

Draft Physical Chemistry and Fate and Transport Assessment for Dibutyl Phthalate (DBP)

Technical Support Document for the Draft Risk Evaluation

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98 KEY ABBREVIATIONS AND ACRONYMS

99	ATSDR	Agency for Toxic Substances and Disease Registry
100	Atm	Atmospheres
101	atm•m³/mol	Atmospheres – cubic meters per mole
102	BAF	Bioaccumulation factor
103	BCF	Bioconcentration factor
104	BMF	Biomagnification factor
105	BSAF	Biota-sediment accumulation factor
106	С	Celsius
107	CASRN	Chemical Abstract Service registry number
108	cP	Centipoise
109	DBP	Dibutyl phthalate
110	DOE	Department Of Energy
111	DOC	Dissolved organic carbon
112	dw	Dry weight
113	DW	Drinking water
114	ECHA	European Chemicals Agency
115	EC/HC	Environment Canada and Health Canada
116	EPA	Environmental Protection Agency
117	F	Fahrenheit (°F)
118	g/cm ³	Grams per cubic centimeter
119	ĞC	Gas chromatography
120	HLC	Henry's Law constant
121	Κ	Kelvin
122	Kaw	Air-water partition coefficient
123	Koa	Octanol-air partition coefficient
124	Koc	Organic carbon-water partition coefficient
125	Kow	Octanol-water partition coefficient
126	М	Molarity (mol/ \dot{L} = moles per Liter)
127	mg/L	Milligrams per liter
128	mL/min	Milliliters per minute
129	mmHg	Millimeters of mercury
130	MBP	Mono butyl phthalate
131	mol	Mole
132	MS	Mass spectrometry
133	N/A	Not applicable
134	NCBI	National Center for Biotechnology Information
135	NIST	National Institute of Standards and Technology
136	NIOSH	National Institute for Occupational Safety and Health
137	NLM	National Library of Medicine
138	nm	Nanometers
139	NR	Not reported
140	·ОН	Hydroxyl radical
141	Pa (hPa)	Pascals (hectopascals; $1 \text{ hPa} = 100 \text{ Pa}$)
142	PA	Phthalic Acid
143	PCF	Plant concentration factor
144	pg/L	Picograms per liter
145	ppm	parts per million
146	QSAR	Quantitative structure activity relationship
	-	· 1

147	RSC	Royal Society of Chemistry
148	RSD	Relative standard deviation
149	SI	Supplemental information
150	STP	Sewage treatment plant
151	TSCA	Toxic Substances Control Act
152	TMF	Trophic magnification factor
153	U.S.	United States
154	UV (UV-Vis)	Ultraviolet (visible) light
155	WHO	World Health Organization
156	WW	Wet weight
157	WWTP	Wastewater Treatment Plant

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- 174 **Docket**
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- 176

177 Disclaimer

- 178 Reference herein to any specific commercial products, process or service by trade name, trademark,
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- 192 leadership.

193 SUMMARY

194	This technical support document is in support of the TSCA Draft Risk Evaluation for Dibutyl Phthalate
195	(DBP) (U.S. EPA, 2024c). EPA gathered and evaluated physical and chemical property data and
196	information according to the process described in the Draft Risk Evaluation for Dibutyl Phthalate (DBP)
197	- Systematic Review Protocol (U.S. EPA, 2024d). During the evaluation of dibutyl phthalate (DBP),
198	EPA considered both measured and estimated physical and chemical property data and information
199	summarized in Table 2-1, as applicable. Information on the full, extracted data set is available in the file
200	Draft Risk Evaluation for Dibutyl Phthalate (DBP) – Systematic Review Supplemental File: Data
201	Quality Evaluation and Data Extraction Information for Physical and Chemical Properties (U.S. EPA,
202	<u>2024b</u>).
203	
204	DBP – Physical Chemistry: Key Points
205	• DBP is a branched phthalate ester used as a plasticizer.
206	• Under standard environmental conditions, DBP is an oily liquid (O'Neil, 2013) with a melting
207	point around -35 °C (Rumble, 2018b).
208	• DBP has a water solubility of 11.2 mg/L at 24°C (Howard et al., 1985) and a log Kow of 4.5
209	(NLM, 2024).
210	• With a vapor pressure of 2.1×10^{-5} mmHg at 25 °C (U.S. EPA, 2019) and a boiling point of 340
211	°C (Rumble, 2018b), DBP has the potential to be volatile from dry, non-adsorbing surfaces.
212	• The selected Henry's Law constant for DBP is 1.81×10^{-6} atm·m ³ /mol at 25 °C (NLM, 2024).
213	,
214	DBP – Environmental Fate and Transport: Key Points
215	EPA evaluated the reasonably available information to characterize the environmental fate and transport
216	of DBP, the key points are summarized below. Given the consistent results from numerous high-quality
217	studies, there is robust evidence that DBP:
218	• Is expected to degrade rapidly via direct and indirect photolysis and will rapidly degrade in the
219	atmosphere ($t_{1/2} = 1.15$ days) (Section 4.3);
220	• Is not expected to hydrolyze under environmental conditions (Section 4.2);
221	• Is expected to have an environmental biodegradation half-life in aerobic environments on the
222	order of days to weeks (Section 4.1);
223	• Is not expected to be subject to long range transport;
224	• Is expected to transform in the environment via biotic and abiotic processes to form phthalate
225	monoesters, then phthalic acid, and ultimately biodegrade to form CO ₂ and/or CH ₄ (Section 1);
226	• Is expected to show strong affinity and sorption potential for organic carbon in soil and sediment
227	(Section 3.2);
228	• Will be removed at rates between 68 to 98 percent in conventional wastewater treatment systems
229	(Section 6.2);
230	• When released to air, will mostly partition to soil and water, and remaining DBP fraction in air
231	will rapidly degrade in the atmosphere (Section 5.1); and
232	• Is likely to be found and accumulate in indoor dust (Section 5.1.1).
233	
234	As a result of limited studies identified, there is moderate evidence that DBP:
235	• Is not expected to biodegrade under anoxic conditions and may be persistent in anaerobic soils
236	and sediments (Section 4.1).
237	• Is not bioaccumulative in fish that reside in the water column (Section 1).
238	• May be bioaccumulative in benthic organisms exposed to sediment with elevated concentrations
239	of DBP proximal to continual sources of release (Section 1).
	•

- Is expected to be partially removed in conventional drinking water treatment systems via sorption to suspended organic matter and filtering media (Section 6.3). 240
- 241

242 1 INTRODUCTION

- DBP is produced by the esterification of phthalic anhydride with isobutyl alcohol in the presence of an
 acid catalyst. DBP is a member of the phthalate class of chemicals that are widely used as adhesives and
 sealants in the construction and automotive sectors. DBP is also commonly used in electronics,
 children's toys, and plastic and rubber materials. DBP is considered ubiquitous in various environmental
 media due to its presence in both point and non-point source discharges from industrial and conventional
 wastewater treatment effluents, biosolids, sewage sludge, stormwater runoff, and landfill leachate (Net
 et al., 2015).
- 250

This assessment was used to determine which environmental pathways to assess further for DBP's risk evaluation. Details on the environmental partitioning and media assessments can be found in Section 4. Based on DBP's fate parameters, EPA anticipates DBP to predominantly be found in water, soil, and sediment. DBP in water is mostly attributable to discharges from industrial and municipal wastewater treatment plant effluent, surface water runoff, and, to a lesser degree, atmospheric deposition. Once in water, DBP is expected to mostly partition to suspended organic matter and aquatic sediments. DBP in soils is attributable to deposition from air and land application of biosolids.

258

259 EPA quantitatively assessed concentrations of DBP in surface water, sediment, and soil from air-to-soil 260 deposition. Ambient air concentrations were quantified for the purpose of estimating soil concentrations 261 from air deposition but were not used for the exposure assessment as DBP was not assumed to be 262 persistent in the air $(t_{1/2} = 1.15 \text{ days} (\text{Peterson and Staples}, 2003))$. In addition, partitioning analysis showed DBP partitions primarily to soil and water when compared to air and sediment, including from 263 264 air releases. Soil concentrations of DBP from land applications were not quantitatively assessed in the 265 screening level analysis since DBP is expected to have limited persistence potential and mobility in soils receiving biosolids. 266

268 2 APPROACH AND METHODOLOGY FOR PHYSICAL AND 269 CHEMICAL PROPERTY ASSESSMENT

EPA completed a systematic review by conducting a literature search of available published articles
through 2019 to find the following physical and chemical property values. After physical and chemical
property data have been extracted and evaluated, values for the endpoints are selected for use in the risk
evaluation as described in the *Draft Systematic Review Protocol for Dibutyl Phthalate (DBP)* (U.S.
EPA, 2024d). Due to the large quantity of available data, only studies with an overall data quality

ranking of "high" were selected for use in this risk evaluation. Empirical data for the octanol:air

276 partition coefficient (log K_{OA}) and the air:water partition coefficient (log K_{AW}) were not available, thus

EPI SuiteTM (U.S. EPA, 2017) was used to estimate a value for each of these parameters.

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2.1 Selected Physical and Chemical Property Values for DBP

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280 **Table 2-1. Selected Physical and Chemical Property Values for DBP**

Property	Selected Value(s)	Reference(s)	Data Quality Rating
Molecular formula	$C_{16}H_{22}O_4$	<u>NLM (2024)</u>	High
Molecular weight	278.35 g/mol	Haynes (2014b)	High
Physical form	Oily liquid	<u>O'Neil (2013)</u>	High
Melting point	−35 °C	<u>Rumble (2018b)</u>	High
Boiling point	340 °C	<u>O'Neil (2013)</u>	High
Density	1.0465 g/cm ³	Rumble (2018b)	High
Vapor pressure	2.01E-05 mmHg	<u>U.S. EPA (2019)</u>	High
Vapor density	9.58	<u>NLM (2024)</u>	High
Water solubility	11.2 mg/L	Howard et al. (1985)	High
Octanol:water partition coefficient (log Kow)	4.5	<u>NLM (2024)</u>	High
Octanol:air partition coefficient (log K _{OA})	8.63 (EPI Suite TM)	<u>U.S. EPA (2017)</u>	High
Air:water partition coefficient (log K _{AW})	-4.131 (EPI Suite TM)	<u>U.S. EPA (2017)</u>	High
Henry's Law constant	1.81E–06 atm·m ³ /mol at 25 °C	<u>NLM (2024)</u>	High
Flash point	157 °C	NLM (2024)	High
Autoflammability	402 °C	NLM (2024)	High
Viscosity	20.3 cP	<u>NLM (2024)</u>	High

282 **2.2 Endpoint Assessments**

283 2.2.1 Melting Point

284 Melting point informs the chemical's physical state, environmental fate and transport, as well as the 285 chemical's potential bioavailability. EPA extracted and evaluated eight high-quality data sources 286 containing DBP melting point information. These sources reported DBP melting points ranging from – 40 to -35 °C (NLM, 2024; NIST, 2022; Elsevier, 2019; U.S. EPA, 2019; Rumble, 2018b; DOE, 2016; 287 ECHA, 2012; NIOSH, 2007; Wang and Richert, 2007; Park and Sheehan, 2000). The mean of the 288 289 reported melting point values within these sources is -35.57°C. Seven of these sources reported a DBP 290 melting point of -35°C (NLM, 2024; NIST, 2022; Elsevier, 2019; U.S. EPA, 2019; Rumble, 2018b; 291 DOE, 2016; NIOSH, 2007), while one source reported a DBP melting point of -40°C (Park and 292 Sheehan, 2000). EPA selected a melting point value of -35 °C (Rumble, 2018b) as a representative 293 melting point value since this value is consistent with the average of the identified information from the 294 overall high-quality data sources. The identified value is consistent with the value proposed in the *Final* 295 Scope for the Risk Evaluation of Dibutyl Phthalate (DBP) (U.S. EPA, 2020).

2.2.2 Boiling Point

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297 Boiling point informs the chemical's physical state, environmental fate and transport, as well as the 298 chemical's potential bioavailability. EPA extracted and evaluated eleven high-quality data sources 299 containing DBP boiling point information. These sources reported DBP boiling points ranging from 338 300 to 340.7 °C (NLM, 2024; NIST, 2022; Elsevier, 2019; U.S. EPA, 2019; Rumble, 2018b, c; DOE, 2016; O'Neil, 2013; ECHA, 2012; NIOSH, 2007; Wang and Richert, 2007; Park and Sheehan, 2000). The 301 302 mean of reported boiling point values within these sources is 340 °C. EPA selected the boiling point 303 value of 340 °C reported by O'Neil (2013) since this value is consistent with the mean of all boiling 304 points measured under standard environmental conditions. The identified value is consistent with the 305 value proposed in the Final Scope for the Risk Evaluation of Dibutyl Phthalate (DBP) (U.S. EPA, 2020).

2.2.3 Density

EPA extracted and evaluated nine high-quality data sources containing DBP density information. These
sources reported DBP density values of 1.042 to 1.0501 g/cm³ (NLM, 2024; Elsevier, 2019; Rumble,
2018b; DOE, 2016; O'Neil, 2013; ECHA, 2012; Cadogan and Howick, 2000; Park and Sheehan, 2000;
WHO, 1997). The mean of the reported density values is 1.0462 g/cm³. EPA selected a density of
1.0465 g/cm³ (Rumble, 2018b) to closely represent the mean of the density values obtained from the
available high-quality data sources. The identified value is consistent with the value proposed in the *Final Scope for the Risk Evaluation of Dibutyl Phthalate (DBP)* (U.S. EPA, 2020).

2.2.4 Vapor Pressure

Vapor pressure indicates the chemical's potential to volatilize, undergo fugitive emissions and other 315 316 releases to the atmosphere, undergo long range transport, and be available for specific exposure 317 pathways. EPA extracted and evaluated eight high-quality data sources containing DBP vapor pressure information. One of these sources reported DBP vapor pressure values of 1.2 to 2.5×10^{-4} mmHg at 25 318 °C (Elsevier, 2019). The remaining seven high-quality data sources reported DBP vapor pressure 319 ranging from 2.01×10⁻⁵ to 7.28×10⁻⁵ mmHg at 25 °C (NLM, 2024; U.S. EPA, 2019; Ishak et al., 2016; 320 ECHA, 2012; Lu, 2009; NIOSH, 2007; Howard et al., 1985; Hamilton, 1980). The mean and mode of 321 these reported vapor pressure values are 4.38×10^{-5} and 2.01×10^{-5} mmHg, respectively, at 25°C. EPA 322 selected the experimentally derived vapor pressure value of 2.01×10^{-5} mmHg (U.S. EPA, 2019) to best 323 represent the mode vapor pressure of DBP obtained from the overall high-quality data sources under 324 standard environmental conditions. The identified value is consistent with the value proposed in the 325 326 Final Scope for the Risk Evaluation of Dibutyl Phthalate (DBP) (U.S. EPA, 2020).

2.2.5 Vapor Density

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EPA extracted and evaluated one high-quality and one medium-quality data source containing DBP vapor density information. These sources reported DBP vapor densities of 9.58 and 9.60 (NLM, 2024; NIOSH, 1976). EPA selected the vapor density value of 9.58 from the one available high-quality data source as a representative value for standard environmental conditions. The identified value is consistent with the value proposed in the *Final Scope for the Risk Evaluation of Dibutyl Phthalate (DBP)* (U.S. EPA, 2020).

2.2.6 Water Solubility

Water solubility informs many endpoints not only within the realm of fate and transport of DBP in the 335 336 environment, but also when modeling for industrial process, engineering, human and ecological hazard, 337 and exposure assessments. EPA extracted and evaluated twelve high-quality data sources containing 338 DBP water solubility information. These sources reported water solubility values from 1.5 to 14.6 mg/L 339 (NLM, 2024; Elsevier, 2019; U.S. EPA, 2019; Rumble, 2018a; EC/HC, 2017; ECHA, 2012; NIOSH, 340 2007; Mueller and Klein, 1992; Defoe et al., 1990; Howard et al., 1985; SRC, 1983b). EPA excluded 341 two of the reported values, 1.5 and 14.6 mg/L (Elsevier, 2019), as those were determined to be potential outliers. These values were higher or lower than the upper (13.00 mg/L) and lower bounds (8.20 mg/L) 342 343 calculated using the interquartile range (1.2 mg/L) rule for potential outliers (U.S. EPA, 2006). The rest 344 of the available data sources reported DBP's water solubility values from 8.7 to 11.4 mg/L. The mean of 345 the reported water solubilities at near ambient temperature is 10.62 mg/L. A water solubility of 11.2 mg/L (Howard et al., 1985) was selected as the empirical value obtained from the overall high-quality 346 347 data sources that best represents DBP's mean water solubility under standard environmental conditions. 348 The identified value is consistent with the value proposed in the Final Scope for the Risk Evaluation of 349 Dibutyl Phthalate (DBP) (U.S. EPA, 2020).

2.2.7 Octanol:Air Partition Coefficient (log KOA)

351 The octanol-air partition coefficient (K_{OA}) provides information on how the chemical will partition 352 between octanol (which represents the lipids or fats in biota) and air. KOA informs on how DBP is likely 353 to partition between air, aerosol particles, foliage, dust, dry surfaces, soil, and animal tissue. No KoA 354 data for DBP were identified in the initial data review for the Final Scope for the Risk Evaluation of 355 DBP (U.S. EPA, 2020). After the final scope was published, EPA extracted and evaluated DBP octanol-356 air partitioning (KoA) data from a single medium quality source. This source reported a predicted log KoA value of 8.45 obtained from a quantitative structure-property relationship (QSPR) model (Lu, 357 2009). The OSPR-derived estimate of 8.45 reasonably aligns with DBP's log K_{OA} value of 8.63 358 estimated using EPI Suite[™] (U.S. EPA, 2017). As such, EPA has selected the EPI Suite[™] derived value 359 of 8.63 as the representative log K_{OA} value for use in risk assessment (U.S. EPA, 2017). The EPI 360 361 SuiteTM modeled value was selected because it closely aligns with the reported predicted value and EPI Suite[™] is considered a highly reliable model. 362

2.2.8 Octanol:Water Partition Coefficient (log Kow)

The octanol-water partition coefficient (Kow) provides information on how the chemical will partition 364 365 between octanol (which represents the lipids or fats in biota) and water. Kow informs on how the 366 chemical is likely to partition in biological organisms as well as for the estimation of other properties including water solubility, bioconcentration, soil adsorption, and aquatic toxicity. EPA extracted and 367 evaluated ten high-quality data sources containing DBP log Kow information. These sources included 368 369 six new additional sources not available for the Final Scope for the Risk Evaluation of Dibutyl Phthalate 370 (DBP) (U.S. EPA, 2020). EPA excluded a reported K_{OW} of 3.74 from two data sources (Howard et al., 371 1985; SRC, 1984) as it was determined to be a potential outlier. This value is higher or lower than the 372 upper (4.41) and lower (4.65) bounds calculated using the interquartile range (0.24) rule for potential

373 outliers (U.S. EPA, 2006). With the exclusion of potential outliers, these sources reported log Kow

- 374 values ranging from 4.25 to 4.79 (NLM, 2024; Elsevier, 2019; U.S. EPA, 2019; EC/HC, 2017; Ishak et
- 375 al., 2016: ECHA, 2012: Verbruggen et al., 1999: Mueller and Klein, 1992: Howard et al., 1985: SRC, 376 1984). The mean of the reported $\log K_{OW}$ values (excluding outliers) is 4.5. EPA selected an
- 377 experimental log Kow value of 4.5 (NLM, 2024) as this is consistent with the mean value obtained from
- 378 the overall high-quality data sources under standard environmental conditions. The identified value
- 379 replaces the value proposed in the Final Scope for the Risk Evaluation of Dibutyl Phthalate (DBP) (U.S. 380 EPA, 2020).

2.2.9 Henry's Law Constant

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Henry's Law constant (HLC) provides an indication of a chemical's volatility from water and gives an 382 indication of potential environmental partitioning, potential removal in sewage treatment plants during 383 air stripping, and possible routes of environmental exposure. EPA extracted and evaluated four high-384 quality data sources containing DBP HLC information. These sources reported DBP HLC values 385 ranging from 8.83×10^{-7} to 1.81×10^{-6} atm m³/mol (NLM, 2024; Elsevier, 2019; U.S. EPA, 2019; 386 <u>Cousins and Mackay, 2000</u>). The mean of the reported HLC values is 1.5×10^{-6} atm·m³/mol. EPA 387 selected the HLC value of 1.81×10^{-6} atm \cdot m³/mol (NLM, 2024) as the value obtained from the overall 388 high-quality data sources that best represents DBP's mean HLC under standard environmental 389 390 conditions. The identified value is consistent with the value proposed in the *Final Scope for the Risk* 391 Evaluation of Dibutyl Phthalate (DBP) (U.S. EPA, 2020).

2.2.10 Flash Point

EPA extracted and evaluated five high-quality data sources containing DBP flash point information. 393 394 These sources reported a DBP flash point of 157 to 171 °C (NLM, 2024; Elsevier, 2019; Rumble, 395 2018c; O'Neil, 2013; NIOSH, 2007). The mean of the reported flash point values is 162 °C. EPA 396 selected a flash point value of 157 °C (NLM, 2024) as the value that best represents the mean flash point value obtained from the available overall high-quality data sources under standard environmental 397 398 conditions. The identified value is consistent with the value proposed in the Final Scope for the Risk 399 Evaluation of Dibutyl Phthalate (DBP) (U.S. EPA, 2020).

400 2.2.11 Autoflammability

401 No autoflammability data for DBP were identified in the initial data review for the Final Scope for the 402 Risk Evaluation of DBP (U.S. EPA, 2020). After the final scope was published, two high-quality and 403 two medium-quality data sources were identified in the systematic review process. The autoflammability 404 values ranged from 402 to 403 °C (NLM, 2024; NCBI, 2020; Rumble, 2018c; NIOSH, 1976). The mean 405 of the reported autoflammability values is 402 °C. EPA selected an autoflammability value of 402 °C for 406 DBP (NLM, 2024) as the value that best represents the mean flashpoint value.

407 2.2.12 Viscosity

- 408 EPA extracted and evaluated three high-quality data sources containing DBP viscosity information. 409 These sources reported viscosity values ranging from 16.63 to 20.3 cP at 20 to 25 °C (NLM, 2024; 410 Elsevier, 2019; Rumble, 2018d). The mean of the reported values is 19.12 cP. EPA selected a value of 20.3 cP at 20 °C as the value that best represents the mean of reported viscosity values under standard 411 412 environmental conditions for this risk evaluation. The identified value is consistent with the value
- 413 proposed in the Final Scope for the Risk Evaluation of Dibutyl Phthalate (DBP) (U.S. EPA, 2020).

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416 The representative physical and chemical property values were selected based on professional

417 judgement and the weight of the scientific evidence, including the overall data quality ranking of the

418 associated references. These physical and chemical property values are then used to inform chemical-

419 specific decisions and model inputs across other disciplines. High and medium quality data are preferred

420 when selecting physical and chemical properties. In some instances where no data were available, 421 models such as EPI SuiteTM were used to estimate the value for the endpoint (*i.e.*, octanol:air partitioning

422 coefficient) and cross-checked with reported data from systematic review. The number and overall

423 quality of the available data sources results in different confidence strength levels for the corresponding

424 selected physical and chemical property values (U.S. EPA, 2021).

426 3 APPROACH AND METHODOLOGY FOR FATE AND 427 TRANSPORT ASSESSMENT

In assessing the environmental fate and transport of DBP, EPA considered reasonably available
environmental fate data including biotic and abiotic biodegradation rates, removal during wastewater
treatment, volatilization from lakes and rivers, and organic carbon:water partition coefficient (log K_{OC}).
The full range of results from data sources that were rated high- and medium-quality were considered
for fate endpoints.

432 433

434 Information on the full extracted data set is available in the file *Draft Risk Evaluation for Dibutyl* 435 *Phthalate (DBP) – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction* 436 *Information for Environmental Fate and Transport* (U.S. EPA, 2024a). When no measured data were 437 available from high- or medium-quality data sources, fate values were obtained from EPI SuiteTM (U.S. 438 EPA, 2017), a predictive tool for physical and chemical properties and environmental fate estimation. 439 Information regarding the model inputs is available in Section 3.2.1.

440

Table 3-1 provides a summary of the selected data that EPA considered while assessing the

442 environmental fate of DBP and were updated after publication of Final Scope of the Risk Evaluation for

443 *Dibutyl Phthalate (DBP)* (U.S. EPA, 2020) with additional information identified through the systematic 444 review process.

445

Property or Endpoint	Value(s)	Reference(s)
	$t_{1/2}$ = approximately 22 years at pH 7 and 25 °C	<u>ATSDR (1999)</u>
The last of the	$K_{\rm H}$ = 1.0 \pm 0.05E-02 $M^{\text{-1}}$ sec^{\text{-1}} at pH 10-12 and 30 $^{\circ}\text{C}$	<u>Wolfe et al. (1980)</u>
Hydrolysis	$t_{1/2} = 45.4$ hours at pH 10 and 30 °C	Zhang et al. (2019)
	$t_{1/2}$ = 3.43 years at pH 7 and 25 °C (estimated); $t_{1/2}$ = 125 days at pH 8 and 25 °C (estimated)	<u>U.S. EPA (2017)</u>
In direct also de care de tice	$t_{1/2} = 1.15$ days (estimated based on a 12-hour day with 1.5E06 OH/cm ³ and OH rate constant of 9.28E-12 OH/cm ³ and OH cm ³ /molecule-sec)	Peterson and Staples (2003)
Indirect photodegradation	$t_{1/2} = 1.13$ days (based on a 12-hour day with 1.5E06 OH/cm ³ and OH rate constant of 9.47E-12 OH/cm ³ and OH cm ³ /molecule-sec)	<u>Lei et al. (2018)</u>
Organic carbon:water partition coefficient (log K _{OC})	3.69 (average of 7 values ranging between 3.14 to 3.94)	Russell and Mcduffie (1986); Xiang et al. (2019)
Aerobic primary biodegradation in water	69% by BOD, 100% by UV-VIS, and 100% by GC after 2 weeks at a concentration of 100 ppm using an unspecified method (most likely Japanese MITI)	<u>NITE (2019)</u>

446 **Table 3-1. Summary of DBP's Environmental Fate Information**

Property or Endpoint	Value(s)	Reference(s)
	100% in 7 days based on loss of test substance in a synthetic medium containing 5 mg yeast extract	<u>Tabak et al. (1981)</u>
	68.3 to >99% (average: 89.8%) primary biodegradation after 28 days using inoculum prepared with soil, domestic, influent sewage microorganisms with a 2-week acclimation period	<u>SRC (1983a)</u>
Aerobic ultimate biodegradation in water	57.4% by theoretical CO_2 (ThCO2) evolution after 28 days	<u>SRC (1983a)</u>
	$84.6 \pm 2.1\%$ (mean \pm SD) after 14 days at 22 °C based on primary biodegradation	Johnson et al. (1984)
Aerobic biodegradation in sediment	16, 56, 73, and 86% after 7 days at 5, 12, 22, and 28 °C, respectively	
	$t_{1/2} = 2.9$ days in natural river sediment collected from the Zhonggang, Keya, Erren, Gaoping, Donggang, and Danshui Rivers in Taiwan.	<u>Yuan et al. (2002)</u>
Anaerobic biodegradation in sediment	$t_{1/2} = 14.4$ days in natural river sediment collected from the Zhonggang, Keya, Erren, Gaoping, Donggang, and Danshui Rivers in Taiwan.	<u>Yuan et al. (2002)</u>
Aerobic biodegradation in soil	88.1–97.2% after 200 days in Chalmers slit loam, Plainfield sand, and Fincastle silt loam soils.	<u>Inman et al. (1984)</u>
	101%, 128%, and 89% after 8 weeks by mean % theoretical gas production in revised anaerobic mineral medium (RAMM), American Society for Testing Materials (ASTM), and supplemental medium from Jackson, MI, respectively	
Anaerobic biodegradation	46%, 59%, and 19% after 8 weeks by % theoretical gas production in RAMM, ASTM, and supplemental medium from Holt, MI, respectively	Union Carbide (1974)
in WWTP sludge	72%, 117%, and 77% after 8 weeks by % theoretical gas production in RAMM, ASTM, and supplemental medium from Ionia, MI, respectively	
	$t_{1/2} = 5.1-6.2$ days in primary sludge from Lundofte municipal wastewater treatment plant acclimated to 10 mg/L di-ethylhexyl phthalate (DEHP) in Lyngby, Denmark	Gavala et al. (2003)
Removal in wastewater treatment	96.6% removal by degradation and decantation based on GC-MS analysis in Fontenay-les-Briis (Essonne-France) WWTP	<u>Tran et al. (2014)</u>

Property or Endpoint	Value(s)	Reference(s)
	Removal efficiency (approximate, based on figure): primary sedimentation: ca50%; chemical enhanced primary treatment: ca 100%; activated sludge: ca. 75%; sand filtration: ca. 95%; chlorination disinfection: ca. 20%	<u>Wu et al. (2017)</u>
	2.9 ± 0.1 and 30.6 ± 3.4 in brown shrimp (<i>Penaeus aztecus</i>) at 100 and 500 ppb, respectively	
Aquatic bioconcentration factor (BCF)	11.7 in sheepshead minnow (<i>Cyprinodon variegate</i>) at 100 ppb	<u>Wofford et al. (1981)</u>
	21.1 ± 9.3 and 41.6 ± 5.1 in American oyster (<i>Crassostrea virginica</i>) at 100 and 500 ppb, respectively	
Aquatic bioaccumulation factor (BAF)	100, 316, 251 and 1259 L/kg dry weight (dw) in bluegill, bass, skygager, and crucian carp, respectively.	Lee et al. (2019a)
	159 (estimated; upper trophic)	<u>U.S. EPA (2017)</u>
	Log BSAF: -1.6, -1.5, -1.5 and -1.4 kg/kg dw, in bluegill, bass, skygager, and crucian carp, respectively	Lee et al. (2019a)
Aquatic biota-sediment accumulation factor (BSAF)	BSAF: 0.2–2 (approximate range from figure) in Oreochromis miloticus niloticus, Liza subviridis, Acanthopagrus schlegeli, Zacco platypus and Acrossocheilus paradoxus	<u>Huang et al. (2008)</u>
	BSAF: 5.5 ± 4.8 , 6.0 ± 2.3 , and 11.8 ± 12.6 in roach, chub, and perch, respectively	<u>Teil et al. (2012)</u>
Aquatic Trophic Magnification Factor (TMF)	0.70 in 18 marine species	Mackintosh et al. (2004)
Terrestrial Biota-Soil Accumulation Factor (BSAF)	0.242 to 0.460 for earthworms	<u>Hu et al. (2005)</u> and <u>Ji and</u> <u>Deng (2016)</u>
Plant Concentration Factor (PCF)	0.26 to 4.78 (Fruit and vegetables)	<u>Sun et al. (2015)</u>

3.1 Tier I Analysis 447

- 448 To be able to understand and predict the behaviors and effects of DBP in the environment, a Tier I
- 449 analysis will determine whether an environmental compartment (*e.g.*, air, water, etc.) will accumulate
- DBP at significant concentrations (i.e., major compartment) or not (i.e., minor compartment). The first 450
- step in identifying the major and minor compartments for DBP is to consider partitioning values (Table 451
- 452 3-1), which indicate the potential for a substance to favor one compartment over another. DBP does not naturally occur in the environment; however, DBP has been detected in water, soil, and sediment in
- 453
 - 454 environmental monitoring studies (NLM, 2024; EC/HC, 2017).

455 3.1.1 Soil, Sediment, and Biosolids

Based on the partitioning values shown in Table 3-1, DBP will favor organic carbon over water or air. 456 457 Because organic carbon is present in soil, biosolids, and sediment, they are all considered major 458 compartments for DBP. This is consistent with monitoring data where higher concentrations of DBP 459 were detected in sediment samples (20–698 ng/g) compared to water samples (114–2,116 ng/L) 460 collected from the Mersey Estuary in the United Kingdom (NLM, 2024).

461 3.1.2 Air

462 DBP is a liquid at standard environmental temperatures with a melting point of -35° C and a vapor pressure of 2.01×10⁻⁵ mm Hg at 25 °C (NLM, 2024). DBP will exist both in the vapor (gaseous) phase 463 and particulate phase in the atmosphere (EC/HC, 1994). The mean concentration of DBP was 1.9 ± 1.3 464 465 ng/m^3 in the vapor phase and $4.0 \pm 2.2 ng/m^3$ in the particulate phase in air samples collected along the Niagara River (EC/HC, 1994). In another monitoring study from Paris, France, higher concentrations of 466 DBP were detected in the vapor phase (2.9 to 59.3 ng/m^3) compared to the particulate phase (0.6 to 4.6 467 468 ng/m³) (NLM, 2024). The octanol:air partition coefficient (KoA) indicates that DBP will favor the 469 organic carbon present in airborne particles. Based on its physical and chemical properties and short 470 half-life in the atmosphere ($t_{1/2} = 1.15$ days (U.S. EPA, 2017)), DBP in the vapor phase is assumed to 471 not be persistent in the air. The AEROWIN[™] module in EPI Suite[™] estimates that a fraction of DBP 472 may be sorbed to airborne particulates and these particulates may be resistant to atmospheric oxidation. 473 DBP has been detected in both indoor and outdoor air and settled house dust in the USA, Europe, 474 Canada and China (NLM, 2024; EC/HC, 2017; Kubwabo et al., 2013; Wang et al., 2013).

3.1.3 Water

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476 A log K_{AW} value of -4.131 indicates that DBP will favor water over air. With a water solubility of 11.2 477 mg/L at 25 °C, DBP is expected to be slightly soluble in water (Howard et al., 1985). In water, DBP is 478 likely to partition to suspended organic material present in the water column based on DBP's water 479 solubility of 11.2 mg/L (Howard et al., 1985) and organic carbon:water partition coefficient of 3.69 480 (Table 3-1). A monitoring study showed that total seawater DBP concentrations, in the False Creek 481 Harbor is a shallow marine inlet in Vancouver, ranged from 50 to 244 ng/L and the dissolved fraction 482 concentrations ranged from 34 to165 ng/L, compared to the suspended particulate fraction concentration 483 which ranged from 9,320 to 63,900 ng/g dry weight (dw) (Mackintosh et al., 2006). Although DBP has 484 low water solubility, surface water will be a major compartment for DBP since it is detected in the ng/L 485 range.

3.2 Tier II Analysis

A Tier II analysis involves reviewing environmental release information for DBP to determine if further 487 488 assessment of specific media is needed. The Toxics Release Inventory (TRI) reported the total on-site 489 releases for DBP in 2022 to be 130,800 pounds with 49,600 pounds released to air, 81,200 pounds 490 released to land, and none released to water. According to production data from the Chemical Data

Reporting (CDR) 2020 reporting period, between one million and ten million pounds of DBP were
produced annually from 2016–2019 for use in commercial products, chemical substances or mixtures
sold to consumers, or at industrial sites. Environmental release data from the Discharge Monitoring
Reports (DMRs) reported total annual releases for DBP from watershed discharge to be 1,224 total lb
per year from 585 watersheds in 2021, 5,149 total lb per year from 588 watersheds in 2022, and 16,555
total lb per year from 568 watersheds in 2023.

497

498 DBP is used mainly as a plasticizer in polyvinyl emulsions and can be used in adhesives, paints and 499 coatings, building materials, printing inks, fabric and textiles, children's toys, and plastic and rubber 500 materials (EC/HC, 2017, 1994). Because DBP is not chemically bound to the polymer matrix and can 501 migrate from the surface of polymer products (EC/HC, 2017), DBP can easily be released to the 502 environment from polymer-based products during their use and disposal. Additionally, DBP may be 503 released to the environment from the disposal of wastewater, and liquid and solid wastes. After 504 undergoing wastewater treatment processes, effluent is released to receiving waters and biosolids 505 (treated sludge) may be landfilled, land-applied, or incinerated and these processes may indicate that media-specific evaluations are necessary (Table 3-2). Releases from landfills and incinerators will occur 506 507 from the disposal of liquid and solid wastes and warrants media specific evaluations.

508

510

Environmental Releases	Key Pathway	Media Specific Evaluations
Wastewater and liquid waste treatment	Effluent discharge to water and land application of biosolids	Air, water, sediment, soil, groundwater, and biosolids
Disposal of liquids and Leachate discharge to water and biogas to air		Air, water sediment, soil, and groundwater
Incineration of liquid and solids	Stack emissions to air and ash to landfill	Air, water, sediment, soil, and groundwater
Urban/remote areas	Fugitive emissions to air	Air, water, sediment, soil, and groundwater
	Deposition	Water and soil
	Partitioning	Water, sediment, soil, and groundwater

509 **Table 3-2. Summary of Key Environmental Pathways & Media Specific Evaluations**

3.2.1 Fugacity Modeling

The approach described by Mackay et al. (1996) using the Level III Fugacity model in EPI SuiteTM V4.11 (LEV3EPITM) was used for this Tier II analysis. LEV3EPITM is described as a steady-state, nonequilibrium model that uses a chemical's physical and chemical properties and degradation rates to predict partitioning of the chemical between environmental compartments and its persistence in a model environment (U.S. EPA, 2017). Environmental degradation half-lives were taken from high and medium quality studies that were identified through systematic review to reduce levels of uncertainties (Table 3-3).

519 Table 3-3. DBP Half-Life Inputs Used in EPI Suite[™] Level III Fugacity Modeling

Media	Half-Life (days)	Reference (s)
Air	1.15	
Water	10	$L_{\rm of}$ at al. (2018), SPC (1082a)
Soil	20	<u>Lei el al. (2018)</u> ; <u>SRC (1985a)</u>
Sediment	90	

520

521 The following input parameters, taken from Table 2-1 and discussed in detail in Section 2.2, were used 522 in LEV3EPITM:

522 523 524

- Melting point = $-35.00 \text{ }^{\circ}\text{C}$
 - Vapor pressure = 2.01×10^{-5} mm Hg
- 525
- Water solubility = 11.2 mg/L
 Log Kow = 4.5
- 526 527
- SMILES: O=C(OCCCC)c(c(ccc1)C(=O)OCCCC)c1 (representative structure)

528

Based on DBP's environmental half-lives, partitioning characteristics, and the results of LEV3EPITM, DBP is expected to be found predominantly in water and soil (Figure 3-1). The model suggests that, under a continuous release scenario, 99.9 percent of releases to soil will remain in soil, 90 percent of releases to water will remain in water with about 10 percent partitioning to sediments, while 58 percent of releases to air will end up in soil with another 6 percent in water. The LEV3EPITM results were consistent with environmental monitoring data. DBP's partitioning behavior are further discussed in the media specific assessment (Section 1).



Figure 3-1. EPI Suite[™] Level III Fugacity Modeling for DBP

539

540 4 TRANSFORMATION PROCESSES

541 When released to the environment, DBP will be transformed to the monoester form (monobuty) 542 phthalate via abiotic processes such as photolysis and hydrolysis of the carboxylic acid ester group (U.S. 543 EPA, 2023). Biodegradation pathways for the phthalates consist of primary biodegradation from 544 phthalate diesters to phthalate monoesters and then to phthalic acid (PA), and ultimately biodegradation 545 of phthalic acid to form carbon dioxide (CO₂) and/or methane (CH₄) (Huang et al., 2013a; Wolfe et al., 546 1980). Monobutyl phthalate is both more soluble and more bioavailable than DBP. It is also expected to 547 undergo biodegradation more rapidly than the diester form. EPA considered DBP transformation 548 products and degradants qualitatively but due to their lack of persistence, these byproducts are not 549 expected to substantially contribute to risk, thus EPA is not considering them further in this risk 550 evaluation. Both biotic and abiotic degradation routes for DBP are described in the sections below.

551 **4.1 Biodegradation**

552 DBP is expected to be readily biodegradable in most aquatic and terrestrial environments. EPA extracted 553 and evaluated 85 biodegradation studies during systematic review. Twenty-six of these studies were 554 extracted and evaluated as overall high-quality data sources (Table 4-1). For the purposes of the

following biodegradation analysis, due to the large number of available high-quality data sources, EPA focused on studies that were given a high data quality rating.

4.1.1 Aerobic Biodegradation in Water

EPA extracted eight high-quality studies evaluating the primary aerobic biodegradation of DBP in water 558 559 (Table 3-1). Studies that used activated sludge inoculums reported primary biodegradation rates greater than 80 percent over 40 days (Desai et al., 1990), 100 percent over 14 days (Fujita et al., 2005), and 100 560 561 percent over 7 days (Tabak et al., 1981). However, there was an additional study using activated sludge as an inoculum with a degradation rate of 0 percent over 100 hours in an unacclimated inoculum, but a 562 563 rate of 100 percent over 100 hours using an inoculum that was acclimated to DBP over the course of 150 564 days using a DBP concentration of 100 mg/L (Jianlong, 2004). These findings suggest that DBP might 565 appear to be persistent when released to aquatic environments with microbial populations that require an adaptation phase to the initial DBP exposure; however, once adapted to DBP exposure, biodegradation 566 567 of DBP is expected to occur. A study using a combination inoculum of soil, activated sludge, and raw 568 sewage reported a primary biodegradation rate ranging from 68.3 to greater than 99 percent over 28 days 569 (SRC, 1983a). Three of the eight identified studies used natural surface water inoculums. Studies 570 reported primary biodegradation rates of 100 percent over 2 days in river water (Cripe et al., 1987), 100 571 percent over 14 days in river water (Fujita et al., 2005), and 100 percent over 14 days in pond water (Fujita et al., 2005); the other reported a half-life of 1.7 to 13 days estuarine and freshwater sites 572 573 (Walker et al., 1984).

574

557

575 Two studies evaluating the ultimate biodegradation of DBP in water were also extracted. One of the 576 studies reported rates of 50 to 70 percent, 40 to 60 percent, and 20 to 50 percent when using inoculums 577 of activated sludge, river water, and pond water, respectively (Fujita et al., 2005). The other reported 578 rates were 47.4 to 74.9 percent, with half-lives of 9.6 to 20.9 days when using a mixture of soil, 579 activated sludge, and raw sewage as the inoculum (SRC, 1983a). While the biodegradation rate of DBP in water will depend on the microbial community and its previous exposures to DBP, most of the data 580 581 on primary and ultimate biodegradation rates suggest an aerobic half-life of less than approximately 30 582 days using a variety of different inoculums. Therefore, for the purposes of this evaluation, DBP is

readily biodegradable under aerobic conditions and will have a half-life on the order of days to weeks.

584 4.1.2 Biodegradation in Sediment 585 EPA extracted twenty-three high-quality studies evaluating biodegradation of DBP in sediment. Out of 586 the twenty-three studies, EPA focused on the nine studies that did not use sediments amended with 587 external inoculums to best represent DBP's biodegradation under natural environmental conditions. One 588 of these studies reported 70.1 to 84.6 percent biodegradation of DBP in freshwater lake sediment under 589 aerobic conditions (Johnson et al., 1984). The same study also evaluated the effects of temperature and 590 DBP initial concentration on DBP's aerobic degradation rates. The results showed 70.1 to 72.6 percent 591 loss at initial DBP concentrations of 0.082 to 8.2 mg/L in 14 days and 73 to 86 percent loss at 22 to 28 592 °C in 7 days. However, the study reported 16 to 56 percent loss of DBP at 5 to 12 °C. These findings 593 showed that temperature could have a significant effect on biodegradation. Three of the selected data 594 sources reported DBP's biodegradation half-lives of 2.7, 2.9, and 46 days in marine sediments under 595 aerobic conditions (Li et al., 2015; Kickham et al., 2012; Yuan et al., 2010). The data source reporting a DBP biodegradation half-life of 46 days in marine sediment, reported higher than expected 596 597 biodegradation half-lives for other phthalates as well. In general, DBP is expected to have a sediment 598 biodegradation half-life of 2.7 to 2.9 days under normal aerobic environmental conditions, but extended 599 half-lives might be possible. 600 601 In contrast to aerobic conditions, phthalate esters are expected to have extended half-lives in sediment

602 under anaerobic conditions. Five of the nine extracted studies reported DBP biodegradation information 603 in sediment under anaerobic conditions (Li et al., 2015; Lertsirisopon et al., 2006; Chang et al., 2005; 604 Kao et al., 2005; Yuan et al., 2002). Chang et. al. (2005) reported 100 percent biodegradation in 28 days 605 at pH 7 and 30 °C in sediments with a nutrient content commonly used to support microbial growth. Kao 606 et. al. (2005) reported 24 percent biodegradation in 30 days at pH 7 and 30 °C in sediment samples with 607 water (not amended with nutrients for microbial growth). Additionally, half-lives in anaerobic sediments 608 have been reported to be 1.2 to 1.6 days in pond sediment (Lertsirisopon et al., 2006), 3.6 days in 609 submerged marine sediment (Li et al., 2015), and 5.1 to 12.7 days in river sediment (Yuan et al., 2002). 610 Overall, the available data show that there is variability in the biodegradation rates and that rates will 611 depend on environmental conditions, such as temperature, redox conditions, and pre-exposure of the 612 microbial communities to DBP. However, most of the studies evaluated suggest that at ambient 613 temperatures DBP will have a half-life of less than one year in both aerobic and anaerobic sediments. 614 Therefore, for the purposes of this evaluation it is assumed that DBP will have a half-life in sediments

on the order of weeks to months.

616

4.1.3 Biodegradation in Soil

EPA extracted eight high-quality studies evaluating biodegradation of DBP under aerobic and anaerobic 617 618 conditions in soil. Studies conducted using aerobic conditions report degradation rates of 88 to 98.6 percent over 200 days in silty loam and sandy soils (Inman et al., 1984); 100 percent over 72 hours in 619 soils of non-specified types from Broome County, New York (Russell et al., 1985); 100 percent over 15 620 621 days (Shanker et al., 1985) in an alluvial garden soil; and 66 percent over 30 days in soil taken from a 622 garden of an unspecified soil type (Wang et al., 1997a). Additionally, studies have reported aerobic biodegradation half-lives of 0.338 to 1.2 days in udic ferrosol and aquic cambisol soils (Cheng et al., 623 2018); 7.8 to 8.3 days in agricultural black soils (Xu et al., 2008); 1.6 days in a sandy clay loam (Yuan et 624 al., 2011); and 17.2 days in loam from a farm (Zhao et al., 2016). Anaerobic biodegradation rates have 625 626 been reported to be 97.8 percent over 200 days in a silt loam (Inman et al., 1984) and 66 percent over 30 627 days in soil taken from a garden of an unspecified soil type (Shanker et al., 1985). Overall, the available data show that there is variability in the biodegradation rates, depending on environmental conditions 628 629 such as temperature, redox conditions, and pre-exposure of the microbial communities to DBP. 630 However, the study with the slowest biodegradation rate in this evaluation suggests a half-life in soil of

approximately 65 days (<u>Inman et al., 1984</u>). Therefore, for the purposes of this evaluation, it is assumed that DBP will have a half-life in soils on the order of weeks to months. 631

632

633

634 Table 4-1. Summary of DBP Biodegradation Information

Environmental Conditions	Degradation Value	Half-life (days)	Reference	Overall Data Quality Ranking
	100%/2 days	N/A	<u>Cripe et al. (1987)</u>	High
	>80%/40 days	N/A	<u>Desai et al. (1990)</u>	High
	100%/14 days	N/A	<u>Fujita et al. (2005)</u>	High
Aerobic primary biodegradation in	100% in acclimated and 0% in unacclimated activated sludge/100 hours	N/A	<u>Jianlong (2004)</u>	High
water	68.3–99%/28 days	N/A	<u>SRC (1983a)</u>	High
	100%/7 days	N/A	<u>Tabak et al. (1981)</u>	High
	N/A	1.7–13 days	<u>Walker et al. (1984)</u>	High
	N/A	45.3–47.5 hours	<u>Wang et al. (1997b)</u>	High
Aerobic ultimate biodegradation in water	50–70% in activated sludge, 40–60% river water, and 20– 50% in pond water/14 days	N/A	<u>Fujita et al. (2005)</u>	High
	47.7–74.9%/28d days	9.6–20.9 days	<u>SRC (1983a)</u>	High
	70.9% at 0.082 mg/L, 70.1% at 0.82 mg/L, and 8.2% at 8.2 mg/L/14 days	N/A	Johnson et al. (1984)	High
Aerobic biodegradation in	16% at 5 °C, 56% at 12 °C, 73% at 22 °C, 86% at 28 °C/7 days	N/A	<u>Johnson et al. (1984)</u>	High
sediment	N/A	46 days in marine inlet sediment	Kickham et al. (2012)	High
	N/A	2.7 days in surface marine sediment	<u>Li et al. (2015)</u>	High

Environmental Conditions	Degradation Value	Half-life (days)	Reference	Overall Data Quality Ranking
	N/A	14.6 days in river sediment	Peng and Li (2012)	High
	N/A	0.6-5.4 days in river sediment	Yuan et al. (2002)	High
	N/A	1.6–2.9 days in mangrove sediment	Yuan et al. (2010)	High
	100%/28 days	9.4 days	<u>Chang et al. (2005)</u>	High
	24%/30 days in river sediment	N/A	Kao et al. (2005)	High
Anaerobic biodegradation in sediment	N/A	3.6 days in non- surface layer marine sediment	<u>Li et al. (2015)</u>	High
	N/A	5.1-12.7 days	Yuan et al. (2002)	High
	N/A	1.2–1.6 days in pond sediment	Lertsirisopon et al. (2006)	High
	N/A	0.338–1.2 days	<u>Cheng et al. (2018)</u>	High
	88–98.6%/200 days (CO ₂ evolution)	N/A	<u>Inman et al. (1984)</u>	High
	100%/72 hours	N/A	Russell et al. (1985)	High
Aerobic biodegradation in	100%/15 days	N/A	<u>Shanker et al. (1985)</u>	High
soil	66%/30 days	N/A	Wang et al. (1997a)	High
	N/A	7.8–8.3 days	<u>Xu et al. (2008)</u>	High
	N/A	1.6 days	Yuan et al. (2011)	High
	N/A	17.2 days	Zhao et al. (2016)	High
Anaerobic biodegradation in	97.8%/200 days (CO ₂ evolution)	N/A	<u>Inman et al. (1984)</u>	High
soil	66%/30 days	N/A	Shanker et al. (1985)	High

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636

4.2 Hydrolysis

The hydrolysis half-life of DBP at neutral pH and temperatures relevant to environmental waters is not
expected to be significant (Lei et al., 2018; Huang et al., 2013a; Wolfe et al., 1980). The hydrolysis halflife was reported to be approximately 22 years (ATSDR, 1999). Hydrolysis under acidic and alkaline

640 conditions is expected to occur with alkaline hydrolysis being more rapid. Alkaline hydrolysis will yield 641 phthalic acid with the monoester as an intermediate (Zhang et al., 2019; Huang et al., 2013a; Wolfe et 642 al., 1980). Zhang et al. (2019) evaluated the hydrolysis of DBP in aqueous alkaline solutions (pH 10) at 643 30 °C. The study reported hydrolysis to be rapid under the tested conditions, reporting a hydrolysis half-644 life of 45.4 hours. Temperature has also been shown to impact hydrolysis rates with hydrolysis rates 645 increasing with an increase in temperature. The hydrolysis half-life for DBP was reported to be 280.2 646 hours in neutral solution at a temperature of 80 °C. Wolfe et al. (1980) evaluated the hydrolysis of DBP 647 in aqueous alkaline solutions at 30 °C. The study reported a hydrolysis rate constant of $1.0 \pm 0.05 \times 10^{-2}$ 648 M^{-1} sec⁻¹ which corresponds to half-lives of 22 years at pH 7 and 8 days at pH 10. In addition, EPI 649 SuiteTM estimated the hydrolysis half-lives of DBP to be 3.43 years at pH 7 and 25 °C, and 125 days at 650 pH 8 and 25 °C (U.S. EPA, 2017) indicating that hydrolysis of DBP is more likely under more caustic 651 conditions and supporting DBP's resistance to hydrolysis under standard environmental conditions.

652

653 When compared to other degradation pathways, hydrolysis it is not expected to be a significant source of

654 degradation under typical environmental conditions. However, the higher temperatures, variations from 655 typical environmental pH, and chemical catalysts present in the deeper anoxic zones of landfills may be

- favorable to the degradation of DBP via hydrolysis (<u>Huang et al., 2013a</u>). This is discussed further in
- 657 Section 5.3.3.

658 **4.3 Photolysis**

659 DBP contains chromophores that absorb light at greater than 290 nm wavelength (NLM, 2013), 660 therefore, direct photodegradation is a relevant but minor degradation pathway for DBP released to air. 661 The major degradation pathway for DBP in air is indirect photodegradation with a measured half-life of 1.13 days (27.1 hours) (calculated from a •OH rate constant of 9.47×10^{-12} cm³/molecule-second 662 and a 12-hour day with 1.5×10^6 OH/cm³) (Lei et al., 2018). Similarly, Peterson and Staples (2003) 663 reported a calculated DBP photodegradation half-life of 1.15 days (≈ 27.6 hours) (calculated from a 664 ·OH rate constant of 9.28×10^{-12} cm³/molecule-second and 1.5×10^{6} OH/cm³). Indirect 665 photodegradation of DBP will yield MBP, PA, di-butyl 4-hydroxyphthalate (m-OH-DBP), and di-butyl 666 4-nitrophthalate (m-NO₂-DBP) (Lei et al., 2018). DBP photodegradation in water is expected to be 667 668 slower than air, due to the typical light attenuation in natural surface water. There is limited information on the aquatic photodegradation of DBP. However, Lertsirisopon et al. (2009) reported 669 670 DBP aquatic direct photodegradation half-lives of 50, 66, 360, 94 and 57 days at pH 5, 6, 7, 8 and 9, 671 respectively, when exposed to natural sunlight in artificial river water at 0.4 to 27.4 °C (average 672 temperature of 10.8 °C). Peterson and Staples (2003) also reported a half-life of 3 hours for aqueous 673 photolysis of DBP in natural sunlight when DBP was present in a surface microlayer on the water at 674 mg/L concentrations. The rate was noted to be stimulated by titanium dioxide and hydrogen peroxide. 675 These findings suggest DBP will be susceptible to photochemical decay in air but that photolysis is not 676 expected to be a significant degradation process in surface water.

678 **5 MEDIA ASSESSMENTS**

DBP has been reported to be present in the atmosphere, aquatic environments, and terrestrial 679 680 environments. Once in the air, DBP will be most predominant in the organic matter present in airborne particles and is expected to have a short half-life in the atmosphere. Based on its physical and chemical 681 properties, DBP is likely to partition to house dust and airborne particles in the indoor environment and 682 683 is expected to have a longer half-life in indoor air as compared to outdoor air. DBP present in surface 684 water is expected to partly partition to aquatic sediments and have an aerobic biodegradation half-life ranging from days to weeks. In terrestrial environments, DBP has the potential to be present in soils and 685 groundwater but is likely to be immobile in both media types. In soils, DBP is expected to be deposited 686 via air deposition and land application of biosolids. DBP in soils is expected to have a half-life on the 687 688 order of weeks to months, and to have low bioaccumulation potential and biomagnification potential in terrestrial organisms. DBP will be released to groundwater via infiltration from wastewater effluent and 689 landfill leachates but is not likely to be persistent in groundwater and/or subsurface environments unless 690 691 anoxic conditions exist.

692 **5.1 Air and Atmosphere**

DBP is a liquid at environmental temperatures with a melting point of -35 °C (Haynes, 2014a) (Rumble, 693 694 2018b) and a vapor pressure of 2.01×10^{-5} mmHg at 25 °C (NLM, 2024). Based on its physical and chemical properties and short half-life in the atmosphere ($t_{1/2} = 1.15$ days (Peterson and Staples, 2003)), 695 DBP is not expected to be persistent in air. The AEROWINTM module in EPI SuiteTM estimated a log 696 697 KOA of 8.63, which suggests that a fraction of DBP may be sorbed to airborne particles and these 698 particulates may be more resistant to atmospheric oxidation. Thus, DBP has the potential to undergo dry 699 deposition and wet deposition into soils and surface water (Zeng et al., 2010; Peters et al., 2008; Xie et 700 al., 2005; Parkerton and Staples, 2003; Atlas and Giam, 1981). Two studies reported a range of 33 to 46 percent of DBP concentration in the air to be associated with suspended particles (Xie et al., 2007; Xie 701 702 et al., 2005). A net deposition of DBP from ambient air into the North Sea was also measured (Xie et al., 703 2005). Based on DBP's short half-life in the atmosphere, it is not expected to be persistent in 704 atmospheric air under standard environmental conditions. 705

Three studies reported DBP to be detected in air at concentrations of greater than 0.002 to 3.4 ng/m^3 over the North Sea (Xie et al., 2005), 0.2 to 0.6 ng/m³ over the Arctic (Xie et al., 2007), and 0.4 to 1.8 ng/m³ over the North Pacific Ocean (Atlas and Giam, 1981). Other studies measured concentrations of DBP in ambient air ranging from 23.7 to 191 ng/m³ in the United States (Wilson et al., 2003; Wilson et al., 2001) and 0.08 to 15 ng/m³ in Sweden (Cousins et al., 2007).

711

5.1.1 Indoor Air and Dust

In general, phthalate esters are ubiquitous in the atmosphere and indoor air. Their worldwide presence in
air has been documented in the gas phase, suspended particles, and dust (Net et al., 2015). A log K_{OA}
value of 8.63 suggests a strong affinity of DBP for organic matter in air particulates. DBP is expected to
be more persistent in indoor air than in outdoor air due to the lack of natural chemical removal
processes, such as solar photochemical degradation.

717

EPA identified several data sources reporting the presence of DBP in indoor air and dust in the United
 States. These studies reported the presence of DBP at higher concentrations in indoor dust samples than

in indoor air, supporting DBP's strong affinity and partitioning to organic matter in dust. Wilson et al.

721 (2001) reported measured samples of indoor air and dust from ten daycare centers located in North

722 Carolina. DBP was detected in all air and dust samples with a mean concentration of 239 ng/m³ (108–

404 ng/m³) in air samples and a mean concentration of 18.4 ppm (1.58–46.3 ppm) in dust samples. In a second study, Wilson et al. (2003) reported measured samples of indoor air and dust from two other daycare centers located in North Carolina with a mean concentration of 488 ng/m³ (222–786 ng/m³) in air samples and a mean concentration of 1.87 ppm (0.058–5.85 ppm) in dust samples. Air and dust samples were collected from residential and office buildings in Massachusetts with a 100 percent detection frequency for DBP. Concentrations of DBP were found to be a mean of 0.251 µg/m³ (0.101– 0.41 µg/m³) in air and a mean of 27.4 µg/g (11.1–59.4 µg/g) in dust (Rudel et al., 2001).

730

731 EPA also identified several data sources reporting the presence of DBP in indoor air and dust outside of 732 the United States. Das et al. (2014) explored the implications of industrial activities by comparing the 733 presence of phthalates in two different cities from India. The study analyzed indoor air and dust samples from the Jawaharlal Nehru University campus (a city with low industrial activities) and Okhla (a city 734 735 with high industrial activities related to the use of phthalates), reporting a general tendency of higher 736 detectable concentrations of DBP in air and dust samples collected in the city of Okhla. This finding 737 suggests that higher concentrations of phthalates in air and dust could be expected near facilities with 738 high use and production of phthalates. Wormuth et al. (2006) determined the indoor air and indoor dust 739 concentrations DBP based on measured concentrations of phthalates in dust of European homes. The study reported DBP mean concentrations of 1,153 ng/m³ and 98 mg/kg for indoor air and indoor dust, 740 respectively. In a study done in Sapporo, Japan, DBP was found to range from 79.6 to 740 ng/m³ in air 741 and 1.8 to 1,476 ng/m³ in indoor dust in residential houses (Kanazawa et al., 2010). DBP was found to 742 743 be the dominating phthalate in a study which analyzed the phthalate content (DBP, BBP, dicyclohexyl phthalate [DCHP], and di-ethylhexyl phthalate [DEHP]) of particulate matter in indoor spaces in 744 745 Norway (Rakkestad et al., 2007).

746 **5.2 Aquatic Environments**

747

751

5.2.1 Surface Water

DBP is expected to be released to surface water via industrial and municipal wastewater treatment plant
effluent, surface water runoff, and, to a lesser degree, atmospheric deposition. DBP has frequently been
detected in surface waters (Zeng et al., 2008a; Tan, 1995; Preston and Al-Omran, 1989).

752 The principal properties governing the fate and transport of DBP in surface water are water solubility 753 (11.2 mg/L, Table 2-1), log K_{AW} (-4.131, Table 3-1), and log K_{OC} (3.14-3.94, Table 3-1). Due to its HLC $(1.81 \times 10^{-6} \text{ atm} \cdot \text{m}^3/\text{mol} \text{ at } 25 \text{ °C}$, Table 2-1), volatilization is not expected to be a significant 754 source of loss of DBP from surface water. A partitioning analysis estimates that about 10 percent of the 755 756 DBP released to water will partition to sediments and approximately 90 percent will remain in surface 757 water (see Section 3.2.1). However, based on its log Koc (3.14–3.94), DBP in water is expected to 758 partition to suspended particles and sediments. DBP is also expected to biodegrade rapidly in most 759 aquatic environments (Section 4.1.1) and thus is not expected to persist in surface water except at areas 760 of continuous release, such as a water body receiving discharge from a municipal wastewater treatment plant, where rate of release exceeds the rate of biodegradation. 761

762

No monitoring data for DBP in surface water was readily available for the United States. Several studies

from outside the U.S. were examined. The available data sources reported the presence of DBP and

other phthalates in surface water samples collected from rivers and lakes. Preston and Al-Omran (<u>1989</u>)

- explored the presence of phthalates within the River Mersey Estuary and reported the presence of DBP
- freely dissolved in water at concentrations ranging from 0.541 to 1.805 μ g/L. Tan (<u>1995</u>) reported the

- presence of DBP in Klang River at concentrations of 0.8 to 4.8 μ g/L. Zeng et al. (2008a) reported the
- presence of DBP in the dissolved aqueous phase of urban lakes in Guangzhou City at mean
 concentrations of 2.03 µg/L. Grigoriadou et al. (2008) reported the presence of DBP in lake water
- samples collected near the industrial area of Kavala city at concentrations of 0.640 to $16 \mu g/L$. The total
- seawater concentrations of DBP in False Creek Harbor, Vancouver ranged from 50 to 244 ng/L with the
- dissolved fraction concentrations ranging from 34 to 165 ng/L. The bottom sediment concentrations
- ranged from 57 to 182 ng/g dw. The concentration in suspended sediment ranged from 9,320 to 63,900
 ng/g dw (Mackintosh et al., 2006). These results show higher concentrations of DBP in the suspended
- sediments than in the dissolved phase or the bottom sediment, which was not expected given the Koc
- value and partitioning analysis results for DBP. This suggests that partitioning of DBP to sediments may
- be much higher than what was predicted in the partitioning analysis and that the concentrations of DBP
- in water may be mostly found in suspended sediment.

5.2.2 Sediments

Based on a log K_{OC} range of 3.14 to 3.94, DBP will partition to the organic matter present in soils and
sediment when released into aquatic environments. Once in water, LEV3EPITMpredicts that close to 90
percent of the DBP will remain in water (U.S. EPA, 2017) (see Section 3.2.1). However, some data
sources have documented higher concentrations of DBP in suspended solids than the dissolved phase
(Mackintosh et al., 2006).

- 786 787 DBP is expected to biodegrade rapidly in aquatic sediments with a half-life of weeks to months (see 788 Section 4.1.2). Due to its strong affinity to organic carbon (log $K_{OC} = 3.14-3.94$), DBP is expected to 789 partly partition to aquatic sediments. This is consistent with the monitoring data sources containing 790 information on the presence of DBP in river sediment samples. DBP concentrations in river sediment 791 ranged between 3 to 3,670 ng/g dw (Cheng et al., 2019; Li et al., 2017b; Li et al., 2017a; Tang et al., 792 2017; Tan, 1995; Preston and Al-Omran, 1989).
- 793

780

No monitoring data for DBP in surface water was readily available for the United States. Several studies
from outside the U.S. were examined. Mackintosh (2006) reported higher concentrations of DBP in the
suspended particles than in deep sediment samples collected from the False Creek Harbor in Vancouver,
Canada. The study reported DBP mean concentrations of 103 and 22,400 ng/g in the deep sediment and
suspended particles, respectively.

799

800 In another study, Kim (2021) evaluated the presence of plasticizers in sediments from highly industrialized bays of Korea. DBP was detected in 95 percent of the collected surface sediment samples 801 802 at a median concentration of 13.2 ng/g dw. The study revealed a gradual decreasing trend in the overall 803 concentration of phthalates toward the outer region of the bays farther away from industrial activities. 804 The findings of this study suggests that industrial activities are a major contributor of phthalates in 805 sediments within the area. It also suggests that DBP has the potential to accumulate in sediments at areas 806 of continuous release, such as a surface water body receiving discharge from a municipal wastewater 807 treatment plant.

808 5.3 Terrestrial Environments

809 **5.3.1 Soil**

810 DBP is expected to be deposited to soil via two primary routes: 1) application of biosolids and sewage

- sludge in agricultural applications or sludge drying applications; and 2) atmospheric deposition. Based
- on DBP's HLC of 1.81×10^{-6} atm·m³/mol at 25 °C and vapor pressure of 2.01×10^{-5} mmHg at 25 °C,
- 813 DBP is not likely to volatilize significantly from soils.
- 814
- B15 DBP is expected to show strong affinity for sorption to soil and its organic constituents based on a log
- 816 Koc of 3.14–3.94 (Xiang et al., 2019; Russell and Mcduffie, 1986), and a log Kow of 4.5 (NLM, 2024).
- 817 Thus, DBP is expected to have slow migration potential in soil environments. In addition, DBP is
- 818 expected to biodegrade rapidly in soil with a half-life of weeks to months. In general, DBP is not
- 819 expected to be persistent in soil as long as the rate of release does not exceed the rate at which820 biodegradation can occur.
- 821

Available data sources have reported the presence of DBP in soil samples. Concentrations ranging from 0.49 to 3.59 mg/kg dw were measured in soil and sediment samples in a vacant tract adjacent to the Union Carbide Corporation's Bound Brook plant in New Jersey (EPM, 1988). Soil samples from waste

- Union Carbide Corporation's Bound Brook plant in New Jersey (ERM, 1988). Soil samples from waste
 disposal sites in Taizhou, China were shown to be contaminated by DBP through improper disposal of
- 826 electronic waste. DBP and DEHP were two of the major phthalates in the study with DBP ranging from
- 1 to 5 mg/kg in the soil samples (<u>Liu et al., 2009</u>). DBP, di-isobutyl phthalate (DIBP), and DEHP were also found to be the main phthalates in agricultural soils in peri-urban areas around Guangzhou, China
- with a 100 percent detection frequency. In New York, DBP was found in soil at concentrations ranging
- from 0.009 to 2.74 μ g/g dw, which exceeds the recommended allowable soil concentrations for DBP set by the state of New York (0.081 μ g/g). The study attributed the source of the phthalates to wastewater
- irrigation, sewage sludge application, disposed plastics and atmospheric deposition ($\underline{\text{Zeng et al., 2008b}}$).
- 833 Similarly, another study found DBP in abundance in Chinese arable soil. Zeng et al. (2009) reported
- BBP concentrations ranging from 0.206 to $30.1 \,\mu g/g$ dw in soils from roadsides, residential areas, and parks in Guangzhou, China.
- 836

5.3.2 Biosolids

Sludge is defined as the solid, semi-solid, or liquid residue generated by wastewater treatment processes.
The term "biosolids" refers to treated sludge that meets the EPA pollutant and pathogen requirements
for land application and surface disposal and can be beneficially recycled (40 CFR Part 503) (U.S. EPA,
1993). Typically, chemical substances with very low water solubility and high sorption potential are
expected to be sorbed to suspended solids and efficiently removed from wastewater via accumulation in
sewage sludge and biosolids.

843

844 As described in Section 6.2, DBP in wastewater has been reported to be mainly removed by particle 845 sorption and retained in the sewage sludge. Based on the STPWIN[™] module in EPI Suite[™], about 55 846 percent of DBP present in wastewater is expected to be accumulated in sewage sludge and discharged 847 into biosolids. The National Sewage Sludge Survey detected DBP in 1998 at a mean concentration of 848 11,200 µg/kg, a standard deviation of 17,800 µg/kg, a maximum concentration of 331,000 µg/kg and a 4 percent detection frequency. Separately, DBP concentrations ranging from 1.7 to 1,260 ng/g dw were 849 850 measured in 20 municipal sewage sludge samples from publicly owned treatment works in the United 851 States (Ikonomou et al., 2012). 852

- 853 Three studies have reported DBP's concentration in sludge in 71 Chinese WWTPs ranging from 0.0004
- to 111 μg/g dw (<u>Zhu et al., 2019</u>; <u>Meng et al., 2014</u>) and 0.58 to 59 μg/g dw in 40 Korean WWTPs (<u>Lee</u>
- 855 <u>et al., 2019b</u>). Two U.S. studies reported sludge concentrations ranging from 0.32 to $17 \mu g/g dw$
- 856 (<u>Howie, 1991</u>) and 966 μ g/L (<u>ATSDR, 1999</u>). When in biosolids, DBP may be transferred to soil during
- land applications. DBP is likely to be more persistent in soil due to its strong sorption potential (Section
- 5.3.1). Land applied DBP is expected to be moderately mobile in the environment despite its strong

sorption to soils. Disposal of sewage effluent has been reported to contaminate groundwater with DBP
 concentrations up to 450 mg/L (<u>ATSDR, 1999</u>).

5.3.3 Landfills

862 For the purpose of this assessment, landfills will be divided into two zones: 1) an "upper-landfill" zone, 863 with standard environmental temperatures and pressures, where biotic processes are the predominant route of degradation for DBP, and 2) a "lower-landfill" zone where elevated temperatures and pressures 864 865 exist, and abiotic degradation is the predominant route of degradation for DBP. In the upper-landfill zone where oxygen may still be present in the subsurface, conditions may still be favorable for aerobic 866 biodegradation, however photolysis and hydrolysis are not considered to be significant sources of 867 868 degradation in this zone. In the lower-landfill zone, conditions are assumed to be anoxic, and 869 temperatures present in this zone are likely to inhibit biotic degradation of DBP. Temperatures in lower landfills may be as high as 70 °C. At temperatures at and above 60 °C, biotic processes are significantly 870 871 inhibited, and are likely to be completely irrelevant at 70 °C (Huang et al., 2013a). 872

873 DBP is deposited in landfills continually and in high amounts from the disposal of consumer products 874 containing DBP. Some aerobic biodegradation may occur in upper-landfills. Similar to other phthalate 875 esters, under anaerobic conditions present in lower-landfills, DBP is likely to be persistent in landfills 876 due to the expected low rates of anaerobic biodegradation in lower-landfills. There is some evidence to support that hydrolysis may be the main route of abiotic degradation of phthalate esters in lower-877 878 landfills (Huang et al., 2013a). Due to the expected persistence of DBP in landfills, it may dissolve into 879 leachate in small amounts based on a water solubility of 11.2 mg/L and may travel slowly to ground 880 water during infiltration of rainwater based on a log Koc of 3.14 to 3.94. For instance, several data 881 sources have reported the presence of DBP in landfill leachate. These sources have reported a rapid 882 decrease in DBP concentration from core to leachate samples (Norin and Strömvall, 2004; Jang and 883 Townsend, 2001; Öman and Hynning, 1993; DERS, 1991). These data sources reported DBP concentrations ranging from 0.4 to 7.8 mg/kg and 1 to 17 µg/L in landfill core and leachate samples, 884 respectively. The reported rapid decrease in DBP's concentration aligns with the expectation that DBP is 885 likely to sorb to organic matter in landfill soils. 886

5.3.4 Groundwater

There are several potential sources of DBP in groundwater, including wastewater effluents and landfill leachates, which are discussed in Sections 5.3.3 and 6.2. Furthermore, in environments where DBP is found in surface water, it may enter groundwater through surface water/groundwater interactions. Diffuse sources include storm water runoff and runoff from biosolids applied to agricultural land.

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887

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893 Even though DBP has a strong affinity to adsorb to organic matter present in soils and sediments (log 894 Koc = 3.14–3.94 (Xiang et al., 2019; Russell and Mcduffie, 1986)), DBP partitioning to groundwater is 895 possible, though will be limited by DBP's low water solubility (11.2 mg/L). For instance, the presence of DBP in groundwater has been reported at concentrations of 0.12 mg/L in Carson, California 896 897 (Geraghty & Miller Inc, 1990). In cases where DBP could reasonably be expected to be present in 898 groundwater environments (proximal to landfills or agricultural land with a history of land applied 899 biosolids), limited persistence is expected based on rates of biodegradation of DBP in aerobic and 900 anaerobic environments (Section 4.1), and DBP is not likely to be persistent in groundwater or 901 subsurface environments unless anoxic conditions exist.

903 6 REMOVAL AND PERSISTENCE POTENTIAL OF DBP

DBP is not expected to be persistent in the environment, as it is expected to degrade rapidly under most environmental conditions, with lower biodegradation potential in low-oxygen media. In the atmosphere, DBP is unlikely to remain for long periods of time as it is expected to undergo photolytic degradation through reaction with atmospheric hydroxyl radicals, with an estimated half-life of 1.15 days. In aquatic

908 environments, DBP is predicted to hydrolyze slowly at ambient temperature, but it is not expected to

- 909 persist since it undergoes rapid aerobic biodegradation (Section 5.2.1). In soil and sediments, DBP has
- 910 the potential to remain for longer periods of time. Due to the rapid biodegradation under most aquatic 911 environments and its estimated BCF of 159.4 L/kg, DBP is expected to have low bioaccumulation
- potential. Using LEV3EPITM (Section 3.2.1), DBP's overall environmental half-life was estimated to be
- approximately 14 days (<u>U.S. EPA, 2017</u>). Therefore, DBP is not expected to be persistent in the
- atmosphere or aquatic and terrestrial environments.

915 6.1 Destruction and Removal Efficiency

916 Destruction and Removal Efficiency (DRE) is a percentage that represents the mass of a pollutant

- 917 removed or destroyed in a thermal incinerator relative to the mass that entered the system. DBP is
- classified as a hazardous substance (40CFR116.4) and EPA requires that hazardous waste incineration

systems destroy and remove at least 99.99 percent of each harmful chemical in the waste, including
 treated hazardous waste (46 FR 7684) (Federal Register, 1981).

920 921

Currently there is limited information available on the DRE of DBP. The available data sources reported

the presence of DBP in the ashes and exhaust gas from hazardous waste incinerators at concentrations of 0 to 1200 μ g/kg and 7.66 to 260 μ g/m³, respectively (Jav and Stieglitz, 1995; Nishikawa et al., 1992;

924 0 to 1200 μg/kg and 7.66 to 260 μg/m³, respectively (Jay and Stieglitz, 1995; Nishikawa et al., 1992;
 925 Shane et al., 1990). These findings suggest that incineration of DBP containing waste has the potential

925 Share et al., 1990). These findings suggest that incineration of DBP containing waste has the potential
 926 to contribute to DBP concentrations in air. However, EPA estimated that highest waste incineration

stack emissions for DBP to be 0.03 tons per year, which corresponds to 0.058 percent of the reported
 DBP TRI air releases in 1990 (Dempsey, 1993). This suggest that DBP present during incineration

929 processes will mainly be released with ash to landfills, with a small fraction released to air as stack

emissions. Based on its hydrophobicity and sorption potential, DBP released to landfills is expected to
 partition to waste organic matter. Similarly, DBP released to air is expected to rapidly react via indirect

photochemical processes within hours (U.S. EPA, 2017) or partition to soil and sediments as described in Section 3.2.1. DBP in sediments and soils is not expected to be bioavailable for uptake and is highly

biodegradable in its bioavailable form (Kickham et al., 2012).

6.2 Removal in Wastewater Treatment

936 Wastewater treatment is performed to remove contaminants from wastewater using physical, biological, 937 and chemical processes. Generally, municipal wastewater treatment facilities apply primary and 938 secondary treatments. During the primary treatment, screens, grit chambers, and settling tanks are used 939 to remove solids from wastewater. After undergoing primary treatment, the wastewater undergoes a 940 secondary treatment. Secondary treatment processes can remove up to 90 percent of the organic matter 941 in wastewater using biological treatment processes such as trickling filters or activated sludge. 942 Sometimes an additional stage of treatment such as tertiary treatment is utilized to further clean water 943 for additional protection using advanced treatment techniques (e.g., ozonation, chlorination, 944 disinfection) (U.S. EPA, 1988).

945

935

946 EPA selected twelve high-quality data sources reporting the removal of DBP in wastewater treatment 947 systems employing both aerobic and anaerobic processes. These sources reported a range of 38 to

948 greater than 99 percent removal of DBP in WWTPs employing secondary and/or tertiary treatment units

949 such as activated sludge, secondary clarifiers, and sand filtration (Wu et al., 2017; Huang et al., 2013b; 950 Shao and Ma, 2009; Peterson and Staples, 2003) (Table 6-1). These studies reported that biodegradation 951 accounted for 27 to 58.9 percent of the overall DBP removal (Shao and Ma, 2009; Peterson and Staples, 952 2003), that the main removal mechanisms are sorption and biodegradation during the primary and 953 secondary treatment, respectively (Wu et al., 2017; Huang et al., 2013b), and that WWTPs employing 954 secondary and tertiary treatment achieve greater than 99 percent removal of DBP (Wu et al., 2017). The 955 median removal of DBP has been reported to be 68 to 98 percent within 50 WWTPs in the United States 956 (U.S. EPA, 1982).

957

958 Based on the available information, the main mechanisms for the removal of DBP in conventional 959 municipal WWTPs are sorption to suspended organic matter, biodegradation during activated sludge 960 treatment, or a combination of sorption and biodegradation. For instance, recent studies have reported 961 greater than 93 percent removal of DBP in three conventional WWTPs with activated sludge treatment 962 in South Africa (Salaudeen et al., 2018a, b). The studies reported that DBP is mainly removed by 963 sorption to suspended particles and sludge. Tran et al. (2014) reported similar findings in a WWTP in 964 France employing a combined decantation and activated sludge tank that achieved 96.6 percent removal 965 of DBP. The study reports that the evaluated phthalate esters (DIBP, DBP, BBP, DEHP, di-isononyl 966 phthalate [DINP], and di-isododecyl phthalate [DIDP]) were mainly removed by sorption to solids. 967 Other studies have reported biodegradation during the activated sludge treatment process to be the main 968 removal mechanism of DBP in two WWTPs in Denmark and India, achieving greater than 91 percent 969 removal of DBP (Saini et al., 2016; Roslev et al., 2007).

- 970 971 In contrast to higher molecular weight phthalate esters, DBP has been reported to be efficiently removed 972 during anoxic and anaerobic wastewater treatment processes (Table 6-1). Gani and Kazmi (2016) 973 evaluated the removal efficiency of DBP in three WWTPs employing anoxic, aerobic, and anaerobic 974 treatment units near the Ganga and Dhamola rivers in India. The wastewater treatment plants 975 investigated were designed as nutrient removal-based sequencing batch reactor (WWTP1-SBR) with 976 anoxic pretreatment zone followed by an activated sludge unit, a conventional activated sludge process 977 (WWTP2-ASP) and up-flow anaerobic sludge blanket (WWTP3-UASB) with a polishing pond. The 978 study reported that biotransformation processes accounted for 70, 67, and 61 percent of the overall DBP 979 removal in the SBR, ASP, UASB treatment plants, respectively. Sorption accounted for less than 5 980 percent of the overall removal of DBP. These findings suggest DBP to be biodegradable under anaerobic 981 conditions. This is supported by a study that explored the efficiency of anaerobic and aerobic sludge 982 post-treatment for the removal of phthalate esters (PAEs) and reported complete removal of DBP during 983 the anaerobic phase (Tomei et al., 2019).
- 984

985 Unlike phthalate esters with longer carbon chains, DBP's water solubility (11.2 mg/L) and log Koc 986 (3.14–3.94) suggest partial removal in WWTP via sorption to sludge. This finding is supported by 987 STPWINTM, which predicted 56 percent of DBP to be removed during conventional wastewater 988 treatment by sorption to sludge with the potential for higher removal via rapid aerobic biodegradation 989 processes (U.S. EPA, 2017). In general, the available information suggests that aerobic processes have 990 the potential to help biodegrade DBP from wastewater, which is in agreement with the expected aerobic 991 biodegradation described in Section 3.1. Air stripping within the aeration tanks for activated sludge 992 processing is not expected to be a significant removal mechanism for DBP present in wastewater. In 993 general, based on the available measured and predicted information, WWTPs are expected to remove 65 994 to 98 percent of DBP present in wastewater.

Data Quality Property Selected Value(s) **Reference**(s) Rating >93% removal; DBP removal in High Salaudeen et al. (2018a); three activated sludge WWTPs in Salaudeen et al. (2018b) South Africa; main removal Removal by sorption mechanism: sorption 96.6% removal, main removal Tran et al. (2014) High mechanism: sorption 91% removal; biodegradation Roslev et al. (2007) High during activated sludge process Removal by biodegradation Saini et al. (2016) 92.67% removal; biodegradation High during activated sludge process 90.10% removal; sorption and High Huang et al. (2013b) biodegradation during the primary and secondary treatment, respectively 85.9% overall removal, 58.9% N2F Shao and Ma (2009) biodegradation, and 11.3% sorption to solids 38 to >99% removal; Wu et al. (2017) High Removal by biodegradation and 85 and 95%; biodegradation Peterson and Staples N2F sorption accounted about 27 percent of the (2003)overall removal >57% Wu et al. (2019) High 70% (SBR), Gani and Kazmi (2016) High 67% (ASP), 61% (UASB); <5% sorption, mainly biodegradation Anaerobic sludge post->99% removal, anaerobic sludge Tomei et al. (2019) High treatment post-treatment WWTP = Wastewater treatment plant; SBR = Sequencing batch reactor; ASP = Activated sludge process; UASB = Up-flow anaerobic sludge blanket

996 Table 6-1. Summary of DBP's WWTP Removal Information

997

6.3 Removal in Drinking Water Treatment

Drinking water in the United States typically comes from surface water (*i.e.*, lakes, rivers, reservoirs)
 and groundwater. The source water flows to a treatment plant where it undergoes a series of water

1000 treatment steps before being dispersed to homes and communities. In the United States, public water

systems often use conventional treatment processes that include coagulation, flocculation,

sedimentation, filtration, and disinfection, as required by law.

- 1004 Limited information is available on the removal of DBP in drinking water treatment plants. A water 1005 concentration of 100 ng/L was measured in the city of Philadelphia's drinking water (Roy F. Weston 1006 Inc, 1980). Several available data sources reported concentrations of DBP in drinking water outside the 1007 United States (1–1,830 ng/L) (Ding et al., 2019; Li et al., 2019; Kong et al., 2017; Shan et al., 2016; Das 1008 et al., 2014; Shi et al., 2012). Kong et al. (2017) explored the presence and removal of phthalate esters in 1009 a drinking water treatment system in east China employing coagulation, sedimentation, and filtration 1010 treatment processes, and reported 64.5 percent removal of DBP from the treated effluent with a drinking 1011 water concentration of 17.2 ng/L. Similarly, Shan et al. (2016) explored the removal of phthalate esters 1012 in two drinking water treatment plants in east China. The first plant employs coagulation, sedimentation, 1013 filtration, and disinfection treatment processes and reported 31 to 48 percent removal of DBP from the 1014 treated effluent while the second plant reported 38 to 56 percent removal of DBP from the treated 1015 effluent in a drinking water treatment system employing peroxidation, coagulation, combined 1016 flocculation and sedimentation, filtration, and disinfection treatment processes. These findings suggest 1017 that conventional drinking water treatment systems have the potential to partially remove DBP present
- 1018 in source water via sorption to suspended organic matter and filtering media.
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1020 7 BIOACCUMULATION POTENTIAL OF DBP

The presence of DBP in several marine aquatic species in North America suggest that the substance is 1021 bioavailable in aquatic environments (Mackintosh et al., 2004). However, DBP can be considered 1022 1023 readily biodegradable in most aquatic environments, and the estimated BCF of 159.4 L/kg (U.S. EPA, 1024 2017) suggests that it is expected to have low bioaccumulation potential. EPA evaluated thirteen overall 1025 high quality data sources reporting the aquatic bioconcentration, aquatic bioaccumulation, aquatic food 1026 web magnification, and terrestrial bioconcentration of DBP (Table 7-1). The available data sources 1027 discussed below suggest that DBP has low bioaccumulation potential in aquatic and terrestrial 1028 organisms (Lee et al., 2019a; U.S. EPA, 2017; Teil et al., 2012), and no apparent biomagnification 1029 across trophic levels in the aquatic food web (Mackintosh et al., 2004). 1030 1031 Several overall high-quality data sources have reported the bioconcentration, bioaccumulation, and food 1032 web magnification of DBP in aquatic species. One of these data sources reported DBP BCF values of 1033 2.9 to 41.6 in sheepshead minnow, American oyster, and brown shrimp after a 24-hour exposure of DBP 1034 (100–500 ppb) (Wofford et al., 1981), suggesting low potential for bioconcentration in aquatic species. This finding agrees with the predicted BCF values of 159.4 to 525 L/kg and monitored BCF values of 1035 1036 0.78 to 7.48 L/kg in fish, respectively (U.S. EPA, 2017; Adeogun et al., 2015; Chemical Manufacturers 1037 Association, 1984). BCF values of 1,500–5,000 have been reported in glass shrimp in a 3-day DBP exposure experiment (Mayer Jr et al., 1973); however, the DBP was rapidly excreted with a 75 percent 1038 1039 loss of DBP during a 7-day depuration period. A monitoring study reported BAF values of 110 to 1247 1040 L/kg dw in crucian carp, skygager, bluegill, and bass samples collected from the Asan Lake in Korea 1041 (Lee et al., 2019a). The highest BAF value reported in Lee et al. (2019a) was 1,247 L/kg dw in crucian 1042 carp. This species is a benthic feeder that generally tends to contain higher levels of phthalate esters due 1043 to greater interaction with sediments. However, the available overall high-quality data sources 1044 containing aquatic biota-sediment accumulation factors (BSAF), reported BSAF values of 0.35 to 11.8 1045 glipid/goc for fish, and 130 glipid/goc for ovsters (Adeogun et al., 2015; Teil et al., 2012; Huang et al., 1046 2008; Mcfall et al., 1985). In addition, the available data sources reported aquatic trophic magnification 1047 factor (TMF) values of 0.70–0.81 (Kim et al., 2016; Mackintosh et al., 2004). Despite the differences in DBP biomonitoring values, DBP is expected to have low bioconcentration potential and low 1048 1049 biomagnification potential across trophic levels in the aquatic food web, but potentially result in higher 1050 uptake by benthic organisms. 1051

1052 There is very limited information on the bioconcentration and bioaccumulation of DBP in terrestrial 1053 environments. EPA extracted and evaluated nine high-quality data sources containing DBP terrestrial 1054 plant concentration factors (PCFs) and biota-soil accumulation factor (BSAF) information for plants and 1055 earthworms, respectively (Table 7-1). Based on DBP's expected strong affinity to organic matter and 1056 rapid biodegradation (on the order of weeks to months in soil), DBP is expected to have limited 1057 bioavailability in soils. This is supported by the reported low BSAF values of 0.242–0.460 in 1058 earthworms (Eisenia foetida) (Ji and Deng, 2016; Hu et al., 2005). Similarly, low PCF values have been reported in the range of 0.02–9.60 for rice, fruits, vegetables, wheat and maize, pond weed, and wetland 1059 1060 grasses. These findings suggest that DBP has a low uptake potential for most edible fruits, vegetables, grasses, and weeds from soil. Therefore, DBP is expected to have low bioaccumulation potential and 1061 1062 biomagnification potential in terrestrial organisms.

1063

1064 Overall, the available data suggest that DBP is expected to have low bioaccumulation potential and low 1065 biomagnification potential in aquatic and terrestrial organisms.

1067 Table 7-1. Summary of DBP's Bioaccumulation Information

Endpoint	Value(s)	Details	Reference (s)	Overall Quality Ranking
	159.4 L/kg (fish)	Estimated steady-state BCF; Arnot-Gobas method, fish upper trophic level.	<u>U.S. EPA (2017)</u>	High
	0.78–7.48 L/kg (fish)	Experimental monitoring sample collection in Nigeria. Tested organisms: Tilapia zillii, <i>Hepsetus odoe</i> , <i>Parachanna obscura</i> and <i>Chrysichthys nigrodigitatus</i> , <i>Mormyrus rume</i> , and a decapod crustacean (African river prawn, <i>Macrobrachium</i> <i>vollenhovenii</i>).	<u>Adeogun et al.</u> (2015)	High
Aquatic bioconcentration factor (BCF)	11.7 (minnow), 21.1–41.6 (oyster), 2.9–30.6 (shrimp)	Experimental laboratory exposure. Organisms Type: (Small Fish) Sheepshead minnow, <i>Cyprinodon</i> <i>variegatus</i> ; American oyster, <i>Crassostrea virginica</i> ; brown shrimp, <i>Penaeus aztecus</i>	<u>Wofford et al.</u> (1981)	High
	500–6,600 (aquatic invertebrates)	Experimental laboratory exposure; BCF of 3,500– 6,600 (Midge larvae, 1–7 days); 2,200–5000 (Water flea, 1–7 days); 1,700–6,500 (Scud, 1–7 days); 500–1,900 (Mayfly, 1–7 days); 1,000– 2,700 (Damselfly, 1–7 days); 1,500–5,000 (Glass shrimp, 1–3 days). 75% DBP loss during 7-day depuration.	<u>Mayer Jr et al.</u> (1973)	High
	525 (fish)	Predicted fish BCF calculated from actual K _{OW} determinations: $log BCF =$ $(0.542 \times log K_{OW}) +$ 0.124	<u>Chemical</u> <u>Manufacturers</u> <u>Association (1984)</u>	High
Aquatic bioaccumulation factor (BAF)	110–1,247 L/kg dw (fish)	Experimental, monitoring study lakes in Korea. Average concentration in fish: $3.3-37.4 \mu g/kg dw$. Log BAF: $2.0-3.1 L/kg dw$. Organisms Type: crucian carp, skygager, bluegill, and bass.	<u>Lee et al. (2019a)</u>	High

Endpoint	Value(s)	Details	Reference (s)	Overall Quality Ranking
	0.56–6.11 (fish)	Experimental monitoring sample collection in Nigeria. Tested organisms (see above for details).	Adeogun et al. (2015)	High
	130 (oyster)	Experimental monitoring Lake Pontchartrain in New Orleans, Louisiana. Calculated from concentration in oysters divided by concentration in sediment. Average: 570 ng/g ww in oyster, <i>Crassotrea</i> <i>virginica</i> .	<u>Mcfall et al. (1985)</u>	High
Aquatic biota- sediment accumulation factor (BSAF)	5.5 to 11.8 g _{lipid} /g _{OC} (fish)	Experimental monitoring sample collection from the Orge river in France. Roach: 5.5 ± 4.8 , Chub: 6.0 ± 2.3 , and Perch: 11.8 ± 12.6 ; BSAF = C _{biota} (ng/g)/C _{sediment} (ng/g)	<u>Teil et al. (2012)</u>	High
	0.35 to 1.35 g _{lipid} /g _{OC} (fish)	Experimental monitoring in 17 out of 21 principal rivers of Taiwan. BSAF = (phthalate in fish/lipid content in fish) / (phthalate in sediment/organic carbon in sediment) Organism type: <i>Oreochromis niloticus, Liza</i> <i>subviridis, Acanthopagrus</i> <i>schlegeli, Zacco platypus</i> and <i>Acrossocheilus paradoxus</i> .	<u>Huang et al. (2008)</u>	High
Aquatic trophic magnification factor (TMF)	0.70	95% confidence interval (lower and upper interval 0.40-1.23) of the reported TMF values in the False Creek food web species including 3 phytoplankton, 1 zooplankton, 10 invertebrates, and 10 fish.	<u>Kim et al. (2016)</u>	High
	0.70–0.81	Food-web magnification factor of 0.70 to 0.81 in 18 marine species in the False Creek food web.	Mackintosh et al. (2004)	High

Endpoint	Value(s)	Details	Reference (s)	Overall Quality Ranking
Terrestrial biota- soil accumulation factor (BSAF)	0.242–0.460	Earthworm from agricultural field in China; 0.23–30 (soil 1) and 0.18–0.23 (soil 2); BSAF = 0.460.	<u>Hu et al. (2005);</u> Ji and Deng (2016)	High
Plant Concentration Factor (PCF)	0.02–0.495 (rice)	Approx. 0.105–0.4 (root), 0.02–0.14 (stem), 0.1–0.495 (leaf), and 0.005–0.255 (grain)	<u>Cai et al. (2017)</u>	High
	0.16–0.19 (radish)	PCF Value: 0.19 (shoot), 0.16 (root)	<u>Cai et al. (2008)</u>	High
	1.38–9.60 (pondweed)	BCF Value: 4.43–8.04 L/kg; Study length: 10 days; root bioconcentration: 9.60 \pm 0.8 (control; lower conc. in found sediment) 1.75 \pm 0.2 (spiked; higher conc. found in sediment); stems and leaves bioconcentration: 7.40 \pm 0.5 (control; lower conc. in sed) 1.38 \pm 0.1 (spiked; higher conc. found in sediment)	<u>Chi and Gao (2015);</u> <u>Wang (2014)</u>	High
	0.33–1.03 (winter wheat and summer maize)	Winter wheat PCF: 0.89 and 0.42 (reclaimed water), 0.80 and 0.33 (mixed water), 0.91 and 0.43 (ground water); Summer maize PCF: 1.03 (reclaimed water), 0.94 (mixed water), 1.01 (ground water)	<u>Li et al. (2018)</u>	High
	0.26–4.78 (fruit and vegetables)	Mean PCF Value: Lettuce leaf: 0.26 ± 0.01 ; strawberry leaf: 0.34 ± 0.08 ; carrot leaf: 1.09 ± 0.21 ; lettuce root: 0.77 ± 0.09 ; strawberry root: 2.61 ± 0.42 ; carrot root 4.78 ± 0.59 ; purchased from the Certified Plant Growers in Temecula, CA; Study length: 28 days.	<u>Sun et al. (2015)</u>	High

Endpoint	Value(s)	Details	Reference (s)	Overall Quality Ranking
	2.11–9.32 (wetland grasses)	Root bioconcentration: 2.11– 9.32 Organisms: <i>P. australis</i> and <i>Typha orientalis</i> ; root systems collected. Study length: 17 days	Wang and Chi (2012)	High

1069 8 OVERALL FATE AND TRANSPORT OF DBP

1070 The inherent physical and chemical properties of DBP govern its environmental fate and transport. 1071 Based on DBP's aqueous solubility, slight tendency to volatilize, and strong tendency to adsorb to organic carbon, this chemical substance will be preferentially sorbed to sediments, soils, and suspended 1072 1073 solids in wastewater treatment processes. Soil, sediment, and sludge/biosolids are predicted to be the 1074 major receiving compartments for DBP as indicated by its physical, chemical, and fate properties and 1075 verified by monitoring studies. Surface water is predicted to be a minor pathway, and the main receiving 1076 compartment for phthalates discharged via wastewater treatment processes. However, phthalates in 1077 surface water will sorb strongly to suspended and benthic sediments. In areas where continuous releases 1078 of phthalates occur, higher levels of phthalates in surface water can be expected, trending downward 1079 distally from the point of release. This also holds true for DBP concentrations in both suspended and 1080 benthic sediments. While DBP undergoes relatively rapid aerobic biodegradation, it is persistent in 1081 anoxic or anaerobic environments (*i.e.*, sediment, landfills) and like other phthalates, it is expected to 1082 slowly hydrolyze under standard environmental conditions.

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When released directly to the atmosphere, DBP is expected to adsorb to particulate matter. It is not expected to undergo long-range transport facilitated by particulate matter due to the relatively rapid rates of both direct and indirect photolysis. Atmospheric concentrations of DBP may be elevated proximal to sites of releases. Off-gassing from landfills and volatilization from wastewater treatment processes are expected to be negligible in terms of ecological or human exposure in the environment due to DBP's low vapor pressure. DBP (not sorbed to suspended particles) released to air may undergo rapid photodegradation and it is not expected to be a candidate chemical for long range transport.

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Under indoor settings, DBP in the air is expected to partition to airborne particles and have an extended
lifetime as compared to airborne DBP in outdoor settings. The available information suggests that
DBP's indoor dust concentrations are associated with the presence of phthalate-containing articles and
proximity to the facilities producing them (Wang et al., 2013; Abb et al., 2009), as well as daily
consumer activities that might introduce DBP-containing products into indoor settings (Dodson et al.,
2017).

DBP has a predicted average environmental half-life of 14 days. DBP is expected to degrade rapidly in situations where aerobic conditions are predominant and be more persistent under anoxic or anaerobic conditions (*e.g.*, in some sediments, landfills, and soils). In anaerobic environments, such as deep

1102 landfill zones, hydrolysis is expected to be the most prevalent process for the degradation of DBP.

9 Weight of the Scientific Evidence Conclusions for Fate and Transport

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

- Given the consistent results from numerous high-quality studies, there is robust confidence that DBP:
 is not expected to undergo significant direct photolysis, but will undergo indirect photodegradation by reacting with hydroxyl radicals in the atmosphere with a half-life of 1.13 to 1.15 days (Section 4.3);
- will partition to organic carbon and particulate matter in air (Section 5.1);
 - will not hydrolyze under standard environmental conditions, but its hydrolysis rate increases with increased pH and temperature in deep-landfill environments (Sections 4.2 and 5.3.3);
 - will biodegrade in aerobic surface water, soil, and wastewater treatment processes (Sections 4.1 and 6.2);
- will not biodegrade under anoxic conditions and may have high persistence in anaerobic soils and sediments (Sections 4.1.2 and 4.1.3);
- will be removed with wastewater treatment and will sorb significantly to sludge, with a small fraction being present in WWTP effluent (Section 6.2);
- has low bioaccumulation potential (Section 1);
 - may be persistent in surface water and sediment proximal to continuous points of release (Section 5.2); and
- is expected to transform to MBP, butanol, and phthalic acid in the environment (Section 1).

1125 As a result of limited studies identified, there is moderate confidence that DBP:

- will be removed in conventional drinking water treatment systems both in the treatment process
 and via reduction by chlorination and chlorination byproducts in post-treatment storage and
 drinking water conveyance with a removal efficiency of 31 to 64.5 percent (Section 6.3).
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