tract – specifically that the weaknesses and uncertainties significantly outweigh the strengths, thereby precluding a definitive conclusion as to whether DEHP exposure causes these effects and the doses at which they occur to confidently provide a POD for quantitative use in risk assessment.

While the commenter recommended that EPA include a table summarizing the study designs and the effects on the female reproductive tract, EPA considers an inclusion of a such a table to be inconsistent with its review of each hazard. EPA identified 50 animal toxicology studies that provided information pertaining to hazard outcomes associated with exposure to less than or equal to 20 mg/kg/day, including: 25 studies on male and female reproduction/development; 16 studies on metabolic endpoints related to glucose/insulin homeostasis and lipid metabolism; 4 studies on cardiovascular/kidney outcomes, 19 studies indicating effects on the liver, 3 studies reporting neurological effects, three studies indicating effects on the immune system, and one study describing effects on musculoskeletal systems, in addition to 5 studies reporting hazards identified by the inhalation route. The evidence supporting each of these hazards was thoroughly evaluated and described in Section 3 of the Non-cancer human health hazard TSD (*i.e.*, hazard identification), including a detailed evaluation of the individual studies and a discussion of the evidence across studies in evidence synthesis and integration. This weight of evidence analysis was organized around modified Bradford-Hill criteria, including examination of concordance regarding dose-response, temporality, strength, consistency, specificity, and biological plausibility and a conclusion as to the

strength of the hazard data. However, EPA elected to depict in tables only data from those studies supporting hazards considered appropriate for dose-response, and these tables are included in Section 4 of the non-cancer hazard assessment.

The commenter also suggested that EPA consider contacting the authors of any publications with incompletely reported results to fill data gaps and resolve uncertainties that prevented their use in dose-response and risk characterization. Although EPA has a mechanism for author-outreach as part of its TSCA systematic review process, in this instance the inability to use certain studies was not because data were not available, but rather because the studies did not generate or present the type of information that EPA considers as part of the risk evaluation (*i.e.*, data that are suitable for quantitative characterization of risks, namely data considered appropriate for dose-response analysis). Therefore, reaching out to the authors to address identified reporting deficiencies would not lend them usable for dose-response analysis. EPA has updated the DEHP non-cancer hazard TSD to clarify why these studies were not included.

Other Non-cancer Hazards

Summary: A public comment (DCHP-0127) stated that "EPA should re-evaluate its conclusion that DCHP liver effects are not adverse in light of new data." The comment continues "Because it did not update its literature search from 2019, EPA did not consider a 2023 study by Aydemir et al. that evaluated both histopathological changes and serum biochemistry to find that DCHP causes liver damage in male and female rats. EPA should include the data in the Aydemir study in its analysis and re-evaluate its conclusion on DCHP and liver toxicity."

EPA Response: EPA thanks the public commenter for providing the Aydemir et al. (2023) reference. EPA has reviewed the reference and integrated discussion of the study into Section 3.3 of the non-cancer human health hazard TSD for DCHP. EPA considers this study to provide additional evidence that oral exposure to DCHP can have liver effects. However, there are uncertainties and inconsistencies in the study. For example, incidence of most histopathological lesions in the liver displayed a flat dose-response, and no corresponding treatment related effects on liver weight or serum chemistry markers of liver toxicity were observed. Overall, EPA considers the liver effects reported by Aydemir et al. to be of questionable toxicological significance given the uncertainties and inconsistencies.

Summary: Several public comments (DCHP-0121, DCHP-0122) provided references (Hyman et al. 2025; Sui et al. 2021; Liu et al. 2023; Tian et al. 2024) regarding the impact of phthalates on cardiovascular disease and metabolic health and encouraged EPA to address these additional human health endpoints in the individual phthalate and cumulative risk evaluations.

EPA Response: EPA conducted an updated literature search for human health hazards related to DCHP exposure. Results of this search included the citations provided by this comment. EPA evaluated the three publications provided by this comment and determined that they are not suitable for dose-response assessment because either they were conducted *in vitro*, did not provide data on apical human health hazard endpoints, tested a single dose, or were not chemical-specific for DCHP. EPA also determined that these publications did not provide any novel mechanistic evidence that would significantly contribute to the mode of action for phthalate syndrome beyond what is already characterized in the human health hazard assessment.

For DEHP, in the draft and final documents, EPA included cardiovascular and metabolic hazards and a discussion of the strengths, limitations, and wight of scientific evidence for these hazards in Section 3.2 Nutritional/Metabolic Effects Related to Metabolic Syndrome and Glucose/Insulin Homeostasis and Lipid Metabolism and in Section 3.3 Cardiovascular and Kidney Toxicity. However, these hazards were ultimately not carried forward for dose-response analysis due to the limitations and uncertainties with these hazards compared to the robust confidence in the effects on the developing male reproductive system.

Summary: A public commenter (SACC25-0132) stated that EPA must account for neurodevelopmental hazards of DEHP when establishing an oral POD. They contend that consideration of observed motor effects in pubertal type 2 diabetes mellitus (P-T2DM) ICR mice at doses as low as 0.18 mg/kg/day when establishing a POD is warranted, and such a POD would be protective of the 1 in 10 people in the U.S. with diabetes. They further state that if "EPA has concerns about the lack of supporting data to use this POD, it should use a database deficiency uncertainty factor of ten."

EPA Response: EPA evaluated the epidemiologic and animal study evidence for neurotoxicity and neurodevelopmental toxicity of DEHP in Section 3.5 of the non-cancer human health hazard TSD for DEHP. As discussed in Section 3.5.3 of the non-cancer human health hazard TSD, EPA evaluated the weight of scientific evidence consistent with modified Bradford Hill criteria (*i.e.*, dose-response, temporality, strength, consistency, and specificity, biological plausibility, and coherence). EPA concluded that experimental animal studies on neurotoxicity have too much uncertainty regarding the limitations in the individual studies and the clinical relevance of the findings for human health to consider them further in dose-response for derivation of an oral POD. EPA also concluded the evidence of association of DEHP exposure with neurological outcomes in human epidemiologic studies were inconsistent among studies or inconclusive. Overall, EPA considers there to be too many scientific uncertainties and limitations in the available database of studies to consider neurodevelopmental outcomes for dose-response assessment or for use in quantitative risk assessment.

Summary: A public comment (SACC25-0148) stated that EPA appropriately concluded that there is too much uncertainty associated with the information available on the potential consequences of DEHP exposure on nutritional/metabolic effects related to glucose/insulin homeostasis to go beyond the qualitative judgment of a likely association and apply this information to dose response and risk characterization, including the derivation of PODs. The public commenter (SACC25-0148) further stated that her support of this conclusion was based on the relatively robust evidence base (16 studies) evaluating this hazard, with many of these studies testing the same range but finding contradictory results, with different responses and/or different magnitude of responses at similar doses. They also pointed out that summarized results for a study Venturelli (2015) incorrectly cited a different study Venturelli (2019) by the same author.

EPA Response: EPA acknowledges the public commenter's support for EPA's conclusions regarding DEHP exposure on nutritional and/or metabolic effects related to glucose/insulin homeostasis and lipid metabolism. EPA agrees with the public commenter that the EPA incorrectly cites Venturelli et al. (2019) instead of Venturelli et al. (2015) and in response to the comment, EPA has revised the DEHP non-cancer human health hazard TSD to cite Venturelli et al. (2015), instead of Venturelli et al. (2019).

Summary: A public comment (DCHP-0128) stated that EPA should evaluate and address evidence linking ortho-phthalates, including DCHP, to developmental neurotoxicity. The comment referenced several epidemiological studies evaluating associations between prenatal exposures to phthalates and outcomes such as attention-deficit hyperactivity disorder (ADHD), while acknowledging that some of the references evaluated "toxicologically similar" phthalates and not DCHP directly. The comment describes an Agency for Toxic Substances and Disease Registry (ATSDR) report identifying sensitive neurodevelopment endpoints for DEHP, as well as European Food Safety and Consumer Product Safety Commission (CPSC) reports that described neurodevelopmental toxicity associated with phthalate exposure. In the absence of sufficient data to conduct a dose-response analysis of neurodevelopmental effects for DCHP, the commenter recommends EPA apply a 10× database deficiency uncertainty factor to the assessment.

EPA Response: EPA's evaluation of neurotoxicity, including developmental neurotoxicity hazard, for DEHP is described in Section 3.5 of the DEHP human health hazard assessment. EPA identified three neurotoxicity studies on rodents reporting sensitive effects (LOAEL 20 mg/kg-day) and evaluated the strengths, weaknesses, and limitations of these studies according to modified Bradford-Hill criteria in a weight of evidence analysis. EPA considers the effects in the three low-dose neurotoxicity studies of mice (Barakat et al., 2018; Feng et al., 2020; Tanida et al., 2009) to be inconsistent with dose-response for neurotoxic endpoints in other studies. Additionally, EPA examined the epidemiological assessments by ATSDR (2022), Health Canada (2018), and Radke et al. (2020). Although ATSDR assessed 26 studies of 13 birth cohorts examining cognitive/mental and psychomotor development and 13 studies of 9 birth cohorts evaluating behavior and attention, a conclusion on the association between DEHP and neurological outcomes could not be reached due to the lack of substantive epidemiological data particularly on adults. The evaluation of the relationship between exposure to DEHP and cognition by Radke et al. (2020) based on 11 medium to high quality studies revealed no discernible trend of greater association in studies with wider ranges or higher exposure levels. Health Canada (2018) determined there is insufficient evidence to associate DEHP metabolites to changes in behavioral, cognitive functioning, or impaired mental and psychomotor neurodevelopment. Overall, EPA determined that the evidence of association of DEHP exposure with neurological outcomes were inconsistent among studies or inconclusive. Further details are included in Section 3.5 of the DEHP human health hazard assessment.

EPA disagrees with the recommendation to apply a database uncertainty factor (UF_D). As stated above, the UF_D is applied for deficiencies in the toxicological database that might lead to a lower POD. The decision to incorporate the database UF in TSCA risk evaluations is determined on a case-by-case basis, and for this case with DCHP, EPA determined that the database of studies did not have deficiencies that might suggest an underprotective POD. Although the number of low dose studies on developmental neurotoxicity are limited, several animal and epidemiology studies were reviewed. EPA does not consider this hazard to warrant inclusion of a UF_D, given the fact that the POD based on developing male reproductive effects is robust and sensitive.

5.2 Cancer Hazard Assessment

Mechanisms of Action

Summary: A public comment (SACC25-0138) "commend[s] EPA's conclusion that the use of developmental effects for the POD is protective of potential carcinogenicity but disagree[s] on the use of peroxisome proliferator-activated receptor alpha (PPARα) as the key molecular initiating event (MIE)." The commenter provides a references to support arguments that 1) phthalates had little to no effect on mRNA expression for fetal liver PPARα and 2) PPARα agonists have not been shown to alter fetal testicular testosterone levels.

Another public commenter (DBP-0129) similarly stated that the PPARα molecular initiating event (MIE) is not the appropriate choice to derive the cancer Point of Departure (POD) for DBP and made the same arguments as the DEHP Consortium above.

EPA Response: In the final risk evaluations for DEHP, DBP, BBP, DCHP, and DIBP and supporting non-cancer and cancer TSDs for each phthalate, EPA has retained the conclusion that the use of developmental effects for the POD is protective of potential carcinogenicity for each phthalate. As discussed in the cancer TSD, DEHP, DBP, BBP, DCHP, and DIBP are all known PPARα activators and PPARα activation in the liver is an early key event in liver tumorigenesis. EPA agrees with the public commenters (SACC25-0138; DBP-0129) that PPARα is not the MIE associated with a disruption of testosterone production and phthalate syndrome. As discussed in Section 3 of the non-cancer human health hazard assessments for DEHP, DBP, DIBP, DCHP, and BBP (and shown in Figure 3-1 in these TSDs), EPA considers the MIE associated with phthalate syndrome to be unknown. However, there is sufficient evidence to demonstrate that androgen receptor antagonism or PPARα activation are not the molecular initiating event.

Summary: A public comment (SACC25-0132) stated that tumorigenesis in DEHP may not be limited to the PPARα MOA, providing citations to support statements that DEHP activates the constitutive androstane receptor and affected cell migration in gastric cancer cells.

EPA Response: EPA discusses the mode of action for liver tumors in rats and mice in Section 4.3.1.1.1 of the *Cancer Human Health Hazard Assessment for DEHP, DBP, BBP, DIBP, and DCHP*, which in includes a discussion of other modes of carcinogenic action. As discussed in Section 4.3.1.1.1 of the cancer TSD, DEHP can activate constitutive androstane receptor (CAR), pregnane X receptor (PXR), and aryl hydrocarbon receptor (AhR) in addition to PPARα. All of these receptors are known to play a role in liver homeostasis and disease. However, gene expression studies in the liver demonstrate that the vast majority of genes modulated following oral exposure to DEHP are regulated by PPARα, and to a lesser extent CAR. For example, Ren et al. (2010) demonstrated that PPARα is required for approximately 94 percent of transcriptional changes, while the remaining 6% were regulated by CAR, providing evidence that PPARα was the predominant nuclear receptor mediating effects in the liver. Additionally, other carcinogenic modes of action such as mutagenicity and cytotoxicity and regenerative proliferation have been ruled out for DEHP.

Overall, EPA concluded that DEHP causes liver tumorigenesis in rats and mice through the PPARα MOA, and this conclusion was supported the SACC during the August 4 through August 9, 2025 peer-review meeting of the phthalates.

Summary: A public comment (SACC25-0148) stated "the cancer classification of Not Likely to be Carcinogenic to Humans should be applied to all seven phthalates because the observed tumors were all driven by the PPARα mechanism of action (MOA), a phenomenon shown to be irrelevant to humans."

Another commenter, DIBP-0133, supports EPA's conclusion of *Suggestive Evidence of Carcinogenic Potential* of BBP based on pancreatic acinar cell adenomas in rats. The commentor stated, "EPA rightfully determined that BBP's evidence would not classify it as *Not Likely to be Carcinogenic to Humans*." The public commenter (DIBP-0133) further stated that "EPA must conduct a dose-response analysis for the BBP cancer endpoint." To further support their stance, the commentor further cites the *Guidelines for Cancer Risk,* which the commentor states, "provide several rationales for conducting a dose-response analysis which EPA failed to consider especially since there are several well-conducted studies by the National Toxicology Program that EPA can utilize for dose-response. Given that EPA is asserting the non-cancer PODs are protective of tumor formation, it must quantitatively support that assertion by conducting a dose-response assessment to affirm its position and determine exposure levels that would be of concern for tumor formation."

EPA Response: EPA has revised its final cancer classifications for DEHP, DBP, and BBP to *Not likely to be carcinogenic to humans*. For DCHP and DIBP, EPA did not derive formal cancer classifications. Instead, EPA used the ReCAAP weight of evidence framework to determine that cancer bioassays are not needed for DIBP and DCHP, and that for these phthalates, lack of formal cancer bioassays is not a source of remaining scientific uncertainty. EPA has previously concluded that DIDP is *Not likely to be carcinogenic to humans* and that DINP is *Not likely to be carcinogenic to humans* at doses below levels that do not result in PPARα activation.

Under the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005), EPA reviewed the weight of evidence for the carcinogenicity of BBP and in the draft BBP cancer assessment (which is found within the draft cancer TSD) concluded that there is *Suggestive Evidence of Carcinogenic Potential* of BBP in rodents based on evidence of pancreatic acinar cell adenomas in male and female F344 rats. However, based on the majority opinion of the SACC, EPA has revised its cancer classification for BBP to *Not Likely to be Carcinogenic to Humans.* As described in Section 4.3.2.4 of the phthalate cancer TSD, there are a number of weight of scientific evidence considerations supporting this revised cancer classification. Given the revised cancer classification, and consistent with EPA cancer guidelines, a cancer dose-response assessment and quantitative cancer risk assessment for BBP are not warranted

Application of the Rethinking Chronic Toxicity and Carcinogenicity Assessment for Agrochemicals Project (ReCAAP) Framework

Summary: A public comment (SACC25-0130) supported the use of the ReCAAP framework to use read across from data-rich toxicologically similar chemicals to the assessment of DIBP and DCHP as a scientifically sound approach to protect human health.

Another comment (SACC25-0154) supported using the ReCAAP method for DIBP and DCHP risk assessments, stating that "its scientific rigor and utility have also been validated by recent OECD IATA case studies." The submitter states that "continued thoughtful application of the ReCAAP framework in TSCA risk evaluations will support informed decision-making."

Another comment (SACC25-0148) commends the EPA for its comprehensive application of the ReCAAP framework in capturing and comparing relevant data on seven phthalates, including DIBP, DCHP, and five additional phthalates used for read-across. They state the EPA successfully addressed data gaps and made informed judgments on the chronic toxicity and carcinogenic potential of DIBP and DCHP.

The commenter (SACC25-0148) also suggests that all seven phthalates should be classified as *Not Likely to be Carcinogenic to Humans* because the observed tumors are driven by the PPARα mode of action, a mechanism considered irrelevant to humans.

In contrast to the majority of comments, one public commentor (DIBP-0133) stated that EPA misapplied ReCAAP to not assess DIBP's potential carcinogenicity and misused the framework to draw the unsupported conclusions. The commentor states, "EPA concluded that its non-cancer POD, based on acute-duration exposure data, is 'appropriate for use in human health risk assessment and is protective of human health' or all durations. EPA must weigh evidence that sensitive effects do occur outside of the developmental exposure window, indicating a need for additional assessment of intermediate and chronic exposure durations… EPA misapplied the results of its cancer risk assessment for other phthalates (for which EPA claims that 'quantitative cancer risk assessment is not needed') to rule out the possibility that cancer is a driver of DIBP's risks. The ReCAAP weight-of-evidence framework is inappropriate to conclude that DIBP does not present cancer risk and EPA must quantify such risk instead of dismissing it out of hand."

EPA Response: EPA acknowledges support from public commenters (SACC25-0130, SACC25-0148; SACC25-0154) for its use of the ReCAAP framework.

Based on SACC recommendations, EPA has revised its final cancer classifications for DEHP, DBP, and BBP to *Not likely to be carcinogenic to humans*. For DCHP and DIBP, EPA did not derive formal cancer classifications. Instead, EPA used the ReCAAP weight of evidence framework to determine that cancer bioassays are not needed for DIBP and DCHP, and that for these phthalates, lack of formal cancer bioassays is not a source of remaining scientific uncertainty. EPA has previously concluded that DIDP is *Not likely to be carcinogenic to humans* and that DINP is *Not likely to be carcinogenic to humans* at doses below levels that do not result in PPARα activation.

EPA disagrees with the public commenter (DIBP-0133) that it misapplied the ReCAAP framework to draw unsupported conclusions. In the non-cancer human health hazard assessment of DIBP, EPA considered all reasonably available information, including developmental studies and repeat-dose studies. As stated in the DIBP hazard assessment, "EPA reviewed and supports the conclusions of the systematic review and hazard identification for DIBP published by Yost et al. (2019). EPA did not identify any literature that would change the conclusions of Yost et al. (2019) pertaining to *slight* evidence for female reproductive effects and liver effects and *indeterminant* evidence for kidney effects. Therefore, EPA did not further characterize these non-cancer hazards in this assessment or carry them forward to dose-response assessment." Instead EPA identified effects on the developing male reproductive system consistent with phthalate syndrome as the most sensitive effect associated with exposure to DIBP and derived and an acute POD based on these effects that is relevant for characterizing acute/intermediate/chronic risks.

The ReCAAP framework was specifically developed as a weight of evidence framework to help determine the need for conducting two-year cancer bioassays for chemicals lacking this data, as is the case for DIBP and DCHP. For DIBP (and DCHP), EPA determined that based on read-across to other

phthalates (DEHP, BBP, DBP, DINP, DIDP), the lack of two-year bioassays is not a remaining source of scientific uncertainty for and that the non-cancer PODs for DIBP and DCHP are health protective.

Summary: A public comment (DCHP-0127) recommended that "EPA should take an established scientific approach to addressing the data gap on DCHP carcinogenicity and not use an unvalidated and inappropriate framework to draw conclusions about cancer hazards." The comment further recommends that "EPA could require tests such as cancer bioassays, or alternatively use the approach recommended by NASEM [in their 2009 publication titled *Science and Decisions: Advancing Risk Assessment*] to make use of data from short term toxicity tests and structure-activity modeling to estimate carcinogenic potency."

Another public commenter (DCHP-0128) stated that the ReCAAP framework has not been used in EPA risk evaluations before, was developed in partnership with pesticide companies, and does not satisfy Congressional mandate to utilize alternatives to traditional animal testing when they "provid[e] information of equivalent or better scientific quality and relevance that will support regulatory decisions." The commenter concludes that application of the ReCAAP framework does not provide sufficient evidence to support EPA's conclusions that the non-cancer POD is protective of DCHP cancer and that cancer risks of DCHP do not need to be quantified.

EPA Response: EPA disagrees with both public commenters (DCHP-0127, DCHP-0128). The purpose of the ReCAAP framework is to determine the need for rodent cancer bioassays for chemicals, such as DIBP and DCHP, lacking the rodent cancer bioassays. As such, EPA applied the ReCAAP framework to determine the need for these studies for DIBP and DCHP. Notably, the ReCAAP framework takes into consideration numerous lines of evidence to inform decision-making, including structural similarity, physical and chemical properties, ADME properties, acute toxicity, evidence of hormone perturbation, developmental and reproductive toxicity, subchronic toxicity, immune systemic perturbation, genotoxicity, mode of action, and evidence of chronic toxicity and carcinogenicity from chemical analogs. Based upon data for DIBP, DCHP, and read-across to other structurally and toxicologically similar phthalates (DEHP, DBP, BBP, DIDP, and DINP), EPA concluded that two-year cancer bioassays are not needed for DIBP or DCHP, and that lack of these studies is not a significant source of scientific uncertainty for these phthalates.

Further, the ReCAAP framework was developed in partnership by U.S. and other international regulatory agencies (U.S. EPA, Health Canada, Australian Pesticides and Veterinary Medicines Authority), as well as other stakeholders (U.S. NTP, agrochemical companies, NGOs, Academics), and OECD has published several Integrated Approach to Testing and Assessment (IATA) case studies demonstrating applicability of the weight of evidence ReCAAP framework (OECD 2024). EPA issued several charge questions to its peer-review committee (SACC), requesting input on the conclusions EPA reached using the ReCAAP Framework. Notably, the committee agreed with EPA's conclusions and its application of the ReCAAP framework for DIBP and DCHP.

5.3 Cumulative Human Hazard Assessment

Proposed Approach to Cumulative Hazard Characterization

Summary: Two public comments (SACC23-0031, SACC23-0065) on the *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substance Control Act* recommended selecting DEHP as the index chemical, if a relative potency factor (RPF) approach is used in the cumulative assessment.

EPA Response: As discussed in Section 2.3 of the TSD for the cumulative risk analysis of DEHP, DBP, BBP, DIBP, DCHP, and DINP under TSCA, EPA considered DEHP and DBP as candidates for the index chemical because both phthalates have high quality toxicological databases demonstrating effects on the developing male reproductive system consistent with a disruption of androgen action and phthalate syndrome, demonstrate toxicity representative of all phthalates in the cumulative chemical group, and are well characterized for the MOA associated with phthalate syndrome. Compared to DEHP and DBP, other phthalates included in the cumulative chemical group (*i.e.*, BBP, DIBP, DCHP, DINP) have considerably smaller databases and fewer dose-response data (see Table 2-3 of the 2025 CRA TSD), and were not considered candidates for the index chemical. Because RPFs were derived using fetal testicular testosterone data, EPA compared the quantity and quality of available dose-response data for this outcome for DBP and DEHP. Overall, DBP has more dose-response data than DEHP in the low-end range of the dose-response curve where the BMD and BMDL estimates at the 5 and 10 percent response level are derived. Therefore, EPA has selected DBP as the index chemical.

Summary: Two public comments (SACC23-0031, SACC23-0065) recommended utilizing lifestage/exposure scenario-specific toxicity values in the cumulative risk assessment.

EPA Response: EPA thanks the commentors for their recommendation. The RPFs derived by EPA are intended to be used to calculate hazard for females of reproductive age/pregnant women, male infants, and male children. Use of the RPFs for other lifestages, such as adult males or women above reproductive age, may be overly conservative and is not intended for use with these age groups. In response to the public comments and feedback from SACC responses to the peer-review charge on the phthalate CRA, EPA has added this clarification to Section 2.6 of the CRA TSD.

Summary: A public comment (DCHP-0122) expressed concern that "the EPA estimate of the relative potency factor (RPF) for DCHP is substantially greater than the RPF for the index chemical DBP. This would indicate that DCHP causes adverse effects at lower doses than DBP. EPA should explain how this may affect the cumulative risk calculations."

EPA Response: EPA agrees with the commentor that the DCHP RPF of 1.66 indicates that DCHP is 66% more potent at reducing fetal testicular testosterone than the index chemical, DBP. For DCHP, multiplying the DCHP dose by the RPF of 1.66 would cause the total dose in DBP equivalents to increase, which in turn would contribute to a lower cumulative MOE, as explained in Section 5 of the CRA TSD. At the August 4-8 SACC meeting, EPA requested the committee provide comment on two approaches for deriving cumulative risk estimates for DCHP as well as the other phthalates. In response to SACC recommendations, EPA developed a Considerations for Determining Confidence in

Cumulative Risk Estimates For CRA Approaches 1 and 2 (see Section 5.4 of the CRA TSD). Based on the weight of scientific evidence considerations outlined in the developed framework, EPA has weighed the strengths and uncertainties associated with the DCHP RPF (Approach 1) and the DCHP POD (Approach 2 and individual DCHP risk evaluation). EPA acknowledges there are strengths and uncertainties of both approaches and concludes that Approach 2 using the POD from the single chemical assessment is the most appropriate for deriving cumulative risks for DCHP. This conclusion is based on the following:

- The POD approach (*i.e.,* Approach 2) is based on 6 studies, including the Li et al. study with testosterone data measured post-natally, and considers the full spectrum of adverse outcomes relevant to phthalate syndrome across a broad degree of dose levels, including multiple studies are or near 10 mg/k/day.
- In contrast, the RPF approach (*i.e.,* Approach 1) is based on 3 studies using a single adverse outcome (fetal testosterone) where only high dose data are available reducing confidence the BMD estimates at the lower end of the dose response curve.
- As a result, both from an adverse outcome pathway perspective and a dose-response perspective, EPA determined that the underlying data for the POD approach are more robust and more appropriate for extrapolating cumulative risk.

Summary: One public comment (SACC23-0060) addressed toxicological distinctions in DINP from other phthalates and requested EPA identify criteria for health effects representing a cumulative risk versus single substance risk. The comments also recommended EPA set criteria for decision making, such as a hazard index (HI) greater than 1 at the 95th percentile.

EPA Response: As discussed in the 2023 draft proposed approach for cumulative risk assessment of phthalate under TSCA, EPA considered data for 7 key outcomes associated with the phthalate syndrome MOA to determine the toxicological similarity of 7 phthalate diesters being evaluated under TSCA (including DEHP, DBP, DIBP, BBP, DCHP, DINP, DIDP), including decreased fetal testicular gene expression of steroidogenic genes; decreased fetal testicular testosterone; decreased male anogenital distance (AGD); increased incidence of male pup nipple retention; increased incidence of hypospadias; formation of multinucleated gonocytes (MNG); and seminiferous tubule atrophy. EPA also considered the full suite of effects associated with phthalate syndrome in the non-cancer human health hazard assessment for DINP. As discussed in both documents, oral exposure to DINP during the critical window of development leads to consistent perturbations in fetal testicular steroidogenic gene expression, reduced fetal testicular testosterone, increased Leydig cell effects (*e.g.,* Leydig cell aggregation), and increased MNG formation. However, DINP less consistently causes decreases in AGD, increased male pup nipple retention, and seminiferous tubule atrophy. Further, severe reproductive tract malformations following oral exposure to DINP are rare and only occur at very high doses (*e.g.,* 750 mg/kg-day or higher), and DINP has not been shown to cause hypospadias or cryptorchidism. The spectrum of effects in rats following oral exposure to DINP are consistent with the development of phthalate syndrome, and DINP was considered toxicologically similar to DEHP, DBP, DIBP, BBP, and DCHP. However, DINP is clearly less potent at inducing effects consistent with phthalate syndrome compared to DEHP, DBP, DIBP, BBP, and DCHP. Consistent with DINP being less potent, EPA calculated a relative potency factor (RPF) for reduced fetal testicular testosterone of 0.21, which indicates DINP is 5 times less potent than the index chemical (DBP) at reducing fetal testicular testosterone.

EPA did not use the hazard index approach to characterize cumulative risk (a relative potency factor approach was used), and therefore EPA did not set any criteria for decision making pertaining to use of a hazard index greater than one.

Summary: A commenter (SACC23-0046) on the *EPA's Draft Proposed Approach for cumulative risk assessment of high-priority phthalates under TSCA*, emphasized that risk management actions for any individual chemical substance or mixture may be control only "to the extent necessary" to mitigate "unreasonable risk" posed by that individual chemical substance or mixture. The comments focus largely on DINP, which they assert is toxicologically distinct relative to the other 5 phthalates (BBP, DBP, DCHP, DEHP, and DIBP) included in the proposal. The comments identify missing elements in the current model, such as the need for a tiered approach to cumulative risk assessment and the inclusion of human relevance as a key concept. Concerns are raised about violations of scientific principles, particularly regarding toxicological similarity, arguing against expanding definitions to include syndromes, and the use of a global relative potency factor (RPF) due to distinct potencies across outcomes (*i.e.*, the dose-response curves are not parallel). They asserted that there are differences in the toxicity of low molecular weight phthalates (*e.g.*, BBP, DBP, and DEHP) and high molecular weight phthalates such as DINP with the high molecular weight phthalates not having adverse male reproductive effects and having a mode of action for reduced testosterone that is not androgen dependent. The document stresses the importance of using the best available science for exposure data, specifically recommending urinary metabolite data. In the context of DINP, the commenter notes its distinct hazard and potency profile compared to other phthalates and highlights missing studies and mechanistic data, suggesting the mode of action does not reflect the best available science.

In a subsequent set of comments (DCHP-0125), the commenter emphasized that EPA should only conduct a cumulative risk assessment for chemicals with a similar mode of action (see Section 5.3).

Conversely, another comment (SACC23-0058) requests that EPA add statements to the draft cumulative approach document clarifying that the level of data available for phthalates goes "well beyond" what is necessary for establishing toxicological similarity, and states that the large amount of data available for the phthalates should not be used to establish a standard of evidence for future cumulative risk assessments.

EPA Response: EPA acknowledges the commentors (SACC23-0046; DCHP-0125; SACC23-0058) opinions regarding EPA's cumulative risk assessment approaches and the *Draft Proposed Approach for cumulative risk assessment of high-priority phthalates under TSCA*. In conducting this cumulative phthalates assessment, EPA extensively discussed the underlying assumptions and available database that resulted in the determination that DEHP, BBP, DBP, DCHP, DIBP, and DINP (but not DIDP due to the determination DIDP does not induce effect consistent with phthalate syndrome) are considered toxicologically similar for phthalate syndrome effects for use in the draft cumulative risk assessment (CRA) and relative potency factor (RPF) derivation. DINP was not found to be toxicologically unique from other phthalates as in the case of DIDP, and EPA disagrees in the commentors of other factors that would change this determination (*e.g.*, molecular weight). In determining RPF derivation approach, EPA discussed available dose-response data for phthalate syndrome-related outcomes (*e.g.*, decreased fetal testicular testosterone production and apical outcomes such as decreased AGD, nipple retention, seminiferous tubule atrophy, and hypospadias) for individual phthalates. EPA's analysis outlines the strength and reliability of using reduced fetal testicular testosterone production and the calculation of RPFs based on different benchmark responses for the included phthalates. In its

discussion, EPA comparatively acknowledged the differences in toxicity of the included phthalates (ultimately selecting DBP as the index chemical for RPF derivation for the CRA of phthalates). EPA considered multiple factors including the widely accepted common phthalate syndrome mode of action (MOA) (initiating key events are not entirely known but reduced fetal testicular testosterone is a well-characterized upstream event tied to reliable dose-response data), the strength of dose-response database for each key outcome (outcomes along the phthalate syndrome MOA pathway), and species-differences (there was determined to be no sufficient differences across mammalian phthalate ADME that would impact this CRA approach). EPA is confident in this CRA approach and believes it has thoroughly discussed and scaled individual chemical potencies based upon reliable data for use in RPF derivation and CRA. Further, EPA believe the CRA of phthalates does not establish an evidentiary standard but rather provides a novel approach and framework for conducting cumulative assessments for high-priority chemicals under TSCA. By taking a cumulative approach to phthalates, which considers TSCA uses as well as the broader background of people's exposure to multiple phthalates, EPA is being protective of susceptible subpopulations (*i.e.*, women of reproductive age, pregnant women, male infants, male children) exposures to phthalates.

Summary: A public commenter (SACC25-0148) highlights that the selected endpoint for toxicity—reduced fetal testicular testosterone content and/or production—is appropriate for estimating relative potency factors (RPFs) for the phthalates cumulative risk assessment (CRA). This endpoint is a common effect observed in five of the six phthalates evaluated (excluding DIDP, which does not exhibit this effect). For five out of the six phthalates, decreased fetal testosterone levels are a consistent endpoint, making it a robust choice for RPF estimation.

Several additional public commenters (DEHP-0135 DIBP-0123) "supports EPA's use of fetal testicular testosterone (FTT) for the relative potency factor (RPF) as opposed to the use of a transcriptomic point of departure (tPOD) given the unconfirmed molecular initiating event (MIE)".

The commenter (SACC25-0148) criticizes EPA's preliminary selection of RPFs based on BMD40 estimates as a conservative choice (likely made because of the absence of BMD10 and BMD5 estimates for some phthalates like BBP and DIBP). The comment suggests that the EPA could make more informed estimates for these values through inference, as the consistency in RPFs across response levels for other phthalates like DEHP and DIBP should bolster confidence in estimating missing values. Specifically, the commenter suggests using 0.52 for BBP's BMD5 and BMD10, and 0.53 for DIBP's BMD5, as these values show consistency across other response levels. The commenter argues that BMD40 estimates are not sufficiently health-protective, and that BMD5 and BMD10 values are more aligned with health-protective goals, recommending their use for RPF determination.

EPA Response: EPA acknowledges the public comments (SACC25-0148, DEHP-0135, DIBP-0123), and agrees with the commentors that decreased fetal testicular testosterone is an appropriate choice for RPF estimation.

For the final phthalate CRA, EPA has retained use of RPFs estimated based on BMD40 estimates. As discussed in Section 2.4 of the CRA TSD, the calculated RPFs using BMD_5, BMD_{10}, and BMD_{40} estimates for DEHP, DCHP, and DINP were nearly identical for each phthalate. RPFs ranged from 0.82 to 0.84 for DEHP, 1.66 to 1.71 for DCHP, and 0.19 to 0.21 for DINP. For DIBP, an RPF of 0.53 was calculated using both BMD_{10} and BMD_{40} estimates; however, no RPF could be calculated using a BMD_5 because a BMD could not be estimated for DIBP at the 5 percent response level. For BBP, an

RPF of 0.52 was calculated using the BMD_{40} estimate. RPFs could not be estimated for BBP at the 5 or 10 percent response levels because BMD_5 and BMD_{10} values could not be estimated for BBP. There is some uncertainty in the applicability of the selected RPFs for DIBP and BBP at the low response levels (*i.e.*, 5% to 10% changes), since RPFs could not be estimated for BBP at the 5 or 10 percent response levels or for DIBP at the 5 percent response level using Metafor Version 4.6.0 BMD modeling results. However, the lack of variability in calculated RPFs for DEHP, DCHP, and DINP across response levels, and the fact that the RPF for DIBP was identical at the 10 and 40 percent response levels, increases EPA's confidence in the selected RPFs for BBP and DIBP. Furthermore, a comparison of the selected RPFs based on BMD_{40} estimates calculated using Metafor Version 4.6.0 to candidate RPFs calculated based on BMD_5 estimates calculated using Metafor Version 2.0.0 demonstrates that RPFs calculated at both response levels using different Versions of Metafor are similar. For example, the selected RPF for DEHP is 0.84 compared to a candidate RPF of 0.88 (4.8% difference); the selected RPF for DIBP is 0.53 compared to a candidate RPF of 0.42 (21% difference); the selected RPF for BBP is 0.52 compared to a candidate RPF of 0.48 (7.7% difference); the selected RPF for DCHP is 1.66 compared to a candidate RPF of 1.83 (10% difference); and the selected RPF for DINP is 0.21 compared to a candidate RPF of 0.19 (9.5% difference). The fact the selected RPFs based on BMD_{40} estimates calculated using Metafor Version 4.6.0 are similar to RPFs based on BMD_5 estimates calculated using Metafor Version 2.0.0 further increases EPA's confidence in the selected RPFs, and indicates that the selected RPFs derived at the 40 percent response level are expected to provide reasonable estimates of potency at the 5 and 10 percent response levels. Further, SACC supported the use of RPF values based on BMD_{40} estimates (U.S. EPA, 2025), once the Committees other recommendations regarding RPF derivation had been addressed. As discussed further in EPA's response to the SACC peer-review report (see EPA responses to Charge Questions 5f, 5g, 9, and 10), EPA considered and integrated SACC's recommendations for deriving RPFs into the final assessment (see the 2025 CRA TSD) and concluded that RPFs based on BMD_{40} estimates are supported by reasonably available information and the best available science.

Finally, it is important to note that although derived RPFs were based on BMD_{40} estimates, EPA selected the $BMDL_5$ for use as the index chemical (DBP) POD, which is health protective.

Summary: A public comment (DCHP-0128) states that "EPA's Focus on Acute Exposures Disregards the Additive Effects Over a Lifetime." The commenter further states that "While EPA correctly recognizes that adverse effects consistent with phthalate syndrome may occur from a single exposure, it unjustifiably limits the scope of its own analysis by only considering cumulative exposures that occur simultaneously….Additionally, developmental reproductive effects caused by early life exposures result in physiological changes that persist throughout a lifetime. Therefore, EPA must consider that later life exposures that can cause reproductive effects may be exacerbated by existing effects resulting from gestational and early-life exposures that affected development."

EPA Response: EPA disagrees with the public commenter. As discussed in Section 1.5 of the CRA TSD, EPA is focusing the application of its phthalate CRA on acute exposure durations which are expected to represent the highest relevant exposures for the common health effect for susceptible populations. Notably, protecting for acute exposure durations will be protective of longer duration exposures, since acute exposures are higher than longer duration exposures.

Summary: A public comment (SACC25-0148) expressed disappointment in the "quality of the sections on the presentation on the nature of, and preference for, an option for the construct of the cumulative risk assessment(s)." They also say "the Agency is struggling to describe the exposure component of risk assessment confidently and clearly. The text in Section 5 CONCLUSION AND NEXT STEPS is simply too muddled and tortured to do a credible review and reach a reasonable conclusion on the option EPA has chosen."

EPA Response: EPA notes that this public commenter did not provide specific recommendations for improving the clarity and presentation of information in Section 5 of CRA TSD. However, EPA has made significant revisions to the final CRA TSD based on SACC recommendations and recommendations from other public commenters. Revisions include (but are not limited to):

- A more detailed discussion of CRA Approaches 1 and 2, including a new table (Table 5-1) that provides a comparison of the two approaches;
- Several example calculations demonstrating implementation of CRA Approaches 1 and 2;
- A discussion of the impact of CRA Approaches 1 and 2 on cumulative risk estimates; and
- A new section that provides a detailed comparison of the two CRA approaches, including a framework of considerations for CRA Approach selection for each phthalate (see Section 5.4 of CRA TSD).

Summary: A public comment (SACC23-0058) stated that EPA should carry forward all outcomes that are considered for RPF development rather than narrowing the list of outcomes. In support of the request, the commenter stated that the ED50s for the phthalates indicate that the relative potencies can vary across outcomes; the commenter provides the example of DBP, which has a lower relative potency for gene expression than nipple retention. The commenter stated that it is not clear whether EPA thinks relative potency should vary depending on outcome, or if there is one relative potency among chemicals that is subject to experimental variability across outcomes.

EPA Response: EPA disagrees with the public commenter (SACC23-0058). As described in OPP's *Guidance on Cumulative Risk Assessment of Pesticide Chemicals that have a Common Mechanism of Toxicity* (U.S. EPA, 2002), RPFs should be developed based on a uniform point of comparison. For chemical substances grouped for CRA, this includes, whenever possible, using the same common effect, same measure of potency, same species/strain and studies that were conducted using relatively comparable methodology. Additionally, consideration should be given to the human relevance of the effect. As discussed in the 2023 CRA proposed approach (U.S. EPA, 2023), EPA outlined six potential options for deriving RPFs that considered use of data from two gestational outcomes (*i.e.*, altered expression of steroidogenic genes in the fetal testis and decreased fetal rat testicular testosterone) and four postnatal outcomes (*i.e.*, reduced anogenital distance (AGD), increased nipple retention, seminiferous tubule atrophy, and hypospadias).

Strengths, limitations, and uncertainties of the available data sets for each of the six key outcomes considered for RPF derivation are discussed in detail in Section 4.4 of the draft 2023 approach (U.S. EPA, 2023), as well as in Section 1.3 of the 2025 CRA TSD. Overall, EPA noted several factors that increased its confidence in using the fetal testicular testosterone data set to derive RPFs, including:

- Reduced testosterone production in the fetal testis plays an early role in the phthalate syndrome MOA.

- Androgen action has a conserved role in the development of the male reproductive system across mammalian species, including humans.
- There are dose-response data available for all six of the toxicologically similar phthalates from multiple studies that are similar in design to support RPF derivation (*i.e.*, utilize the same species/strain of rat, same route/method of exposure, similar exposure durations, similar timing and method (*i.e., ex vivo* testosterone production via radioimmunoassay or fetal testicular testosterone content) of measurement.
- During the 2023 peer-review meeting, SACC supported fetal testosterone production as an outcome for phthalate syndrome (U.S. EPA, 2023).

In contrast, EPA noted several factors that decreased its confidence in using postnatal outcomes to derive RPFs, including variability in outcome reporting; limited dose-response data; inconsistent findings for several outcomes (*e.g.*, reduced anogenital distance, increased nipple retention) for DINP; variability in study design and duration. Given the limitations and uncertainties associated with studies of postnatal outcomes, EPA did not attempt to derive RPFs for any postnatal outcome.

Given the strengths, limitations, and uncertainties of each key outcome discussed in Section 4.4 of the draft 2023 approach (U.S. EPA, 2023) and Section 1.3 of the 2025 CRA TSD, EPA selected reduced fetal testicular testosterone as the basis for deriving RPFs.

Summary: A public comment (SACC23-0052) supported the use of a relative potency factor (RPF) approach to assessing cumulative risk of phthalates, as well as the evaluation of RPF using fetal testicular testosterone data, but recommended additional endpoints be considered. An additional public comment (SACC23-0058) supported the use of the RPF approach, noting that EPA should include further discussion of the advantages of RPF such as the ability to estimate relevant exposure units (mg/kg-day) rather than the unitless Hazard Index, and that RPF can be used in conjunction with probabilistic dose-response methods.

EPA Response: EPA acknowledges the support from the public commenter (SACC23-0052) to use RPFs based on reduced fetal testicular testosterone data. As discussed in the 2023 CRA proposed approach (U.S. EPA, 2023), EPA outlined six potential options for deriving RPFs that considered use of data from two gestational outcomes (*i.e.*, altered expression of steroidogenic genes in the fetal testis and decreased fetal rat testicular testosterone) and four postnatal outcomes (*i.e.*, reduced anogenital distance (AGD), increased nipple retention, seminiferous tubule atrophy, and hypospadias). Given the strengths, limitations, and uncertainties of each key outcome discussed in Section 4.4 of the draft 2023 approach (U.S. EPA, 2023) and Section 1.3 of the 2025 CRA TSD EPA, selected reduced fetal testicular testosterone as the basis for deriving RPFs.

During the 2023 peer-review of the CRA proposed approach, SACC also recommended that EPA consider adding a second endpoint in addition to phthalate syndrome for demonstrating toxicological similarity and conducting CRA (U.S. EPA, 2023). Specifically, SACC recommended including liver toxicity, developmental neurotoxicity, or female reproductive effects. While EPA acknowledges that there are varying amount of data demonstrating that certain phthalates can cause these effects, EPA did not consider these effects as the basis for deriving RPFs or a CRA for several reasons. First, although DEHP, BBP, DBP, DCHP, DIBP, DINP, and DIDP have all been shown to cause liver toxicity, most of the observed liver effects in experimental animal models are mechanistically linked to peroxisome proliferator-activated receptor alpha (PPARα) activation, which can vary between species raising questions about human relevance. Additionally, the non-cancer POD based on

phthalate syndrome-related effects is a more sensitive outcome than liver toxicity for most phthalates (with DINP and DIDP being exceptions). Further, there are limited data demonstrating female reproductive effects or developmental neurotoxicity for DCHP and DIBP, while data for other phthalates varies in quality and quantity such that definitive conclusions about exposure-response relationships cannot be established. Therefore, EPA did not consider liver toxicity, developmental neurotoxicity, or female reproductive effects further as the basis for deriving RPFs or a CRA.

Summary: Regarding the key outcomes in the AOP for the development of phthalate syndrome, the public comment (SACC23-0042, SACC23-0045) stated that:

1. EPA should provide more detailed justification for selecting male reproductive effects characterizing phthalate syndrome as the most appropriate hazard endpoint on which to base toxicologic similarity in the CRA.
2. All seven key outcomes are relevant for inclusion in the hazard analysis and development of relative potencies, where appropriate, given the robustness of the database for each of these outcomes.
3. If there are effects on other hazard endpoints (*e.g.*, female reproductive tract, cancer, neurodevelopment, etc.) at lower doses than those that affect the male phthalate syndrome endpoints, then EPA needs to have a plan to adjust for this circumstance (*e.g.*, application of additional uncertainty factors when deriving margins of exposure).
4. The importance of clarifying whether there is just one or more than one MOA (androgen-independent and androgen-dependent) will depend upon whether the Agency chooses to apply the data on the adverse outcome at the end of the MOA/adverse outcome pathway or a molecular initiating event or key event within the pathway(s) to support the relative potency comparisons.

EPA Response: EPA acknowledges the commenters (SACC23-0042, SACC23-0045) suggested considerations in evaluating the phthalate syndrome mode of action (MOA) in its risk assessment. EPA believes it has appropriately considered the hypothesized phthalate syndrome MOA and the key events along the MOA (and associated dose-response data) in conducting its human hazard assessment of individual phthalates and in the CRA.

1. EPA believes it has adequately discussed the supporting database of the hypothesized phthalate syndrome MOA and associated dose-response data of various key outcomes (*e.g.*, reduced fetal testicular testosterone production, reduced AGD, hypospadias, cryptorchidism, sperm effects, etc.). In reviewing available human health effects data, EPA determined effects on the developing male reproductive system were the most sensitive endpoint for use in risk characterization of phthalate exposure, which was in agreement with multiple other regulatory bodies, including Health Canada, NASEM, NICNAS, and U.S. CPSC. Available dose-response data for phthalate syndrome-related outcomes were discussed in the CRA and toxicological similarly was based on these most sensitive outcomes were determined for DEHP, BBP, DBP, DCHP, DIBP, and DINP (but not DIDP due to the determination DIDP does not induce effect consistent with phthalate syndrome).
2. EPA discussed all considered key outcomes and associated dose-response for considered in the CRA of phthalates. EPA considered data for 7 key outcomes associated with the phthalate syndrome MOA to determine the toxicological similarity, including decreased fetal testicular gene expression of steroidogenic genes; decreased fetal testicular testosterone; decreased male

anogenital distance (AGD); increased incidence of male pup nipple retention; increased incidence of hypospadias; formation of multinucleated gonocytes (MNG); and seminiferous tubule atrophy. Strengths, limitations, and uncertainties of the available data sets for each of the six key outcomes considered for RPF derivation are discussed in detail in Section 4.4 of the draft 2023 approach (U.S. EPA, 2023), as well as in Section 1.3 of the 2025 CRA TSD. Given the strengths, limitations, and uncertainties of each key outcome discussed in Section 4.4 of the draft 2023 approach (U.S. EPA, 2023), EPA believes the selected endpoint of reduced fetal testicular testosterone as the basis for deriving RPFs is the most robust approach.

3. For the toxicologically similar phthalates (DEHP, BBP, DBP, DCHP, DIBP, and DINP), EPA is confident that effects on the developing male reproductive system are the most sensitive endpoint on which to base its risk characterization. EPA discussed additional identified health effects (*e.g.*, female reproductive outcomes, liver effects, kidney effects) for each phthalate within the individual human hazard TSDs. However, these effects were not determined to be more sensitive than phthalate syndrome-related outcomes on the developing male reproductive system. For DIDP, which was found to not induce effect consistent with phthalate syndrome, the point of departure was based upon reduced offspring survival in rats.
4. EPA's discussion of the hypothesized phthalate syndrome MOA (see Figure 3-3 of the draft 2023 approach (U.S. EPA, 2023)) indicated that the exact molecular initiating events preceding cellular and tissue/organ level responses remain unknown. Although both androgen receptor antagonism and PPARα activation have been hypothesized, there is conflicting evidence. Nevertheless, cellular-, organ, and organism-level responses are generally characterized, and the available health effects studies and/or dose-response data on these key outcomes along this MOA were discussed. There is not currently evidence to suggest multiple MOAs for phthalates based upon differing molecular initiating events (*i.e.*, such as androgen-dependent or –independent as suggested by the commentor), and thus, EPA is confident in its evaluation of the hypothesized MOA and key outcomes for used in relative potency comparison.

Summary: A public commenter (DCHP-0125) did not support the use of RPFs based on reduced fetal testicular testosterone or the overall cumulative risk assessment (CRA) approach. These comments are a follow-up to earlier comments (SACC23-0046) on the EPA draft proposed approach for cumulative risk assessments (see Section 1.1.2). They (DCHP-0125) state that EPA claims "…testosterone reduction is an early marker in the phthalate syndrome mode of action (MOA), that androgen action is a conserved role in development, and that there is a sufficient body of data to support RPF derivation for a total of six phthalates [BBP, DBP, DCHP, DEHP, DIBP, and DINP] (US EPA, 2024c, p. 14)." The commenter quoted SACC, "… 'that there seems to be some controversy regarding whether the antiandrogenic mode of action is indeed shared between all reported adverse effects' (US EPA, 2023b, p. 67)." The commenter referenced Li et al. 2021 in support of two distinct adverse outcome pathways with discrete key events for phthalate syndrome. One pathway is androgen dependent but only applicable to rats, whereas the other is androgen-independent with applicability to both rodents and humans. The recent work of Lea et al. (2025), and Rogers et al. (2025) related to DINP was cited. The commenter concludes that, "…the inclusion of DINP in the CRA, based on an anti-androgenic MOA being shared with the other phthalates, is not supported when using the MOA network that meets EPA standards and incorporates the best available science." Recent MOAs evidence for phthalate syndrome does not support inclusion of DINP in the CRA.

The commenter also noted that the dose-response curves used to derive the RPF are not parallel among the phthalates. Therefore, the RPFs are dependent on the response level at which the phthalates are

compared. To show the importance of parallelism in RPF derivation, fetal testosterone production data from (Furr et al., 2014) were modeled by the commenter. DBP was more potent than DEHP and less potent than DCHP and BBP in reducing testosterone production. For BBP, the RPF varied from 1.67 for 50% reduction in testosterone production to 6.68 for a 5% reduction in testosterone production. The commenter also noted the importance of heterogeneity in results, stating, "Further, in a qualitative comparison of the ED5, ED10, ED50s derived from Furr et al., (2014), between the DCHP Blocks 23 and 33 there is apparent intra-lab heterogeneity." The commenter stated, "For these reasons [*e.g.,* strain, timing of outcome measurement, testosterone production vs. content], EPA should perform sensitivity analyses to assess and identify sources and impacts of heterogeneity by including covariates in the meta-regression models."

They continue by saying, "The SACC specifically recommended that EPA consider adding a second endpoint in addition to phthalate syndrome, and even suggested options such as liver toxicity, developmental neurotoxicity, and female reproductive effects." They point to NASEM including evaluation of fetal testosterone and anogenital distance (AGD) as another alternative or second endpoint. They stated, "EPA should more fully describe why they did not follow NASEM's assessment or the recommendations from the SACC, instead using a single endpoint and single life stage; they should also describe any uncertainty that may have resulted from this decision."

EPA Response: EPA disagrees with the public commenters that it was inappropriate to derive RPFs based on decreased fetal testicular testosterone. As discussed in EPA's 2023 draft proposed approach for CRA of phthalates, as well as in the 2025 CRA TSD and the non-cancer human health hazard TSDs for DEHP, DBP, DIBP, BBP, and DCHP, the MOA underlying phthalate syndrome is not fully characterized. There are well-established aspects of phthalate syndrome that are mechanistically linked to decreased androgen action and decreased fetal testicular testosterone, such as reduced AGD, increased male pup nipple retention, and male reproductive tract malformations such as cryptorchidism, hypospadias, and others. There are also effects within the syndrome that occur through a less well characterized mechanism(s), such as multinucleated gonocyte formation, Sertoli cell effects, and sperm effects. Regardless, reduced fetal testicular testosterone is a well-established effect associated with phthalate syndrome, and decreased fetal testicular testosterone is an early, upstream event in the MOA that precedes downstream apical outcomes such as male nipple retention, decreased anogenital distance, and male reproductive tract malformations (*e.g.,* hypospadias and cryptorchidism). Decreased fetal testicular testosterone should occur at doses that are lower than or equal to doses that cause downstream apical outcomes associated with a disruption of androgen action.

EPA disagrees with the public commenter that DINP should not be included as part of the phthalate CRA. In its 2023 proposed approach for CRA of phthalates under TSCA, EPA demonstrated the DINP is toxicologically similar to DEHP, DBP, DIBP, DCHP, and DIBP. Notably, DINP consistently perturbs steroidogenic gene expression in the fetal testis, and reduces fetal testis testosterone production, albeit at higher doses than DEHP, DBP, DIBP, DCHP, and DIBP. Dose-response modeling clearly demonstrates that DINP is a less potent antiandrogen than DEHP, DBP, DIBP, DCHP, and DIBP, which is reflected in the smaller DINP RPF of 0.21. Notably, in 2023 SACC agreed with EPA's conclusion that DINP is toxicologically similar to, but less potent than, DEHP, DBP, DIBP, DCHP, and DIBP.

As discussed by the National Research Council in 2008 (NRC 2008), there may be challenges associated with the RPF approach because phthalate dose-response curves may lack "parallelism." For parallel dose-response curves the RPF is constant, regardless of the response level (that is, 5%, 10%, or 40%). However, different chemical dose-responses may have differing shape and slope dose-

response curves leading to variability in RPFs across different BMRs. This concern was echoed by the SACC during the 2023 peer-review of EPA's *Draft Proposed Approach for Cumulative Risk Assessment (CRA) of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substances Control Act* and the August 2025 peer-review of this CRA TSD. Although SACC noted that parallel dose-response curves are not required for estimating RPFs, they are preferred, and demonstrating parallel dose-response curves would increase confidence in EPA's derived RPFs.

Consistently, EPA's *Advances in Dose Addition for Chemical Mixtures: A White Paper* (U.S. EPA, 2023) states "*In the Agency-wide guidance on dose addition, there is an assumption of constant relative potency (U.S. EPA, 1987; U.S. EPA, 2000), but a demonstration of empirical evidence, such as similar DRC [dose-response curve] shapes, is not required."* Thus, RPFs can be applied for chemicals with dissimilar dose-response curves, as the establishment of a known or suspected common MOA shared by members of the class of compounds is considered more fundamental. It is common practice to estimate RPFs closer to the low-dose range of the dose-response function. This practice is intended to reduce possible high-dose influences on estimated RPFs that may arise due to saturation of certain kinetic processes (*e.g.*, receptor binding, metabolic elimination). However, this approach also carries an implicit assumption that dose-response curve shapes will be similar below the selected response level (U.S. EPA, 2023).

For parallel dose-response curves, the RPF is constant regardless of the response level (that is, 5%, 10%, or 40%). As discussed earlier in Section 2.4.1.3 of the CRA TSD, candidate RPFs calculated using BMD_5, BMD_{10}, and BMD_{40} estimates derived using Metafor Version 4.6.0 were nearly identical across response levels for DEHP (RPFs ranged from 0.82–0.84), DCHP (RPFs ranged from 1.66–1.71), and DINP (RPFs ranged from 0.19–0.21), providing evidence of parallel dose-response curves with the index chemical DBP. For DIBP, an RPF of 0.53 was calculated at both the 10 and 40 percent response levels, providing evidence of parallel dose-response curves with the index chemical; however, no RPF could be calculated at the 5 percent response level because a BMD_5 could not be estimated for DIBP. For BBP, an RPF of 0.52 was calculated using the BMD_{40} estimate. RPFs could not be estimated for BBP at the 5 or 10 percent response levels because BMD_5 and BMD_{10} values could not be estimated for BBP.

For use in the CRA, EPA selected RPFs based on BMD_{40} estimates calculated using Metafor Version 4.6.0, since this was the only the only response level at which a full set of RPFs could be derived for all phthalates included in the CRA. Because candidate RPFs could not be derived for BBP or DBP at the 5 percent response level, or for BBP at the 10 percent response level, there is some uncertainty regarding constant proportionality for these two phthalates in the low-end range of the dose-response curve. However, this uncertainty was addressed by calculating candidate RPFs using BMD estimates derived via Metafor Version 2.0.0, which allowed BMD estimates to be calculated for all phthalates at all response levels. As discussed earlier in Section 2.4 of the CRA TSD, there was little variability in candidate RPFs calculated using BMD_5, BMD_{10}, and BMD_{40} estimates derived using Metafor Version 2.0.0, providing evidence of parallel dose-response curves for DEHP, DBP, BBP, DCHP, DIBP, and DINP. Further, candidate RPFs calculated using BMD_5 estimates derived using Metafor Version 2.0.0, were similar to the selected RPFs calculated using BMD_{40} estimates derived using Metafor Version 4.6.0. This indicates that the selected RPFs derived from the 40 percent response level are expected to provide a reasonable estimates of potency at the 5 and 10 percent response levels, and provides evidence of parallel dose-response curves for all the phthalates included in the CRA.

During the 2023 peer-review of the CRA proposed approach, SACC also recommended that EPA consider adding a second endpoint in addition to phthalate syndrome for demonstrating toxicological

similarity and conducting CRA (U.S. EPA, 2023c). Specifically, SACC recommended including liver toxicity, developmental neurotoxicity, or female reproductive effects. While EPA acknowledges that there are varying amount of data demonstrating that certain phthalates can cause these effects, EPA did not consider these effects as the basis for a CRA for several reasons. First, although DEHP, BBP, DBP, DCHP, DIBP, DINP, and DIDP have all been shown to cause liver toxicity, most of the observed liver effects in experimental animal models are mechanistically linked to peroxisome proliferator-activated receptor alpha (PPARα) activation, which can vary between species raising questions about human relevance. Additionally, the non-cancer POD based on phthalate syndrome-related effects is a more sensitive outcome than liver toxicity for most phthalates (with DINP and DIDP being exceptions). Further, there are limited data demonstrating female reproductive effects or developmental neurotoxicity for DCHP and DIBP, while data for other phthalates varies in quality and quantity such that definitive conclusions about exposure-response relationships cannot be established. Therefore, EPA did not consider liver toxicity, developmental neurotoxicity, or female reproductive effects further as the basis for a CRA. However, these effects are discussed further, as relevant, in the cancer human health hazard assessment of phthalates (U.S. EPA, 2025a) and each individual non-cancer human health hazard assessments for DEHP (U.S. EPA, 2025x), DBP (U.S. EPA, 2025v), DIBP (U.S. EPA, 2025y), BBP (U.S. EPA, 2025u), DCHP (U.S. EPA, 2025w), DINP (U.S. EPA, 2025z), and DIDP (U.S. EPA, 2024d).

EPA considered deriving candidate RPFs using the one postnatal outcome supported by SACC (*i.e.*, reduced AGD). However, given the limitations and uncertainties discussed in the 2025 CRA TSD, EPA considered there to be too much uncertainty associated with the data set to derive candidate RPFs for all six of the phthalates included in the CRA. Further, reduced rat AGD is a less sensitive outcome than reduced rat fetal testicular testosterone. This is demonstrated by the 2017 NASEM meta-analysis and BMD analysis of reduced fetal rat testicular testosterone and reduced rat AGD for DEHP, DBP, and BBP, which provides BMD_5 estimates of 15 (reduced fetal testis testosterone) and 270 (reduced AGD) mg/kg-day for DEHP; 12 (reduced fetal testis testosterone) and 150 (reduced AGD) mg/kg-day for DBP; 23 (reduced fetal testis testosterone) and 250 (reduced AGD) mg/kg-day for BBP (NASEM, 2017). Further, NASEM judged the animal database for AGD to not be amenable to meta-analysis for DIBP and DINP. EPA did not identify any new information that would change the conclusions drawn from the NASEM meta-analysis.

Mechanism of Action

Summary: Two public comments (SACC23-0031, SACC23-0065) expressed mixed opinions on the robustness of the mode of action proposed in *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substance Control Act* for selecting the most sensitive effect.

EPA Response: The relevance of the phthalate syndrome mode of action (MOA) in humans based upon the spectrum of phthalate-syndrome antiandrogenic effects across phthalates and cross-species differences was thoroughly discussed in the *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substance Control Act*. It was recognized in Section 3.1.4.1 that human explant and xenograft studies were found to have limitations and were variable in terms of antiandrogenic or steroidogenesis effects of phthalates in human fetal tissue. However, discussion of evidence and existing assessments (ECCC/HC; EFSA; ECHA; NICNAS; U.S. CPSC) determined effects of phthalates on the human developing male reproductive system through antiandrogenic MOA cannot be ruled out, and rat

models are appropriate for characterizing risk to human health. Some species differences in phthalate metabolism and toxicokinetic are reported, but metabolite excretion profiles are similar across mammalian species, including humans (Section 3.1.5.1). As discussed in Section 3.1.5, although there are some species differences in sensitivity (*e.g.,* variability in reproductive tract steriodogenic or critical window of susceptibility across some mammalian models), antiandrogenic effects are generally conserved across mammalian species. Overall, EPA acknowledged that the molecular initiating event(s) of phthalate syndrome are not entirely know (*e.g.,* Figure 3-3 of the cumulative risk assessment). However, evidence integration across key outcomes for grouping phthalates (Section 3.1.2 discussion on consistent sensitive hallmarks of phthalate syndrome-related effects across phthalates) and species-specific considerations (Section 3.1.4 – 3.1.7) support that DEHP, DBP, BBP, DIBP, DCHP, and DINP are toxicologically similar, and the phthalate syndrome MOA is applicable to humans and provides the most sensitive effects for phthalate exposure. EPA is confident in the robustness and sensitivity of the phthalate syndrome MOA for characterizing risk to human health. To account for potential human- and species-related variability, EPA applied the animal to human extrapolation factor (*i.e.,* interspecies extrapolation; UF_A) of 3× and a within human variability extrapolation factor (*i.e.,* intraspecies extrapolation; UF_H) of 10×.

Summary: A public comment (SACC23-0052) recommended EPA base the cumulative assessment on phthalate syndrome as a whole, rather than the most sensitive effect and stated that the 6 phthalates may act by slightly different mechanisms and not share a single most sensitive endpoint.

EPA Response: Notably, in 2023 SACC agreed with EPA, that focusing on "the most sensitive effect of phthalate exposure is the best approach given the limitations of available data and the challenges of using other approaches" (U.S. EPA, 2023). As discussed in EPA's 2023 draft proposed approach for CRA of phthalates under TSCA, EPA considered addressing phthalate syndrome by focusing on the syndrome as a whole (see Section 4.1.1 of the 2023 CRA proposal) and by focusing on the most sensitive effect (see Section 4.1.2 of the 2023 CRA proposal). As discussed in the 2023 CRA proposal, there are limitations associated with addressing the syndrome as a whole (namely, lack of comprehensive dose-response models for addressing the syndrome as a whole). Further, proposed dose-response models (*e.g.*, Blessinger et al. 2020) for addressing phthalate syndrome as a whole require individual pup level data, which is infrequently available and would limit the number of studies available to EPA for BMD modeling. Due to this limitation and others, EPA determined that addressing phthalate syndrome as a whole is not supported by reasonably available information and does not represent the best available science.

Summary: Another comment (BBP-0120, DIBP-0123, DEHP-0136, DBP-0130), commends the EPA for advancing the cumulative risk assessment (CRA) for phthalates and peer review by the Scientific Advisory Committee on Chemicals (SACC), stressing the need for adherence to the best available science per the Toxic Substances Control Act (TSCA). However, the comment points out deficiencies in the draft CRA, such as issues with benchmark dose modeling, points of departure (POD), and RPF. It criticizes EPA's deviation from its Benchmark Dose (BMD) Technical Guidance, particularly using a 10% benchmark response (BMR) for continuous endpoints like fetal testicular testosterone (FTT), suggesting a BMR based on one control standard deviation instead. The comment agrees that products with multiple phthalates are unlikely to significantly expose workers due to low concentrations and non-overlapping usage. It emphasizes the need for a consistent health endpoint and recommends using the best available science while avoiding combining disparate effects among phthalates. It urges EPA to address deficiencies, reconsider occupational exposure calculations, recalibrate BMRs, and validate

detection methods before setting regulatory measures to comply with TSCA guidelines.

EPA Response: EPA disagrees with commentors (BBP-0120, DIBP-0123, DEHP-0136, DBP-0130), that state or suggest the BMR of 1 control SD is more appropriate than the selected BMR of 5 percent. For BMD modeling of continuous data, such as reduced fetal testicular testosterone, EPA's BMD Technical Guidance (U.S. EPA , 2012) recommends a BMR of 1 control SD "always be presented for comparison purposes," however, EPA's BMD technical guidance also states that "The ideal is to have a biological basis for the BMR for continuous data." For reduced fetal testicular testosterone, EPA evaluated BMRs of 5, 10, and 40% based on biological and statistical considerations, as outlined in the appendix titled "Considerations for Benchmark Response (BMR) Selection for Reduced Fetal Testicular Testosterone," which is included in the non-cancer human health hazard assessments for DEHP, DBP, DIBP, DCHP, and BBP. As discussed in this appendix in each non-cancer TSD, EPA has reached the conclusion that a BMR of 5 percent is the most appropriate and health protective response level for evaluating decreased fetal testicular testosterone when sufficient dose-response data are available to support modeling of fetal testicular testosterone in the low-end range of the dose-response curve. As such EPA considers $BMDL_5$ estimates for reduced fetal testicular testosterone appropriate for determining the point of departure (POD) for each high-priority phthalate.

It is also important to note that BMD modeling of reduced fetal testicular testosterone data was used to support not only POD determination, but also determination of RPFs for use in the phthalate CRA. RPFs must be based on a constant response level. A BMR of 1 control SD is inappropriate to use for estimating RPFs because SD will vary from study-to-study and from phthalate-to-phthalate and therefore does not represent a constant response level appropriate for deriving RPFs.

As described in the occupational exposure value derivation appendices, which are included in the risk evaluations of DEHP, BBP, DIBP, DCHP, and DBP, EPA has calculated 8-hour existing chemical occupational exposure value to summarize the occupational exposure scenario and sensitive health endpoints into a single value. The calculated values may be used to support risk management efforts under TSCA section 6(a), 15 U.S.C. § 2605. TSCA requires risk evaluations to be conducted without consideration of costs and other non-risk factors, and thus the calculated occupational exposure values represents risk-only numbers. If risk management for DEHP, BBP, DIBP, DCHP, and/or DBP follows the finalized risk evaluations, EPA may consider costs and other non-risk factors, such as technological feasibility (*e.g.*, availability of validated detection methods), the availability of alternatives, and the potential for critical or essential uses. Any existing chemical exposure limits used for occupational safety risk management purposes could differ from the occupational exposure value presented in the final risk evaluations for DEHP, BBP, DIBP, DCHP, and DBP based on additional consideration of exposures and non-risk factors consistent with TSCA section 6(c).

Benchmark Dose Modeling

Summary: Two public commentors (SACC25-0137, SACC25-0155) commented on the use of meta-analysis software package Metafor to derive relative potency factors (RPFs) of phthalates for the cumulative risk assessment (CRA) of fetal testicular testosterone. They cited several issues with EPA's use of Metafor:

- Metafor was designed for meta-analysis of linear models, not non-linear models like dose-response models.
- Metafor does not properly account for potency heterogeneity in the phthalate dose-responses,

evidenced by background heterogeneity and a lack of "parallelism" in the slopes.

- Metafor does not use the current state-of-the-science methods for meta-analysis of dose-response, suggesting an approach that "incorporates potency heterogeneity, allows for evaluation of parallelism, and enables model averaging."
- NASEM's meta-analysis was "adequate for illustrative purposes" but "should not be used in a regulatory risk assessment."
- Use of BMD_{40} to calculate RPFs leads to both over- and underestimation of risks, due to the lack of parallelism at the benchmark response.
- Concerns over selection of DBP datasets for inclusion in the meta-analysis, citing "inconsistencies and concerns in data quality."

The comments further criticized both options presented by EPA for characterizing cumulative risk and calculating margin of exposure (MOE).

- For Option 1, they expressed concerns over scaling point of departures (PODs) for the phthalates into DBP equivalents using RPFs because of their above criticisms of the EPA models to derive the RPFs.
- For Option 2, they expressed concerns that "not all phthalate-specific PODs are derived from the same endpoint (*i.e.*, reduced fetal testicular testosterone), nor does this approach account for uncertainty in the POD when the NOEL (rather than the BMD) is used."

The comments suggested Bayesian Hierarchical Modeling (BHM) as an alternative method to derive RPFs that address the cited issues. The comment further compared EPA's approach to BHM in deriving RPFs and MOEs, noting how Option 1 and Option 2 lead to similar values under BHM but dissimilar values under EPA's approach.

The comments expressed concerns of using cumulative risk analysis (CRA) for risk management purposes. They stated, "it remains unclear how these multiple single-substance CRA assessments will inform risk management decisions or support refined assessments beyond the screening-level analyses…" and recommend that EPA develop "an integrated CRA risk communication tool to support informed risk management decisions."

EPA Response: EPA acknowledges the commentors' criticisms of its use of the Metafor package and of parallelism in the dose-response curves. Although parallel dose-response curves are not required for derivation of RPFs (U.S. EPA, 2023a), EPA agreed with commentors and SACC comments that additional analyses would be helpful to determine if phthalate curves are parallel to increase confidence in EPA's derived RPFs and therefore performed additional analyses in response, as described below.

To address the limited number of models available in Metafor, EPA conducted additional BMD analyses of fetal testicular testosterone (FTT) data using EPA's BMD Software (see Section 2.4.1.1 of the 2025 CRA TSD). Although the linear model did not frequently provide the best-fitting model amongst those used in BMDS, the linear model provided an adequate fit for reasonable BMD/BMDL estimates in many cases. The BMD estimates across response levels (5, 10, and 40%) were similar across modeling approaches for each phthalate, supporting EPA's use of Metafor for meta-analysis and BMD analysis of FTT.

In response to the suggestions from commentors and SACC, EPA considered Bayesian Hierarchical Modeling as an alternative method to estimate BMD and RPF values (see Section 2.4.1.4 of the CRA TSD). However, the RPF integration method (Ring et al., 2023) is not yet available as open-source software and therefore is not reasonably available to EPA at this time.

Regarding the two options for characterizing cumulative risk (termed Approaches 1 and 2), EPA considered the strengths, limitations, and uncertainties of each approach in Section 5 of the 2025 CRA TSD. EPA developed a framework of considerations for CRA Approach selection for each phthalate (see Section 5.4 of the 2025 CRA TSD). Following a consideration of the weight of scientific evidence for both approaches for each phthalate, EPA concluded that Approach 2 was the most scientifically defensible for DCHP, DIBP, BBP, and DEHP, while Approach 1 and 2 were mathematically identical for DBP, and Approach 1 was selected for DINP.

The selection of DBP datasets for the meta-analysis is discussed in Section 2.4.1.1 of the 2025 CRA TSD (as well as in Section 5.1 and Appendix B of the meta-analysis TSD), detailing the reasons why datasets were included or excluded from the analysis.

Risk management is outside of the scope of this response to comments document. Following finalization of the DEHP, DBP, DIBP, BBP, and DCHP risk evaluations, the agency will initiate risk management rulemaking to address the unreasonable risks of injury presented by BBP, DBP, DCHP, DEHP, and DIBP to human health and the environment. As part of any future rulemaking process, EPA will publish a notice of proposed rulemaking, seeking public comment before any final rule is issued.

Summary: A public comment (SACC23-0031) expressed mixed opinions on whether a BMD_5 or BMD_{40} is most appropriate.

Another public comment (BBP-0120, SACC25-0135, SACC25-0153, DIBP-0123) indicated that a benchmark response (BMR) of 5 percent change (BMR_5) is recommended by EPA's Benchmark Dose Technical Guidance for quantal endpoints in reproductive and developmental studies, but not for continuous endpoints such as fetal testicular testosterone and that the BMR_{10} is not sufficiently justified. The commenter (SACC25-0135, DIBP-0123) also states, "the use of a BMR of 40% deviation relative to control (BMR_{40}) is no longer supported given data from Howdeshell et al., 2015 [as] … a 25% reduction in fetal testosterone was associated with 17% incidence of reproductive tract malformations (RTM) in F1 males."

EPA Response: The EPA acknowledges the commentor's opinions about EPAs choice of BMR for fetal testicular testosterone data. EPA includes an Appendix in each of the non-cancer human health hazard TSD for each phthalate titled, "Considerations for Benchmark Response (BMR) Selection for Reduced Fetal Testicular Testosterone," where EPA describes its consideration of BMRs of 5, 10, and 40, as well as its rationale for its selection of a BMR of 5 percent for decreases in fetal testicular testosterone in each. EPA developed a weight of scientific evidence conclusion in Appendix E of each respective phthalate. Therein, EPA concludes that a BMR of 5 percent is the most appropriate and health protective response level for evaluating decreased fetal testicular testosterone. EPA points out that although some publications have suggested that reproductive tract malformations are observed in male tats when fetal testicular testosterone was reduced by about 40 percent, there is no scientific consensus on the biologically significant response level and no other authoritative or regulatory agencies have endorsed the 40 percent response level as biologically significant.

Additionally, as described in EPA's *Benchmark Dose Technical Guidance* (U.S. EPA, 2012), a BMR based on relative deviation is not strictly for quantal endpoints; it can also apply to continuous endpoints if statistically or biologically warranted, which is the case for most developmental and reproductive studies.

Summary: A public comment (DCHP-0122) expressed concern regarding "EPA's use of RPF based on BMD_{40} because $BMDL_5$ and $BMDL_{10}$ could not be estimated for BBP and DIBP", stating that it "potentially underestimates the risk calculations". The comment recommended that EPA explain how this may affect the cumulative risk calculations.

EPA Response: As described in Section 2.4 of the CRA TSD, EPA calculated candidate RPFs at the 5, 10, and 40% response levels. EPA's updated analysis (see Table 2-4 and Table 2-5 in the CRA TSD) for fetal testicular testosterone demonstrated that RPFs did not significantly vary across the range of BMRs (5%, 10%, and 40%). Therefore, EPA is confident that using the RPFs calculated from BMD_{40} estimates would not significantly affect the cumulative risk calculations, as compared to use of RPFs based on BMD_5 or BMD_{10} estimates.

Summary: A public comment (BBP-0120, SACC25-0135, SACC25-0153) stated "EPA's use of FTT [fetal testicular testosterone] production for its BMD modeling is supported as a suitable choice in selecting PODs." However, the same commenter later stated, "The consortia believe that EPA's inconsistent use of endpoints (AGD, RTM, and fetal testosterone) for deriving OEV or RPF values is not compliant with TSCA."

Another public comment (DCHP-0126) stated that "While the consortia support EPA's use of a relative potency factor (RPF) in its phthalate cumulative risk analysis, the consortia suggest that EPA ensure it is using a consistent endpoint (*e.g.,* fetal testosterone production) rather than calculating RPFs for individual phthalates using other endpoints (*e.g.,* anogenital distance). Doing so ensures that RPFs are derived consistently across all phthalates."

EPA Response: The selection of fetal testicular testosterone as the endpoint for BMD modeling to calculate RPF was well supported by public comments and the SACC. The endpoint was consistently used across chemicals to estimate RPFs. See EPA response to the 2025 SACC Report (U.S. EPA, 2025), Charge Question 5f for more discussion of the selection of fetal testicular testosterone. See Section 7.1 for discussion of OEVs.

Summary: A public comment (SACC25-0138) raised concerns with the fetal testicular testosterone (FTT) data used in the BMD modeling for DEHP does not report time of tissue sampling and states there is evidence of fluctuations in testosterone throughout the day in adult humans and rats, while acknowledging that "no studies identify this variable in males during gestation." The comment also states that FTT in rats fluctuates during gestational day (GD) 18 through 20, making it an unreliable endpoint. Finally, the comment states that DEHP exposures in the selected studies covered a range of gestational days and durations.

EPA Response: As stated in the comment, no studies were available to indicate time of sampling is impactful in evaluation of fetal testicular testosterone, indicating this is not a deficiency in the studies

used. Additionally, the reference provided in the discussion of fluctuating fetal testicular testosterone (FTT) during gestation (Tsukahara et al., 2009) does not describe changes in FTT levels during GS18 through 20, which is in fact the window for FTT sampling in the cited study. Therefore, EPA maintains that there is no clear rationale to discount the selection of studies that measured FTT from GS18 through 21 used in the BMD modeling for DEHP.

DEHP exposure consistently covered the sensitive window of exposure for male development (GD 16-18). Most of the studies were previously used in a similar meta-analysis by NASEM () and the only new study (Gray et al., 2021) had the same exposure as several studies previously included. Additionally, EPA conducted a sensitivity analysis to determine if the meta-analysis was sensitive to leaving out the results of individual studies (*e.g.*, the sole study that dosed from GD2-20) and did not see a significant impact on the meta-analysis (CRA TSD). Therefore, EPA maintains these studies are appropriate for the BMD analysis of DEHP.

Summary: A public comment (BBP-0120, SACC25-0135, SACC25-0153, DIBP-0123) disagreed with EPA's selection of the best fitting BMD model for DIBP based on Akaike information criterion (AIC) and recommended model averaging with ToxicR or Bayesian Benchmark Dose (BBMD) System.

EPA Response: EPA acknowledges the commentors criticisms (BBP-0120, SACC25-0135, SACC25-0153, DIBP-0123) but disagrees with the comment and offers several clarifications. EPA adhered to the *Benchmark Dose Technical Guidance* (U.S. EPA, 2012) for its BMD analyses. Contrary to what the commentor states, EPA did not select BMD models based solely upon AIC; multiple BMD variable outputs for continuous and quantal data were considered for model selection, which are summarized in Table 4-3 of the non-cancer human health hazard TSD. Per EPA BMD guidance, if BMDL estimates from individual models are comparable, the model with the lowest AIC is recommended to be used for calculating the BMDL for the POD.

Although the SACC was supportive of EPA's BMD model approaches and model interpretations for BBP and DIBP during the 2025 peer-review meeting, SACC still recommended EPA consider alternative approaches. Therefore, in response to the suggestions from public commentors as well as the SACC, EPA considered the use of BMDS models that use Bayesian fitting procedures and Bayesian model averaging for fetal testicular testosterone data, but ultimately did not apply them for the individual phthalate assessments. EPA did not apply Bayesian model averaging because this model is not yet available as open-source software for EPA's use in dose-response analysis of non-quantal endpoints.

Summary: A public comment (DCHP-0126) stated that "Use of BMR40 values is unsuitable in deriving OEVs because all of the BMR40 values exceed the lowest LOAEL. The use of these values incorrectly suggests DCHP as the most hazardous substance, when currently available data based on the NOAEL and EPA's POD selected for phthalate syndrome suggest that DEHP would be the lowest POD based on the current data set. The consortia question EPA's inconsistent use of endpoints (AGD and fetal testosterone) for deriving OEV or RPF values. We suggest that EPA reconsider the selected RPF values and be consistent and transparent with its determinations.

EPA Response: EPA agrees with the public commenter (DCHP-0126) that use of BMDLs derived at the 40% response level (*i.e.*, BMR of 40%) are unsuitable for deriving OEVs because these BMDL

estimates exceed the lowest LOAEL for each phthalate. EPA has used the BMD_{40} estimates to derive RPFs for use in the cumulative assessment. However, for the index chemical (DBP), the POD used in both the individual DBP risk evaluation and the cumulative assessment is based on a $BMDL_5$ of 9 mg/kg-day for reduced fetal testicular testosterone.

EPA has not proposed to derive occupational exposure values based on BMDL estimates at the 40% response level. As described in the appendix titled "Considerations for Benchmark Response (BMR) Selection for Reduced Fetal Testicular Testosterone", which is included in the non-cancer human health hazard assessments for DEHP, DBP, DIBP, BBP, and DCHP, EPA considers a BMR of 5% to be the most appropriate and health protective response level for evaluating decreased fetal testicular testosterone when sufficient dose-response data are available to support modeling of fetal testicular testosterone in the low-end range of the dose-response curve. In the individual phthalate risk evaluations, EPA calculated occupational exposure values for each phthalate using the acute/intermediate/chronic POD selected for characterizing risk for each phthalate. For DEHP, DBP, DIBP, BBP, and DCHP, the POD was based on effects on the developing male reproductive system consistent with phthalate syndrome.

Summary: A public comment (SACC23-0040) addressed several points regarding the "Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substances Control Act" (Draft CRA Approach). Co-authors provided related comments (SACC23-0058, SACC23-0066). Regarding the Draft CRA Approach, the commentors generally agreed with EPA's approach to implementing CRA for phthlate syndrome. They offered several comments and suggestions:

1. They advised that, in addition to the six phthalates presented, EPA should include other chemical and non-chemical stressors in the CRA. For such other stressors, toxicological similarity evidence could be based on similar key cellular- and organ-level events and shared postnatal outcomes. They agreed with the EPA using DINP as being toxicologically similar to the other phthalates and likely contributor to cumulative phthalates risk, but they suggested including other phthalates and pesticides or adjustment factors in the risk evaluation. They disagreed with EPA's definition of non-chemical stressors, suggesting a broader definition like the ones used by the CDC or WHO.
2. They disagreed with EPA that dose addition is a "conservative" assumption and argued that reported evidence shows dose addition underestimates risk.
3. They asked EPA to conduct CRA for other health endpoints, not just the "most sensitive effects," and model multiple outcomes and approaches to inform the RPFs, even if not all phthalates are associated. Most sensitive effects may preclude better suited endpoints for RPFs, so EPA should consider factors other than sensitivity. Furthermore, EPA should consider multiple outcome measures in "addressing phthalate syndrome as a whole" instead of a single outcome analysis and use statistical techniques that combine data from multiple outcomes to derive RPFs.
4. Between relative potency factor (RPF) and hazard index (HI) for quantifying risk of exposures to multiple phthalates, they suggested discussing the advantages and disadvantages of both. For HI, they argued that the threshold of HI < 1 incorporates a flawed assumption of a "safe" level of exposure.
5. For RPF, they agreed RPF is the more scientifically appropriate method. However, they asked EPA to incorporate more stressors and approaches into the RPF development and to explore

how uncertainty factors affect RPFs. Furthermore, EPA should consider how RPF variability affect CRA risk estimates and decisions regarding unreasonable phthalate risk, and EPA should analyze all datasets and endpoints and compare RPFs, even if effects are not statistically significant for all phthalates, rather than pre-selecting the endpoint(s) for RPF development.

6. Regarding EPA's six options for deriving RPFs, they agreed with "combining data across outcomes for the postnatal effects" but asked EPA to clearly describe statistical methods for combining RPFs. They asked EPA to consider integration of all datasets into a single model construct, such as nested or multi-level models or structural equation models.
7. They asked for clearer descriptions of aggregate exposure. EPA should discuss individuals with combinations of general, consumer, and occupational exposures. They asked EPA to clearly define who are considered fenceline communities, how model considers aggregate exposure within and across fenceline communities, and consider which data sources contribute to identification of fenceline exposures. They recommended EPA broadened its consideration of PESS, evaluate the potential for individual exposure to multiple phthalate-containing consumer products, and develop understanding of the uses and limitations of NHANES data.
8. They asked EPA to complete both scenario-based and reverse dosimetric approaches to estimate cumulative phthalate exposures. They recommended: 1) specify the assumptions for each approach and how to treat data gaps, 2) define "reasonable combinations" when estimating exposures, 3) identify where approaches could underestimate exposures and how to overcome this, 4) ensure that parameters for reverse dosimetry account for human variability, 5) address combinations of exposures, and 6) incorporate non-attributable and non-TSCA sources.
9. They argued that margin of exposure (MOE) approach for risk characterization is not consistent with "best available science" as it does not estimate the proportion of exposed population. They suggested the approach used by the WHO: combine outputs with index chemical equivalent exposures to estimate exposed population.
10. They advised EPA to use a validated systematic review method for CRA and consider risks to PESS. They argued that several recommendations from NASEM and SACC for improving the TSCA Method have yet to be addressed.

EPA Response: Many of the comments made in (SACC23-0040, SACC23-0058, SACC23-0066) raised questions in 2023 that have been addressed in the draft risk evaluations of DEHP, DBP, DIBP, DCHP, and BBP, SACC peer review, and in the final risk evaluations of DEHP, DBP, DIBP, DCHP, and BBP. In particular, what information would be used to inform cumulative risk assessment (CRA) and how CRA would be applied to inform the phthalates risk evaluations has been presented and discussed as advice from the SACC and the public has been considered throughout the phthalates risk evaluation effort.

- Response to Point 1: EPA focused its phthalate CRA on the 6 toxicologically similar phthalates (DEHP, DBP, DIBP, BBP, DIBP, DINP) that are currently undergoing risk evaluations under TSCA. Other chemicals not prioritized for risk evaluation under TSCA were not considered for inclusion in the CRA. As discussed in EPA's draft proposed approach for CRA of phthalates under TSCA (U.S. EPA, 2023) and in the 2025 CRA TSD (U.S. EPA, 2025) few scientific methods have been developed that allow for a quantitative analysis of cumulative risk from combined exposure to chemical and non-chemical stressors. EPA continues to actively work to strengthen the scientific underpinning for assessing cumulative impacts, including impacts

from non-chemical stressors. Until Agency-wide guidance and established methodologies have been developed, EPA does not expect to quantitatively evaluate non-chemical stressors under TSCA, which have not been carefully defined for applicability to EPA laws and policies.

- Response to Point 2: EPA disagrees with the public commenter. EPA states that dose addition is more conservative and health protective than response addition, which is a consistent with EPA's chemical mixtures risk assessment guidance (U.S. EPA, 2000; U.S. EPA, 2023). Further, as discussed in Section 4.3 of the draft proposed approach for CRA of phthalates under TSCA (U.S. EPA, 2023), mixtures studies of phthalates clearly demonstrate that dose-addition models accurately predict the observed effects following exposures to mixtures of phthalates in rats.
- Response to Point 3: EPA acknowledges that the public commenter and SACC during the 2023 pee-review meeting recommended that EPA consider adding a second endpoint to the cumulative risk assessment. Specifically, SACC recommended including liver toxicity, developmental neurotoxicity, or female reproductive effects. While EPA acknowledges that there are varying amounts of data demonstrating that certain phthalates can cause these effects, EPA did not consider these effects as the basis for a CRA for several reasons. First, although DEHP, BBP, DBP, DCHP, DIBP, DINP, and DIDP have all been shown to cause liver toxicity, most of the observed liver effects in experimental animal models are mechanistically linked to PPARα activation, which can vary between species raising questions about human relevance. Additionally, the non-cancer POD based on phthalate syndrome-related effects is a more sensitive outcome than liver toxicity for most phthalates (with DINP and DIDP being exceptions). Further, there are limited data demonstrating female reproductive effects or developmental neurotoxicity for DCHP and DIBP, while data for other phthalates varies in quality and quantity such that definitive conclusions about exposure-response relationships cannot be established. Therefore, EPA did not consider liver toxicity, developmental neurotoxicity, or female reproductive effects further as the basis for a CRA. However, these effects are discussed further, as relevant, in the cancer human health hazard assessment of phthalates (U.S. EPA, 2025a) and each individual non-cancer human health hazard assessments for DEHP (U.S. EPA, 2025x), DBP (U.S. EPA, 2025v), DIBP (U.S. EPA, 2025y), BBP (U.S. EPA, 2025u), DCHP (U.S. EPA, 2025w), DINP (U.S. EPA, 2025z), and DIDP (U.S. EPA, 2024d).
- Response to Points 4-5: As described in the 2025 CRA TSD, EPA developed relative potency factors (RPFs) for phthalates based on decreased fetal testicular testosterone production. EPA acknowledges that the hazard index (HI) is another approach that can be used to characterize cumulative risk; however, as discussed in EPA's mixtures and cumulative risk assessment guidance (U.S. EPA, 2000; U.S. EPA, 2023), the RPF approach is a more refined approach for assessing cumulative risk. EPA selected a total uncertainty factor of 30 (intraspecies UF of 3 and interspecies UF of 10) for DEHP, DBP, DIBP, BBP, and DCHP (see Section 4 of the non-cancer human health hazard TSDs for DEHP, DBP, DIBP, BBP,DCHP). Similarly, for the cumulative assessment, a total uncertainty factor of 30 was also selected. Since the same uncertainty factors were selected for each phthalate and the CRA, it was unnecessary to explore how uncertainty factors affect RPFs. Additionally, SACC supported EPA's approach for deriving phthalate RPFs (U.S. EPA, 2025). The commenter is directed to EPA's response to the 2025 SACC report for further details pertaining to how EPA addressed recommendations provided by SACC during the 2025 peer-review meeting.
- Response to Point 6: In the 2023 draft phthalate CRA proposal, EPA considered six options for deriving RPFs based on both gestational and postnatal effects consistent with the development

of phthalate syndrome. The strengths, limitations, and uncertainties associated with the reasonably available data for each outcome considered for deriving RPFs is provided in Section 1.3 of the 2025 CRA TSD. For postnatal outcomes (*i.e.*, decreased male anogenital distance, male nipple retention, seminiferous tubule atrophy, incidence of hypospadias), EPA considered there to be too many limitations and uncertainties to support RPF derivation. In contrast, a number of factors increased EPA's confidence in using the fetal testicular testosterone data set to derive RPFs. Therefore, EPA calculated RPFs using decreased fetal testicular testosterone data using Option 2 (meta-analysis and BMD analysis of combined data sets). The public commenter is directed to Section 1.3 of the 2025 CRA TSD for further details.

- Response to Point 7: EPA responded in detail to public comments pertaining to aggregate exposure assessment in Section 4.4 of this document.
- Response to Point 8: As discussed in the final CRA TSD, EPA utilized CDC's NHANES urinary biomonitoring data and reverse dosimetry to estimate non-attributable cumulative exposure to DEHP, DBP, BBP, DIBP, and DINP. EPA considered scenario-based approaches for estimating cumulative exposure, however, EPA determined that reasonably available data do not support doing so due to limitations and uncertainties with available data. For example, due to the wide range of cumulative exposure scenarios that may exist in phthalate-containing workplaces, it was not possible to provide a robust quantitative assessment of cumulative risk for workers who may be exposed to multiple phthalates based on reasonably available data. EPA did not have data on specific use patterns, facility campaigns, or quantitative estimates of co-exposure in an occupational setting necessary for development of probabilistic exposure models. The public commenter is directed to Section 3 of the 2025 CRA TSD for further discussion of the limitations and uncertainties associated with scenario-based approaches considered by EPA.
- Response to Point 9: EPA has used MOE approaches for all TSCA risk evaluations. As for a probabilistic approach specifically, EPA is in the early stages of research associated with developing probabilistic methods and guidance for use in human health hazard assessment. Until this research is matured and completed, EPA will continue to use the approaches described in existing EPA guidance documents for using default values (U.S. EPA, 2002) and for developing refined values (*e.g.*, 2014 Guidance for Applying Quantitative Data to Develop Data-Derived Extrapolation Factors for Interspecies and Intraspecies Extrapolation; (U.S. EPA, 2014). There is no current policy for determining appropriate regulatory thresholds for results of a probabilistic analysis.
- Response to Point 10: The public commenter is directed to EPA's systematic review protocol for DEHP, DBP, DIBP, BBP, and DCHP for a description of how EPA identified, evaluated, and integrated literature in the final risk evaluation for each phthalate. More detailed responses to comments on EPA's systematic review processes are also provided in Section 9 of this document. Additionally, EPA has considered PESS throughout all aspects of the final phthalate risk evaluations for DEHP, DBP, DIBP, DCHP, and BBP, including in the hazard and dose-response assessment, as well as the exposure assessments. EPA has provided more detailed responses to public comments regarding PESS in Section 6.5 of this document.

Summary: A commenter (DIBP-0131) raises concerns about EPA's derivation of the point of departure (POD) for DIBP, stating that EPA used a human equivalent dose (HED) of 5.7 mg/kg-day based on a $BMDL_5$ from fetal testicular testosterone (FTT) in Gray et al., 2021, modeled with EPA BMD software v3.3.2 rather than the Metafor tool (v4.6.0) used for other phthalate evaluations. They assert the inconsistency in BMD modeling tools affects occupational exposure values (OEVs) that rely on shared technical support documents (TSDs). The commenter further notes that EPA omitted relevant data in its BMD modeling, including portions of Furr et al., 2014 (blocks 2, 14, and 30). They highlight a lowest NOAEL of 100 mg/kg-bw/day for FTT across Howdeshell et al., 2008; Gray et al., 2021; and Hannas et al., 2011, and a lowest LOAEL of 125 mg/kg-bw/day for testicular pathology (Saillenfait et al., 2008), with the lowest LOAEL for FTT at 200 mg/kg-bw/day (Furr et al., 2014), and request that EPA either justify the exclusions or include the omitted data.

A commenter (DIBP-0131) proposes that POD derivation for the continuous FTT endpoint should follow EPA's Benchmark Dose Technical Guidance by using a BMDL1SD (a change in the mean equal to one control standard deviation) with model averaging, implemented in ToxicR and/or Bayesian BBMD, consistent with advice from the SACC. They recommend a hierarchy for POD selection to ensure consistency across phthalates: first, a model-averaged BMDL1SD; if unavailable or not supported (*e.g.*, below the NOAEL or above the LOAEL), then the NOAEL; and finally the LOAEL. Using this approach, the commentor performs their own BMD modeling and analysis effort and report the parameters and methodologies they utilize for this effort. Based on their modeling, the commenter proposes a POD of 113.4 mg/kg-bw/day, the BMDL1SD from Hannas et al., 2011. They state this POD aligns with their hierarchy, falls between the NOAEL (100 mg/kg-bw/day) and LOAELs (125 mg/kg-bw/day for testicular pathology; 200 mg/kg-bw/day for FTT), and is protective of the lowest LOAELs identified in the available literature.

EPA Response: As discussed in Section 4 of the human health hazard TSD for DIBP, EPA utilized two approaches for BMD modeling of FTT data for DIBP. First, EPA used Metafor to BMD modeled combined fetal testicular testosterone data from 3 studies (Gray et al., 2021; Hannas et al., 2011; Howdeshell et al., 2008). However, no BMD_5/$BMDL_5$ could be derived using this approach. Since no $BMDL_5$ could be derived through the updated meta-analysis and BMD modeling analysis, EPA modeled individual fetal testicular testosterone data from the three studies included in the updated meta-analysis using EPA's BMD Software (BMDS version 3.3.2). This analysis using EPA's BMDS included the full suite of standard continuous models (Exponential, Hill, Polynomial, Power, Linear), compared to the meta-analysis that only included the linear and linear-quadratic models. For the final human health hazard assessment of DIBP, EPA selected the $BMDL_5$ of 24 mg/kg-day based on reduced fetal testicular testosterone from the study by Gray et al. (2021). Notably, the SACC supported EPA's selection of a $BMDL_5$ of 24 mg/kg-day from Gray et al. (2021) for use as the basis for the POD, given the lack of studies evaluating doses of DIBP less than 100 mg/kg-day (U.S. EPA, 2025o). EPA considered the POD derived from the BMD analysis of data in this study to have the least uncertainty and highest confidence upon examination of the weight of scientific evidence. This POD is more sensitive than the lowest NOAEL of 100 mg/kg-day based on fetal testicular testosterone data from 3 studies (Gray et al., 2021; Hannas et al., 2011; Howdeshell et al., 2008) and LOAEL of 125 mg/kg-day based on increased incidence of testicular pathology (Saillenfait et al., 2008), which are likely under-protective due to the limited number of studies and lack of testing at doses lower than 100 mg/kg-day.

EPA notes that the fetal testosterone data reported by Furr et al. (2014) is reported as three separate studies, each which evaluated a single dose level of DIBP. The block 2 study included doses of 0 and an unknown dose of DIBP (administered dose in mg/kg-day does not appear to be reported in the main

publication or supplemental files), while block 14 included doses of 0 and 500 mg/kg-day DIBP, and Block 30 included doses of 0 and 200 mg/kg-day DIBP. EPA discusses this data in the DIBP human health hazard TSD, however, this data was not BMD modelled using EPA's BMDS because each study only evaluated a single dose level.

For BMD modeling of continuous data, such as reduced fetal testicular testosterone, EPA's BMD Technical Guidance (U.S. EPA , 2012) recommends a BMR of 1 control SD "always be presented for comparison purposes," however, EPA's BMD technical guidance also states that "The ideal is to have a biological basis for the BMR for continuous data." For reduced fetal testicular testosterone, EPA evaluated BMRs of 5, 10, and 40% based on biological and statistical considerations, as outlined in the appendix titled "Considerations for Benchmark Response (BMR) Selection for Reduced Fetal Testicular Testosterone," which is included in the non-cancer human health hazard assessments for DEHP, DBP, DIBP, DCHP, and BBP. As discussed in this appendix in each non-cancer TSD, EPA has reached the conclusion that a BMR of 5 percent is the most appropriate and health protective response level for evaluating decreased fetal testicular testosterone when sufficient dose-response data are available to support modeling of fetal testicular testosterone in the low-end range of the dose-response curve. As such EPA considers $BMDL_5$ estimates for reduced fetal testicular testosterone appropriate for determining the point of departure (POD) for each high-priority phthalate.

5.4 Other Comments on Human Hazard

Summary: A public comment (DCHP-0122,) recommended that "EPA should clarify whether its dose selection estimates are only protective of male reproductive development, or if they also protect other toxicity endpoints". This comment, as well as additional comments, further specified that EPA should consider whether the cumulative and individual phthalate risk assessments are protective of neurodevelopmental endpoints in animals and humans.

EPA Response: EPA acknowledges the consideration of multiple health domains during the risk assessment of high-priority phthalates under TSCA (DCHP-0122). EPA considers its risk assessment and human equivalent (HED) dose for point of departure (POD) selection for phthalates to be health protective of developing male reproductive effects, pregnant women, and adults of any age (*e.g.*, adult males and females). The selected PODs for each phthalate are based upon the most sensitive health effects consistent with phthalate syndrome for DEHP, BBP, DBP, DCHP, DIBP, and DINP (but not DIDP due to the determination DIDP does not induce effects consistent with phthalate syndrome). In the individual human health hazard TSDs for each phthalate, additional health effects (effects other than effects on the developing male reproductive system) were discussed, but were not found to be more sensitive than phthalate syndrome-related outcomes. EPA is confident of the health protectiveness of the selected PODs for individual phthalates and in the CRA analysis for selecting the most conservative health effect level and across acute, intermediate, and chronic exposure duration scenarios.

Summary: Several public commenters (DCHP-0127, DEHP-0138) stated, "EPA's non-cancer hazard identification and dose-response assessment[s] for…" DBP, DCHP, and DEHP are "…not consistent with the best available science." The commenters stated that "EPA improperly excluded ['all' for DBP and DEHP commenters] human epidemiology studies from dose-response assessment." The commenters also stated, "EPA failed to make appropriate use of benchmark dose modeling to specify a non-cancer point of departure for risk characterization."

The commenter (DCHP-0127) continues, "In the DCHP hazard assessment, EPA inappropriately and without valid justification decided to not even assess study quality for 22 out of the 24 identified epidemiological studies, thereby eliminating them from further consideration." The comment then provides an example in comments DCHP-0127 and SACC25-0145 : "For example, the use of spot-urine samples is a limitation that is expected to result in some degree of exposure misclassification, but to the extent this occurs, it is likely to result in imprecision in effect estimates. In general, the uncertainties in exposure characterization may result in exposure misclassification, but that does not mean the studies are not useful or informative and potentially strong candidates for determination of the point of departure (POD)."

The commenters (DCHP-0127, DEHP-0138, SACC25-0145) concluded, "By excluding relevant epidemiology studies of…" DBP, DCHP, and DEHP "…from dose-response analysis, EPA has violated TSCA's requirement to use the best available science. EPA cannot broadly exclude epidemiologic studies from dose-response assessment in the…" DBP, DCHP, and DEHP "…Draft Risk Evaluation[s] and must consider each relevant study on an individual basis as a candidate for POD derivation."

EPA Response: EPA disagrees and believes that the non-cancer dose-response assessments for DEHP, DBP, BBP, DIBP, and DCHP are consistent with the best available science. As discussed in Section 5.5.2 of the systematic review protocols for DEHP, DBP, DIBP, BBP, and DCHP, PECO-relevant epidemiologic studies were evaluated for data quality and EPA considered using human epidemiology studies quantitatively as part of the risk evaluations for each of these five phthalates. However, due to limitations and uncertainties in the phthalate database of epidemiology studies, primarily related to exposure characterization, EPA does not consider it appropriate to use epidemiology studies of DEHP, DBP, BBP, DIBP, and DCHP quantitatively to characterize risk (see Section 1.1 of the non-cancer human health hazard assessments of DEHP, DBP, DIBP, BBP, and DCHP for further details). Primary sources of uncertainty include the source(s) of exposure; timing of exposure assessment that may not be reflective of exposure during outcome measurements; measured urinary metabolites may represent exposure to more than one parent phthalate; and use of spot-urine samples, which due to rapid elimination kinetics may not be representative of average urinary concentrations that are collected over a longer term or calculated using pooled samples. Additional uncertainty results from co-exposure to mixtures of multiple phthalates that may confound results for the majority of epidemiologic studies, which examine one phthalate and one exposure period at a time such that they are treated as if they occur in isolation. Limitations and uncertainties associated with human epidemiologic studies that preclude their use in quantitative dose-response assessment are discussed in Section 1.1 of the Human Health Hazard Assessment of DINP. Instead, EPA considered epidemiologic evidence for DEHP, DBP, BBP, DIBP, and DCHP qualitatively as part of hazard identification and characterization steps of the hazard assessment. EPA's use of human epidemiologic studies of DEHP, DBP, BBP, DIBP, and DCHP is discussed further in Section 1.1 of each non-cancer human health hazard TSD for DEHP, DBP, BBP, DIBP, and DCHP. Notably, EPA's decision to use epidemiologic studies of DEHP, DBP, BBP, DIBP, and DCHP qualitatively is consistent with existing assessments of by Health Canada, U.S. CPSC, ECHA, EFSA, Australia NICNAS, as well as ATSDR for DEHP.

Summary: A public comment (SACC23-0052) on the *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substances Control Act* encouraged EPA to consider health effects other than male reproductive toxicity as part of the cumulative risk assessment and each individual phthalate risk

evaluation.

EPA Response: As discussed in each of the individual final non-cancer human health hazard assessments for DEHP, DBP, DIBP, BBP, and DCHP, EPA considered reasonably available information for each phthalate pertaining to health effects other than male reproductive toxicity as part of each individual phthalate risk evaluation, including effects on female development/reproduction, immune effects, liver effects, kidney effects, and neurological effects, as appropriate.

As discussed in Section 1.2 of the TSD for the cumulative risk analysis of DEHP, DBP, BBP, DIBP, DCHP, and DINP under TSCA, EPA considered other health effects recommended by SACC during the 2023 peer-review meeting, including liver toxicity, developmental neurotoxicity, or female reproductive effects. The amount and quality of data demonstrating these effects varies across phthalates, and EPA did not consider these effects as the basis for a CRA. Additionally, although DEHP, BBP, DBP, DCHP, DIBP, DINP, and DIDP have all been shown to cause liver toxicity, most of the observed liver effects in experimental animal models are mechanistically linked to peroxisome proliferator-activated receptor alpha (PPARα) activation, which can vary between species and therefore the human relevance is questionable. Additionally, the non-cancer POD based on phthalate syndrome-related effects is a more sensitive outcome than liver toxicity for most phthalates (with the chronic non-cancer POD for DINP being one exception). Further, there are limited data demonstrating female reproductive effects or developmental neurotoxicity for DCHP and DIBP and reasonably available data for other phthalates vary in quality and quantity such that definitive conclusions about exposure-response relationships cannot be established. Therefore, EPA did not consider liver toxicity, developmental neurotoxicity, or female reproductive effects further as the basis for a CRA.

6 HUMAN HEALTH RISK CHARACTERIZATIONS

Comments associated with this topic are summarized in the subsections below.

6.1 Occupational Risk Characterization

Summary: A public commenter (DCHP-0 124) provided a table describing their uses of DCHP and other phthalates (BBP, DBP, and DEHP) in manufacturing operations, "so that EPA can confirm or update its assumptions about exposures and uses."

EPA Response: EPA thanks the commenter for providing the information. The information was evaluated but no changes were made as a result of the information in the risk evaluation.

Summary: A public comment (BBP-0120, SACC25-0135, SACC25-0153, DIBP-0123) stated that the PODs used to derive occupational exposure values (OEVs) were not clear or consistent across phthalates. "The consortia believe that EPA's inconsistent use of endpoints (AGD, RTM, and fetal testosterone) for deriving OEV… is not compliant with TSCA."

EPA Response: The PODs derived for the individual chemicals as presented in each risk evaluation were used to determine OEVs and the values were clearly stated in equations presented for each calculation. Public commenters are directed to the appendix titled "Occupational Exposure Value Derivation" in the risk evaluations for DEHP, DBP, DIBP, BBP, and DCHP for further details.

Summary: A public commenter (SACC25-0138) recommended a different POD based on a BMD_{1SD} be used in the OEV calculation for DEHP, consistent with a proposed hierarchy of POD selection by that commenter.

EPA Response: EPA disagrees with the public commenter (SACC25-0138). For DEHP, EPA selected a POD of 4.8 mg/kg-day based on a NOAEL for increased incidence of reproductive tract malformations at the LOAEL of 14 mg/kg-day. This POD was used for risk characterization in the DEHP risk evaluation and is the basis of the DEHP OEV. EPA does not consider the $BMDL_{1SD}$ of 17.96 mg/kg-day based on reduced fetal testicular testosterone in the study by Saillenfait et al. (2013), which was derived by the public commenter, to be health protective or appropriate for use in risk assessment or for deriving an OEV. This is demonstrated by the fact that the suggested $BMDL_{1SD}$ of 17.96 mg/kg-day is greater than the LOAEL of 14 mg/kg-day for increased incidence of male reproductive tract malformations, clearly indicating that the suggested $BMDL_{1SD}$ is not health protective. Further, as discussed Appendix E of the non-cancer human health hazard TSD for DEHP, EPA selected a BMR of 5% for BMD modeling of decreased fetal testicular testosterone based on biological and statistical considerations. Although the public commenter is correct that EPA's BMD Technical Guidance (U.S. EPA , 2012) recommends a BMR of 1 control SD "always be presented for comparison purposes", BMD technical guidance also states that "The ideal is to have a biological basis for the BMR for continuous data."

Summary: A public comment (DBP-0127) stated "the statute [Section 6 of TSCA] does not require that the only reasonable risk is no risk". In the context of Appendix F of the DBP draft risk evaluation, (DBP-0127) stated "EPA has described the threshold for the OEV as 'Any [sic] appreciable risk of adverse toxicological outcomes' in this and several other risk evaluations and that "EPA believes that 'any appreciable risk of adverse toxicological outcomes' represents an *unreasonable risk of injury to health* under Section 6 of TSCA." In addition, the commenter recommended that EPA develop a clear and transparent process for developing OEVs and existing chemical exposure limits (ECELs) under TSCA that incorporates peer review of the approach and the ultimate outcome (value). The commenter stated that EPA has not sought public dialogue with stakeholders, experts, or the regulated community on OEV and ECEL derivations.

EPA Response: EPA has established a process to provide transparency on how an occupational exposure value (OEV) is calculated. For chemicals undergoing TSCA section 6 review, EPA currently publishes an OEV as part of the risk evaluation. The OEV is derived from the available chemical-specific information including, but not limited to hazard, relevant occupational epidemiology, and exposure information The chemical-specific information that makes up the OEV appendix (*e.g.*, hazard endpoint and uncertainty factors) are available for both peer review (as part of the SACC peer review) and public comment for a given chemical. Beyond TSCA peer review, EPA encourages public comment on the OEV, proposed ECEL and any other information pertaining to a chemical's occupational risk profile including hazard, occupational exposure, occupational monitoring limitations, and technological feasibility of workplace controls. The OEV takes in consideration uncertainty factors used in the risk evaluation, that reflect the uncertainties in the hazard values. EPA does not use the OEV threshold as a sole basis for decision-making. As mentioned in Appendix F, "if risk management for DBP follows the finalized risk evaluation, EPA may consider costs and other nonrisk factors." when deriving a regulatory exposure limit (ECEL). If EPA determines unreasonable risk, a chemical-specific risk management rule may follow that includes a proposed or final ECEL value. The ECEL value is intended to address the unreasonable risk, and while it is informed by the OEV appendix, it also includes consideration of cost and other non-risk factors, such as analytical and technological feasibility. EPA concurs with the commenter about the value of meaningful engagement and values public comment during the various stages of the Section 6 process – prioritization, risk evaluation, and/or risk management phases.

Summary: A public comment (DBP-0131) encouraged EPA to consider Limits of Quantitation (LOQ values) relative to proposed OEVs. The commenter stated that "if the current methods such as OSHA 104 yield too many sampling results below the LOQ and the OEV is also low, then the impact on using conservative assumptions for left censored data for the exposure assumptions justifies the need for detailed discussion."

EPA Response: As discussed the risk evaluations of DEHP, DBP, BBP, DIBP, and DCHP, EPA derived OEVs for each phthalate based on the selected acute POD based on effects on the developing male reproductive system. The derived OEVs for DEHP, DBP, DIBP, BBP, and DCHP are 0.3, 0.61, 1.5, 3.1, and 0.63 mg/m^3 (8-hour TWA). These calculated values represent the exposure concentration below which exposed workers and ONUs are not expected to exhibit any appreciable risk of adverse toxicological outcomes, accounting for PESS. It is derived based on the most sensitive human health effect (*i.e.*, decreased fetal testicular testosterone and/or other phthalate syndrome related effects) and exposure duration (*i.e.*, acute) relative to benchmarks and a standard occupational scenario assumption of an 8-hour workday. TSCA requires risk evaluations to be conducted without consideration of costs and other non-risk factors, and thus this occupational exposure value represents a risk-only number. If

risk management for BBP follows the finalized risk evaluation, EPA may consider costs and other non-risk factors, such as technological feasibility and the LOQ. Any existing chemical exposure limit 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).

6.2 Consumer and Indoor Risk Characterization

Summary: A public commenter (SACC25-0145) stated that EPA improperly disregarded high-intensity exposure and risk estimates in its unreasonable risk determination for consumer exposures to DBP in paints and toys by revisiting any exposure estimates resulting in margins of exposure (MOEs) less than 30, and therefore below the "benchmark MOE" used by EPA in finding unreasonable risks. The commenter goes on to say that EPA found that consumer paints and coatings contribute to the unreasonable risk of DBP based on dermal exposure, but it disregarded potential high risks to infants and toddlers from inhalation exposures based on assertions about infrequent use of metal coating products, and that aggregate exposure should be considered for children's toys.

EPA Response: EPA recalculated all liquid products dermal exposure doses and MOEs from draft to final. The new approach uses a metabolically active human skin dermal absorption study Beydon et al., (2010) rather than guinea pig skin dermal absorption study (Doan et al., 2010) used in the draft. The new liquid products dermal doses all resulted in MOEs above the benchmark of 30. The liquid products dermal absorption approach used in the draft had uncertainties due to the differences between animal and human skin absorption. The liquid products dermal absorption approach used in the final assessment using human skin dermal absorption studies provides a stronger confidence that the results are more representative of human dermal exposures to paints and other products and articles.

With respect to the determination of risk or no risk for scenarios with MOEs below the benchmark, EPA relied on Agency precedent when determining how to apply benchmarks. In the draft risk evaluation for DBP, and in previous risk evaluations, EPA explained that the benchmark does not represent a bright line for unreasonable risk determination. EPA has discretion to make unreasonable risk determinations based on other considerations as appropriate. Other considerations include the endpoint under consideration, the reversibility of effect, exposure-related considerations (*e.g.*, duration, magnitude, or frequency of exposure, or population exposed), and the confidence in the information used to inform the hazard and exposure values. These considerations and their impacts on EPA's use of calculated risk estimates and benchmarks are explained in the unreasonable risk determination for DBP and are consistent with past risk evaluations where EPA considered MOEs below the benchmark.

Summary: A public commenter (SACC25-0151) equated EPA's tiered screening approaches and refinement of exposure estimates for consumer exposure to be equivalent to manipulating the process to avoid finding unreasonable risk from high-end exposure. The commenter provided the example that "infants and toddlers exposed to DEHP in furniture—even though the high-intensity, aggregate acute exposure scenario produced an MOE of 30, equal to EPA's benchmark MOE, EPA excluded this scenario in its final risk determination. EPA's rationale, that infants and toddlers are unlikely to have prolonged contact with synthetic leather furniture, is unsupported and overlooks common real-world behaviors such as napping, screen time, or reading while seated. This selective dismissal of high-risk

exposure estimates *after* characterizing risk goes against best practices and contradicts TSCA's mandate to rely on the best available science and reasonably available data."

EPA Response: EPA disagrees with the commenter about inappropriate manipulation and selection of exposure estimates to avoid unreasonable risk determination. The screening approach used inputs for use patterns that were applied across age groups and leaned on the conservative side, then revisited the application of the inputs when MOEs were below the benchmark to corroborate appropriateness and representativeness for each age group. If the high intensity use scenario resulted in MOEs above the benchmark, then any combination of lower intensity use patterns would result in less exposures. For example, when considering exposures to synthetic leather furniture, EPA used a scenario that would result in the highest exposures. At first, EPA considered all age groups laying on the furniture for 8 hours, knowing that such behavior may apply to some age groups but for other age groups would be considered a misuse or an overestimation. Infants laying on furniture for 8 hours is considered a misuse per furniture warning labels and professional health care recommendations. Toddlers may have naps, screen time, and reading time on furniture, but 8 hours would also be considered an extreme assumption. As such, when MOEs for any scenario were below the benchmark, EPA revisited inputs and assumptions applicability and representativeness for each age group.

Summary: A public commenter (SACC25-0132) stated that EPA should not dismiss the high-intensity scenarios for individuals using air-beds within the DEHP risk evaluation. The commenter stated that 14 hours of exposure without sheets is a reasonably foreseen exposure scenario for susceptible populations, such as infants, toddlers, and pregnant people. The "refinement" of a sheet and 8-hour exposure added to the scenario when risk estimates were below benchmark was not consistent with other assumptions in the risk evaluation and deemed arbitrary, capricious and contrary to TSCA by the commenter.

EPA Response: EPA disagrees about inappropriate selective dismissal of high intensity use exposure estimates. Any removal of high intensity use exposures were discussed in the technical support document (see Section 2) if MOEs were above the benchmark and in the risk characterization section in the risk evaluation document if the MOEs were below the benchmark. The screening approach used inputs for use patterns that were applied across age groups and leaned on the conservative side, then revisited the application of such inputs when MOEs were below the benchmark to corroborate appropriate application and representativeness of inputs. If the high intensity use scenario resulted in MOEs above the benchmark, then any combination of lower intensity use patterns would result in less exposures and would not warrant a refinement or recalculation.

For DEHP air beds dermal exposures, the approaches moved from a flux-limited to a concentration-dependent approach. The refinement is expected to improve the accuracy and relevance of the exposure estimates. In the high intensity use exposure scenarios, EPA used inputs that would result in the highest exposures. At first, EPA considered all age groups laying on the air bed for 14 hours with and without a bedsheet cover. Some use pattern inputs may apply to some age groups but for other age groups would be considered a misuse or an overestimation. For example, 14 hours of sleep for adults is likely an overestimation for the average adult, however the input was used because MOEs for either screening or refined approach were above the benchmark of 30 for adults. Infants, who are likely to sleep for 14 hours, laying on air beds is considered a misuse per air beds warning label and professional health care recommendations. The air bed age limit warning label goes up to 15 and 18-month-old children. EPA did not opt to remove the screening level results from the risk calculator for all populations but recognized that infants in this assessment (< 12 months) should not have exposures

because these articles are not targeted for their use, and neither are children up to 18 months. Readers can find that air beds dermal screening and refined exposures for all age groups above 18 months had MOEs above the benchmark. See Section 2.3.2 in the DEHP consumer TSD for a description of the refined air beds dermal absorption approach and Appendix B for summary tables for the screening and refined air beds dermal absorption doses and MOE results.

6.3 General Population Risk Characterization

No comments are associated with this topic.

6.4 Cumulative Risk Characterization

Summary: A public commenter (SACC23-0047) stated that the purpose of their comments was to provide recommendations to improve EPA's development of cumulative risk assessments (CRA) under TSCA. The commenter recommended that EPA:

1. The commenter's primary concern with the *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-requested Phthalate under TSCA* is the timing of its release:
    a. At the time of the comment, the draft risk evaluations for the individual phthalates had not been released, so the commenter considered the draft CRA document to include numerous preliminary scientific conclusions which may change following completion of systematic review, public comment, and peer review.
    b. The commenter stated that EPA did not sufficiently explain why the relative potency factor (RPF) approach is more appropriate than other approaches, such as the hazard index (HI) and therefore considers EPA's proposal to use the RPF approach to be premature.
    c. The commenter noted that there may have been changes in uses and recommended that EPA identify current COUs and potential for co-exposures.
2. The commenter agreed with EPA's stated intent to consider refinements to risk estimates when default screening approaches suggest unreasonable risk but noted that this is not consistent with EPA's past practices for individual risk evaluations (*e.g.,* EPA did not consider information on workplace controls from the semiconductor industry association in its final risk evaluation for n-methylpyrrolidone).
3. The commenter recommended that EPA use both the RPF and HI approach for the CRA due to data gaps on dermal and inhalation routes of exposure.
4. The commenter recommended that EPA characterize the scientific conclusions in the *Draft Proposed Approach for CRA on Phthalates* as preliminary and not apparent conclusions of law.

EPA Response:

- Response to Points 1 and 4: EPA acknowledges receipt of these comments. EPA's Draft Proposed Approach for CRA of Phthalates under TSCA was publicly released as a draft document in 2023 and all conclusions reached in the 2023 draft proposal were characterized as draft/preliminary conclusions. The final 2025 CRA TSD and final risk evaluations for DEHP,

DBP, DIBP, BBP, and DCHP reflect EPA's final scientific conclusions, which take into considerations recommendations from multiple SACC peer-review meetings held in 2023, 2024, and 2025, as well as multiple public comment periods.

- Response to Point 2: EPA acknowledges the commenter's support for making refinements to screening level approaches. Throughout the final risk evaluations of DEHP, DBP, BBP, DCHP, and DIBP, when screening-level approaches indicated potential risk (*i.e.,* MOEs below the benchmark), EPA conducted refinements and sensitivity analyses as supported by reasonably available data and the best available science.
- Response to Point 3: As described in the 2025 CRA TSD, EPA developed relative potency factors (RPFs) for phthalates based on decreased fetal testicular testosterone production. EPA acknowledges that the hazard index (HI) is another approach that can be used to characterize cumulative risk; however, as discussed in EPA's mixtures and cumulative risk assessment guidance (U.S. EPA, 2000; U.S. EPA, 2023), the RPF approach is a more refined approach for assessing cumulative risk. Additionally, SACC supported EPA's approach for deriving phthalate RPFs (U.S. EPA, 2025). The commenter is directed to EPA's response to the 2025 SACC report for further details pertaining to how EPA addressed recommendations provided by SACC during the 2025 peer-review meeting.

Summary: A public commenter (SACC25-0132) states flaws in the cumulative risk assessment (CRA) related to the selected point of departure (POD) for DEHP resulted in illogical conclusions with respect to the consideration of background exposures. The commenter states that the selection of a less sensitive POD in the CRA resulted in a finding of higher risks for certain TSCA COUs without the background exposures than with the background exposures. The commenter states that the current process for accounting for background exposures underestimates real-world risks for DEHP and DBP.

EPA Response: Based on SACC feedback, EPA considered the strengths and limitations of two approaches described in the CRA TSD and the applicability of each for assessing cumulative risk for each phthalate. EPA has revised the DEHP RE to reflect the SACC's recommendation to use Approach 2 for estimating cumulative risk for DEHP. Use of CRA approach 2 for DEHP results in cumulative MOEs that are lower than MOEs from aggregate exposure to DEHP for each individual COU. The lower cumulative MOEs are a result of the addition of background cumulative phthalate exposure from NHANES, which adds 6.2 to 15.5% to the risk cup, depending upon the population and age group being assessed.

Summary: One comment (DCHP-0122) recommends that "EPA must ensure that the cumulative risk assessment is protective from both high acute exposures and lower life-long exposures." The comment further specifies "EPA should clarify how the cumulative risk assessment would protect the health of male infants, toddlers, and children form ongoing co-exposures to phthalates."

EPA Response: As discussed in Section 1.5 of the CRA TSD, EPA focused its phthalate CRA on acute exposure durations which are expected to represent the highest relevant exposures for the common health effect for susceptible populations (*i.e.,* women of reproductive age, male infants, male children). Notably, protecting for acute exposure durations will be protective of longer duration exposures, since acute exposures are higher than longer duration exposures and the points of departure are the same for acute and chronic exposure. By taking a cumulative approach to phthalates, which considers TSCA uses as well as the broader background of people's exposure to multiple phthalates,

EPA is being protective of susceptible subpopulations (*i.e.*, women of reproductive age, pregnant women, male infants, male children) exposures to phthalates.

Summary: A public commenter (SACC25-0145) provided detailed comments on the CRA, as summarized as follows:

1. EPA's selection of the fetal testicular testosterone endpoint to derive relative potency factors (RPFs) for estimating cumulative risks to multiple phthalates is appropriate.
2. EPA's benchmark dose and relative potency factor estimates are appropriate for the phthalates cumulative risk assessment.
3. EPA's "Option 2" for computing cumulative risk is a potentially useful concept, but EPA's implementation of the option is deeply flawed.
4. EPA must ensure that the CRA is protective for both higher, acute exposures and for longer and lower exposures that can also cause adverse effects.
5. EPA should include other chemicals and non-chemical stressors in the phthalates CRA.
6. EPA's use of National Health and Nutrition Examination Survey (NHANES) data to represent background phthalate exposure is generally appropriate but must be supplemented by additional data to avoid underestimating cumulative exposures.
7. EPA should use reasonably available data to develop scenarios for worker exposures and combine worker and consumer exposures.
8. EPA should use reasonably available data to develop scenarios for consumer exposures.
9. EPA should include inhalation and dermal uptake from vapor in the cumulative risk from exposure to indoor dust. Cumulative risk from dust should be integrated into consumer and worker exposure scenarios, with consideration for sub-populations that experience higher exposures from dust, such as people in environmental justice and low-income communities.

EPA Response:

- Points 1 & 2: EPA acknowledges the public commenters support for using fetal testicular testosterone data to derive RPFs.
- Point 3: EPA disagrees with the public commenter's characterization of Approach 2. For Approach 2, cumulative non-attributable exposure and risk as estimated from NHNAES is added to the individual aggregate exposure estimate for individual COUs for each phthalate. Depending on the population being evaluated, the addition of cumulative non-attributable exposure from NHANES adds 6.2 to 15.5% to the risk cup, regardless of which phthalate is being evaluated.
- Point 4: As discussed in the 2025 CRA TSD (see Section 1.5), EPA is focusing the application of its phthalate CRA on acute exposure durations which are expected to represent the highest relevant exposures for the common health effect for susceptible populations. Notably, protecting for acute exposure durations will be protective of longer duration exposures, since acute exposures are higher than longer duration exposures.
- Point 5: EPA disagrees with public commenter that it should consider non-chemical stressors. Few methods have been developed that allow for a quantitative analysis of cumulative risk from combined exposure to chemical and non-chemical stressors. Until Agency-wide guidance

and established methodologies have been developed, EPA does not expect to quantitatively evaluate non-chemical stressors when conducting CRAs under TSCA.

EPA considered five high-priority (DEHP, DBP, DIBP, DCHP, BBP) and two manufacturer requested phthalates (DINP, DIDP) for inclusion in its cumulative chemical group. DIDP was excluded because it is not toxicologically similar to the other phthalates and does not cause effects consistent with a disruption of androgen action or phthalate syndrome. Although there are potentially other phthalates and chemical substances that can disrupt androgen action and cause phthalate syndrome, these chemicals are not actively being evaluated under TSCA and therefore were not further considered for inclusion in the cumulative risk assessment.

- Point 6: As discussed in Section 4 of the CRA TSD, EPA acknowledges that there are limitations associated with use of NHANES urinary biomonitoring data, namely that national biomonitoring data does not oversample highly exposed subpopulations, this conclusion cannot be extrapolated to low-frequency, high-exposure scenarios. EPA did not identify any additional reasonably available biomonitoring data to supplement NHANES, as suggested by the public commenter. EPA also notes the public commenter did not provide any specific references for EPA to consider to supplement its analysis.
- Points 7 & 8: EPA recognizes that consumers, workers, and the general population may be exposed to multiple phthalates associated with TSCA COUs. However, EPA did not combine exposure scenarios in its assessment for workers and consumers because the Agency did not find any evidence to support such an aggregate analysis based on the reasonably available information, such as exposure from various exposure scenarios within a given relevant exposure duration, particularly given the quick metabolism and elimination of phthalates. Instead, as stated in the phthalate CRA, EPA used NHANES urinary biomonitoring data, which provides an estimate of aggregate exposure via all routes and pathways that cannot be attributed to specific sources or pathways of exposure, to capture additional exposures experienced by workers and consumers.
- Point 9: EPA disagrees with the public commenter. Section 3.3 of the 2025 CRA TSD provides a summary of the physical chemical and fate parameters of phthalates. Based on the physical and chemical parameters of phthalates, EPA does not expect significant volatilization of phthalates from house dust. EPA has however quantified cumulative exposure to phthalates from ingestion of house dust. Notably, exposure of phthalates through ingestion of house dust was not found to be a significant source of exposure. For example, for the highest exposed population (children 3-6 years of age), exposure to phthalates through ingestion of house dust only contributed 3.4% to the risk cup.

Summary: A public commenter (SACC23-0066) criticized several aspects of EPA's approach to cumulative risk.

First, the commenter asked for a clearer conceptual model on aggregating multiple exposures routes, particularly consumer, occupational, and general population estimates. For consumer exposure in particular, the commenter advised a revised model that recognizes potential overlap in exposures to users of two products. Regarding the exposure approach, the commenter advised using both reverse dosimetry and scenario-based approaches and evaluate the risk of underestimating exposure. In addition, they advise EPA account for human variability in reverse dosimetry parameters, including the potential influence of small sample sizes or particular aspects of the study population.

Second, the commenter criticized EPA's use of margin of exposure (MOE) approach to characterize non-cancer risks. The commenter argued the MOE approach is overly simplistic and "perpetuates the flawed assumption that there is a 'safe' or 'no risk' level of a chemical exposure." The commenter advised considering methods that estimate the size of the exposed population who may experience adverse health outcomes.

EPA Response: Many of the comments made in (SACC23-0066) raised questions in 2023 that have been addressed in the draft risk evaluations of DEHP, DBP, DIBP, DCHP, and BBP, SACC peer review, and in the final risk evaluations of DEHP, DBP, DIBP, DCHP, and BBP. In particular, what information would be used to inform cumulative risk assessment (CRA) and how CRA would be applied to inform the phthalates risk evaluations has been presented and discussed as advice from the SACC and the public has been considered throughout the phthalates risk evaluation effort. Many of the specific issues raised in the comments, such as the development of conceptual models and use of the MOE approach, are addressed elsewhere in this response-to-comments document and in more detail in the 2025 CRA TSD. Conceptual models based on the draft 2023 approach are discussed in the beginning of Section 5 in the CRA TSD. As discussed further in the 2025 CRA TSD, to estimate cumulative risk from exposure to phthalates, EPA combined aggregate exposure from a single COU for each individual phthalate with non-attributable background phthalate exposures, which was estimated using NHANES urinary biomonitoring data and reverse dosimetry.

EPA has used MOE approaches for all TSCA risk evaluations and disagrees that it implies a safe limit. EPA is tasked under TSCA with identifying *unreasonable risk*. EPA has responded to similar comments about using probabilistic risk assessment in Section 6.6 of this response to comments document.

Summary: A public comment (DCHP-0122) recommends that "EPA should use high-end occupational phthalate exposures instead of the 50th percentile in cumulative risk estimates to protect more than 50% of the worker population." The comment explains that "Relying on the median fails to account for workers who may be exposed to significantly higher levels of phthalates and fails to protect half of the occupational population. EPA then combines MOEs derived from the 50th percentile for workers with MOEs derived from the 95th percentile NHANES cumulative exposure estimates. EPA should provide their rationale for using the 50th percentile for workers and the 95th percentile for cumulative exposure estimates. Additionally, EPA should clarify whether this example for risk estimation will actually be applied in the individual phthalate risk evaluations."

EPA Response: In the risk evaluations for DCHP (as well as DEHP, DBP, BBP, and DIBP), EPA estimated 50th and 95th percentile exposure estimates for all evaluated occupational exposure scenarios. For characterizing exposures among COUs, there may be various sources of data and various approaches used to characterize each COU. It is important to consider how the range of exposure estimates was developed for each COU individually and how EPA determined the applicability of the high-end or central tendency exposure estimates to be representative of exposures for the COU. For COUs where higher exposures are potentially expected, the high-end exposure values are more relevant. For COUs where low to medium exposures are potentially expected, the central tendency estimates are more relevant. Also, depending on the data or approach used to create the range of exposure estimates, there may be greater confidence in the central tendency than the high-end estimates, or vice versa. For example, for spray application of Adhesives and Sealants or Paints and Coatings, high-end levels of exposure may occur when factors like spray equipment type, spray booth ventilation configuration, product concentration, and spray duration contribute to unusually

elevated exposure levels. But for some other COUs like PVC Plastics Compounding or Converting, the central tendency values of exposure are expected to be more reflective of worker exposures: 1) Because of conservative assumptions in the analysis like assuming concentration of phthalate in workplace dust is equal to that in PVC or non-PVC products; and 2) Because liquid plasticizers are generally added to dry mixtures during the compounding process, and any dust generated would come from the dry material rather than the plasticizer.

"Cumulative MOE"/"Risk Cup" Approach Under TSCA

Summary: A public comment (DBP-0127) was submitted on the use of the "cumulative MOE"/"risk cup" approach under TSCA. The comment noted that the concept was introduced late in the process without adequate stakeholder engagement or detailed explanation. The comment also questioned if this approach, despite its use "for pesticides for over two decades", still represents the "best available science" under TSCA and called for its stand-alone consideration by the Science Advisory Committee on Chemicals (SACC).

The public comment (DBP-0127) also stated "cumulative assessments under TSCA should not necessarily be done in the same manner as cumulative assessments under the FQPA-amended FIFRA/FFDCA because the statutes have different regulatory standards ("no unreasonable risk" vs. "reasonable certainty of no harm")" and pointed out that "commodity chemicals regulated under TSCA do not generally have food uses and are not intentionally engineered to have biological activity, unlike pesticides, which are engineered to kill a pest and intended for application to food crops."

EPA Response: EPA agrees with the public commenter that CRAs under TSCA should not necessarily be done in the same exact manner as CRAs under FIFRA. As the commenter correctly points out, both programs have different statutory requirements. Further, TSCA does not statutorily require EPA to conduct cumulative risk assessments. However, TSCA does require EPA to conduct risk evaluations using reasonably available information and the best available science, and draw conclusions based on the weight of scientific evidence. As discussed in EPA's 2023 *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act*, for some chemical substances the best available science may indicate that a cumulative risk assessment is necessary for EPA to protect human health and the environment.

EPA disagrees with the public commenter that the risk cup concept requires stand-alone consideration by the SACC. The risk cup concept is a common concept in risk assessment, which has been used to describe aggregate exposure estimates under FQPA (https://www.epa.gov/pesticide-registration/prn-97-1-agency-actions-under-requirements-food-quality-protection-act). As described in Section 1.7 of the 2025 CRA TSD, the "risk cup" term is used to help conceptualize the contribution of various phthalate exposure routes and pathways to overall cumulative risk estimates and serves primarily as a communication tool.

Additionally, although EPA did not ask a specific charge question regarding the risk cup, during peer-review SACC stated that the risk cup concept is "a useful approach that is consistent with the approach used for crop protection products, and it is a good way to determine when the risk level is getting too great or if there is still a good amount of space in the risk cup so that it would take a lot to increase the risk to a level of concern" (see p.81 of U.S. EPA, 2025).

6.5 Potentially Exposed or Susceptible Subpopulations (PESS)

Risk Characterization for PESS

Summary: Several comments (SACC23-0044, SACC23-0052) supported the characterization of "fenceline" communities located near industrial facilities as PESS, but recommended that the definition be expanded to consider communities near other significant sources of TSCA chemicals.

One of these public comments (SACC23-0044) recommended engaging with Tribal and PESS communities to identify relevant data.

One of these public comments (SACC23-0052) stated the EPA should consider non-attributable and non-TSCA exposures that may vary by race/ethnicity and socioeconomic status.

One of these public comments (SACC23-0052) recommended EPA consider non-industrial occupational groups PESS, including firefighters, nail salon workers, and hairdresser worker.

Another public comment (DIBP-0130, DIBP-0133) stated that EPA must evaluate risks to all potentially exposure or susceptible subpopulations, including tribal populations, truck drivers, ride share drivers, and others who work in vehicles, warehouse workers, fire fighters, and pregnant people.

EPA Response: EPA considers, consistent with statutory authority, PESS throughout the risk evaluation of phthalates, including but not limited to the exposure assessment, hazard identification, and dose-response analysis supporting this assessment. EPA incorporated the following potentially exposed and susceptible populations (PESS) into its assessment—women of reproductive age/pregnant women, infants, children and adolescents, people who frequently use consumer products and/or articles containing high-concentrations of phthalates, people exposed to phthalates in the workplace, and tribes whose diets include large amounts of fish. These subpopulations are PESS because some have greater exposure to phthalates per body weight (*e.g.*, infants, children, adolescents) or due to age-specific behaviors (*e.g.*, mouthing of toys, wires, and erasers by infants and children, assessed in the consumer exposure scenarios), while some experience aggregate or sentinel exposures. EPA also considered aggregate exposure to phthalates using NHANES urinary biomonitoring data. NHANES provides an estimate of non-attributable (*i.e.*, can't be attributed to specific TSCA or non-TSCA sources) aggregate exposure to phthalates. As part of this national-scale analysis, EPA considered race (*i.e.*, White non-Hispanic, Black non-Hispanic, Mexican-American, Other) and socioeconomic status (*i.e.*, above or below poverty level). For phthalates, at the national-scale, race and socioeconomic status did not appear to lead to higher exposure for either phthalate. However, EPA acknowledges that certain uses or exposure scenarios for phthalates may not be reflected in NHANES at the national scale for women of color of reproductive age and economically insecure individuals, and that these groups of individuals may experience higher exposures to phthalates under certain conditions. By evaluating each of the phthalates individually as well as viewing them through the lens of cumulative risk, and focusing the individual and cumulative assessments on sensitive populations (*e.g.*, women of reproductive age, male infants, male children), EPA is confident that its assessments are protective of human health and the environment.

Summary: A public comment (SACC23-0052) stated that EPA should consider all infants, toddlers and children PESS, not just males.

EPA Response: EPA acknowledges that all infants, toddlers and children may be considered PESS based on greater exposure on a per body weight basis. However, male infants, male toddlers, and male children are more susceptible to the most sensitive effects association with phthalate exposure (*i.e.*, effects on the developing male reproductive system consistent with phthalate syndrome). Because male infants, toddlers, and children have greater exposure and greater susceptibility, protecting this age group would also be protective of female infants, toddlers, and children.

Summary: A public comment (SACC25-0132) stated that EPA should use a higher respiration rate for pregnant people to account for differences in pulmonary function, ventilatory pattern, and respiration rates in pregnancy to appropriate characterize this PESS.

Another public comment (DIBP-0130, DIBP-0133) also stated that on pregnant people, EPA must consider higher respiration rates for potentially exposed or susceptible subpopulations specific to pregnant people, when calculating risk for this group. "Research shows that pregnancy affects pulmonary function and ventilatory pattern, and that minute ventilation (the volume of air that moves into and out of the lungs in one minute) increases significantly (up to 48%) during pregnancy."

EPA Response: EPA used a breathing rate based on average worker breathing rates. The breathing rate accounts for the amount of air a worker breathes during the exposure period. The typical worker breathes about 10 m^3 of air in 8 hours or 1.25 m^3/h (U.S. EPA, 1991).

For consumers, EPA used different inhalation rates for each lifestage and considered inhalation rates during product use and after product use. The highest inhalation rate corresponds to youth (11 to 15 years of age) during product use, 0.78 m^3/hr and the inhalation rate value for adults during product use is 0.74 m^3/hr. The Exposure Factors Handbook Tables 6-53, 6-54, and 6-55 provides inhalation rates for pregnant people for various age groups, at different pregnancy stages, and body mass (U.S EPA, 2011). The inhalation rates ranged from 0.55 m^3/hr (for underweight, 11 to 23 years of age, 9th week of pregnancy) to 2.0 m^3/hr (overweight, 30 to 55 years of age, 36th week of pregnancy). While the range provided represents the 5th and 99th percentiles respectively the mean values for the different age groups, pregnancy stage, and body mass ranged from 0.71 m^3/hr to 1.0 m^3/hr. The mean inhalation rate values for pregnant women are captured in the current analyses.

Summary: A public comment (DIBP-0089) stated that EPA should "explicitly quantify risk to children, pregnant individuals, and people in fenceline communities disproportionately exposed to DIBP from air, food contact materials, and consumer products."

EPA Response: EPA agrees with the comment with the caveat that TSCA does not require EPA to look at exposures to chemical substances from food contact materials or other sources not subject to TSCA, which EPA lacks authority to regulate. Nevertheless, EPA considers vulnerable populations including children and pregnant individuals as a PESS as stated in Section 5.2 of the final non-cancer human health hazard TSD for DIBP. This section describes how susceptibility was addressed in the risk evaluation for these populations through the use of uncertainty factors, which are reflected in the benchmark MOE or captured in the choice of POD, which is based on reproductive and developmental effects. EPA used previously peer-reviewed methodology for fenceline communities to evaluate

exposures via the ambient air pathway as described in the final environmental media and general population exposure TSD for DIBP (see EPA-HQ-OPPT-2025-0260-0031).

Summary: A public comment (DIBP-0089) asked, "Will EPA include mapping of demographic overlays using EJScreen to identify low-income or BIPOC communities near production or disposal sites of DIBP-containing materials?"

EPA Response: EPA is required to consider potentially exposed or susceptible subpopulations (PESS) in its TSCA risk evaluations. In its phthalates risk evaluations, EPA utilized the information, tools, models, and approaches reasonably available and most appropriate for considering PESS in its risk evaluations. In doing so, EPA is confident that it has appropriately accounted for exposure and risk to PESS in its phthalate risk evaluations. EJScreen is no longer an available EPA tool and therefore it was not used for including demographic overlays.

Summary: A public comment (DIBP-0130, DIBP-0133) stated that regarding tribal populations, EPA did not account for other practices that expose tribal citizens to higher levels of contaminants. Unique tribal exposures can include harvesting subsistence foods such as plants and berries for food and medicine, a cultural practice that can involve exposures to sediment, water, and soil. Other unique exposures include mastication of plants for basket weaving, hauling local water, occupations that result in daily exposure to surface water, steam from water used in sweat lodge ceremonies, living proximate to unlined, uncovered community dumps with open burning of trash, and residing in older and substandard housing with poor ventilation. Additionally, EPA failed to calculate BBP and DIBP's risks to Indigenous children. For tribal populations, EPA only considered fish ingestion of adults. EPA asserted that children's consumption rates were generally either lower than adults or approximately the same for adults and children, and therefore assessment of children's consumption rates was not necessary. However, among the Squaxin Island Tribe in Washington, children under the age of five have a reported fish ingestion rate of 2.9 g/kg-day, a rate higher than adult central tendency ingestion rate.

EPA Response: Although EPA only explicitly detailed Tribal exposure through fish ingestion, many of the screening-level analysis for the general population were inclusive of Tribal populations who may be near releasing facilities. Exposure from swimming in surface water impacted by releasing facilities was assessed and would be a conservative estimate of exposure through other activities occurring in surface water. Exposures resulting from soil or the applications of biosolids for use in agricultural activities is limited by the strong sorption of the phthalates to soil with limited uptake to plants and limited mobility within soil and landfills. Conservative estimates of phthalates in agriculturally-applied biosolids were provided for informational purposes in the general population exposure assessment which may be used for the tribal-specific scenarios. Similarly, discussion of communities adjacent to landfills has also been discussed in the landfill sections of the aforementioned general population exposure assessment and may be applied to tribal communities residing near or around DEHP containing landfills. Due to the low volatilization, phthalates would not be expected to volatilize in uncovered community dumps, but would likely be immobilized through sorption to refuse, organics, and soil as stated in the assessment. For other scenarios that may impact Tribal populations, EPA continues to work on improving methods to assess Tribal exposure and engage with stakeholder for better exposure data.

While children's fish ingestion rate of 2.9/kg-day for the Squaxin Island Tribe is slightly higher than the mean adult ingestion rate of 2.7 g/kg-day for the Suquamish Tribe, EPA also quantitatively

evaluated the highest 95th percentile adult ingestion rate. That value is 10.9 g/kg-day for the Shoshone-Bannock Tribes, and 3 to 4 times greater than the children ingestion rate.

Summary: A public comment (DBP-0129) identified risks to PESS in the DBP draft Risk Evaluation, but stated that they anticipate that no unreasonable risks will be identified in the final Risk Evaluation due to updated analysis. The commenter stated that in the draft DBP RE, there was risk to teenage consumers due to dermal exposure, but that they anticipate changes in the dermal analysis will lead to no risk in the final Risk Evaluation. Additionally, the commenter stated that EPA identified risk to infants from drinking water under certain flow conditions in the draft Risk Evaluation, but they anticipate that the application of wastewater treatment removal efficiency to the risk will not exceed the Margin of Exposure (MOE) in the final Risk Evaluation.

EPA Response: EPA agrees with the commenter in that the DBP dermal exposure approach between draft and final was revisited and resulted in some MOEs that were below the benchmark of 30 shifting to above the benchmark. Please see Section 4.6 of this document for more details about the changes between draft and final DBP dermal exposure approach. Briefly, after further review of available dermal absorption data of DBP, EPA has identified an *ex vivo* study [Beydon et al. (2010)] that used metabolically active human skin for estimating dermal absorption of DBP. The data represent more biologically relevant estimates of dermal absorption in humans.

EPA calculated MOEs below the benchmark for infants for its screening-level assessment for certain scenarios. When considering factors such as wastewater treatment and additional flow metrics, EPA determined that drinking water was not a pathway of concern for infants.

Summary: Public commenters (DCHP-0127, DEHP-0138, SACC25-0145) stated "EPA failed to adequately identify potentially exposed or susceptible subpopulations (PESS), as required by TSCA." These commenters noted that the Draft Risk Evaluations for DBP, DCHP, and DEHP did "…include consideration of various categories of 'biological susceptibility' in Table 5-1 of the draft hazard assessment document[s], which is a useful initial step towards developing a consistent, structured approach to identifying PESS in TSCA risk evaluations." Commenter (DCHP-0127) added, "However, the evaluation is still deficient in identifying PESS, particularly in its failure to include a similar table that identifies PESS on the basis of greater exposure as required by statute." Commenters (DEHP-0138) also expressed that the evaluations for DBP and DEHP were deficient in , identifying PESS, and had "…taken a step backwards with the exclusion of more detailed evaluations of PESS based on both greater exposure and greater susceptibility that were included in recent risk evaluations." Additionally, the commenters noted that "…EPA failed to fully consider all PESS within each category identified for..." DBP, DCHP, and DEHP.

Another public commenter (SACC23-0051) stated that EPA should consider real-world impacts of cumulative exposures on Environmental Justice communities, and stated that chemical exposures inflict disproportionate harms on communities of color, Indigenous peoples, and other overburdened populations. Another comment (SACC-0059) stated similarly that the EPA's definition of vulnerable subpopulations is too narrow and that EPA should consider exposure from non-attributable and non-TSCA exposures in these populations, because they stated that diet and personal care products are a significant source of phthalate exposure in such population, and vary by race/ethnicity and socioeconomic status. This commenter also states that "certain non-industrial occupational groups known to have high occupational exposures to phthalates" should be considered PESS, and name

firefighters, nail salon workers, and hairdressers as examples. They suggest examining the demographics of these occupations to investigate racial and ethnic disparities in phthalate exposures, giving the example that nail salon workers in the U.S. are disproportionately of Asian descent and experience high levels of occupational phthalate exposures.

EPA Response: EPA disagrees with the public commenters (DCHP-0127, DEHP-0138, SACC25-0145). EPA considers PESS throughout the risk evaluation of DBP, DCHP, DEHP, BBP, and DIBP, including but not limited to the exposure assessment, hazard identification, and dose-response analysis supporting these assessments. EPA incorporated the following potentially exposed and susceptible populations (PESS) into its assessment—women of reproductive age/pregnant women, infants, children and adolescents, people who frequently use consumer products and/or articles containing high-concentrations of DBP, DCHP, DEHP, BBP, and DIBP, people exposed to DBP, DCHP, DEHP, BBP, and DIBP in the workplace, and tribes whose diets include large amounts of fish. These subpopulations are PESS because some have greater exposure to DBP, DCHP, DEHP, BBP, and DIBP per body weight (*e.g.*, infants, children, adolescents) or due to age-specific behaviors (*e.g.*, mouthing of toys, wires, and erasers by infants and children, assessed in the consumer exposure scenarios), while some experience aggregate or sentinel exposures. EPA also considered aggregate exposure to DBP, DEHP, BBP, and DIBP using NHANES urinary biomonitoring data. NHANES provides an estimate of non-attributable (*i.e.*, can't be attributed to specific TSCA or non-TSCA sources) aggregate exposure to DBP, DEHP, BBP, and DIBP. As part of this national-scale analysis, EPA considered race (*i.e.*, White non-Hispanic, Black non-Hispanic, Mexican-American, Other) and socioeconomic status (*i.e.*, above or below poverty level). For DBP, DCHP, DEHP, BBP, and DIBP, at the national-scale, race and socioeconomic status did not appear to lead to higher exposure for either phthalate. However, EPA acknowledges that certain uses or exposure scenarios for DBP, DEHP, BBP, and DIBP may not be reflected in NHANES at the national scale for women of color of reproductive age and economically insecure individuals, and that these groups of individuals may experience higher exposures to DBP, DCHP, DEHP, BBP, and DIBP under certain conditions. EPA also considered cumulative exposure to DBP, DEHP, BBP, DIBP, and DINP for male children and women of reproductive age. Based on the analysis by race, black non-hispanic women of reproductive age had slightly higher cumulative exposure than other populations. EPA used the 95th percentile cumulative exposure for black non-hispanic women of reproductive age as the basis of its cumulative risk assessment of phthalates (i.*e.*, EPA used the highest exposure estimate [95th percentile] for the highest exposed population (black non-hispanic women of reproductive age)). By evaluating each of the phthalates individually as well as viewing them through the lens of cumulative risk, and focusing the individual and cumulative assessments on sensitive populations (*e.g.*, women of reproductive age, male infants, male children), EPA is confident that its assessments are protective of human health and the environment.

Summary: A public comment (DCHP-0122) expressed concern that "The decision to evaluate risk using a factor of 10 for human variability is based on institutional guidance, not the best available science." The comment further states that "the National Research Council, the World Health Organization's International Programme on Chemical Safety (IPCS), and California EPA's Office of Environmental Health Hazard Assessment (OEHHA) have all recommended uncertainty factors larger than 10 to adequately account for the range of variability in human response to chemical exposures. Notably, the IPCS recommended a 42-fold factor to account for human variability based on studies conducted with healthy human adults." The comment concludes that "considering the ongoing sensitivity to phthalate effects through multiple life stages, and the 'possibility that the available data might not be representative of individuals who are most susceptible to the effect' (p.25), uncertainty

factors of *at least* 42 is warranted to address response variability, with additional factors necessary to address differences in age-related susceptibility from the prenatal to postnatal periods."

Similarly, public commenters (DEHP-0138, SACC25-0145, SACC25-0151) stated, "The 10X default human variability (UFH) uncertainty factor that EPA relies on to account for intra-species variability is based on a scientific recommendation made nearly 70 years ago." They continued, "The World Health Organization's International Programme on Chemical Safety (IPCS) examined human variability in toxicokinetic and toxicodynamic responses to chemical exposures using a probabilistic method and found that variability at the 99th percentile across the general population was up to more than four times higher than what is reflected in EPA's default intra-species adjustment factor. Accordingly, the WHO recommends using larger uncertainty factors, up to 42X, just to account for normal variability in the human response to chemical exposures among healthy adults." One of these public commenters (SACC25-0151) provided examples of the impact of an application of a 42X intraspecies uncertainty factor on the risk to consumers from DEHP, stating that the application of this uncertainty factor would result in 17 consumer uses below the benchmark MOE, including various high-end and aggregate exposures from furnishings, cords, toys, and shower curtains affecting infants and children up to age 15.

An additional comment (DCHP-0127) made a similar point: "The WHO and other authoritative bodies have demonstrated that the traditional 10X uncertainty factor is insufficient for fully accounting for risk in sensitive groups and recommend the use larger uncertainty factors." The comment suggested that "When such [chemical specific] data are absent, the application of appropriate adjustment factors (beyond the customary 10x factor for human variability) should be applied to ensure that risks to PESS are not underestimated". One public commenter (SACC25-0151) specified that EPA should consider factoring an additional 10X UF to account for increased susceptibility of Black women due to intrinsic and extrinsic factors such as the increased likelihood of experiencing poverty, racism, healthcare inequities, certain disease disparities, and disproportionate chemical exposures compared to other racial groups in the U.S. population, and provided the example that the application of this additional 10X safety factor would result in nearly 65 distinct consumer uses of DEHP falling below the benchmark.

Conversely, another public comment (DEHP-0139) stated that EPA's use of an uncertainty factor of 10 for workers was overly conservative, because worker populations are less heterogenous than the general population and exclude children and the elderly. The commenter requested that EPA use a UF of 3 to 5 instead for workers.

Another public comment (SACC23-0051) recommended the use of additional uncertainty factors for cumulative human health assessment, but did not provide specific suggested uncertainty factors or further justification to support the recommendation.

EPA Response: EPA disagrees with the public commenters (DEHP-0138, SACC25-0145, SACC25-0151, DCHP-0122, DCHP-0127, DEHP-0139, SACC23-0051). For the risk evaluations of DEHP, DBP, DCHP, BBP, and DIBP, EPA selected uncertainty factors consistent with U.S. EPA Guidance (U.S. EPA, 1994; U.S. EPA, 2002; U.S. EPA 2022). For DEHP, DBP, DCHP, BBP, and DIBP, the total uncertainty factor selected as the benchmark MOE was 30×, which includes application of a intraspecies uncertainty factor (UF_H) of 10× to account for variability in toxicokinetics and toxicodynamics within the human population to account for differences in sensitivity and a interspecies uncertainty factor (UF_A) of 3× to account for species differences in toxicodynamics. Consistent with EPA guidance (U.S. EPA, 2011) the UF_A was reduced from a value of 10× to 3×

because allometric body-weight scaling to the ¾ power was used to derive a human equivalent dose (HED) from the POD, which was derived from a rodent toxicology study. EPA recognizes that other organizations (*e.g.*, California OEHHA, WHO IPCS, or other agencies/stakeholders that set worker OELs [occupational exposure limits]) may have different recommendations for the application of uncertainty factors. However, in the risk evaluations of DEHP, DBP, DCHP, BBP, and DIBP selected uncertainty factors based on the Agencies own long-standing guidance.

Characterization of Children's Exposure

Summary: A public commenter (SACC25-0132, SACC25-0145, DIBP-0133) stated that EPA must quantify exposure to DEHP, DBP, DIBP, and BBP from human breast milk, including for children older than 12 months, given that there is biomonitoring evidence for concentrations of these phthalates in breast milk. The commenter stated that EPA's assertion that this pathway can't be considered due to uncertainties in break milk half-life are unfounded, as EPA could consider values for other similar chemicals, or use the Verner model that is has used in the past (in the Risk Evaluation for tris(2-chloroethyl) phosphate).

A public comment (DCHP-0128, DIBP-0133) recommended that DCHP levels in breast milk be estimated using a range of elimination half-lives from similar phthalates within the Kapraun model developed at EPA to estimate transfer to infants during gestation and lactation.

EPA Response: EPA disagrees with the public commenters (SACC25-0132, SACC25-0145, DIBP-0133, DCHP-0128). As discussed in Section 4.1.4 of the DEHP, DBP, DIBP, BBP, and DCHP risk evaluations (and in more detail in the Environmental Media, General Population, and Environmental Exposure TSDs for DBP, DEHP, DCHP, BBP, and DIBP), EPA identified biomonitoring studies of each phthalate that measured concentrations of each phthalate or its metabolites in human milk. EPA explored the potential to model milk concentrations and concluded that there is insufficient information (*e.g.*, sensitive and specific half-life data) available to support modeling of the milk pathway, the Agency also concluded that modeling is not needed to adequately evaluate risks associated with exposure through milk. This is because the POD used in this assessment is based on male reproductive effects resulting from maternal exposures throughout sensitive phases of development in multigenerational studies. EPA, therefore, has confidence that the risk estimates calculated based on maternal exposures are protective of a nursing infant's greater susceptibility during this unique lifestage, whether due to sensitivity or greater exposure per body weight. Further discussion of the human milk pathway is provided in the risk evaluations and Environmental Media, General Population, and Environmental Exposure TSDs for DEHP, DBP, DIBP, BBP, and DCHP.

Summary: A public commenter (SACC25-0132, DIBP-0130, DIBP-0133) stated that EPA must quantify ingestion and dermal exposures from swimming in children under 6. The commenter expressed support for EPA's inclusion of this pathway for older children and adults, and cited evidence that younger children also participate in water recreation activities, and that children, including children under six, may experience higher water ingestion during these activities than adults.

DIBP-0133 commenter stated that "Children younger than six years old can and do swim or otherwise interact with surface waters. A recent study found that approximately 90% of children aged four to seven at studied beach locations "waded, swam, or played in the water." Further, 25% of children aged one to three "swallowed water" while swimming in freshwater and 22% swallowed marine

water. The Exposure Factors Handbook also acknowledges that children can start swimming years before they reach six years of age, and the American Academy of Pediatrics recommends swim lessons starting at age one and states that most children are capable of basic swimming by four years of age."

EPA Response: EPA focused its general population assessment on the most exposed subpopulations based on intake rate for its screening level assessment. For dermal exposure while swimming, adults (21+ years) had the greatest exposure based primarily on having greater skin surface area exposed. For incidental ingestion while swimming, youth (11–15 years) had the greatest exposure as compared to adults (21+ years), youth (16–20 years), and children (6–10 years) based on recommended values for water ingestion while swimming as provided in Table 3-7 of the Exposure Factors Handbook and alongside assumed swimming duration. EPA acknowledges that children under 6 can and do swim or otherwise interact with surface waters, but EPA did not have reasonably available data to determine water ingestion while swimming for children under 6 to assess swimming for that age group. The comment cites an EPA-funded study that found that children evaluated swallowed 36 ml of water on average compared to adults who swallowed on average 9 ml of water. However, EPA's screening-level assessment already utilized intake values greater than the value in provided reference. As part of a conservative screening assessment EPA utilized the upper percentile for water ingestion while swimming which was 152 mL/hour for youth (11–15 years) for two hours of swimming which exceeds the cited value. EPA also incorporated other protective assumptions in its swimming assessment including assuming high-end direct discharge into a low flow receiving waterbody with no wastewater treatment or dilution and swimming directly at the outflow of release. With the many compounding protective assumptions, EPA did not find incidental ingestion and dermal contact through swimming to be a pathway of concern for the general population and is confident that the assessment is protective of all subpopulations.

Summary: A public commenter (SACC25-0132, DIBP-0133) stated that EPA must consider the exposure pathways related to tire crumb rubber on playgrounds, including those experiences by young children under 3. The commenter was supportive of EPA's evaluation of tire crumb (sports) fields, but noted that for young children exposure on playgrounds may be significant and should be considered.

EPA Response: EPA agrees with the commenter that children under 3 can be exposed to tire crumb in playgrounds, however the current analysis considerations include people 3 years of age and older during high performance activities that would rival the intensity toddlers 1 to 2 years of age or even infants with some mobility would show in terms of exposure duration and frequency. EPA is confident that the 3- to 5-year-old children scenario is already an upper bound or overestimation based on activity patterns such as duration of exposure and frequency. Since MOEs for children 3 to 5 years were various orders of magnitude above the benchmark, any scenario with shorter durations of exposures and frequencies, and inhalation rates would result in larger MOE values.

Summary: A public commenter (SACC25-0132, DIBP-0133) stated that EPA must consider the exposure pathways related to synthetic leather clothing for young children under 3 years of age, noting that many clothing items made of these materials are available for young children.

EPA Response: EPA assessed exposures to children from clothing depending on the identified information for each phthalate. Generally, the synthetic leather clothing scenario was based on a reasonable assumption that these materials may be used in synthetic leather clothing as well.

However, not every phthalate assessment had confirmation of synthetic leather clothing content. See section 2.1.1 section describing solid article phthalate content in each consumer TSD for a description of the identified information. Most of the examples identified for synthetic clothing were not targeted for infants and toddler use. In the absence of specific information about phthalate content in children's clothing EPA assessed exposures using the solid articles with potential for semi-routine dermal exposure scenario as a proxy. Infants and toddlers' exposure to clothing materials were assessed based on reasonably available information confirming the presence of the phthalate in items of clothing targeted for these lifestages.

Summary: A public commenter (SACC25-0132, DIBP-0133) stated that EPA must consider mouthing exposures for children under 3 for additional products not considered, including mouthing of outdoor furniture and synthetic clothing. Furthermore, SACC25-0132 and DIBP-0133 state that fraction of mouthing time associated with soft plastic items used in the existing evaluations is likely underestimated, due to the fact that it is from a 2003 study and that plastic use has increased significantly in the last 2 decades.

EPA Response: EPA assessed mouthing exposures to indoor furniture for children, however the only phthalate with outdoor seating was DCHP. For DCHP the outdoor seating is not the typical synthetic leather or other soft material expected in indoor furniture used for the mouthing exposure. The seating examples for DCHP were for concrete, masonry, plaza decks, roof decks, balconies, terraces and stadiums. While children under 3 years can "mouth" these seating examples, EPA is confident that the indoor furniture scenario offers a higher exposure due to higher frequency and duration of exposure. While EPA agrees that one of the few sources used to calculate mouthing exposure is over 20 years old, there is no similar study performed recently, and age of a study is not the only basis to disqualify its applicability in this assessment. The consumer TSD provides a description of the weaknesses and strengths and sources of uncertainty in using the study in Section 5.1. EPA recognized variability in the study and what that entails for the confidence in the estimated exposures.

Characterization of Exposure in Tribal Communities

Flow Rates for Fish Consumption

Summary: A public comment (DCHP-0121) considered the flow scenario selection arbitrary, stating that the modelling utilized a large number of assumptions and that the selected flow rate reduced risk to Tribal populations.

A public commenter (SACC25-0132) stated that EPA improperly revised risk determinations for tribal populations via indefensible assumptions related to fish consumption levels (and dismissal of heritage consumption rates) and stream flow metric percentiles, and via a change in the selection of relevant OES.

Additionally, another public commenter (DCHP-0128) stated that EPA did not sufficiently justify the refinements to the release estimates, waste water treatment assumptions, and flow rates used to determine risk estimates. For example, EPA did not provide support for the assumption that "high-end releases [of DCHP] discharge to surface waters with high flow conditions" when selecting 75 and 90th percentile flow rates. The public comment did not support the assumption a wastewater removal value

taken from a study of a plant in China should be applied to US treatment plants. The public commenter considered EPA's "refinements" arbitrary and capricious.

Another public comment (DIBP-0133) raised concerned about tribal populations fish ingestion risk determination. "Historically, EPA has found that a "calculated MOE that is less than the benchmark MOE supports a determination of unreasonable risk of injury to health, based on noncancer effects." However, rather than making that determination, EPA readjusted its calculations and results until it ultimately reached a conclusion that there was no unreasonable risk from fish ingestion for tribal populations. EPA revised its modeling approach and recalculated risks to tribal communities using estimated surface water concentrations of DIBP and BBP based on modeled DIBP and BBP release amounts from the occupational exposure scenario with the highest modeled surface concentration—plastics compounding—and expected flow rates of receiving water bodies so that risk would fall below its MOE benchmark. For BBP, EPA further adjusted the calculations using the 75% and 90% flow rates instead the 50th percentile flow metric so the risk would fall above the benchmark.

Another comment (EPA-HQ-OPPT-2018-0434-0130) stated that DIBP and BBP assessments ignored including ingestion of fish for infants under that age of one and children living in subsistence fisher families.

Another comment (SACC25-0132) stated that EPA underestimated fish ingestion for Indigenous children and inappropriately considered all children under 5 as a group (ignoring potentially higher consumptions per unit body weight in younger children).

EPA Response: EPA disagrees that it made an arbitrary flow selection that reduced risk to Tribal populations. EPA assessed surface water concentrations and fish tissue concentrations based on modeled water concentrations using a range of flows (P50, 75, 90) as supported by the SACC as an informative, scenario-based risk characterization. Because DCHP, DIBP, and BBP did not have reported releases to water for all OES, EPA estimated a wide range of fish tissue concentrations using a various surface water concentrations including the water solubility limit, highest monitored surface water concentration, and modeled surface water concentrations using various flow rates (P50, 75, 90) for receiving water bodies paired with both high-end modeled releases and central tendency modeled releases. EPA did not select a single flow rate to determine risk to Tribal populations. Instead, EPA looked across the range of fish tissue concentrations. MOEs were above the benchmark for most scenarios except for scenarios where many conservative assumptions, such as high-end releases to a low flow receiving water body with no assumed dilution and uptake by fish directly at the outflow, were compounded. As stated by the SACC, "using P50 of a multi-year low-flow condition is likely to provide a conservative upper-bound, but pairing this with high-end emissions can amplify conservatism to the point of limiting utility." EPA utilized upper-bounds estimates using P50 as a starting point for a screening-level analysis and refined surface water concentrations further as necessary by including other flow metrics, applying assumptions of wastewater treatment, and considering plausibility of estimated concentrations based on monitoring data or physical chemical properties. In the absence of reported release data for all OES, EPA supported its modeling result contextually with environmental monitoring data of surface water concentrations and fish tissue concentrations. Please see Section 7 of the Environmental Media and General Population Exposure and Environmental Exposure TSDs for all phthalates for monitoring data. All fish tissue concentrations estimated using the various scenarios assessed by the EPA, except for the lowest modeled tissue concentration using a P90 flow, exceeded the measured tissue concentration for DCHP and DIBP. All modeled fish tissue concentrations for BBP exceeded any measured fish tissue concentrations from monitoring studies for BBP. Moreover, phthalates generally have low potential

for bioaccumulation, biomagnification, and uptake. Therefore, EPA is confident that it appropriately assessed fish ingestion including for Tribal populations using reasonably available information.

Regarding fish ingestion rates, EPA referred to the Office of Water's "Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health" (U.S. EPA, 2000) for the adult subsistence fisher consumption rate. This reference did not provide any rates for children from subsistence fish consumption. However, EPA considered tribal populations' consumption rates, including the 95th percentile of ingestion, which are higher than subsistence fish consumptions rates and, therefore, consideration of those should be protective of subsistence fisher families and infants under the age of one for the general population. For adult tribal populations, the use of the highest 95th percentile adult ingestion rate of 10.9 g/kg-day for the Shoshone-Bannock is 3-4 times greater than any fish ingestion rate for children. While EPA acknowledges the importance of heritage fish ingestion rates for many reasons, including for cultural and treaty-obligation reasons, EPA considers current ingestion rates to be more representative of contemporary rates of fish consumption. Heritage rates refer to ingestion rates that existed prior to non-indigenous settlement on Tribal fisheries resources, as well as changes in culture and lifeways. EPA reviewed existing literature for ingestion rates that reflect heritage rates. Based on available literature, EPA identified heritage rates for four Tribes. The heritage rate identified is much higher than any of the available 95th percentile current ingestion rates in the suggested literature. For example, the 770 g/day for the Alaskan communities is a maximum rather than a mean ingestion rate and still lower than the heritage consumption rate of 1646 g/day. If members of a tribe are consuming fish at heritage rates for a chemical undergoing TSCA risk evaluation, such information would be important for using heritage rates in risk assessment. EPA welcomes continued discussion with the tribes on the use of fish consumption information in risk assessment.

Summary: Two public comments (SACC23-0043, DCHP-0121) expressed support for EPA's identification of tribal populations as potentially exposed or susceptible subpopulations (PESS), outlining the ways that exposure of Tribal communities may differ from the general public, including differences in diet, infrastructural, and cultural practices. Both comments recommended EPA fund a forum on tribal risk and the form a panel of experts on tribal considerations in TSCA risk assessments. A similar public comment (SACC23-0063) reiterated the request in (SACC23-0043) that "consultation be offered to tribes on any proposed principles, guidelines, or rules that involve how EPA/OPPT assess risk to vulnerable communities." Another comment (SACC23-0044) recommended that EPA develop data collection projects to account for Tribal communities.

EPA Response: Engagement with the tribes has been an important part of EPA's existing chemical risk evaluation activities. EPA's relationship with the National Tribal Toxics Council is productive and has added great value to our risk assessments. Regarding formal tribal consultation, from September 2024 to January 2025, EPA's "Draft Considerations and Resources for Assessing Tribal Exposures in TSCA Risk Evaluations" underwent formal tribal consultation (see tribal consultation materials available at: Tribal Consultation)

Summary: Two public comments (DCHP-0121, DCHP-0128) expressed support for the consideration of heritage fish consumption rates, but recommended that current fish consumption be estimated and presented with the 95th percentile value for that range. One of these public comments (DCHP-0128), expressed concern that EPA's assessment of current Tribal fish consumption (216 g/day) is not sufficiently justified and underestimates tribal population's exposure to DCHP by relying on a mean value from the Suquamish Tribe and disregarding higher mean tribal fish consumption rates.

Additionally, the public comments (DCHP-0121, (DCHP-0128) provided references to indicate that children may consume more fish and drinking water per body weight than adults, and one of the comments (DCHP-0121) recommended heritage fish consumption rates be used to estimate consumption in children separately.

EPA Response: Throughout its phthalates assessments, EPA made assumptions to be protective of children. Regarding fish ingestion rates for children, EPA referred to the Office of Water's "Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health" (U.S. EPA, 2000) for the adult subsistence fisher consumption rate. This reference did not provide any rates for children from subsistence fish consumption. However, EPA considered tribal populations' consumption rates, including the 95th percentile of ingestion, which are higher than subsistence fish consumptions rates and, therefore, consideration of those should be protective of subsistence fisher families and infants under the age of one for the general population. For adult tribal populations, the use of the highest 95th percentile adult ingestion rate of 10.9 g/kg-day for the Shoshone-Bannock is 3-4 times greater than any fish ingestion rate for children. EPA acknowledges the importance of heritage fish ingestion rates for many reasons, including for cultural and treaty-obligation reasons. To support a risk determination under TSCA, EPA would need information conforming to the scientific standards of section 26 indicating that populations are consuming fish at heritage rates. Although not specific to DCHP, EPA is using consumption rates that, based on reasonably available information, are protective of children.

Summary: A public comment (DCHP-0128) recommended that EPA consider discharge from multiple OES known to be reasonably foreseen to co-occur (based on North American Industry Classification System code) into a single water body to estimate surface water concentrations used to estimate exposure through fish ingestion.

One of the public comments (DCHP-0128) also stated that aggregate and cumulative exposures were not considered for Tribal populations, which may have co-exposure to multiple phthalates during fish ingestion.

EPA Response: EPA has no reasonably available data for DCHP on location of releasing facilities or information on whether facilities are discharging based on multiple OES. EPA makes assumptions using EPA standard models to estimate discharges associated with various individual OES for DCHP but has no reasonably available data to consider discharge from multiple OES to a single waterbody for DCHP. However, as discussed in EPA's draft proposed approach for CRA of phthalates under TSCA (U.S. EPA, 2023), EPA recognized that the general population, those impacted by facility release of phthalates, could be exposed to multiple phthalates from single facilities that release more than one phthalate or be exposed to multiple phthalates due to living in close proximity to co-located facilities. EPA analyzed the co-location of all the known phthalate-releasing facilities within common watersheds. EPA identified DMR, NEI, and TRI data for DEHP, DBP, and BBP, but not for DCHP, DINP, and DIBP. These EPA databases provide information on facilities releasing phthalates to

various environmental media and provide latitude and longitude data for releasing facilities. Using the release information, EPA identified 1,461 facilities that report use of a single phthalate, while 461 report use of multiple phthalates (*i.e.*, any combination of DEHP, DBP, or BBP). Using the available location data, EPA mapped the reporting facilities to look for geographic patterns or hotspots. Individual facilities are broadly dispersed around the United States. EPA also analyzed the locations of the identified facilities by watershed or hydrologic units. A hydrologic unit represents the area of the landscape that drains to a portion of the stream network and is identified by a unique Hydrologic Unit Code (HUC). EPA identified 21 HUC12 watersheds with four or more releasing facilities.

Even where co-located facilities within watersheds have been identified, there is difficulty in estimating the cumulative exposures in those locations. First, the programmatic data from DMR, NEI, and TRI are reported per facility for a single reporting year. Although information such as the highest release is reported, there is no information on the timing of release of phthalates into the environment, making it difficult to identify any areas that are impacted by multiple phthalates concurrently.

Although EPA identified 461 facilities reporting the use of multiple phthalates, the reporting data does not state whether the multiple phthalates are used concurrently within the facility and released simultaneously to the environment. Often, use or production of multiple chemicals, such as the phthalates, do not occur concurrently. Instead, a single phthalate is used for a determined period of time before the facility uses another phthalate for another period of time. In these instances, phthalates would not be released from the facility concurrently and, therefore, may not pose a cumulative exposure to surrounding communities based on the fate parameters of the phthalates. EPA recognizes that the lack of data on the timing of the releases makes it difficult to quantify cumulative exposure from facilities reporting use of multiple phthalates.

In general, EPA recognizes that there may be discrete locations impacted by the release of multiple phthalates, either through single facilities releasing multiple phthalates or multiple facilities within the same watershed or releasing to the same wastewater facility. Releases would need to be continuous to lead to ongoing exposure given the relatively low persistence in the environment. In the risk evaluations for the individual phthalates, the general population exposures from pathways such as drinking water, recreational swimming, ambient air, incidental soil ingestion, and fish ingestion for each phthalate are estimated and found to be much lower than exposures for consumer and occupational populations, even when quantified using a screening-level assessment using conservative (*e.g.*, low tier, high risk) assumptions.

The commenter is directed to Section 3.3.2 of the CRA TSD for further discussion of TSCA releasing facilities.

Summary: Several public commenters (DCHP-0121, DCHP-0128 and SACC25-0132) requested the consideration of additional exposure pathways in the consideration of phthalate exposure for Tribal Populations.

One commenter (DCHP-0121) requested consideration of additional exposure pathways for Tribal populations, including exposure associated with the harvesting and use of aquatic plants and riverine sediments, as well as the ingestion of other aquatic animals (including shellfish and marine mammals). The comment states that phthalate levels in shellfish would likely be similar to sediment and provides a reference on phthalate contamination in seals. The comment requests that EPA use a modeling approach and data from toxicologically similar phthalates to estimate exposure.

Another public comment (DCHP-0128) noted that EPA did not consider sweat lodge ceremonies, which could increase inhalation exposures.

Similarly, in SACC25-0132 a commenter, while supporting EPA's identification of Tribal Communities as PESS, stated that EPA irrationally failed to consider specific tribal practices or living conditions that result in higher exposure to DEHP and DBP, including: harvesting of subsistence foods that may have been exposed to contaminated soil, sediment, or water; mastication of plants used in basket-weaving; manual hauling of potentially contaminated water; and participation in sweat-lodge ceremonies using contaminated steam; residing proximate to dump or trash-burning; and residing in substandard housing with poor ventilation. Furthermore, the commenter contends that EPA must consider aggregate exposures to tribal communities in addition to fish consumption.

EPA Response: The Agency estimated potential dermal exposure from water in section 5 of the Draft Environmental Media, General Population, and Environmental Exposure Technical Support Document. Though the scenario is not specific to tribal populations, it considers a high release scenario (PVC plastic compounding without wastewater treatment). The Agency believes this to be inclusive of scenarios where tribal populations may be harvesting and using aquatic plants.

EPA did not explicitly consider sweat lodge ceremonies which could lead to inhalation exposures based on phthalates in the water. However, EPA's screening-level surface water exposure assessment included exposure through swimming in untreated water and drinking water sourced from surface water that is wastewater treated but not drinking water treated based on many conservative assumptions. Therefore, the Agency believes this is inclusive of all scenarios based on exposure to surface water.

EPA explicitly detailed Tribal exposure through fish ingestion, and many of the screening-level analyses for the general population were inclusive of Tribal populations who may be near releasing facilities. Exposure from swimming in surface water impacted by releasing facilities was assessed and would be a conservative estimate of exposure through other activities occurring in surface water. Exposures resulting from soil or the applications of biosolids for use in agricultural activities are limited by the strong sorption of the phthalates to soil with limited uptake to plants and limited mobility within soil and landfills. Conservative estimates of phthalates in agriculturally-applied biosolids were provided for informational purposes in the general population exposure assessment which may be used for the tribal-specific scenarios. Similarly, discussion of communities adjacent to landfills has also been discussed in the landfill sections of the aforementioned general population exposure assessment and may be applied to tribal communities residing near or around DEHP containing landfills. Due to the low volatilization, phthalates would not be expected to volatilize in uncovered community dumps, but would likely be immobilized through sorption to refuse, organics, and soil as stated in the assessment. For other scenarios that may impact Tribal populations, EPA continues to work on improving methods to assess Tribal exposure and engages with stakeholders in order to receive improved exposure data.

Summary: Two public comments (SACC23-0044, DCHP-0121) noted that unregulated landfills may not be as protective in Tribal communities as those servicing the general public.

One comment (DCHP-0121) requested additional characterization of DCHP exposure through leachate from landfills and variations in wastewater and drinking water filtration systems. References

are provided indicating that many Tribes are unable to comply with federal waste and drinking water standards. The comment argues that failure to consider this means "best available science" was not used in consideration of Tribal communities.

EPA Response: Because of its presence in consumer products like rubber and plastic articles, DCHP may be introduced to municipal solid waste landfills through the disposal of these products. However, the transport and fate properties of DCHP indicate that DCHP will not leach significantly from these locations and will likely sorb to particulates and organic matter present at the disposal site. DCHP is likely to degrade biologically and through photolysis in upper parts of the landfills, but lower sections of the landfills may not be suitable for biologic degradation. In deeply interred landfills, hydrolysis will likely be the sole method of degradation and may result in higher persistence. This line of reasoning can also be applied to landfills or waste piles present in Tribal communities that may be less tightly regulated like those in the state of Alaska. DCHP introduced to these landfills are not likely to be interred and will degrade through biological mechanisms and through photolysis. Similarly, DCHP would be expected to partition primarily to the soil compartment rather than water. For these reasons, the Agency does not anticipate that tribes living near less tightly regulated landfills would have increased exposure from surface water, groundwater, or drinking water.

Monitoring data for DCHP in surface water, groundwater, and drinking water is limited. As mentioned in the *Draft Environmental Media, General Population, and Environmental Exposure Technical Support Document*, DCHP was queried using all known names and CAS numbers related to DCHP with no results. In addition, a 2021 survey of phthalates in Washington state water bodies did not measure concentrations of DCHP in water above reporting limits (0.51 ug/L) (WA DOE, 2022). One reference measured DCHP in dissolved water concentrations in surface waters in the United States and Canada. Seawater samples with column concentrations of up to 15 ng/L DCHP were measured in Puget Sound, WA and Barkley Sound, BC (Keil et al., 2011). This monitoring data further supports that exposure to DCHP via water would be limited.

The combination of transport and fate properties as well as limited monitoring data suggests people living around less regulated landfills will have limited exposure to DCHP from these pathways.

Summary: A public comment (SACC23-0063) described the Arctic as "fence-line" community based on global distillation of chemicals in colder environments, the potential for increased dietary exposure, and increased indoor exposures to volatile and semi-volatile chemicals through inhalation.

The same commenter (SACC23-0063) stated that EPA should consider exposure to volatile and semi-volatile chemicals through inhalation, not just ingestion of indoor dust to better represent exposures of Artic Indigenous peoples.

EPA Response: The phthalate esters under consideration in the TSCA Risk Evaluations are not likely candidates for long range transport due to the nature of the phthalate esters to both biodegrade, degrade via photolysis, and sorption to organic carbon present in the environment. Furthermore, the phthalate esters are not likely to appreciably bioaccumulate. In Section 3.3.2 of the 2025 CRA TSD, EPA identified DMR, NEI, and TRI data for DEHP, DBP, and BBP, but not for DCHP, DINP, and DIBP. These EPA databases provide information on facilities releasing phthalates to various environmental media and provide latitude and longitude data for releasing facilities. Using the release information, EPA identified 1,461 facilities that report use of a single phthalate, while 461 report use of multiple phthalates (*i.e.*, any combination of DEHP, DBP, or BBP). Individual facilities are broadly

dispersed around the United States. Of note, no releasing facilities are reported in Alaska. Therefore, "fence-line" exposure to the phthalate esters in Arctic communities was not considered in Arctic communities due to lack of releases in Alaska and Arctic regions of the United States reported in the above databases. However, EPA's screening-level assessment considered exposure directly near releasing facilities, including ingestion of fish from surface waters impacted by direct discharge, is inclusive of communities not near releasing facilities. Majority of exposures to phthalates by these communities are more likely attributable to use of consumer products and occupational exposures, in Arctic communities.

The indoor dust exposure assessment used the stay-at-home scenario parameter of 21 hours per day spent in the home and also evaluated inhalation and ingestion of suspended and settled dust. The commenter may have focused on the comparison between monitoring and modeling data that was performed for ingestion only to simplify calculations and to streamline the comparison. See Section 4 in the Consumer and Indoor Dust TSD documents for DEHP, DBP, DIBP, BBP, and DCHP. However, EPA assessed inhalation and ingestion for all indoor dust scenarios (see Section 2 and Table 2-1 in the Consumer and Indoor Dust TSD documents for DEHP, DBP, DIBP, BBP, and DCHP) and used the modeling results to assess risk. The stay-at-home parameter was applied to all the articles assessed in the indoor dust high, medium, and low intensity use exposure scenarios to ensure a protective duration of time spent in the home. The Consumer and Indoor Dust TSDs have clarifying statements in the indoor assessment section to explain that the indoor consumer articles exposure scenarios were modeled with stay-at-home parameters that consider use patterns similar to or higher than those in other indoor environments, including vehicles.

Cumulative Exposure of Potentially Exposed or Susceptible Subpopulations (PESS)

Summary: A public comment (DCHP-0121) expressed surprise and concern over the lack of special consideration for the PESS of Tribal populations in the cumulative risk assessment, stating "consideration of cumulative tribal risk is required under TSCA."

EPA Response: EPA also disagrees with the public commenter (DCHP-0121) that "consideration of cumulative tribal risk is required under TSCA." As described in the *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act*, TSCA does not explicitly require EPA to conduct cumulative risk assessments. While EPA is not required to consider cumulative risk, in applying cumulative principles to the phthalates the Agency considered high-end exposures that, based on available information, would appear to account for tribal exposures to phthalates. In addition, EPA did consider tribal exposures to each individual high-priority phthalate (DEHP, DBP, DIBP, BBP, DCHP) through ingestion of fish as part of its general population exposure assessment. However, as part of the general population screening-assessment EPA utilized high-end conservative assumptions, which are not appropriate for combining to estimate cumulative risk without further refinement.

Summary: Two public commenters (SACC25-0132, DCHP-0128) stated that using NHANES biomonitoring data from children age three to less than six underestimates background exposures to phthalates for infants within the CRA. One commenter (SACC25-0132) recommends using modeled exposure estimates for children under 3 years of age.

A public comment (DCHP-0121) notes that the NHANES dataset does not cover children <3 year of age. Given that "young children have distinct toxic exposures, consumptions per unit body weight, and corresponding metabolic rates", the comment concludes that "the current assessment may not protect children in the general population, let alone tribal children."

One of these public comments (DCHP-0128) notes that daily intake of phthalates trended down with age from 3–15, and therefore, the NHANES data from 3–5 years may underestimate exposures of infants and toddlers.

In DIBP-0133 the commenter stated that using biomonitoring data for children as a proxy for infants and toddlers, a subpopulation that is likely to face higher background exposures to phthalates, is inadequate. The commenter raised concern in using the NHANES urinary biomonitoring data for male children (three to less than six years of age) as a proxy for male infants (less than one year old) and male toddlers (one to two years old) to derive urinary metabolite concentrations for both BBP and DIBP. The cumulative risk assessment shows that exposure *decreases* as age increases, suggesting that daily intake of phthalates for an infant may be higher than for a child between the ages of three and six.

EPA Response: NHANES does not include urinary biomonitoring data for children less than three years of age and EPA did not identify any reasonably available national scale studies that provided recent urinary biomonitoring data for infants and children under 3 years of age. EPA used biomonitoring data from children aged three to less than six as a surrogate for these younger lifestages. Although one commenter (SACC25-0132) suggests the use of exposure modeling to predict exposures for children under the age of three, the commenter did not identify any specific models that they recommend EPA use to estimate cumulative background exposure for infants and EPA is unaware of any such reasonably available model or data to accomplish the public commenters suggestion.

As part of its phthalate risk assessment, U.S. CPSC (2014) used urinary biomonitoring data from 258 infants (0–37 months of age) collected between 1995 to 2005 (see Table 2.7 in U.S. CPSC 2014) to calculate phthalate daily intake values via reverse dosimetry. Using the infant daily intake values reported by U.S. CPSC (2014), and the RPFs and index chemical POD calculated as part of EPA's current CRA, EPA calculated cumulative MOEs of 192 (indicates risk cup is 16% full) and 48 (indicates risk cup is 63% full) for infants based on mean and 95th percentile exposure estimates, respectively (see Table 1 below). As can be seen from Table 1 below, DEHP and DBP were the primary contributors to the risk cup for infants. However, temporal trends analysis of NHANES urinary biomonitoring data demonstrates large declines in human exposure to phthalates, including DEHP and DBP. For example, creatinine corrected 95th percentile urinary concentrations of DBP metabolites and DEHP metabolites have declined 59 to 88% in adults from the 2003–2004 NHANES cycle to the 2017–2018 NHANES cycle (see Table 2 below). Similar large declines in urinary phthalate metabolite concentrations would also be expected for infants.

Although there is some uncertainty in current urinary concentrations of phthalate metabolites for infants (since EPA didn't identify any recent, national scale infant urinary biomonitoring data), EPA is confident in its risk conclusions and is confident that it has not meaningfully underestimated cumulative exposure or risk for infants by using children aged three to less than six as a surrogate for infants.

Table 1. Cumulative Risk Estimates for Infants Using Biomonitoring Data from U.S. CPCS (2014)

Population	Exposure Estimate	Phthalate	Intake (ug/kg-day)	RPF	RPF Adjusted (µg/kg-day)	% Contribution	Cumulative Daily Intake (µg/kg-day)	Cumulative MOE (Benchmark = 30)	% Contribution to Risk Cup
Infants (0-37 months)	Mean	DBP	2.6	1	2.6	23.7	10.9612	192	16%
		DIBP	0.44	0.53	0.2332	2.1			
		BBP	1.9	0.52	0.988	9.0			
		DEHP	7.6	0.84	6.384	58.2			
		DINP	3.6	0.21	0.756	6.9			
	95th	DBP	10.4	1	10.4	23.7	43.821	48	63%
		DIBP	2.1	0.53	1.113	2.5			
		BBP	8.5	0.52	4.42	10.1			
		DEHP	28.7	0.84	24.108	55.0			
		DINP	18	0.21	3.78	8.6			

Table 2. Decline in DBP and DEHP Urinary Metabolite Concentrations in NHANES*

Parent Phthalate	Urinary Metabolite	NHANES Cycle	Metabolite	Age Group	Sample Size	Creatinine Corrected 95th Percentile (95% CI) (ng/mL)	% Decrease
DBP	MnBP	2003–2004	MnBP	Adults	1,889	83.64	59
		2017–2018	MnBP	Adults	1,896	34.4	
DEHP	MECPP	2003–2004	MECPP	Adults	1,889	241.83	85
		2017–2018	MECPP	Adults	1,896	36.59	
	MEHHP	2003–2004	MEHHP	Adults	1,889	174.4	88
		2017–2018	MEHHP	Adults	1,896	21.7	
	MEHP	2003–2004	MEHP	Adults	1,889	22.47	77
		2017–2018	MEHP	Adults	1,896	5.18	
	MEOHP	2003–2004	MEOHP	Adults	1,889	111.06	86
		2017–2018	MEOHP	Adults	1,896	15.15	

* NHANES Urinary Metabolite Concentration data from EPA's General Population Exposure Assessments for DEHP and DBP.

6.6 Other Comments on Human Risk Characterization

Summary: One commentor (SACC-23) expressed general concern with EPAs use of biomonitoring data, which they believe should be used on a case-by-base bases and should not be generalized in the CRA.

"While [the commentor] recognizes that biomonitoring data, like other kinds of data, may have challenges, [the commentor] encourages caution regarding generalized statements about biomonitoring limitations and instead encourages EPA to consider biomonitoring data on a case-by-case basis, in accordance with TSCA's scientific standards."

The commentor recommended looking to Germany (Schultz, 2007) and Canada (Faure, 2020) guidance literature on how to best apply biomonitoring data within a regulatory context and avoid over-generalization.

The commentor does not believe that the current application of biomonitoring data within the CRA is in line with TSCA's scientific standards.

EPA Response: As described in the general population exposure TSDs for DEHP, DBP, DIBP, BBP, and DCHP, as well as the 2025 phthalate CRA TSD, EPA used 2017–18 NHANES urinary biomonitoring data and reverse dosimetry to estimate aggregate exposure to DEHP, DBP, DIBP, and BBP, as well as cumulative exposure to DEHP, DBP, DIBP, BBP, and DINP for the general U.S. population. Notably, aggregate and cumulative MOEs were greater than the benchmark of 30 for all assessed populations at the 95th percentile.

As part of its phthalate CRA, background cumulative exposure, as estimated from CDC's NHANES dataset, was combined with aggregate exposure from individual COUs for each individual phthalate risk evaluation. EPA believes that it has sufficiently documented the strengths as limitations of NHANES biomonitoring data (see the 2025 CRA TSD and the 2023 phthalate CRA proposal) and has used NHANES biomonitoring data in a fit for purpose manner consistent with the best available science in the risk evaluations for DEHP, DBP, DIBP, BBP, and DCHP.

Summary: A public commenter (SACC25-0132, SACC25-0145) maintained that the current margin of exposure (MOE) approach used by EPA for assessing non-cancer risk is not the "best available science" and states that EPA should conduct a probabilistic dose-response assessment using methods of the World Health Organization's IPCS to estimate the risk of adverse effects at various levels of exposure. The commenter (SACC25-0145) provides specific examples of the application of probabilistic risk analysis in the comments for DBP and in attachment 3 for DEHP. Furthermore, they contend that to the extent EPA relies on the MOE approach, it is that essential that uncertainty factors are included to prevent the underestimation of risk.

Public commenters (DCHP-0127, DEHP-0138) stated that "EPA should apply best available methods to generate quantitative estimates of non-cancer risks for varying levels of exposure to..." DBP, DCHP, and DEHP. The commenters explained that "The National Academies and the World Health Organization (WHO) have outlined more robust methods for risk estimation that more accurately account for variability in the human population and have been demonstrated in published case studies. The commenter (DCHP-0127) "…applied the WHO methodology to the DCHP endpoint of reduced

fetal testosterone, using the BMD and BMDL values derived by EPA through application of the NASEM meta-regression model, to estimate risk-specific doses for several levels of incidence (*e.g.*, 1%, 0.1%, etc.)" and supplied results. The commenter concluded that "EPA should apply the WHO framework to the DCHP male reproductive toxicity and liver toxicity endpoints, using appropriate BMD estimates." The commenters (DEHP-0138) provided similar analyses for the DBP endpoint of reduced fetal testosterone and the DEHP endpoint of increased male reproductive tract malformations. These commenters also suggested that the WHO framework be applied for other DBP and DEHP health endpoints.

EPA Response: EPA disagrees with the public commenters. EPA will utilize current policies, models, and screening methods, but is committed to being consistent with the best available science and weight of scientific evidence approaches to guide the Agency in using this information. EPA recognizes the advancing science to inform risk evaluation and will not discourage the use of new methods as long as they are consistent with the standards in section 26 of TSCA. EPA also recognizes that different approaches require different types and amounts of data and will select and employ methods that are fit for purpose within the context of a particular risk evaluation. In some cases, it may be necessary to utilize default parameters in modeling and risk calculations, and to utilize conservative assumptions, whereas in other cases assumptions may be replaced with specific or specialized data. EPA has utilized the MOE approach in previous risk assessments, citing its utility. However, EPA does agree with comments that there are numerous ways to characterize risk, of which the MOE is just one. There will be risk scenarios where one approach may be better than another and, as commenters correctly pointed out, the science of risk characterization is still evolving, particularly for non-cancer hazards. For the risk evaluations of DEHP, DBP, DIBP, BBP, and DCHP, EPA used the MOE approach to characterize risk, which EPA felt was the most suitable approach for the risk evaluations of DEHP, DBP, DIBP, BBP, and DCHP given the reasonably available data, the utility of the MOE approach, and the precedent for using the MOE approach in previous TSCA risk evaluations. EPA does not agree with the commenter to use another approach.

As for a probabilistic approach specifically, EPA is in the early stages of research associated with developing probabilistic methods and guidance for use in human health hazard assessment. Until this research is matured and completed, EPA will continue to use the approaches described in existing EPA guidance documents for using default values (U.S. EPA, 2002) and for developing refined values (*e.g.*, 2014 Guidance for Applying Quantitative Data to Develop Data-Derived Extrapolation Factors for Interspecies and Intraspecies Extrapolation; (U.S. EPA, 2014). There is no current policy for determining appropriate regulatory thresholds for results of a probabilistic analysis. Until probabilistic methods are standardized into guidance, the EPA does not wish to speculate on estimates of percent population affected.

Non-Chemical Stressors

Summary: Two public comments (DCHP-0121, DCHP-0128) recommended consideration of non-TSCA and non-chemical stressors in the cumulative assessment of phthalates, especially as PESS may be disproportionally impacted by multiple non-chemical stressors.

One comment (DCHP-0121) acknowledged that the *Draft Risk Evaluation of DCHP* did not identify statistical differences in "cumulative exposure between races and socioeconomic status for women of reproductive age," but states that "the relative effects of these exposures may be incredibly different

due to the synergistic effects of phthalate exposure and other chemical and non-chemical stressors that these subpopulations are exposed to."

The other comment (DCHP-0128) recommended application of an uncertainty factor in the absence of sufficient data to evaluate this quantitatively.

EPA Response: EPA disagrees with public commenters (DCHP-0121, DCHP-0128) that it needs to quantitatively consider non-chemical stressors and synergistic effects. Few methods have been developed that allow for a quantitative analysis of cumulative risk from combined exposure to chemical and non-chemical stressors. Until Agency-wide guidance and established methodologies have been developed, EPA does not expect to quantitatively evaluate non-chemical stressors when conducting CRAs under TSCA. Further, as discussed in Section 4.2 of the *Draft Proposed Approach for Cumulative Risk Assessment of High-Priority Phthalates and a Manufacturer-Requested Phthalate under the Toxic Substances Control Act* (U.S. EPA, 2023), empirical data from *in vivo* phthalate mixtures studies demonstrate that dose addition models accurately predict observed mixtures effects, indicating that synergistic effects resulting from phthalate exposure are not expected.

EPA identified a range of factors that may have the potential to increase biological susceptibility to DEHP, DBP, BBP, DIBP, and DCHP, for example lifestage, pre-existing diseases, physical activity, diet, stress, and co-exposures to other environmental stressors that contribute to related health outcomes. As applicable, EPA has included qualitative discussion of these factors in Section 5 of the human health hazard TSDs for DEHP, DBP, DIBP, BBP, and DCHP.

EPA disagrees with the public commenter (DCHP-0128) that the application of an uncertainty factor is needed in the absence of sufficient data to evaluate non-chemical stressors quantitatively. As discussed above, Agency-wide guidance and established methodologies have been developed to address non-chemical stressors, this includes application of a potential uncertainty factor.

As outlined in the *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances* (U.S. EPA, 2021), the study quality rating for epidemiological studies accounts for the consideration of non-chemical stressors, including confounders (*e.g.*, smoking, stress) and comorbidities (*e.g.*, diabetes) in risk estimates.

7 ENVIRONMENTAL RISK ASSESSMENT

Comments associated with this topic are summarized in the subsections below.

7.1 Environmental Exposures

No comments are associated with this topic.

7.2 Environmental Hazards

Addressing Data Gaps

Summary: A public comment (DCHP-0121) expressed concern that "EPA's determination of no unreasonable risk [to the environment for DCHP] is not based on "substantial evidence" as required under 15 U.S.C. § 2625(h) and 2618(c)(1)(B)." The comment recommended considering phthalates such as DEHP as analogues to fill data gaps for environmental hazards.

EPA Response: EPA has identified hazard thresholds for DCHP for chronic exposures to aquatic invertebrates and terrestrial vertebrates based on the reasonably available and substantial evidence. The resulting COC and hazard threshold is protective of vertebrates, invertebrates, algae, and terrestrial mammals. The environmental hazards of DCHP were considered and presented in the *Environmental Hazard Assessment for Dicyclohexyl Phthalate (DCHP) Technical Support Document for the Draft Risk Evaluation* and the *Risk Evaluation for Dicyclohexyl Phthalate (DCHP).* EPA also considered additional studies received from public comments between the draft and final versions of this technical support document, but none reported DCHP hazards.

EPA also considered the physical and chemical properties and the available environmental hazard data of DCHP in comparison to potentially analogous phthalates, including DEHP. The physical and chemical properties and existing data for DCHP are substantially different for the other phthalates (*e.g.,* DCHP is a solid at room temperature, while DEHP is a liquid), making direct extrapolations from DEHP to DCHP hazards (including dietary exposure by wildlife) tenuous.

EPA has indicated no risk to aquatic organisms under realistic release, flow, wastewater treatment, and solubility scenarios and provides the weight of evidence in the risk evaluation. The only exception was one scenario resulting RQ>1 in which the highest release volume was used, no DCHP was removed by wastewater treatment, and the slowest flow rate was modeled. This scenario resulted in DCHP concentration in the water that were over 2 orders of magnitude measured concentrations from U.S water bodies. Other scenarios with anything but the highest inputs and less conservative assumptions (*e.g.,* faster flow rates) resulted in RQ values <1.

DEHP Environmental Hazard

Summary: A public comment (DBP-0127) on Charge Question 2 regarding EPA's derivation of an avian hazard value for DEHP stated that "EPA should provide greater transparency for these *ad hoc* screening approaches before a *draft* risk evaluation is released". The comment stated that while "the higher level observation of low exposure and subtle effects at a high dose may be sufficient to make a qualitative decision that DEHP is not likely to present unreasonable risk to birds", the relevance of aspects of the analysis were "questionable" – specifically, the relevance in using a single (acute) dose to eggs when compared to environmental concentrations (*i.e.,* chronic concentration), as well as the

use of a non-standard behavioural endpoint ("a decrease in imprinting preference scores, in the newly hatched chicks (14 to 24 hours old)").

Another public comment (SACC- 0148) stated that, in their opinion, there are no adequate data available for use in developing an Avian Hazard value. For the study by Wood and Bittman (1980), the commenter recommended that EPA calculate the achieved oral dose of DEHP by interpolating the food consumption data from the graph and using it along with the body weight data reported in a table, and the body weight of the chicken hen (*Gallus gallus domesticus*) to calculate the dose in mg/kg-day. However, they ultimately agree that EPA recognized that the Wood and Bitman study suffers from a number of significant inadequacies and rightly excluded it as a candidate. The Abdul-Ghani et al study "combined the results from the DEHP-treated groups and with those in the DBP-treated groups, making it impossible to attribute the findings to DEHP, DBP or both." The EPA did not provide criteria or justification for its declared LOAEL of 100 mg/kg. The commenter also pointed out errors in the text for the charge questions (claiming they incorrectly reference section 5.1 instead of 5.2 of the draft environmental hazard assessment for DEHP), and selection of the data set for the terrestrial mammal hazard threshold.

EPA Response: The study used for avian hazard threshold determination was from Abdul-Ghani (2012). However, the hazard threshold was revised and based developmental malformations including gastroschisis and omphalocele in the chicken (*Gallus gallus domesticus*) from pre-hatch single dose egg injections into the albumen of 0, 5, 20, 50, and 100 mg DEHP resulting in a NOAEL/LOAEL of 5/20 mg/kg of egg (Abdul-Ghani, 2012). EPA has identified the geometric mean of the NOAEL/LOAEL of 10 mg/kg of egg from egg injection for the avian hazard threshold. The risk evaluation has been updated with language and justification for the modified avian hazard threshold value.

Although Wood and Bittman (1980) did not report achieved dose, EPA has estimated a mean achieved dose of approximately 578 mg/kg-day in the treated groups using the mean body weight data presented in a table and an interpolated estimate of the feed consumption presented in a graph. However, the DEHP incorporated into the feed apparently altered palatability, with feed consumption significantly decreased by 10% compared to controls over the 4-week period. This study was excluded from quantitative use in hazard determination due to significant apparent food aversion occurring in chicken exposed to DEHP in feed at this concentration, and some uncertainty around the estimation of the achieved dose of approximately 578 mg/kg-day.

In addition, supporting studies have been introduced further characterizing the effects of DEHP on avian species on cardiac and renal tubular epithelial cells, respectively, in addition to a multitude of mechanistic effects recorded in both studies (Wang, 2019; Wang, 2020). In both studies, the NOAEL/LOAEL were <250/250 mg/kg-day based on swelling and dilation of cardiac cells (Wang et al., 2019), and disorganized renal structure, a partially dilated glomerulus, renal interstitial congestion, and an atrophied Bowman's space (Wang et al., 2019). These studies further describe the effects of DEHP on avian species.

Summary: A public comment [DBP-0127] on Charge Question 3 regarding the methodology used to derive a DEHP chronic concentration of concern for aquatic invertebrates stated that "EPA needs to consider the weight of the scientific evidence in conducting an ecological hazard characterization and put it in the context of likely exposures for each environmental medium" – especially in the context of developing a quantitative hazard threshold for DEHP in which 73 (out of 82) studies were not considered as they "either demonstrated no acute or chronic effects at any concentration tested, or the reported hazard values exceeded the limit of solubility of 3.0 µg/L selected by EPA (U.S. EPA, 2024b)."

Another comment (SACC- 0148) stated that "it appears the agency has mistakenly substituted "aquatic invertebrates" for "aquatic vertebrates" in this question." The commenter also said the EPA should provide an explanation for dismissing one aquatic species hazard study (Corradetti et a. 2013). The majority of studies showed no effects up to the limit of solubility, so the level of EPA confidence that DEHP poses a chronic hazard should be reduced to "moderate."

EPA Response: While many acute toxicity studies showed no effects up to and exceeding the limit of solubility, chronic effects were seen in most studies received by systematic review. Only studies that had endpoints less than the EPA-selected limit of solubility were considered for hazard threshold determination. The quality of the database and relevance of confidence metrics were two of the five measures classified as robust to determine overall confidence for chronic aquatic endpoints. Consistency of hazard metrics was classified as robust for acute aquatic organism endpoints in addition to quality of the database and relevance. Given the number of unbound hazard values for both acute and chronic aquatic studies indicating either on effects observed greater than the highest concentration tested, or effects observed at concentrations less than the lowest concentration tested, the biological gradient (dose-response) received slight confidence, while the strength and precision of acute aquatic study hazard values was robust and for chronic studies it was moderate. Given the overall strength of the data, both acute and chronic aquatic studies received robust confidence rankings.

When the confidence for the different evidence categories is unequal and vary in weight, professional judgement, a descriptive narrative of the weight of scientific evidence, and uncertainties factor into the rationale for weighing and weighting the evidence that supports the overall confidence in the risk characterization. It is important to note that the overall confidence in the risk characterization is not a statement of confidence on whether there is risk (RQ ≥ 1), or the magnitude of risk, *i.e.*, how much the RQ is greater than one. Rather, it is a statement of confidence in the inputs used to calculate the RQ. Hazard confidence is only one part of the overall risk characterization/risk estimation confidence. The exposure confidence for aquatic organisms at a chronic exposure basis was rated as robust (acute aquatic organisms was not assessed quantitatively). Therefore, the overall confidence was rated as robust for aquatic organisms on a chronic exposure basis in the risk characterization.

Corradetti (2013) was not dismissed from consideration within the *Environmental Hazard Assessment for DEHP Technical Support Document for the Draft Risk Evaluation.* That Technical Support Document reports, "A chronic fish study that received a medium-quality ranking was conducted with DEHP on zebrafish (*D. rerio*) over 21 days to measure reproductive effects at 0.2 and 20 µg/L (Corradetti et al., 2013). At the end of the study, significant increases in GSI and decreases in embryo number and hatching rate percentage were observed at both concentrations tested. The study authors concluded that exposure to DEHP at environmentally relevant concentrations could negatively affect fish reproduction (Corradetti et al., 2013)." The results of hazard values and endpoints from Corradetti (2013) are presented among a summary of other relevant studies in Table 3-1 of the *Environmental*

Hazard Assessment for DEHP.

Consistent with the recommendations of the public commenter (SACC- 0148), EPA revised charge question 3 prior to the SACC peer-review meeting to ensure appropriate use of the terms aquatic vertebrates and aquatic invertebrates.

Use of Omics Data for Ecological Hazard Identification

Summary: A public comment [DBP-0127] on Charge Question 4 regarding the use of transcriptomics data as supporting information for ecological hazard identification for BBP, DBP, and DIBP stated three recommendations: 1) "EPA should develop a validated approach for tPOD/mPOD derivation before 'omics-based PODs are integrated into EPA environmental hazard assessments" (*e.g.*, referencing the process followed by the EPA ETAP study), 2) "EPA should make available the unpublished report that contains the details of the development of the 'omics-based PODs for DBP, BBP, and DIBP" (*e.g.*, referencing the unpublished report of Bencic et al. 2024, not being made available in EPA dockets for review and comment and requiring a FOIA request), and 3) "EPA should clarify why an ontology-based method was used over gene-based method for defining the ecological tPOD".

Another public commenter (SACC25-0148) also critiqued EPA's draft hazard assessments for DBP, BBP, and DIBP, highlighting a lack of meaningful integration of transcriptomic and metabolomic Points of Departure (POD) into the determination of Concentrations of Concern (COC) for acute exposure scenarios. The commenter explains that EPA did not employ metabolomic PODs (mPODs) or transcriptomic PODs (tPODs) for deriving hazard thresholds, citing uncertainties about their ecological relevance. Despite claims that multiomics PODs for DBP align with SSD-derived COCs, the commenter argues this confidence is overstated given current knowledge. The commenter concludes that tPOD and mPOD data are insufficient to support chemical-specific COCs, either alone or with SSD methods, contributing minimally to confidence in acute hazard values. Furthermore, no suitable 'omics data exist for chronic exposure assessments, leaving them uninfluential in determining confidence in chronic hazard values.

EPA Response: EPA notes that the development of validated methods for tPOD/mPODs is a goal of the workgroup that produced the referenced transcriptomic data, and that further development is progressing under the research group identified in the Bencic et al. 2024 paper. A manuscript outlining this work is being prepared for publication in a peer reviewed journal, but is not yet available to the public.

While the tPOD and mPODs were not used to derive hazard thresholds for these chemicals because high-quality apical studies were available, EPA believes that the inclusion of these methods improves our confidence in the acute hazard values, because the tPOD and mPODs align well in most cases with the chosen COC. EPA disagrees that, in principle, a tPOD or mPOD would be insufficient to support a chemical-specific COC in the absence of high-quality apical studies, because the forthcoming Bencic et al. 2024 paper demonstrates that tPOD and mPODs in question align well with hazard thresholds derived from traditional apical endpoints.

Regarding the use of the Gene Ontology framework, there are many acceptable methods for the derivation of tPODs, and method development remains an active area of research and debate. Several studies have sought to compare methods with the goal of determining the best approach, however, the

relative rankings of methods appear to differ with experimental parameters (exposure durations, chemical class, etc), suggesting that there isn't a one-size-fits all best approach. EPA used gene set summarization, which aggregated genes based on biological processes. Because this approach relies on a common biological context for the group of genes used to derive the BMD, it has intuitive appeal while also comparing favorably to other methods in head-to-head comparisons. More importantly, this method was also employed in EPA's Transcriptomic Assessment Product (ETAP), which had recently undergone a favorable review by the EPA's Board of Scientific Counselors (https://www.epa.gov/etap/about-epa-transcriptomic-assessment-product-etap). As the ETAP served as the most up to date standard methodology that had been reviewed for the specific purpose of deriving points of departure for use in chemical risk assessments, we judiciously adhered to the methods described in the ETAP for data processing and the derivation of BMDs.

Summary: A public comment [SACC25-0148] highlights a possible error in EPA's draft document on DEHP's chemical risk assessment substituting "aquatic invertebrates" for "aquatic vertebrates" in a Charge Question. The commenter assumes the agency intended to address chronic hazard values for aquatic vertebrates, as the document suggests adequate data for chronic values but insufficient data for acute values. Despite the majority of studies showing no effects up to the solubility limit, the EPA maintains robust confidence in DEHP's chronic hazard potential below this limit. The commenter suggests this introduces uncertainty and recommends adjusting the confidence level to "moderate." The consistency across dose-response results in the selected studies provides a strong foundation for the NOAEC and LOAEC values, indicating strengths in the assessment process, yet the comment calls for improved clarity on study exclusions and confidence rationale to enhance the robustness and transparency of the EPA's risk assessment.

EPA Response: Section 5 of the *Environmental Hazard Assessment for DEHP Technical Support Document for the Draft Risk Evaluation* details the Weight of Scientific Evidence for Environmental Hazard with narrative on the following criteria for assessing confidence for aquatic hazard studies: Quality of the Database; Consistency; Strength (Effect Magnitude), and Precision; Biological Gradient/Dose-Response; Biological, Physical/Chemical, Environmental Relevance. EPA has robust confidence that DEHP poses hazard to aquatic vertebrates from chronic duration exposures. Robust confidence for hazard below water solubility is supported by two studies in which effects on mortality, growth, and development were observed in Japanese medaka exposed to 0.1 µg/L DEHP for 21-d (Chikae et al., 2004a; Chikae et al., 2004b) and further supported with additional chronic hazard studies conducted by Golshan et al., 2015, Corradetti et al., 2013, and Zanotelli et al., 2010. These studies reported effects on mortality, growth, reproduction, and development at concentrations ranging from 0.01 to 10 µg/L and exposure durations ranging from 21 to 91 days. For benthic sediment dwelling invertebrates, EPA has moderate confidence based on effects observed on growth and development. This confidence is supported by one study in which effects on growth were observed in midge exposed to 0.3 µg/L DEHP (Kwak et al., 2005). Although not used for COC determination, a pelagic invertebrate study with the marine copepod (*Parvocalanus crassirostris*) also showed effects around a similar threshold of less than 0.3 µg/L (Heindler et al., 2017). This study was not considered for COC calculations due to analytical measurement concerns and background concentrations of DEHP.

Aside from three studies (two vertebrate and one invertebrate study), the remaining non-dietary aquatic studies not considered quantitatively were almost exclusively conducted at concentrations greater than the EPA-determined limit of water solubility.

While most acute toxicity studies (*i.e.*, less than 96-hours in duration) showed no effects, aquatic studies of chronic duration were more inconclusive. Six chronic aquatic fish studies showed no effects up to the highest concentration tested, three studies showed effects less than the lowest concertation tested, and five studies showed effects within the range of test concentrations.

Six chronic aquatic invertebrate studies showed no effects up to the highest concertation tested while seven studies demonstrated effects within the concentration range.

Four chronic and subchronic sediment-dwelling invertebrate studies showed no effects at the highest test concentration while one study showed effects less than the lowest concertation tested.

EPA asked SACC to provide input on this topic in Charge question 3. In its response SACC stated, "There are clearly documented adverse effects of chronic exposure to phthalates including toxic and endocrine disruption, especially DEHP in fish (see the Draft Environmental Hazard Assessment for Diethylhexyl Phthalate (DEHP)). There was very strong evidence comprised of 82 studies of high or medium quality that provided data useful for deriving a CoC and verifying the potential adverse effects of exposure to DEHP, including one study that employed direct exposure to its metabolite, MEHP (Ye et al. 2014)". Furthermore, SACC supported the selection of the Chikae et al. 2004 for its chronic aquatic vertebrate as a Concentration of Concern and noted "…confidence in this decision is increased by the fact that the other three studies, in different species, found similar NOAECs and LOAECs (Table 3-1 of the *Environmental Hazard Assessment)"*. The SACC conducted an SSD on chronic exposure data as presented within a recent published study by Hong et al. (2024), however, the application of SSD methods for chronic exposure datasets are not entirely appropriate. Chronic toxicity data are not as amenable to Species Sensitivity Distributions (SSD) model fitting and other probabilistic procedures as are toxicity data from acute experiments. Chronic toxicity data encompass different endpoints (*e.g.*, reproduction, growth, development, molecular, cellular, etc.), different exposure durations, and nonequivalent statistical estimates. EPA is unaware of any scientifically validated procedures for standardizing across these endpoints, durations, and statistics. Also, the available chronic endpoint data have less taxonomic coverage and may have different modes of action acting across different physiological effects.

Other Environmental Hazard Comments

Summary: A public comment (DCHP-0117) asked how DCHP effect honeybees. The comment noted a decline of pollinators and posited that impacts on male humans may correspond to impacts on drone bees.

EPA Response: EPA found no reasonably available evidence of the hazardous effects of DCHP on honeybee survival, development, reproduction, or behavior.

BBP and DIBP Environmental Hazard

Summary: A public comment (DIBP-0125) indicated that EPA should follow its own guidelines when conducting risk evaluations, specifically to both the scale of the assessment and the critical ecological endpoints and ecosystem and receptor characteristics. The scale of the assessment is local and site specific with emissions being limited to the point source. EPA should make it clear the scale of the sites being assessed and how it will conduct the assessment at those sites. The ecological receptors in freshwater, sediment, soil, and terrestrial ecosystems have not been explicitly stated in any scoping document. Regarding unreasonable risk to the environment, EPA should indicate the decision criteria that are being applied and the level of exposure that constitutes unreasonable risk.

EPA Response: EPA considered all available studies to characterize the environmental hazards of BBP and DIBP to surrogate species representing various receptor groups, including aquatic vertebrates, aquatic invertebrates, amphibians, aquatic plants, algae, and avians. EPA applies a systematic review process to identify information and derive hazard thresholds across taxonomic groups for both aquatic and terrestrial organisms with a focus on apical endpoints (*e.g.*, those affecting survival, growth, or reproduction) that most affect populations of organisms. In other words, the ecological receptors were populations of aquatic and terrestrial organisms most likely to be exposed to BBP and DIBP in various media. The risk characterization and risk assessment approach that use hazard thresholds for these ecological receptors are described in Section 5.3 of the *Risk Evaluation for Butyl Benzyl Phthalate (BBP)* and the *Risk Evaluation for Diisobutyl Phthalate (DIBP)*. This section also describes the use of Risk Quotients (RQs) in risk decisions and characterizations. The relevant exposure pathways of BBP and DIBP into the environment and the scale of the assessment are summarized in Section 5.3.2 of the *Risk Evaluation for Butyl Benzyl Phthalate (BBP)* and Section 5.3.1 of the *Risk Evaluation for Diisobutyl Phthalate (DIBP)* and described in the more detail in Section 3.3 of the *Risk Evaluation for Butyl Benzyl Phthalate (BBP)* and the *Risk Evaluation for Diisobutyl Phthalate (DIBP)* which also contains links to the Technical Support Documents that further describe the scale of the assessments.

Summary: A public comment (DIBP-0125) stated that EPA should be consistent for deriving a concentration of concern. DBP was used as an analog and read across for DIBP. EPA used a probabilistic approach to derive an aquatic acute COC for DIBP from an SSD that contained 96-h LC50s for nine species. However, EPA used a deterministic approach for chronic hazards, selecting the toxicity study of invertebrates and vertebrates from read across from the DBP environmental hazard data. The commenter stated that given the abundance of chronic aquatic toxicity data, EPA should prepare a chronic SSD to determine an appropriate COC for DBP that can be used for both DBP and DIBP.

EPA Response: Chronic toxicity data are not as amenable to Species Sensitivity Distributions (SSD) model fitting and other probabilistic procedures as are toxicity data from acute experiments. Chronic toxicity data encompass different endpoints (*e.g.*, reproduction, growth, development, molecular, cellular, etc.), different exposure durations, and nonequivalent statistical estimates. EPA is unaware of any scientifically validated procedures for standardizing across these endpoints, durations, and statistics. Also, the available chronic endpoint data have less taxonomic coverage and may have different modes of action acting across different physiological effects. Using SSDs derived from chronic endpoints would introduce myriad additional uncertainties that might obscure the interpretations needed for clear risk characterizations. Thus, EPA relied on population-level and

relevant endpoints from studies with clear dose-responses, exposure durations, and adverse effects to determine protective chronic COCs.

Summary: A public comment (BBP-0123) supported EPA using 1.9 µg/L of BBP as the concentration of concern (COC) for risk to aquatic species.

EPA Response: EPA acknowledges the public comment for support of the chronic COC for aquatic vertebrates.

Summary: A public comment (DIBP-0131) agreed with EPA's use of the Species Sensitivity Distribution (SSD) for acute aquatic toxicity but submits that EPA overlooked a relevant study in its assessment, and instead relied on read-across from data on DBP as an analog. The commenter reported that this 72- hour growth inhibition study in algae (LyondellBasell, 2010), submitted under TSCA Section 8(e), established a NOEC at 0.35 mg/L and a LOEC at 0.90 mg/L, from which the commenter derived a ChV of 0.56 mg/L as the geometric mean of the NOEC and LOEC.

For a chronic CoCs, the commenter recommended using DIBP specific data to derive chronic CoCs as follows: (1) for vertebrates, an acute study in fish (Bencic, 2024) with a 24-hour LC50 of 5300 µg/L divided by an acute-to-chronic ratio (ACR) of 10 to get a ChV of 530 µg/L, divided by an assessment factor (AF) of 10 to derive a chronic CoC of 53 (compared to EPA's ChV of 1.56 µg/L based on read-across from DBP chronic aquatic toxicity data); (2) for invertebrates, the commenter recommended an acute study in copepods (Linden, 1979).with a 96-hour LC50 of 3000 µg/L divided by an acute-to-chronic ratio (ACR) of 10 to get a ChV of 300 µg/L, divided by an assessment factor (AF) of 10 to derive a chronic CoC of 30 (compared to EPA's ChV of 12.23 µg/L based on read-across from DBP chronic aquatic toxicity data; and (3) for plants/algae, the commenter recommended that the POD be based on a 72-hour algal growth inhibition study for which the chronic endpoints (*i.e.*, NOEC and LOEC) may be used to derive directly the COC without application of an ACR.

EPA Response: The commenter stated that EPA should use empirical data on the chemical of interest when available over analogue data but also acknowledges that there are only two other acute studies on DIBP, while at the same time saying that they supported the use of an SSD for acute aquatic toxicity. EPA's guidance on SSD requires at least 8 species, and the durations and endpoints should all be similar (*e.g.*, 96-hour mortality). Therefore, using only three studies exclusive to DIBP would provide insufficient data to run an SSD, and it would be inappropriate to include the algal study on growth inhibition, both regarding the duration relative to the life history of this taxa and the inclusion of an endpoint different than mortality. EPA states in section 6.1 of the DIBP Eco Hazard TSD "Plant and algae data 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". EPA derived the aquatic acute COC for DIBP from an SSD that contained 96-h LC50s for nine species identified in systematic review (two species with DIBP hazard data and seven species with DBP hazard data), bolstered by an additional 72 predicted LC50 values from the Web-ICE toxicity value estimation tool. All studies included in the SSD were rated high or medium quality. The best fit curve for the low-end of the statistical distribution, resulting in an HC05 of 406 µg/L. After taking the lower 95th percentile of this HC05 as an alternative to the use of assessment factors, the acute aquatic COC for vertebrates and invertebrates is 287 µg/L. EPA considers the CoC based on this analysis to be both scientifically defensible and robust, especially considering DBP's appropriateness as an analogue, given that it is an isomer of DIBP and has very similar p-chem and

fate properties and toxicity. Further, the study suggested by the commenter (LyondellBasell, 2010) lacked significant methodological information, including sample size and composition of negative control.

Examination across the multiple lines of evidence DBP is an appropriate analog with high and medium quality aquatic, sediment-dwelling, and terrestrial hazard data to be used in a read-across to DIBP. Detailed within Appendix A (Analog Selection for Environmental Hazard) of the *Environmental Hazard Assessment for Diisobutyl Phthalate (DIBP)*, DBP was indicated as structurally similar to DIBP in AIM, OECD QSAR Toolbox, and the Cheminformatics Search Module. Further, the similarity in the physical, chemical, and environmental fate and transport properties support the ability to read across DBP hazard as well to supplement the DIBP environmental hazard data set. Lastly, comparison of the DBP analog empirical hazard data to corresponding ECOSAR toxicity predictions for DIBP shows agreement of hazard values well within 10-fold (Appendix A.3 of the *Environmental Hazard Assessment for Diisobutyl Phthalate [DIBP])*. Therefore, EPA has high confidence in the use of DBP as an analog in the absence of DIBP data as well as where limited DIBP data is available, including for chronic aquatic vertebrates and chronic aquatic invertebrates.

BBP Environmental Hazard

Summary: A public comment (BBP-0125) The findings of unreasonable risk to the environment are driven by freshwater emissions and potential effects to aquatic vertebrates from chronic exposures to those emissions. EPA should use appropriate hazard endpoints when evaluating potential risk of injury to the environment. It is unclear why EPA conducted omics-based points of departure when there was significant acute and chronic apical endpoints data available.

EPA Response: As stated in the *Environmental Hazard Assessment for Butyl Benzyl Phthalate (BBP) Technical Support Document for the Risk Evaluation*, TSCA section 4(h)(1)(B) requires EPA to encourage and facilitate the use of scientifically valid test methods and strategies that reduce or replace the use of vertebrate animals while providing information of equivalent or better scientific quality and relevance that will support regulatory decisions. In accordance with the EPA's New Approach Methods Work Plan, the EPA Office of Pollution Prevention and Toxics (OPPT) is developing new methods for use in TSCA risk evaluations. Specifically, a project was conducted to generate omics-based PODs and compared them to traditional endpoints using fathead minnow as the model organism for three of the phthalates undergoing a TSCA risk evaluation, including BBP (Bencic et al. 2024). The results suggest that fathead minnow larvae exhibited changes in gene expression, metabolite levels, and swimming behavior at sublethal concentrations of BBP. While hazard thresholds are usually calculated with *in vivo* data measuring an apical endpoint (*e.g.*, mortality, reproduction, growth), these mechanistic (transcriptomic and metabolomic) and behavior points of departure represent potential information that may be used for reducing the time needed for toxicity testing *in vivo* and provide an alternate method to characterize hazard as well as provide evidence for mechanisms of action. At this time, EPA has not used the omics-based PODs in the BBP risk evaluation. There are uncertainties with respect to the extent to which these sub-organismal and individual-level effects (*e.g.*, behavior) at short exposure durations are comparable to population-level outcomes, such as survival and reproduction, in wild fish populations.

Summary: A public comment (BBP-0125) EPA identified zebrafish (*Danio rerio*) as the most sensitive aquatic vertebrate for which a clear population-level fitness endpoint was available under conditions of chronic exposure (see blue box in Table 1). However, the draft risk evaluation indicates that the effects identified are 3% lower fecundity, 2% lower fertilization success, 100% increase in plasma vitellogenin and reduced gonad weight in males. It is unclear whether these effects are in fact adverse effects.

EPA Response: The characterization of this study was expanded and corrected between the draft and final versions to highlight that the BBP effects were more pronounced than initially stated in the draft. Specifically, chronic BBP exposure resulted in 17 percent lower fecundity (*i.e.,* Table 8 in Battelle, 2018) instead 3 percent previously stated in the draft version. The zebrafish (*Danio rerio*) was the most sensitive aquatic vertebrate to chronic BBP exposure. This 21-day reproduction test of zebrafish exposed to measured concentrations of BBP found 17 percent lower fecundity and 2 percent lower fertilization success in females in treatments with 33 µg/L BBP (LOEC). No effects were observed at 11 µg/L BBP (NOEC). These BBP effects on female zebrafish occurred in a monotonic dose-response manner with greater effects at higher BBP concentrations. Male zebrafish had higher gonad weight, gonadal-somatic index values, and body weight in treatments with 3.6 µg/L BBP (LOEC). These BBP effects did not increase at higher BBP concentrations but were consistently higher than in fish from control treatments. This combination of reproductive effects on multiple female and male zebrafish endpoints over chronic BBP exposures signified potential adverse outcomes to fish populations. A 17 percent reduction in fecundity may be an adverse effect on a fish population as it directly lowers population growth rates which compounds over multiple generations of exposure.

Summary: A public comment (BBP/DBP-0125) indicated that in the draft BBP hazard assessment EPA uses probabilistic approaches (*e.g.,* SSD) when data from at least eight species are available. When then acuate aquatic COC was determined, EPA used an SSD based on five fish species and six invertebrate species identified during systematic review. However, the chronic aquatic COC was not based on an SSD despise having eight high or medium quality studies for chronic aquatic toxicity. EPA should have used an SSD or other probabilistic analysis of chronic aquatic toxicity data. Additionally, DIBP used a deterministic approach when using the invertebrate and vertebrate data from DBP, even though DBP is even more data rich than BBP. EPA should prepare an SSD to determine the COC for DBP than can also be used for DIBP.

EPA Response: EPA uses SSDs to derive COCs when acute toxicity data from at least eight species are available and does not use SSDs for chronic toxicity COCs. The draft BBP Risk Evaluation did not specify “acute toxicity data”, but the final version has been amended to read “For aquatic species, EPA uses probabilistic approaches (*e.g.,* SSD) when acute toxicity data from at least eight species are available (Raimondo, 2010)”.

Chronic toxicity data are not as amenable to Species Sensitivity Distributions (SSD) model fitting and other probabilistic procedures as are toxicity data from acute experiments. Chronic toxicity data encompass different endpoints (*e.g.,* reproduction, growth, development, molecular, cellular, etc.), different exposure durations, and nonequivalent statistical estimates. EPA is unaware of any scientifically validated procedures for standardizing across these endpoints, durations, and statistics. Also, the available chronic endpoint data have less taxonomic coverage and may have different modes of action acting across different physiological effects. Using SSDs derived from chronic endpoints would introduce myriad additional uncertainties that might obscure the interpretations needed for clear

risk characterizations. Thus, EPA relied on population-level and relevant endpoints from studies with clear dose-responses, exposure durations, and adverse effects to determine protective chronic COCs.

Summary: A public comment (SACC25-0132) stated that EPA should have considered non-reproductive adverse effects of DBP and DEHP on terrestrial mammals beyond rodents. The commenters provided references to support statements that that developmental effects in rats and non-rodents showed similar sensitivity to DBP; similarly, studies evaluating the effects of DEHP on cows, boars, ferrets, sheep were provided for DEHP. The commenter states that EPA fails to meet statutory obligations under TSCA and may underestimate broader ecological risks by not considering this "reasonably available information."

EPA Response: EPA has robust confidence that DEHP poses potential hazard to terrestrial mammals. This conclusion is supported by evidence obtained from 26 laboratory rodent studies conducted for use as human health models. It is important to note that a hazard value was derived from the most sensitive ecologically-relevant endpoint from the data set. The terrestrial mammalian hazard threshold of 80.79 mg/kg-day is the geometric mean of the NOAEL of 48.58 mg/kg-day and LOAEL of 140.15 mg/kg-day based on a decrease in pup survival during lactation (Tanaka et al., 2002). Nearly all other rodent studies considered for hazard threshold determination were within an order of magnitude of the selected value.

EPA typically focuses on apical endpoints (*i.e.*, mortality, growth, reproduction, development, behavior). DEHP exposure to lactating cows resulted in deleterious effects on ovarian function, nuclear maturation, and oocyte developmental competence at a LOAEL of 100 mg/kg-day. In boars, DEHP exposure has been linked to impaired reproductive outcomes at doses of 300 mg/kg-day. A study in ferrets documented changes in liver weight and histological changes in reproductive organs at 650 mg/kg-day. These effects may not all be apical endpoints and are above (less conservative) than the selected hazard value for hazard to terrestrial mammals. The one value that is below the EPA-hazard value is in the domestic sheep of 25 mg/kg, resulting in reduced ovarian size and slowed ovarian growth rates. Although these effects are within the same order of magnitude of the current mammalian hazard threshold identified within the *Environmental Hazard Assessment for DEHP Technical Support Document for the Draft Risk Evaluation*, the route of exposure to DEHP within Herreros et al. (2013) was administered via intramuscular injections at 25 and 50 mg/kg three times a week for two months. The relevance of DEHP injected versus taken orally is important when examining hazard thresholds and comparisons to the landscape of studies with oral routes of exposure are difficult given large differences in how the chemical is introduced to the body.

7.3 Environmental Risk Characterization

Summary: A public comment (SACC25-0132) stated that EPA's failure to consider chronic risks to marine mammals is a serious omission from the risk evaluation as DEHP has been detected in the serum of marine mammals and has been linked to toxicity in marine mammals (*e.g.*, endocrine and oxidative stress response, thyroid hormone disruption, and modulation of transcriptional activity of thyroid receptors).

EPA Response: The environmental risk characterization was performed with the landscape of reasonably available hazard data from DEHP toxicity studies to evaluate risk to environmental receptors. For DEHP, environmental hazard data from environmental or toxicology studies identified during systematic review have used evidence that characterizes apical endpoints (*e.g.*, mortality, reproduction, growth); that is, endpoints that could have population-level effects such as reproduction, growth, and/or mortality. Biota monitoring studies within marine mammals have reported relationships with molecular and cellular endpoints, however, the ability to designate a hazard threshold from biomonitoring studies in marine mammals introduces many uncertainties that would limit confidence. EPA did use a mammalian endpoint to characterize risk to terrestrial mammals, which was derived from the NOAEL/LOAEL of 48.58/140.15 mg/kg-day (representing the maternal achieved intake during lactation), resulting in a geometric mean of 80.79 mg/kg-day as the hazard value for terrestrial mammals. This study was selected based on the most conservative value from a pool of potential studies that received an overall quality determination of high or medium from systematic review.

The DEHP Risk Evaluation conducts screening level assessments with respect to trophic transfer and ingestion rates of prey, sediment, soil, and water with high concentrations of DEHP to determine if exposure estimates approach concentrations that illicit harm within terrestrial and aquatic mammals and birds. The screening level assessments used a number of assumptions including maximum values for absorption from media and prey and also assume that the representative organisms are within the area affected (Area Use Factor) all the time. In addition, concentrations of DEHP used within these screening level assessments are above monitoring values reported within biota, waters, soil, and sediment.

The revised Terrestrial Risk Characterization section within the DEHP Risk Evaluation performed a screening level trophic transfer analysis to examine DEHP concentrations and ingestion rates for a bird species with maximum DEHP concentrations within biosolid amended soils and resulting prey items. Estimated DEHP soil concentrations following application of biosolids are detailed within EPA's *Environmental Media and General Population and Environmental Exposure for DEHP*. Using the highest calculated topsoil concentration of 6.25 mg/kg following an agricultural application of biosolids on soybeans, EPA assumed 100 percent uptake by a worm, so that the concentration of DEHP in the earthworm is equivalent to the soil concentration. The role of DEHP within water versus DEHP within sediment and prey can be examined using a maximum DEHP concentration from a groundwater sample within the EPA's Six-Year Review of Drinking Water Standards. The EPA Wildlife Exposure Factor Handbook (U.S. EPA, 1993) provides an estimated water ingestion rate (g/g-day) is 0.10, which would result in a total daily intake of DEHP from water of 1.3×10-2 mg/kg-day with a maximum absorption fraction of 1 as a screening level assumption. The contribution of potential DEHP from water is significantly less than intake from DEHP contaminated prey and incidental soil ingestion. Using the insectivorous Woodcock as a representative species with a daily feed intake rate (FIR) of 0.77 and incidental soil intake rate (SIR) of 0.16 as wildlife exposure factors (U.S. EPA, 1993) and assuming a 100% absorption fraction of that soil and the prey items (earthworms) resulted in a maximum daily concentration for oral uptake of DEHP of 6.22 mg/kg-day. This DEHP from soil and prey for the insectivorous bird is two orders of magnitude (~40 times) lower than concentrations resulting in subapical effects (250 mg/kg-day) from chronic feeding studies in Japanese qual (Wang et al. 2019; Wang et al. 2020) and one order of magnitude less than the DEHP mammal hazard threshold of 80.79 mg/kg-day from Tanaka et al. (2002).

For terrestrial mammals the EPA employed screening level concentrations within soil and prey to determine concentrations of DEHP from soil and prey ingestion and compare that to the hazard

threshold for mammals. Concentrations of DEHP in soil following agricultural application of municipal biosolids were not identified from TRI or the NEI release data nor were any monitoring studies identified during systematic review. As such, DEHP concentrations in soil were estimated using the concentrations identified in sludge, ranging from 0.657 to 0.31 mg/kg (U.S. EPA, 2009). The maximum biosolid topsoil concentration was estimated at 6.15 mg/kg based on application to soybeans (*Environmental Media and General Population and Environmental Exposure for Diethylhexyl Phthalate (DEHP)*). The mammalian hazard threshold was 80.79 mg/kg-day in the *Environmental Hazard Assessment for Diethylhexyl Phthalate (DEHP)*. Terrestrial organisms, such as mammals, would need to consume over 13 kg of DEHP-tainted soil or prey items to reach the threshold for toxicity.

Summary: A public comment (SACC25-0132) recommended that EPA evaluate the environmental risks of the DBP metabolite monobutyl phthalate (MBP) in the final risk evaluation. The comment states that MBP is expected to be photogenerated and released into the environmental upon disposal of DBP, but EPA did not provide sufficient justification to support the statement that MBP is "not expected to substantially contribute to risk." The comment also provided references to support a statement that MBP has ecological toxicity to aquatic organisms.

Similarly, DIBP-0130 and DIBP-0133, stated that EPA failed to consider the environmental toxicity of BBP's known degradation product MBP, which is generated and released into the environment during the disposal phase of BBP's lifecycle. DIBP-0133 further explained that "as a photodegradation product of BBP, MBP is expected to be generated and released into the environment during the disposal phase of BBP's lifecycle. EPA claimed that MBP is both more soluble and more bioavailable than BBP and is also expected to undergo biodegradation more rapidly than the diester form. However, these characteristics alone do not support the conclusion that MBP poses no substantial environmental risk. Even if MBP degrades more rapidly, its higher solubility and bioavailability may increase short-term exposure and toxicological relevance. Several studies that are not considered by EPA have documented MBP's ecological toxicity as to aquatic organisms, such as inducing energy metabolism disturbances in the gills. These studies demonstrate that MBP can exert toxic effects on aquatic organisms at environmentally relevant concentrations."

EPA Response: EPA evaluated the mechanistic studies provided by the commenters and has included a new section in the *Environmental Hazard Assessment for Dibutyl Phthalate* discussing EPA's evaluation of MBP toxicity (Section 3.1.2.1).

MBP is not expected to contribute significantly to environmental hazard due to lack of environmental persistence and lower environmental toxicity compared to the parent compound DBP. The toxicity data submitted by the commenters support this conclusion. Zhang et al. (2021) found non-apical effects in zebrafish (*Danio rerio*) after 96 hours of exposure to 5 mg/L of MBP, including appearance of apoptotic bodies and autophagosomes in the liver along with transcriptomic effects. Tao et al. (2020) also found non-apical effects in zebrafish (*D. rerio*) gills after 96 hours of exposure to 10 mg/L of MBP, including mitochondrial damage and transcriptomic effects. Because the exposure levels of MBP in these studies are four orders of magnitude higher than the most sensitive chronic effects in aquatic vertebrates for the parent compound DBP, and are approximately an order of magnitude higher than acute LC50 values causing mortality in zebrafish for the parent compound DBP despite causing only cellular-level effects, EPA does not consider MBP a significant contributor to aquatic vertebrate toxicity relative to the parent compound DBP.

7.4 Other Comments on Environmental Risk Assessment

Summary: A public comment (SACC25-0132, DIBP-0130, DIBP-0133) stated that EPA should include domestic animals in environmental risk evaluations under TSCA. The comment states that pets are exposed to phthalates through the same indoor exposures as humans as well as pet-specific activities (*e.g.*, chewing toys) as evidenced by urinary metabolites identified in cats and dogs.

EPA Response: EPA is aware of data evaluating levels of phthalate metabolites in urine of domestic dogs and cats. For examples, Karthikraj et al. (2019) measured 21 phthalate urinary metabolites in 50 domestic dogs and 50 domestic cats. Levels of urinary phthalate metabolites reported by Karthikraj et al. (2019) are similar to those reported for humans in NHANES during the same sampling period (*e.g.*, 2017–18). For example, for DEHP, the phthalate with the most sensitive POD, Karthikraj et al. (2019) report mean urinary levels of MEHHP to be 7.1 ng/mL for domestics cats and 7.4 ng/mL for domestic dogs (see Table 1 in Karthikraj et al.). Comparatively, NHANES reports 50th percentile urinary levels of MEHHP for the most exposed population (*i.e.*, black non-hispanic women of reproductive age) to be 7.8 ng/mL (see Table_G-2 in the DEHP general population exposure TSD). For the phthalate CRA, EPA evaluated combined background exposure to DEHP, DBP, DIBP, BBP, and DINP from NHANES with aggregate exposure estimates from individual TSCA COUs for each phthalate to provide conservative estimates of cumulative exposures. Further, EPA did not identify any unreasonable risk from exposure to any consumer COUs for DEHP, DBP, DIBP, BBP, and DCHP. Therefore, EPA considers its phthalate risk evaluations and phthalate CRA to be protective of humans, as well as their domestic cats and dogs.

Summary: A public commenter (DIBP-0130, DIBP-0133) stated that EPA failed to assess cumulative risk to animals in the ecological risk assessment. The commenter stated that, though EPA acknowledged the lack of formalized Agency guidelines for ecological CRA, EPA has a TSCA mandate to conduct a cumulative ecological assessment and describe potential risk arising from cumulative exposures. The commenter stated that there is "ample documentation supporting such conclusions for phthalates."

EPA Response: As described in EPA's *Draft Proposed Principles of Cumulative Risk Assessment under the Toxic Substances Control Act* (U.S. EPA, 2023), at this time, EPA is focusing its CRA efforts on human health, not on ecological taxa. This is because established Agency cumulative risk guidance documents are available to support human health, but not yet ecological CRA. Currently, EPA does not have a peer-reviewed framework for assessing environmental cumulative risks of chemical substances, however, it may develop an approach for conducting CRA under TSCA for ecological taxa in the future. Therefore, EPA is not actively pursuing a cumulative risk assessment of phthalates for terrestrial or aquatic organisms.

Additionally, EPA disagrees that it has a TSCA mandate to conduct a cumulative ecological assessment. There are no regulatory text mandating cumulative ecological assessments under TSCA; to the contrary, the 2024 *Procedures for Chemical Risk Evaluation Under the Toxic Substances Control Act (TSCA)* (40 CFR part 702) states that "…EPA is finalizing this rule without an explicit requirement related to cumulative risk assessment. EPA is nonetheless committed to considering and applying cumulative risk assessment approaches for future chemicals undergoing risk evaluation, where supported by the reasonably available information and best available science." EPA believes that its current ecological risk evaluations, which represents the best available science, does not support the development of a cumulative ecological risk evaluation for this group of phthalates with

the information that is reasonably available. Unlike in the human health hazard assessment, ecological hazard assessment includes consideration of many plant and animal species with substantially different physiological characteristics, life histories, and environmental exposures. Due to the lack of an established common denominator for developing a measure of comparative exposure across phthalates for environmental taxa, or a peer-reviewed framework for obtaining such a measure, EPA does not believe that it is reasonable to pursue a cumulative ecological assessment in the timeframe of this risk evaluation.

8 UNREASONABLE RISK DETERMINATION

Comments associated with this topic are summarized in the subsections below.

General Comments

Summary: A public comment (DCHP-0124) opposed EPA's "whole chemical" approach to its risk determinations for chemicals, including DCHP. The comment explains that "[t]his approach violates Section 6 of TSCA because EPA is making one determination of risk for a chemical 'as a whole' rather than for each condition of use and is resulting in unscientific risk determinations that are based on only certain conditions of use (or even a single condition of use) that supposedly 'contribute to' the unreasonable risk determination."

Another public comment (DIBP-0131) argues that a "whole chemical" determination of unreasonable risk is inappropriate when only unlikely worst-case scenarios drive the outcome, using pre-catalyst manufacturing for DIBP as an example.

In contrast, other public comments (DCHP-0128, SACC25-0132, DIBP-0133) advocated for EPA using a "whole chemical" approach as consistent with EPA's framework rule and TSCA section 6(b) mandate to "determine whether *a chemical substance* presents an unreasonable risk of injury to health or the environment . . . under the conditions of use." The public comments further state PESS may be exposures to combination of conditions of use and thus TSCA should review chemicals holistically. Two of these comments, SACC25-0132 and DIBP-0133, point to the 2024 Risk Evaluation Framework Rule as EPA identifying a single determination as to whether a chemical presents unreasonable risk as both practicable and in line with TSCA.

EPA Response: A single determination that a chemical substance presents an unreasonable risk does not mean that the entirety or whole of that chemical's uses – or even a majority of uses – presents an unreasonable risk. The final phthalate risk evaluations considered the chemicals' COUs (*i.e.*, the intended, known and reasonably foreseen circumstances under which the chemical is manufactured, processed, distributed in commerce, used or disposed of). The potentially different exposure scenarios presented by different COUs are reflected in the risk evaluation's exposure assessment. In these final risk evaluations, EPA listed the COUs that significantly contribute to the unreasonable risk. Further, EPA's risk management rule(s) based on these final risk evaluations will focus on those COUs that contribute to the unreasonable risk.

This approach does not lead to "unscientific risk determinations" because the risk determinations for all of the phthalate risk evaluations are based on the peer reviewed risk characterization in the Risk Evaluation, based on reasonably available information pursuant to TSCA section 26(k) and 40 CFR 702.33, and developed in accordance with EPA's obligation under TSCA sections 26(h) and (i) to make decisions in a manner consistent with the best available science and based on the weight of scientific evidence. Further, contrary to some commenters' intimation that EPA ignore that some conditions of use do not contribute to the unreasonable risk from one of the phthalates, EPA identifies in each Risk Evaluation which conditions of use significantly contribute to the unreasonable risk and which conditions of use do not contribute to the unreasonable risk of DCHP. This information can be found in the Executive Summary of each Risk Evaluation, at the beginning of each risk determination, and in Table 6-1 and Table 6-2 (as applicable) of each Risk Evaluation.

Summary: A public comment (SACC25-0132, DIBP-0133) raised concerns that "EPA arbitrarily departed from risk benchmarks in the Draft Risk Evaluations, finding no contribution to unreasonable risk from COUs that exceed those benchmark levels." According to the comment, EPA provided a generic justification that benchmarks are not "bright-lines" and should be considered with other "risk-related factors," but did not provide sufficient justification for each COU. For example, EPA preliminarily determined that Recycling of products containing BBP did not contribute to unreasonable risk when worker risks exceeded the benchmark MOE.

In addition, the comment (SACC25-0132, DIBP-0133) states that in the absence of unreasonable risk, EPA did consider the same benchmarks a "bright-line" and did not apply the same evaluation of other risk factors. The comment argues that the assumption that EPA assessments can only overestimate risk reduces the evaluation to a screening level assessment. "If EPA does not believe its benchmarks are "bright-line rules," then it must evaluate the potential for unreasonable risk when the calculated risks fall below an applicable benchmark, as opposed to solely when the risks exceed the benchmark level."

DIBP-0133 states that the "one-sided application of EPA's benchmarks effectively reduces the Draft Risk Evaluations to screening level assessments, where calculations that do not exceed EPA's benchmarks are taken as conclusive evidence of no unreasonable risk and risk calculations above those levels trigger further inquiry and contextualization.

The public comment (SACC25-0132) stated that EPA does not provide sufficient justification within the DBP Risk Evaluation for not determining unreasonable risk to the environment for 3 COUs with robust confidence and RQs of 1.04. The commenter considers the provided explanation as arbitrary and in violation of mandates to utilize all reasonably available information.

EPA Response: These commenters (SACC25-0132, DIBP-0133) note places where EPA determined that a COU and/or exposure pathway does not significantly contribute unreasonable risk to a particular phthalate even though a risk estimate exceeded the benchmark. The commenters also expressed concern that EPA's use of benchmarks was a "one way" approach, whereby additional factors are considered only when benchmarks are exceeded. Because EPA makes conservative assumptions when deriving a risk estimate, such as making upper-bound exposure assumptions, EPA has high confidence that exposures will not result in unreasonable risk if a benchmark is not exceeded. Where risk estimates do exceed benchmarks, this is a first step in informing EPA's whether those estimates support an unreasonable risk determination under TSCA. The unreasonable risk determination must be informed by science, and in making a finding of "presents unreasonable risk," as the commenter notes, EPA considers risk-related factors beyond exceedance of benchmarks. Risk-related factors include the type and severity of health effects under consideration, the reversibility of the health effects being evaluated, exposure-related considerations (*e.g.*, duration, magnitude, frequency of exposure), or population exposed—particularly populations with greater exposure or greater susceptibility (PESS) and the confidence in the information used to inform the hazard and exposure values. EPA also considers the confidence in the exposure and hazard inputs that informed the estimates, as well as any potentially conservative assumptions. In general, when the reasonably available information for the chemical substance is limited, EPA uses conservative assumptions that overestimate the risk, and therefore, as the comment argues, EPA explains how the assessment can overestimate the risk. For example, the assessment of the COU Consumer use – Floor coverings – Floor coverings for DIBP resulted in MOEs below the benchmark based on the high intensity exposure scenario for acute and chronic inhalation and aggregate exposure. However, in the high intensity use scenario for inhalation exposure, one of the two different article scenarios assessed (vinyl flooring) assumed among other things, that the entirety of the house flooring contained DIBP, that the vinyl flooring contained the

maximum reported value of 0.074 w/w, and that exposed children spent 20 hours per day in the home (*i.e.*, 20 hours in the environment where the flooring is present). Although this high-intensity exposure scenario is possible, the confluence of these factors (*e.g.*, 100% DIBP vinyl flooring and the highest weight fraction identified of 0.074 w/w) may be an upper-bound; ultimately, EPA is uncertain and lacks supporting evidence of the widespread use of vinyl flooring coverage in homes. The medium- and low-intensity use scenarios allow for the presence of other floor coverings in addition to vinyl flooring (50 and 25% of floor coverage respectively) and flooring with lower weight fractions of DIBP ($5.6\times10-5$ and 0.026 w/w), which may be a better representation of average U.S. homes. The MOEs from the medium use scenarios are all almost twice the benchmark or greater (*i.e.*, 57 or above) across all exposure routes, durations, and populations, and therefore does not support considering that such COU contributes to the unreasonable risk of DIBP. As indicated by the commenter, in the case of the DIBP risk evaluation, EPA conducted a sensitivity analysis of the modeled generic scenarios with multimedia releases, which reduced the uncertainty in the risk estimates for the environment. For some of the COUs, the RQs for a number of the different scenarios modeled were less than 1, and some of the RQs greater than 1 are not so much greater than 1 that the RQs can be certain to indicate risk even in light of conservatisms and uncertainties, in keeping with the fact that the benchmark is not a bright line for risk. Conversely, for two COUs, EPA does not have information to support a definitive percentage of release to surface water versus release to other pathways; however, EPA's sensitivity analysis shows RQs well above 1 even when only 10 percent of the release is assumed to go to surface water. And when applying potential wastewater treatment efficiency of 90 percent, these COUs indicate risk starting at 25 percent release to surface water. Even though EPA has slight confidence in the release estimates for the COUs with multimedia releases, there is overwhelming evidence to support that these two COUs significantly contribute to unreasonable risk of injury to the environment.

One of the main concerns raised in the commenter's (DIBP-0133) submission is EPA's reliance on central tendency exposure estimates. EPA does not believe it is appropriate to rely exclusively on the highest modeled exposures in determining significant contributions to unreasonable risk as a default as it does not align with EPA's mandate under TSCA to consider reasonably available information, operate in a manner that is consistent with the best available science, and make decisions based on the weight of the scientific evidence (15 U.S.C. 2625(h), (i), (k)). EPA's finding of "presents unreasonable risk" is an exhaustive conclusion based on all of the risk-related factors presented in EPA's assessment. Although, the high-end scenario is the basis of many of EPA's determinations under the COUs, using high-end values which indicate risk without additional considerations could result in inaccurate determinations under the COUs.

In response to commenter's concerns, EPA has added additional language in the risk determination to provide additional context and justification (*see* Section 6 of each risk evaluation).

Summary: A public comment (DIBP-0133) stated that "EPA calculated risks from cumulative exposures to BBP and other phthalates that exceed EPA's established benchmarks, but did not propose an unreasonable finding because the analysis supporting that finding is still undergoing peer review. The Draft Risk Evaluations rest upon hazard assessments that are subject to ongoing peer review. There is no rational basis for EPA to propose unreasonable risk determinations based on those hazard assessments while excluding cumulative risks from the determinations."

EPA Response: As discussed in the CRA TSD, EPA considered two approaches for cumulative risk characterization (termed Approaches 1 and 2). CRA Approaches 1 and 2 are compared in Section 5 of

the CRA TSD. During SACC peer-review, the SACC recommended that EPA retain both approaches, as the two approaches complement one another and can provided a more complete picture of cumulative risk. SACC also recommended that, if possible, EPA selected the most scientifically defensible approach for risk characterization of each phthalate.

As discussed in Section 5.4 of the CRA TSD, EPA developed a framework of considerations for CRA Approach selection for each phthalate. Based on the weight of scientific evidence considerations outlined in the developed framework EPA has weighed the strengths and uncertainties associated with the BBP RPF (Approach 1) and the BBP POD (Approach 2 and individual BBP risk assessment). Given the strengths and uncertainties associated with the BBP RPF and the BBP POD, EPA has more confidence in the BBP POD compared to the BBP RPF and has concluded that Approach 2 is more appropriate for use in risk characterization in the *Risk Evaluation of BBP*. Consistently, for the final BBP risk evaluation, EPA has utilized CRA Approach 2 to characterize cumulative risk, and findings from this characterization are used to support the unreasonable risk determination, as appropriate (note: in the draft BBP risk evaluation, Approach 1 was used to characterize cumulative risk).

Summary: A public comment (DIBP-0089) stated that "EPA should include analysis of safer alternatives to DIBP and recommend restrictions consistent with the "unreasonable risk" designation. Alternatives assessments should be publicly transparent and prioritize hazard reduction over regrettable substitution."

EPA Response: EPA acknowledges this comment and notes that an alternatives analysis and any proposed regulations to manage any unreasonable risks of injury to health or the environment from these chemical substances will be encompassed in a future risk management rulemaking process (or processes) pursuant to TSCA section 6(a). As part of any future rulemaking process, EPA will publish a notice of proposed rulemaking, seeking public comment on the proposal before finalizing any risk management regulations.

Human Health Risk Determination

Summary: Several public commenters (DCHP-0127, DEHP-0138, SACC25-0132, DIBP-0133) expressed concerns over the use of central tendency exposure to characterize risks for occupational and consumer COUs, stating that EPA does not provide sufficient justification to disregard high-end exposures in the unreasonable risk determination. The comments state that high-end exposure estimates (95th to 99th percentile of exposure) are required under TSCA to protect PESS with greater exposures, which is separate from PESS with increased susceptibility (female workers of reproductive age). The comments overall recommended identifying uncertainties early in the exposure assessment, use conservative assumptions in the absence of data, and addressing all reasonably foreseeable exposures, including high-end exposures.

One of the public comments (SACC25-0132) pointed to specific examples of where high end estimates resulted in MOEs below the benchmark but were not used to support an unreasonable risk finding for that COU: commercial use of furnishing, cleaning, and treatment care products in the DBP and risk evaluation and disposal in the DEHP risk evaluation. The comment noted an inconsistency in high end exposure estimates being used to inform the risk determination for all consumer COUs with MOE above the benchmark, but central tendency being used for infants and preschooler exposures for to furnishing, cleaning, treatment/care products, and airbeds with sufficient justification when high-

end estimates were below benchmark; in this case, the commenter also states that the high-end estimates were not conservative in this case for infants, due to discounting dermal exposures.

One of these public comment (SACC25-0132) identified an inconsistency between assumptions being considered reasonable by EPA to estimate exposure and then discounted by EPA as unreliable during risk determination, providing as an example assumptions used to estimate dust generation within the DEHP occupational assessment (100% of dust is abraded product and weight fraction of DEHP in dust is equal to the highest reported SDS for that product). Addressing a similar concern in the DCHP risk evaluation, another public comment (DCHP-0127) recommended that "If EPA [does] have evidence that its current 'high-end' estimates are not representative of high-end exposures for a given COU, the appropriate action would be to then develop new high-end estimates rather than relying only on the central tendency estimates." Similarly, a public comment (SACC25-0132) questioned why EPA did not calculate high-intensity scenarios for some consumer uses of DBP and DEHP (*e.g.*, dermal exposure in furniture, clothing, and auto interiors) with justification that these scenarios are unlikely and uncertain. The comment notes that in some cases, the medium intensity uses of these scenarios were near benchmark MOE and no justification was provided to describe why medium intensity scenarios had more certainty.

Another commenter (DIBP-0130, DIBP-0133) stated that EPA arbitrarily departed from its risk benchmarks, finding no contribution to unreasonable risk from multiple COUs that exceed those benchmark levels. "For multiple COUs of both BBP and DIBP, EPA failed to justify its finding of no contribution to unreasonable risk. For instance, as described in greater detail below, EPA found no contribution to unreasonable risk from the use of DIBP in children's clothing, despite calculating risks exceeding the non-cancer risk benchmark." Additionally, the commenter, DIBP-0133, raised concern that EPA also "calculated risks from DIBP in flooring that are up to two times worse than EPA's non-cancer risk benchmark. EPA discounted that risk because "the high-intensity exposure scenario assumed that the entirety of the house flooring contained DIBP, that the vinyl flooring contained the maximum reported value of 7.4 percent DIBP, and that exposed children spent 20 hours per day in the home." EPA acknowledged that "this scenario is possible," but states that it is "uncertain of the widespread confluence of these factors. And finally, DIBP-0133, stated that EPA used central tendency exposures to minimize risks to consumers who use paints and coating containing BBP. EPA never explained why it views the inputs as conversative. Nor did EPA consider the possibility that its exposures assumption may understate risks to someone who gets the spray on more than 10 percent of their hand. Instead, EPA simply asserted that "the medium-intensity dermal exposure scenarios for sealing and refinishing sprays are reasonably expected to occur."

EPA Response: In order to illustrate the full range of possible worker exposures based on the reasonably available data, EPA provided occupational exposure results for central tendency and high-end exposure conditions. However, there may be various sources of data and various approaches used to characterize central tendency and high-end occupational exposures for each COU. It is important to consider how the range of exposure estimates was developed for each COU individually and determine the applicability of the high-end or central tendency exposure estimates to be representative of exposures for the COU. For COUs where higher exposures are potentially expected, the high-end exposure values may be most relevant (*e.g.*, spray applications). For COUs where low to medium exposures are potentially expected, the central tendency estimates may be most relevant (*e.g.*, dust from plastic manufacturing). Also, depending on the data or approach used to create the range of exposure estimates, there may be greater confidence in the central tendency than the high-end estimates, or vice versa. See the risk characterization specific to each OES within each individual phthalate risk evaluation for details on how representative the central tendency vs high-end exposure

estimates are for that OES.

As one example, for spray application of adhesives and sealants or paints and coatings, high-end levels of exposure may occur when factors like spray equipment type, spray booth ventilation configuration, product concentration, and spray duration contribute to unusually elevated exposure levels. But for some other COUs like waste handling, treatment, and disposal, the central tendency values of exposure are expected to be more reflective of worker exposures due to conservative assumptions in the analysis like assuming phthalate concentrations in workplace dust are equal to that in waste materials. Therefore, for greater clarity, the utility of central tendency and high-end occupational exposure estimates are described as part of the occupational risk characterizations and risk determinations in each risk evaluation.

With regards to consumer exposures to furnishing, cleaning, and treatment care products in the DBP and risk evaluation and disposal in the DEHP risk evaluation mentioned by commenters, EPA disagrees about inappropriate selective dismissal of high intensity use exposure estimates. The consumer DEHP disposal COU discussion is available in Section 3.1.4 of the DEHP risk evaluation document and Section 2.1 (Qualitative Assessments Section) in the Consumer TSD. Briefly, EPA considered down-the-drain releases and exposures to the general population from consumer products and articles disposal in its evaluations. EPA described a qualitative discussion of disposal and subsequent environmental exposure. EPA considers the specific uses and physical chemical properties of the chemical being assessed to determine the pathways of exposures for humans and the environment when conducting individual chemical risk evaluations.

With regards to the furnishing, cleaning, and treatment care products in DBP and DEHP refinements. The commenter mentioned article examples like air beds within the furnishing, cleaning, treatment/care products which was identified for DEHP and not DBP. For airbeds, the screening approach used inputs for use patterns that were applied across age groups and leaned on the conservative side, then revisited the application of such inputs when MOEs were below the benchmark to corroborate appropriate application and representativeness of inputs. If the high intensity use scenario resulted in MOEs above the benchmark, then any combination of lower intensity use patterns would result in less exposures. For example, when considering exposures to DEHP from air beds, EPA used a scenario that would result in the highest exposures. At first, EPA considered all age groups laying on the air bed for 14 hours, knowing that such behavior may apply to some age groups but for other age groups would be considered a misuse or an overestimation. Infants laying on air beds for 14 hours is considered a misuse per CSPC mandated furniture warning labels[1] and independent third-party professional health care guidance(Doering and Salm-Ward, 2017[2]). In addition, the National Institutes of Health (NIH) has infant sleeping guidances that do not recommend air beds.[3] Toddlers may have naps, screen time, and reading time on air beds (intermittent exposures during a 24-hour period), but 14 or 8 hours would also be considered an extreme assumption for toddlers 1 to 2 years old. As such, when MOEs for any scenario were below the benchmark, EPA revisited inputs and assumptions applicability and representativeness for each age group accordingly.

For DBP furnishing, cleaning, and treatment care product COU, EPA identified synthetic leather furniture. EPA used inputs for this scenario that would result in the highest exposures. At first, EPA considered all age groups laying on the furniture for 8 hours, knowing that such behavior may not apply to some age groups. For example, infants laying on furniture for 8 hours is considered a misuse per NIH infant sleep guidances[3]. Toddlers may have naps, screen time, and reading time on furniture (intermittent exposures during a 24-hour period), but 8 hours would also be considered an extreme assumption for toddlers 1 to 2 years old. As such, when MOEs for any scenario were below the

benchmark, EPA revisited inputs and assumptions applicability and representativeness for each age group accordingly.

For DIBP children's clothing, EPA did additional refinement to consider migration from the solid clothing material to skin to demonstrate reduced dermal uptake. See DIBP risk evaluation Section 4.3.3 for the refinement approach description and results in Table 4-17.

For DIBP vinyl flooring high-intensity use exposure scenario, EPA had high uncertainty in the widespread use of vinyl flooring in US residences. EPA modeled the high-intensity use exposure scenario that considers 100% surface coverage. Although the high-intensity use exposure scenario may be possible it may be an upper-bound or an overestimation and EPA is uncertain and lacks supporting evidence of the widespread use of vinyl flooring coverage in homes.

For spray paints and coating containing BBP, the consumer exposure assessment did not find any MOEs for the high, medium, or low intensity use exposure scenarios below the benchmark for this COU for individual or cumulative assessments.

[1] https://publichealthpost.org/health-equity/air-mattresses-infant-deaths/
[2] https://ajph.aphapublications.org/doi/full/10.2105/AJPH.2017.303709
[3] https://safetosleep.nichd.nih.gov/reduce-risk/safe-sleep-environment#:~:text=Soft%20surfaces%2C%20like%20couches%2C%20sofas,at%20an%20angle%20or%20inclined)

Summary: A public comment (DCHP-0121) expressed support for the determination of unreasonable risk from occupational exposure to DCHP.

Similarly, another public comment (DIBP-0131) expressed support for EPA's "determination that DIBP does not pose an unreasonable risk to health for consumers or the general population under any COU."

EPA Response: EPA acknowledges the support for the unreasonable risk determinations of DCHP and DIBP expressed by these public commenters.

Summary: Regarding environmental risk, a commenter (DIBP-0131) disagrees with the EPA's determination of unreasonable risk to the environment for the COU "processing -- incorporation into formulation, mixture, or reaction product -- pre-catalyst manufacturing (*e.g.*, catalyst component for polyolefins production)", citing EPA's use of DBP read-across rather than DIBP-specific aquatic toxicity data. They assert EPA overestimated releases by assuming all U.S. polypropylene uses DIBP-containing catalysts, suggesting 60% as a more appropriate assumption based on industry estimates, and they state that DIBP consumption for this use has declined since 2019. They report recalculated risk quotients using EPA's modeled surface water concentrations and their corrected chronic COC of 30 µg/L; for the pre-catalyst scenario, they note that EPA's DIBP surface water concentration of 6.57 µg/L was stated to be supported by "existing facility release data" and request the source for this data be provided. Further, using the 6.57 µg/L value as a worst-case concentration in tandem with the commenter's own derived COC of 30 ppb, the commenter's calculations yield an RQ of 0.22, indicating no unreasonable environmental risk. They conclude that corrected RQs do not indicate unacceptable chronic aquatic toxicity for pre-catalyst manufacturing and that applying the acute-to-

chronic ratio approach with a 30 µg/L COC results in RQs <1 for plasticizer and pre-catalyst manufacturing conditions of use.

EPA Response: Examination across the multiple lines of evidence support DBP as an appropriate analog with high and medium quality aquatic, sediment-dwelling, and terrestrial hazard data to be used in a read-across to DIBP. EPA has high confidence in the use of DBP as an analog in the absence of DIBP data as well as where limited DIBP data is available, including for chronic aquatic vertebrates and chronic aquatic invertebrates. The release value provided for the COU "processing -- incorporation into formulation, mixture, or reaction product -- pre-catalyst manufacturing (*e.g.*, catalyst component for polyolefins production)" is comparable to actual DIBP release, specifically the high end P75 value. However, the exact value is confidential business information. EPA is confident in the RQ calculated for this COU. Lastly, the commenter states that only 60 percent of U.S. polypropylene uses DIBP-containing catalysts; however, there were no data submitted to support this claim. Therefore, EPA used the best reasonably available data, including the national production volume of polypropylene and the DIBP-specific concentrations from industry-provided process information, to inform the release assessment of DIBP.

8.1 Comments Related to Risk Management

Summary: A public comment (DCHP-0124) recommends the following considerations if EPA finalizes the unreasonable risk determination for DCHP and proceeds to risk management:

1. EPA must provide exemptions for "critical uses of DCHP such as medical, defense, aerospace, automotive, semiconductors, and filtration applications across industrial and commercial markets."
2. "If EPA implements a workplace protection program, EPA should ensure that the program is feasible for companies and aligns with OSHA requirements to prevent confusion and a patchwork of conflicting requirements in the workplace. "
3. "EPA should establish a *de minimis* level of DCHP that is allowed in adhesive and sealant products, such as of 0.1%. A *de minimis* level is necessary to account for impurities of DCHP in products and will avoid restricting the use of DCHP for uses that do not result in exposures that present an unreasonable risk. "
4. "EPA should more clearly define each condition of use and draft them as broadly as possible given the complexity of adhesive and sealant products and downstream applications."
5. EPA should ensure any risk management rule provides appropriate compliance timelines to allow stakeholders to have enough time to comply with the new requirements.

Similarly, another public commenter (DIBP-0124) asserted that regulating phthalates as a class is appropriate, given similar mechanism of action, and cited the CPSC's limits of 0.1% for BBP and DIBP in child-care articles and toys, and noted that the EU's REACH Annex XVII applies a 0.1% sum limit for phthalates across most articles. The commenter recommended that EPA meet or exceed this limit under TSCA through issuing unreasonable-risk determinations for both BBP and DIBP that cover consumers including PESS (pregnant people, fetuses, infants/children, workers, and people with obesity/metabolic disease), followed by §6(a) risk management that prohibits BBP and DIBP in consumer products and articles with a default ≤0.1% w/w limit (alone and, where appropriate, in sum with other restricted phthalates). The commenter stated that this recommendation was based on these phthalates having widespread exposure and identified reproductive/developmental hazards (evidence

EPA already categorized as robust), including exposure via transplacental transfer and to infants (presumably via lactation), in addition to cardiotoxicity.

EPA Response: EPA will pursue risk management after finalization of the risk evaluations to address the unreasonable risk of BBP, DBP, DCHP, DEHP, and DIBP to human health and/or the environment.

Under TSCA section 6(b), EPA determines through the risk evaluation process whether a chemical substance, presents an unreasonable risk to human health or the environment, under the conditions of use. The risk evaluation is not applicable to the manufacture, processing, or distribution in commerce of substances excluded from the definition of "chemical substance" under TSCA section 3(2)(B)(ii) through (vi). These include, but are not limited to, any pesticide (as defined by the Federal Insecticide, Fungicide, and Rodenticide Act) and any food, food additive, drug, cosmetic, or device, as defined in section 201 of the Federal Food, Drug, and Cosmetic Act.

Under TSCA section 6(a), EPA may apply one or more of the risk management options provided to ensure that the chemical substance no longer presents an unreasonable risk.

Summary: A public commenter (SACC25-0125) recommended that EPA limit production and use of these phthalates. Several other public comments (DEHP-0081, DEHP-0082, DEHP-0083, DEHP-0084, DEHP-0085, DEHP-0119, DEHP-0140) stated support for regulating DEHP and other phthalates. Commenters stated that the use of phthalates in articles and products including PVC water pipes, general plastic articles, food products, IV bags, and others should be eliminated. Commenters stated several potential health effects of phthalate exposure.

EPA Response: EPA will pursue risk management after finalization of the risk evaluations to address the unreasonable risk of BBP, DBP, DCHP, DEHP, and DIBP to human health and the environment.

Summary: A public commenter (DIBP-0126) recommended that EPA "ban the production and use of these toxic substances" and raised concerns about politics and industry profits impacting EPA's mission to protect the environment and public health.

EPA Response: EPA thanks the commenter for submission of its comments, and the agency will initiate risk management rulemaking to address the unreasonable risks of injury presented by BBP, DBP, DCHP, DEHP, and DIBP to human health and the environment. As part of any future rulemaking process, EPA will publish a notice of proposed rulemaking, seeking public comment on the proposal before finalizing any risk management regulations.

Summary: Two public comments (DEHP-0141, DIBP-0132) raise concerns that the recycling materials industry does not control the chemical substances that manufacturers use to make products and materials that eventually become available for reuse or recycling. Thus, TSCA regulation of certain chemical substances could impede or stop the reuse or recycling of certain products and materials.

EPA Response: EPA thanks the commenter for submission of its comments and will initiate risk management rulemaking to address the unreasonable risks of injury presented by BBP, DBP, DCHP,

DEHP, and DIBP to human health and the environment. As part of any future rulemaking process, EPA will publish a notice of proposed rulemaking, seeking public comment on the proposal before finalizing any risk management regulations.

Summary: A public comment (BBP-0122) requests "EPA identify a *de minimis* level for BBP and DIBP below which EPA has no reasonable basis to conclude that there is an unreasonable risk, and we encourage EPA to give high priority to this issue. This *de minimis* or regulatory threshold would be applicable to articles, replacement parts, and non-dimensional chemical uses such as adhesives and greases." The commenter's suggested *de minimus* threshold is of 0.1% (by weight) for DIBP and BBP in manufactured products similar to those of the International Material Data System (IMDS), EPA's PCE Risk Management Rule, Proposed NMP Risk Management Rule, and EPA's June 2024 Risk Management Rule.

EPA Response: EPA will set forth proposed regulations to minimize risks to phthalates (including DIBP and BBP) in a notice of proposed rulemaking for public comment before the issuance of a final rule as required by TSCA. EPA also notes that any determination of exemptions from regulatory requirements developed for risk management, such as a *de minimus* threshold, would be presented in a notice of proposed rulemaking allowing for public comment before a final rule is issued.

Summary: A public comment (BBP-0122) states that use of PPE and the "industrial hygiene hierarchy of controls" in the automotive industry reduces the risk posed by BBP and DIBP to workers in these sectors. The commenter request that "any final risk determination for any COU related to automotive manufacturing facilities should reflect the consistent use of engineering controls and PPE, including, but not limited to, OSHA requirements."

EPA Response: The risk evaluations of BBP and DIBP assessed exposures to workers from applications of paints/coatings and adhesives/sealants, and these uses are relevant for the automotive industry. For DIBP, there were no chemical-specific data available for estimating occupational exposure from applications of paints/coatings or adhesives/sealants. While for BBP, the only chemical-specific monitoring data available for application of paints/coatings were for non-mist generating activities. Consequently, EPA relied on surrogate mist monitoring data presented in the ESD on Coating Application via Spray-Painting in the Automotive Refinishing Industry (OECD, 2011). The underlying mist concentration data considered in the ESD reflected a variety of industrial and commercial automotive refinishing scenarios, including various booth configurations. Therefore, engineering controls (*i.e.*, spray booths) were used during the collection of the mist monitoring data, and these data were used in the exposure assessment of DIBP and BBP for uses relevant to the automotive industry. Further, EPA presents various levels of respiratory protection from DIBP and BBP exposure, as well as potential risk reduction from proper use of the PPE, in the Section 4.3.2 of the Risk Evaluations for DIBP and BBP. However, the unreasonable risk determination is not based on the use of PPE. Engineering controls, PPE, and other risk mitigations will be considered during risk management.

Summary: A public comment (DCHP-0123) requests that the EPA consider the complexities related to semiconductor manufacturing and related equipment (SMRE) and the SMRE operational context during the development of their rule regarding DCHP. The comment further explains, "We believe that DCHP could be present in components used in SMRE, and we encourage EPA to consider this possibility in the risk evaluation process. Nonetheless, we do not believe that the potential presence of DCHP in SMRE poses an unreasonable risk. If DCHP is indeed present in components of SMRE, it would represent a very limited exposure concern. First, only a very limited number of people work with SMRE when in use. Second, those employees wear personal protective equipment (PPE) when interacting with the equipment. Accordingly, there is little to no risk of the general public being exposed to DCHP that may be present in SMRE. Therefore, SEMI kindly requests that EPA adopt a nuanced approach that considers the full complexity of the semiconductor industry, including the use of PPE and related occupational safety measures, to support the successful implementation of future rules regarding DCHP.

EPA Response: EPA will initiate risk management to address the unreasonable risk of DCHP to human health after finalization of the risk evaluation. EPA acknowledges the commenter's concern that the complexities of the semiconductor manufacturing industry be considered during rulemaking to address the unreasonable risk. EPA will propose risk management regulations to mitigate the unreasonable risks to DCHP in a notice of proposed rulemaking before the issuance of a final rule as required by TSCA. The public is encouraged to comment on EPA's approach in the notice of proposed rulemaking.

9 SYSTEMATIC REVIEW

Summary: Public commenters (DCHP-0127, DEHP-0138, SACC25-0145, SACC25-0151, DIBP-0133) asserted that EPA did not employ the best available science in POD selection for BBP, DIBP, DEHP, and DBP. The commenters outlined the following reasons why EPA "did not apply the best available science to identify and evaluate relevant and useful health effects studies..." for DBP, DCHP, and DEHP. These are summarized below.

1. Recent guidance recommends that environmental health-relevant systematic reviews should be updated no more than 12 months before publication of a review. For all evidence streams, EPA relied on a literature search that hasn't been updated since 2019.
2. The commenters considered EPA's incorporation of existing assessments (and the "further-filtering process" for streamlining and reserving full data quality evaluation to those studies considered in dose-response) to be equivalent to an exclusion of consideration of the data:
    a. The commenter (DCHP-0127) explains "For earlier toxicology studies (before 2014), EPA relied entirely on the Health Canada assessment and did not apply its own search, inclusion/exclusion and study evaluation procedures. Toxicology studies published after September 2019 were not included at all, with one exception. This is not a clear, comprehensive or consistent approach to identifying the toxicology evidence relevant to assessing the health effects of DCHP."
    b. The commenters (DCHP-0127, DEHP-0138, DEHP-0138, and SACC25-0145) stated, "EPA relied on assessments conducted by Health Canada [or other agencies] to exclude studies, without supporting justification and inconsistent with the best available science."
    c. The commenters stated, "EPA used deficient inclusion and exclusion criteria for health effects evidence that inappropriately excluded important toxicity endpoints."
    d. Public commenter (SACC-0151) stated that: (1) for epidemiologic evidence, studies published between 2019-2023 were only considered if they were submitted to the EPA docket; (2) for toxicological evidence, all but a handful of studies published since 2020 were excluded from consideration altogether; and (3) EPA did not consider any of the relevant toxicological studies of DBP and DEHP from its literature search for studies published between 2014-2019 for dose-response assessment, only relying on a narrow subset of studies identified by previous regulatory assessments, including by Health Canada and ATSDR.
    e. The commenters contend that EPA inappropriately excluded at least 37 PECO-relevant health effects studies for DCHP, 446 PECO-relevant studies for DBP, and 733 PECO-relevant studies for DEHP from evidence integration.
3. Public commenter (SACC-0151) stated that EPA inappropriately excluded all epidemiology studies from dose-response analysis without adequate scientific justification, and narrowed the body of toxicological evidence to 50 studies for DEHP and only 7 studies for DBP.
4. They state, "EPA's methods for evaluation of study quality need to incorporate further improvements that have been recommended by the National Academies."
5. They additionally state, "EPA continues to use unclear terminology regarding evidence synthesis and integration."
6. They contend that EPA released incomplete draft systematic review protocols for DBP, DCHP,

and DEHP that were not made publicly available in advance of the draft risk evaluations.

7. Lastly, they stated, "EPA should prepare a new TSCA systematic review handbook that is aligned with the best available scientific methods and issue updated draft systematic review protocols for all risk evaluations currently in development."

EPA Response: EPA disagrees with the public commentors that it did not employ the best available science in POD selection for its human health hazard assessments of BBP, DIBP, DEHP, and DBP. EPA was guided by its *Draft Systematic Review Protocol Supporting TSCA Risk Evaluations for Chemical Substances*, which was reviewed by the SACC and by the NASEM, and developed a specific protocol for each of the phthalates to document any chemical-specific updates. Use of existing assessments was appropriately included in protocols to ensure that the best available information was used while at the same time responsibly using government resources to not unnecessarily duplicate work done by other authoritative bodies. Where necessary, EPA updated its literature search to capture any information developed after the initial search, which is documented in the chemical-specific protocol for each phthalate. In addition, the scientific evidence that EPA used to inform its phthalate risk evaluations was reviewed and commented on by the SACC in August 2025.

At the time of the draft release, EPA relied primarily on literature searches from 2014 to 2019. However, EPA conducted additional literature updates between issuance of the draft and final risk evaluations to include studies identified during the public comment periods for each phthalate and by the SACC during the peer review meeting in August 2025.

EPA disagrees that it, "continues to use unclear terminology regarding evidence synthesis and integration." EPAs approach to evidence integration is detailed in Section 6 of the chemical-specific systematic review protocol for each phthalate. Because of the wealth of existing assessments for the phthalates, especially for DBP, DEHP, and BBP, a modified fit-for-purpose approach was employed, again to ensure appropriate use of government resources. Rather than evaluating and integrating all evidence examining exposure to DBP, DEHP, BBP, DIBP, and BBP and all health outcomes, EPA focused on identifying studies that could inform an updated dose response assessment or that supported identification of a new human health hazard. As described in the systematic review protocol, for key human health outcomes (*e.g.*, developmental/reproductive and nutritional/metabolic effects (DEHP only), EPA integrated evidence streams (*i.e.*, evidence from human studies, *in vivo* animal studies, and mechanistic data) as part of its analysis.

EPA acknowledges the suggestion that it should prepare a new TSCA Systematic Review Handbook. Instead of updating the 2021 Draft Systematic Review Protocol, EPA issued chemical-specific protocols for each phthalate that documents chemical-specific changes and updates to the systematic review process for phthalate.

Summary: A public comment (SACC23-0030) stated they are concerned about the lack of transparency of the Systematic Review process and that any changes to the SR process should be put to the public in a transparent manner and allow for public comment.

EPA Response: EPA released draft systemic review protocols for DCHP, DEHP, DBP, DIBP, and BBP for public comment in 2025.

Summary: A public commenter (SACC25-0145) stated that EPA should have included a clear "summary judgement" based on specific hazard descriptors from the draft systematic review protocol (2021), such as:

- Evidence demonstrates
- Evidence indicates likely
- Evidence suggests but is not sufficient to conclude
- Evidence is inadequate

EPA Response: EPA's approach to evidence integration is detailed in Section 6 of the Systematic Review Protocols for DEHP, DBP, DIBP, BBP, and DCHP. Because of the wealth of existing assessments for DEHP, DBP, DIBP, BBP, and DCHP, a modified fit for purpose approach was employed. Rather than evaluating and integrating all evidence examining DEHP, DBP, DIBP, BBP, and DCHP exposure and all health outcomes, EPA focused on identifying studies that could inform an updated dose response assessment or that supported identification of a new human health hazard. For DEHP, DBP, DIBP, BBP, and DCHP, EPA made judgments based on the weight of scientific evidence to determine which health outcomes to focus its dose-response assessment on.

10 OTHER COMMENTS

Summary: A public comment (DCHP-0120) stated "CHEMICALS Are good if they are NATURAL, not man made with chemical bonds that are unrelenting and permanent, we are being eaten by PLASTICS and now the Dicyclohexyl Phthalate. This chemical needs testing in a closed environment and not produced as is for use and distribution. I am not against chemicals, but 2 many are being created without sensitivity to the permanence of them and their life cycle on EARTH, Our Earth.!!! c2it."

EPA Response: All reasonably available data on both the hazardous effects of DCHP and the distribution of DCHP within the environment was considered in the DCHP Risk Evaluation using the best available science. Details on the process of identifying relevant literature are available in the *Draft Systematic Review Protocol for Dicyclohexyl Phthalate (DCHP)*. Toxicology studies in controlled laboratory environments were used to derive hazard values for DCHP for both human health and the environment, as detailed in sections Section 4.2 and 5.2 of the *DCHP Risk Evaluation*. The evaluation of the fate and transport of DCHP within the environment and the environmental release of DCHP from conditions of use under TSCA are detailed in Sections 2 and 3 of the DCHP Risk Evaluation. EPA determined the DCHP does not present an unreasonable risk the environment under the TSCA COUs. EPA will pursue risk management after finalization of the risk evaluation to address the unreasonable risk of DCHP to human health.

Summary: A public commenter (DIBP-0088) stated that EPA and Bureau of Land Management (BLM) regulatory requirements are a hindrance to oil sands operations, due to the designation of certain lands as navigable waterways and the application of the Endangered Species Act.

EPA Response: The conditions of use under which the EPA evaluated the phthalates did not include conditions of use related to oil sands operations, or oil production in general. Also, the comment relates to regulatory burden, and EPA's TSCA risk evaluations are not regulations, but rather provide determinations as to whether the Agency needs to initiate regulatory action.

Summary: A public commenter (DIBP-0127) stated "PROPIN CONFIDENTIAL EPA-HQ-OPPT-2018-0434 Butyl Benzyl Phthalate (BBP); Diisobutyl Phthalate (DIBP); Draft Risk Evaluations Under the Toxic Substances Control Act (TSCA); Notice of Availability and Request for Comment The Petition for Motion should proceed with the Hearings based on the posed harm and health risks from SACC and EPA Report. I am requesting assistance and support from all applicable agencies and departments involved, including the Attorney General and Chief of Counsel, as appropriate."

EPA Response: Under TSCA section 6, EPA is required to conduct risk evaluations for "chemical substances" and manage any unreasonable risk(s) those chemicals present to health or the environment, under the chemical's "conditions of use." TSCA section 6 sets up a three-phase process for evaluating existing chemical substances. EPA is following this statutory process, as described below:

- **Prioritization** is the process of deciding if an existing chemical requires risk evaluation. EPA designates chemicals as either "high-priority" or "low-priority" based on factors like hazard,

exposure potential, persistence, and bioaccumulation. High-priority chemicals move to risk evaluation.

- **Risk evaluation** is the process of assessing whether the chemical poses an unreasonable risk to human health or the environment, under its conditions of use. EPA must comply with TSCA's science and information standards and seeks public input and peer review to ensure transparency and scientific rigor. Chemicals that present unreasonable risk move to risk management.
- **Risk management** is the process of issuing rules to regulate conditions of use of a chemical that present unreasonable risk. EPA also seeks public input during the rulemaking stage to ensure transparency and sufficiency of its regulations.

More information on the Toxic Substance Control Act can be found here: https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/frank-r-lautenberg-chemical-safety-21st-century-act