



United States
Environmental Protection Agency

EPA Document# EPA-740-R-25-037

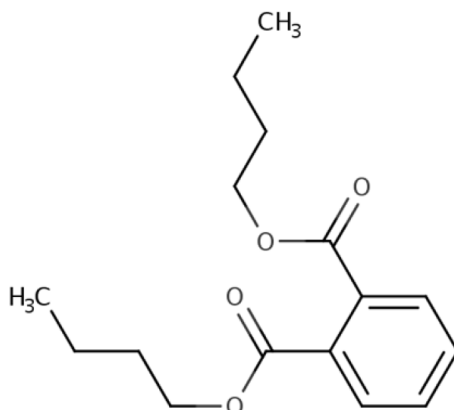
December 2025

Office of Chemical Safety and
Pollution Prevention

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

Technical Support Document for the Risk Evaluation

CASRN 84-74-2



December 2025

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KEY ABBREVIATIONS AND ACRONYMS

BAF	Bioaccumulation factor
BCF	Bioconcentration factor
BMF	Biomagnification factor
BSAF	Biota-sediment accumulation factor
CASRN	Chemical Abstract Service Registry Number
CDR	Chemical Data Reporting
cP	Centipoise
DBP	Dibutyl phthalate
DMR	Discharge Monitoring Reports
DOE	Department of Energy (U.S.)
DOC	Dissolved organic carbon
DRE	Destruction and removal efficiency
EC/HC	Environment Canada and Health Canada
EPA	Environmental Protection Agency (U.S.)
HLC	Henry's Law constant
K _{AW}	Air-water partition coefficient
K _{OA}	Octanol-air partition coefficient
K _{oc}	Organic carbon-water partition coefficient
K _{ow}	Octanol-water partition coefficient
MBP	Mono butyl phthalate
NCBI	National Center for Biotechnology Information
NIST	National Institute of Standards and Technology
NIOSH	National Institute for Occupational Safety and Health
·OH	Hydroxyl radical
PCF	Plant concentration factor
QSPR	Quantitative structure-property relationship
RSC	Royal Society of Chemistry
RSD	Relative standard deviation
STP	Sewage treatment plant
TRI	Toxics Release Inventory
TSCA	Toxic Substances Control Act
TSD	Technical support document
TMF	Trophic magnification factor
U.S.	United States
WHO	World Health Organization
WWTP	Wastewater treatment plant

SUMMARY

This technical support document (TSD) accompanies the *Risk Evaluation for Dibutyl Phthalate (DBP)* ([U.S. EPA, 2025c](#)). EPA (or the Agency) gathered and evaluated physical and chemical property data and information according to the process described in the *Systematic Review Protocol for Dibutyl Phthalate (DBP)* ([U.S. EPA, 2025d](#)). During the risk evaluation of DBP (CASRN 84-74-2) under the Toxic Substances Control Act (TSCA), EPA considered measured and estimated physical and chemical property data and information (see Table 2-1). Information on the full, extracted data set is available in the *Risk Evaluation for Dibutyl Phthalate (DBP) – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties* ([U.S. EPA, 2025b](#)).

DBP – Physical Chemistry

EPA evaluated the reasonably available information to characterize the physical and chemical properties of DBP, the key points are summarized below:

- DBP is a branched phthalate ester used as a plasticizer.
- Under standard environmental conditions, DBP is an oily liquid ([O'Neil, 2013](#)) with a melting point around $-35\text{ }^{\circ}\text{C}$ ([Rumble, 2018b](#)).
- DBP has a water solubility of 11.2 mg/L at 24°C ([Howard et al., 1985](#)) and an octanol-water partition coefficient ($\log K_{ow}$) of 4.5 ([NLM, 2024](#)).
- With a vapor pressure of 2.1×10^{-5} mmHg at $25\text{ }^{\circ}\text{C}$ ([U.S. EPA, 2019](#)) and a boiling point of $340\text{ }^{\circ}\text{C}$ ([Rumble, 2018b](#)), DBP has the potential to be volatile from dry, non-adsorbing surfaces.
- The selected Henry's Law constant (HLC) for DBP is 1.81×10^{-6} atm·m³/mol at $25\text{ }^{\circ}\text{C}$ ([NLM, 2024](#)).

DBP – Environmental Fate and Transport

EPA evaluated the reasonably available information to characterize the environmental fate and transport of DBP. Given consistent results from numerous high-quality studies, there is robust evidence that DBP:

- Is expected to degrade rapidly via direct and indirect photolysis and will rapidly degrade in the atmosphere ($t_{1/2} = 1.15$ days) (Section 4.3);
- Is not expected to hydrolyze under environmental conditions (Section 4.2);
- Is expected to have an environmental biodegradation half-life in aerobic environments on the order of days to weeks (Section 4.1);
- Is not expected to be subject to long-range transport;
- Is expected to transform in the environment via biotic and abiotic processes to form phthalate monoesters, then phthalic acid, and ultimately biodegrade to form CO₂ and/or CH₄ gas;
- Is expected to show strong affinity and sorption potential for organic carbon in soil and sediment (Section 3.2);
- When released to air, DBP will mostly partition to soil and water and the remaining DBP fraction in air will rapidly degrade in the atmosphere (Section 5.1);
- Is likely to be found and accumulate in indoor dust (Section 5.1.1); and
- Will be removed at rates between 68 to 98 percent in conventional wastewater treatment systems (Section 6.2).

As a result of limited studies identified to characterize the transformation, removal, persistence, and bioaccumulation potential of DBP, there is moderate evidence that DBP:

- Is not expected to biodegrade under anoxic conditions and can persist in anaerobic soils and sediments (Section 4.1).
- Is not bioaccumulative in fish that reside in the water column (Section 7).

- May bioaccumulate in benthic organisms exposed to sediment with elevated concentrations of DBP proximal to continual sources of release (Section 7).
- Is expected to be partially removed in conventional drinking water treatment systems via sorption to suspended organic matter and filtering media (Section 6.3).

1 INTRODUCTION

DBP is produced by the esterification of phthalic anhydride with isobutyl alcohol in the presence of an acid catalyst. It 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. Considered ubiquitous in various environmental media, DBP is commonly found 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](#)).

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 Sections 4 and 5. 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. Its occurrence in soils is attributable to DBP deposition from air and land application of biosolids.

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

2 APPROACH AND METHODOLOGY FOR PHYSICAL AND CHEMICAL PROPERTY ASSESSMENT

EPA completed a systematic review by conducting a literature search of available published articles through 2019 to identify the physical and chemical property values provided in this TSD (Table 2-1). After physical and chemical property data are extracted and evaluated, values for the endpoints are selected for use in the risk evaluation, as described in the *Systematic Review Protocol for Dibutyl Phthalate (DBP)* ([U.S. EPA, 2025d](#)). 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 partition coefficient (log K_{OA}) and the air:water partition coefficient (log K_{AW}) were not available; thus, EPI Suite™ ([U.S. EPA, 2017](#)) was used to estimate a value for each of these parameters.

2.1 Selected Physical and Chemical Property Values for DBP

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

Property	Selected Value(s)	Reference	Data Quality Rating
Molecular formula	C ₁₆ H ₂₂ O ₄	NLM (2024)	High
Molecular weight	278.35 g/mol	Haynes (2014b)	High
Physical form	Oily liquid	O'Neil (2013)	High
Melting point	−35 °C	Rumble (2018b)	High
Boiling point	340 °C	O'Neil (2013)	High
Density	1.0465 g/cm ³	Rumble (2018b)	High
Vapor pressure	2.01E−05 mmHg	U.S. EPA (2019)	High
Vapor density	9.58	NLM (2024)	High
Water solubility	11.2 mg/L	Howard et al. (1985)	High
Octanol:water partition coefficient (log K_{OW})	4.5	NLM (2024)	High
Octanol:air partition coefficient (log K_{OA})	8.63 (EPI Suite™)	U.S. EPA (2017)	High
Air:water partition coefficient (log K_{AW})	−4.131 (EPI Suite™)	U.S. EPA (2017)	High
Henry's Law constant	1.81E−06 atm·m ³ /mol at 25 °C	NLM (2024)	High
Flash point	157 °C	NLM (2024)	High
Autoflammability	402 °C	NLM (2024)	High
Viscosity	20.3 cP	NLM (2024)	High

2.2 Endpoint Assessments

2.2.1 Melting Point

Melting point is one of the inherent properties that informs the chemical's physical state (liquid, solid, or gas) at normal environmental conditions. The melting point is the temperature at which a chemical

substance coexists as its solid and liquid forms in equilibrium under atmospheric pressure (1 atmosphere) ([Rumble, 2021](#)). The physical state information helps assessors understand the environmental fate and transport, as well as the chemical substance's potential exposure routes, and bioavailability to both aquatic organisms and humans (occupational and general population). EPA extracted and evaluated eight high-quality data sources 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](#); [ECHA, 2012](#); [NIOSH, 2007](#); [Wang and Richert, 2007](#); [Park and Sheehan, 2000](#)). The mean of the reported melting point values within these sources is -35.57 °C. Seven of these sources reported a DBP melting point of -35 °C ([NLM, 2024](#); [NIST, 2022](#); [Elsevier, 2019](#); [U.S. EPA, 2019](#); [Rumble, 2018b](#); [DOE, 2016](#); [NIOSH, 2007](#)), while one source reported a DBP melting point of -40 °C ([Park and Sheehan, 2000](#)). EPA selected a melting point value of -35 °C ([Rumble, 2018b](#)) as a representative melting point value because this value is consistent with the average of the identified information from the overall 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)*; CASRN 84-74-2 (also referred to as the "final scope") ([U.S. EPA, 2020](#)).

2.2.2 Boiling Point

Boiling point is one of the inherent properties that informs the chemical's physical state (liquid or gas) at normal environmental conditions. The boiling point is the temperature at which a chemical substance coexists in equilibrium as vapor and liquid under atmospheric pressure ([Rumble, 2021](#)). The physical state helps inform multiple aspects of the risk evaluation. Boiling point informs the chemical's physical state, environmental fate and transport, as well as the chemical's potential bioavailability. EPA extracted and evaluated 11 high-quality data sources containing DBP boiling point information. These sources reported DBP boiling points ranging from 338 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 mean of reported boiling point values within these sources is 340 °C. EPA selected the boiling point value of 340 °C reported by O'Neil ([2013](#)) because this value is consistent with the mean of all boiling points measured under standard environmental conditions. The identified value is consistent with the value proposed in the final scope for DBP ([U.S. EPA, 2020](#)).

2.2.3 Density

Density is the chemical substance mass per unit volume (*i.e.*, g/mL) ([Rumble, 2021](#)) and is one of the inherent properties that informs assessors about the chemical's mobility in the environment under normal conditions. This information assists the assessors in understanding whether the environmental release of DBP is likely to sink or float in aquatic systems or ambient air. 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 document ([U.S. EPA, 2020](#)).

2.2.4 Vapor Pressure

Vapor pressure indicates the pressure exerted by a liquid or solid chemical in equilibrium with its vapor at a given temperature (25 °C for standard environmental conditions) (40 CFR 796.1950). This information informs the assessors if a chemical substance has the potential to volatilize and be released into the atmosphere, undergo long-range transport, and be available for specific exposure pathways. This information helps assessors understand the predicted concentration and environmental releases of a chemical substance to air. A chemical's potential to be present in air is expected to increase as vapor

pressure increases. 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°C (Elsevier, 2019). The remaining seven high-quality data sources reported DBP vapor pressure 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; ECHA, 2012; Lu, 2009; NIOSH, 2007; Howard et al., 1985; Hamilton, 1980). The mean and mode of these reported vapor pressure values are 4.38×10^{-5} and 2.01×10^{-5} mmHg, respectively, at 25°C . EPA selected the experimentally derived vapor pressure value of 2.01×10^{-5} mmHg (U.S. EPA, 2019) to best represent the mode vapor pressure of DBP obtained from the overall high-quality data sources under standard environmental conditions. The identified value is consistent with the value proposed in the final scope for DBP (U.S. EPA, 2020).

2.2.5 Vapor Density

The vapor density is the mass of a vapor per unit volume of the vapor relative to air (U.S. EPA). The relative vapor density is the ratio between the vapor density of a chemical substance and the vapor density of air (1.0). Relative vapor densities greater than 1 will indicate a higher tendency to sink while vapor densities lower than 1 indicate a higher tendency to float in ambient air. This information may inform the fate assessors about the predicted fate and transport of chemical substances (as vapors) when released to ambient air. 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 document (U.S. EPA, 2020).

2.2.6 Water Solubility

The water solubility indicates the maximum amount of DBP that will be dissolved in pure water. Water solubility informs many endpoints—not only within the realm of fate and transport of DBP in the environment, but also when modeling for industrial process, engineering, as well as human and ecological hazard, and exposure assessments. Chemical substances that are soluble in water can be expected to readily disperse through the environment. EPA extracted and evaluated 12 high-quality data sources containing DBP water solubility information. These sources reported water solubility values ranging from 1.5 to 14.6 mg/L (NLM, 2024; Elsevier, 2019; U.S. EPA, 2019; Rumble, 2018a; EC/HC, 2017; ECHA, 2012; NIOSH, 2007; Mueller and Klein, 1992; Defoe et al., 1990; Howard et al., 1985; SRC, 1983b). EPA excluded 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) calculated using the interquartile range (1.2 mg/L) rule for potential outliers (U.S. EPA, 2006). The rest of the available data sources reported DBP's water solubility values ranging from 8.7 to 11.4 mg/L. The mean of 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 data sources that best represents DBP's mean water solubility under standard environmental conditions. The identified value is consistent with the value proposed in the DBP final scope document (U.S. EPA, 2020).

2.2.7 Octanol:Air Partition Coefficient (Log K_{OA})

The octanol-air partition coefficient (K_{OA}) provides information on how the chemical, under equilibrium, will partition between octanol and air (U.S. EPA, 2025). This informs fate assessors on how the chemical is likely to partition in the environment between air and aerosol particles, air and foliage, and air and soil. This information also helps to understand how chemical substances are expected to behave in the atmosphere and in human respiratory tissues. Chemical substances with high K_{OA} are more likely to sorb onto solid surfaces as well as to bioaccumulate in the human respiratory tissue. No

K_{OA} data for DBP were identified in the initial data review for the DBP final scope ([U.S. EPA, 2020](#)). After the final scope was publicly released, EPA extracted and evaluated DBP octanol-air partitioning (K_{OA}) data from a single medium-quality source. This source reported a predicted log K_{OA} value of 8.45 obtained from a quantitative structure-property relationship (QSPR) model ([Lu, 2009](#)). The QSPR-derived estimate of 8.45 reasonably aligns with DBP's log K_{OA} value of 8.63 estimated using EPI Suite™ ([U.S. EPA, 2017](#)), which is considered a highly reliable model. As such, EPA has selected the EPI Suite™-derived value of 8.63 as the representative log K_{OA} value for use in risk assessment ([U.S. EPA, 2017](#)).

2.2.8 Octanol:Water Partition Coefficient (Log K_{ow})

The octanol-water partition coefficient (K_{ow}) provides information on how the chemical, under equilibrium, will partition between octanol (which represents the lipids or fats in biota) and water ([U.S. EPA, 2025](#)). This informs on how the chemical is likely to partition in the environment as well as for the estimation of other properties including water solubility, bioconcentration, soil adsorption, and aquatic toxicity. EPA extracted and evaluated 10 high-quality data sources containing DBP log K_{ow} information. These sources included six new additional sources not available for the DBP final scope ([U.S. EPA, 2020](#)). EPA excluded a reported K_{ow} of 3.74 from two data sources ([Howard et al., 1985](#); [SRC, 1984](#)) as it was determined to be a potential outlier. This value is higher or lower than the upper (4.41) and lower (4.65) bounds calculated using the interquartile range (0.24) rule for potential outliers ([U.S. EPA, 2006](#)). With the exclusion of potential outliers, these sources reported log K_{ow} values ranging from 4.25 to 4.79 ([NLM, 2024](#); [Elsevier, 2019](#); [U.S. EPA, 2019](#); [EC/HC, 2017](#); [Ishak et al., 2016](#); [ECHA, 2012](#); [Verbruggen et al., 1999](#); [Mueller and Klein, 1992](#); [Howard et al., 1985](#); [SRC, 1984](#)). The mean of the reported log K_{ow} values (excluding outliers) is 4.5. EPA selected an experimental log K_{ow} value of 4.5 ([NLM, 2024](#)) as this is consistent with the mean value obtained from the overall high-quality data sources under standard environmental conditions. The identified value replaces the value proposed in the *Final Scope for the Risk Evaluation of Dibutyl Phthalate (DBP)*; CASRN 84-74-2 ([U.S. EPA, 2020](#)).

2.2.9 Henry's Law Constant

The Henry's Law constant (HLC) of a chemical substance is the ratio of the substance's vapor pressure and water solubility ([U.S. EPA, 2025](#)). The HLC (atm·m³/mol) provides an indication of a chemical's volatility from water and an indication of potential environmental partitioning, potential removal in sewage treatment plants during air stripping, and possible routes of environmental exposure. EPA extracted and evaluated four high-quality data sources containing DBP HLC information. These sources reported DBP HLC values ranged from 8.83×10⁻⁷ to 1.81×10⁻⁶ atm·m³/mol ([NLM, 2024](#); [Elsevier, 2019](#); [U.S. EPA, 2019](#); [Cousins and Mackay, 2000](#)). The mean of the reported HLC values is 1.5×10⁻⁶ atm·m³/mol. EPA selected the HLC value of 1.81×10⁻⁶ atm·m³/mol ([NLM, 2024](#)) as the value obtained from the overall high-quality data sources that best represents DBP's mean HLC under standard environmental conditions. The identified value is consistent with the value proposed in the final scope for DBP ([U.S. EPA, 2020](#)).

2.2.10 Flash Point

The flash point is the lowest temperature at which a substance produces sufficient vapor to form an ignitable mixture with air ([U.S. EPA, 2025](#)). This helps fate assessors understand the fire hazards of the chemical. EPA extracted and evaluated five high-quality data sources containing DBP flash point information. These sources reported a DBP flash point of 157 to 171 °C ([NLM, 2024](#); [Elsevier, 2019](#); [Rumble, 2018c](#); [O'Neil, 2013](#); [NIOSH, 2007](#)). The mean of the reported flash point values is 162 °C. EPA 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

conditions. The identified value is consistent with the value proposed in the DBP final scope document ([U.S. EPA, 2020](#)).

2.2.11 Autoflammability

Autoflammability is the lowest temperature a chemical substance will spontaneously ignite without a spark or flame ([Rumble, 2021](#)). Like the flash point, this helps fate assessors to understand the fire hazards of a chemical. No autoflammability data for DBP were identified in the initial data review for the scope of the risk evaluation ([U.S. EPA, 2020](#)). After the final scope was publicly released, two high- and two medium-quality data sources were identified in the systematic review process. The autoflammability values ranged from 402 to 403 °C ([NLM, 2024](#); [NCBI, 2020](#); [Rumble, 2018c](#); [NIOSH, 1976](#)). The mean of the reported autoflammability values is 402 °C. EPA selected an autoflammability value of 402 °C for DBP ([NLM, 2024](#)) as the value that best represents the mean flashpoint value.

2.2.12 Viscosity

Viscosity is the expected flow resistance of a chemical substance due to molecular friction within a fluid ([U.S. EPA, 2025](#)). This helps fate assessors to understand the transport of a chemical substance directly released to the environment. EPA extracted and evaluated three high-quality data sources containing DBP viscosity information. These sources reported viscosity values ranging from 16.63 to 20.3 cP (centipoise) at 20 to 25 °C ([NLM, 2024](#); [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 environmental conditions for this risk evaluation. The identified value is consistent with the value proposed in the *Final Scope for the Risk Evaluation of Dibutyl Phthalate (DBP); CASRN 84-74-2* ([U.S. EPA, 2020](#)).

2.3 Strengths, Limitations, Assumptions, and Key Sources of Uncertainty for the Physical and Chemical Property Assessment

The representative physical and chemical property values were selected based on professional judgement and the weight of the scientific evidence, including the overall data quality ranking of the associated references. These physical and chemical property values were then used to inform chemical-specific decisions and model inputs across other disciplines. High- and medium-quality data are preferred when selecting physical and chemical properties. In some instances where no data were available, models such as EPI Suite™ were used to estimate the value for the endpoint (*i.e.*, octanol:air partitioning coefficient, K_{OA}) and cross-checked with reported data from systematic review. The number and overall quality of the available data sources results in different confidence strength levels for the corresponding selected physical and chemical property values ([U.S. EPA, 2021](#)).

3 APPROACH AND METHODOLOGY FOR FATE AND 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.

Information on the full extracted data set is available in the file *Risk Evaluation for Dibutyl Phthalate (DBP) – Systematic Review Supplemental File: Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport* ([U.S. EPA, 2025a](#)). When no measured data were available from high- or medium-quality data sources, fate values were obtained from EPI Suite™ ([U.S. EPA, 2017](#)), a predictive tool for physical and chemical properties and environmental fate estimation. Information regarding the model inputs is available in Section 3.2.1.

Table 3-1 provides a summary of the selected data that EPA considered while assessing the environmental fate of DBP and were updated after the public release of the *Final Scope of the Risk Evaluation for Dibutyl Phthalate (DBP); CASRN 84-74-2* ([U.S. EPA, 2020](#)), along with additional information identified through the systematic review process.

Table 3-1. Summary of Environmental Fate Information for DBP

Property or Endpoint	Value(s)	Reference(s)
Hydrolysis	$t_{1/2}$ = approximately 22 years at pH 7 and 25 °C	ATSDR (1999)
	K_H = $1.0 \pm 0.05E-2 \text{ M}^{-1} \text{ sec}^{-1}$ at pH 10–12 and 30 °C	Wolfe et al. (1980)
	$t_{1/2}$ = 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.S. EPA (2017)
Indirect photodegradation	$t_{1/2}$ = 1.15 days (estimated based on a 12-hour day with $1.5E6 \cdot \text{OH}/\text{cm}^3$ and $\cdot \text{OH}$ rate constant of $9.28E-12 \cdot \text{OH}/\text{cm}^3$ and $\cdot \text{OH} \text{ cm}^3/\text{molecule-sec}$)	Peterson and Staples (2003)
	$t_{1/2}$ = 1.13 days (based on a 12-hour day with $1.5E6 \cdot \text{OH}/\text{cm}^3$ and $\cdot \text{OH}$ rate constant of $9.47E-12 \cdot \text{OH}/\text{cm}^3$ and $\cdot \text{OH} \text{ cm}^3/\text{molecule-sec}$)	Lei et al. (2018)
Organic carbon:water partition coefficient (log K_{OC})	3.69 (average of 7 values ranging between 3.14–3.94)	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)	NITE (2019)
	100% in 7 days based on loss of test substance in a synthetic medium containing 5 mg yeast extract	Tabak et al. (1981)
	68.3 to >99% (average: 89.8%) primary biodegradation after 28 days using inoculum	SRC (1983a)

Property or Endpoint	Value(s)	Reference(s)
	prepared with soil, domestic, influent sewage microorganisms with a 2-week acclimation period	
Aerobic ultimate biodegradation in water	57.4% by theoretical CO ₂ (ThCO ₂) evolution after 28 days	SRC (1983a)
Aerobic biodegradation in sediment	84.6 ± 2.1% (mean ± SD) after 14 days at 22 °C based on primary biodegradation	Johnson et al. (1984)
	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	Yuan et al. (2002)
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	Yuan et al. (2002)
Aerobic biodegradation in soil	88.1–97.2% after 200 days in Chalmers slit loam, Plainfield sand, and Fincastle silt loam soils.	Inman et al. (1984)
Anaerobic biodegradation in WWTP sludge	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	Union Carbide (1974)
	46%, 59%, and 19% after 8 weeks by % theoretical gas production in RAMM, ASTM, and supplemental medium from Holt, MI, respectively	
	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) wastewater treatment plant (WWTP)	Tran et al. (2014)
	Removal efficiency (approximate, based on figure): primary sedimentation: ca. –50%; chemical enhanced primary treatment: ca. –100%; activated sludge: ca. 75%; sand filtration: ca. 95%; chlorination disinfection: ca. 20%	Wu et al. (2017)

Property or Endpoint	Value(s)	Reference(s)
Aquatic bioconcentration factor (BCF)	2.9 ± 0.1 and 30.6 ± 3.4 in brown shrimp (<i>Penaeus aztecus</i>) at 100 and 500 ppb, respectively	Wofford et al. (1981)
	11.7 in sheepshead minnow (<i>Cyprinodon variegatus</i>) at 100 ppb	
	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.S. EPA (2017)
Aquatic biota-sediment accumulation factor (BSAF)	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)
	BSAF: 0.2–2 (approximate range from figure) in <i>Oreochromis niloticus niloticus</i> , <i>Liza subviridis</i> , <i>Acanthopagrus schlegelii</i> , <i>Zacco platypus</i> and <i>Acrossocheilus paradoxus</i>	Huang et al. (2008)
	BSAF: 5.5 ± 4.8, 6.0 ± 2.3, and 11.8 ± 12.6 in roach, chub, and perch, respectively	Teil et al. (2012)
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	Hu et al. (2005) and Ji and Deng (2016)
Plant concentration factor (PCF)	0.26 to 4.78 (fruit and vegetables)	Sun et al. (2015)

3.1 Tier I Analysis

To understand and predict the behaviors and effects of DBP in the environment, a Tier I 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 step in identifying the major and minor compartments for DBP is to consider partitioning values (Table 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 environmental monitoring studies ([NLM, 2024](#); [EC/HC, 2017](#)).

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. Because organic carbon is present in soil, biosolids, and sediment, they are all considered major compartments for DBP. This is consistent with monitoring data where higher concentrations of DBP were detected in sediment samples (20–698 ng/g) compared to water samples (114–2,116 ng/L) collected from the Mersey Estuary in the United Kingdom ([NLM, 2024](#)).

3.1.2 Air

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 has been detected in the vapor (gaseous)

phase and particulate phase in the atmospheric samples collected along the U.S.-Canadian border of the Niagara River ([EC/HC, 1994](#)). The study reported mean DBP concentrations of 1.9 ± 1.3 ng/m³ in the vapor phase and 4.0 ± 2.2 ng/m³ in the particulate phase in air samples collected near Lake Ontario in the Niagara River ([EC/HC, 1994](#)). The same study reported mean DBP concentrations of 4.5 ± 3.5 ng/m³ in the vapor phase and 6.2 ± 2.6 ng/m³ in the particulate phase in air samples collected near the American side of the Niagara River.

In another monitoring study from Paris, France, higher concentrations of DBP were detected in the vapor phase (2.9 to 59.3 ng/m³) compared to the particulate phase (0.6 to 4.6 ng/m³) ([NLM, 2024](#)). The octanol:air partition coefficient (K_{OA}) indicates that DBP will favor the organic carbon present in airborne particles. Based on its physical-chemical properties and short half-life in the atmosphere ($t_{1/2} = 1.15$ days ([U.S. EPA, 2017](#))), DBP in the vapor phase is assumed to not be persistent in the air. The AEROWIN™ module in EPI Suite™ estimates that a fraction of DBP may be sorbed to airborne particulates and these particulates may be resistant to atmospheric oxidation. DBP has been detected in both indoor and outdoor air and settled house dust in the United States, Europe, Canada, and China ([NLM, 2024](#); [EC/HC, 2017](#); [Kubwabo et al., 2013](#); [Wang et al., 2013](#)).

3.1.3 Water

A log K_{AW} value of -4.131 indicates that DBP will favor water over air. With a water solubility of 11.2 mg/L at 25 °C, DBP is expected to be slightly soluble in water ([Howard et al., 1985](#)). In water, DBP is likely to partition to suspended organic material present in the water column based on DBP's water solubility of 11.2 mg/L ([Howard et al., 1985](#)) and organic carbon:water partition coefficient of 3.69 (Table 3-1). A monitoring study showed that total seawater DBP concentrations in the False Creek Harbor, a shallow marine inlet in Vancouver, ranged from 50 to 244 ng/L and the dissolved fraction concentrations ranged from 34 to 165 ng/L compared to the suspended particulate fraction concentration that ranged from 9,320 to 63,900 ng/g dry weight (dw) ([Mackintosh et al., 2006](#)). Although DBP has low water solubility, surface water will be a major compartment for DBP because it is detected in the ng/L range.

3.2 Tier II Analysis

A Tier II analysis involves reviewing environmental release information for DBP to determine if further assessment of specific media is needed. The Toxics Release Inventory (TRI) reported the total on-site releases for DBP in 2022 to be 130,800 pounds (lb) with 49,600 lb released to air, 81,200 lb released to land, and none released to water. For the risk evaluation, EPA used TRI data from reporting years 2017 to 2022 to provide a basis for estimating releases ([U.S. EPA, 2022d](#)). There is a decreasing general trend from 2017 to 2022 for total releases of DBP reported to TRI, though the rate of the decrease has reduced in recent years. EPA did not consider data from the 2023 or 2024 TRI because the data were not finalized at the time of this draft evaluation, though the preliminary data show a trend that is generally consistent with previous years. According to production data from the Chemical Data Reporting (CDR) 2020 reporting period, between 1 million and 10 million lb of DBP were produced annually from 2016 to 2019 for use in commercial products, chemical substances or mixtures sold to consumers, or at industrial sites. Review of preliminary 2024 CDR data shows that total production volume for 2023 is similar to the previously reported range. Environmental release data from the Discharge Monitoring Reports (DMRs) reported total annual releases for DBP from watershed discharge as 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.

DBP is used mainly as a plasticizer in polyvinyl emulsions and can be used in adhesives, paints and coatings, building materials, printing inks, fabric and textiles, children's toys, and plastic and rubber

materials ([EC/HC, 2017, 1994](#)). Because DBP is not chemically bound to the polymer matrix and can migrate from the surface of polymer products ([EC/HC, 2017](#)), it can easily be released to the environment from polymer-based products during their use and disposal. Additionally, DBP can be released to the environment from the disposal of wastewater as well as liquid and solid wastes. After undergoing wastewater treatment processes, effluent is released to receiving waters and biosolids (treated sludge) may be landfilled, land-applied, or incinerated. The existence and use of these processes indicate that media-specific evaluations are necessary (Table 3-2). Releases from landfills and incinerators will occur from the disposal of liquid and solid wastes and warrants media-specific evaluations.

Table 3-2. Summary of Key Environmental Pathways and Media-Specific Evaluations

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 solids to landfills	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

3.2.1 Fugacity Modeling

The approach described by Mackay et al. ([1996](#)) using the Level III Fugacity model in EPI Suite™ V4.11 (LEV3EPI™) was used for this Tier II analysis. LEV3EPI™ is described as a steady-state, non-equilibrium 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).

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

Media	Half-Life (days)	References
Air	1.15	Lei et al. (2018) ; SRC (1983a)
Water	10	
Soil	20	
Sediment	90	

The following input parameters, taken from Table 2-1 and discussed in detail in Section 2.2, were used in LEV3EPI™:

- Melting point = -35.00 °C
- Vapor pressure = 2.01×10^{-5} mm Hg
- Water solubility = 11.2 mg/L
- Log K_{ow} = 4.5
- SMILES: O=C(OCCCC)c(c(ccc1)C(=O)OCCCC)c1 (representative structure)

Based on DBP's reported environmental half-lives, partitioning characteristics, and the results of LEV3EPI™, 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; and 58 percent of releases to air will end up in soil with another 6 percent in water. The LEV3EPI™ results were consistent with environmental monitoring data. DBP's partitioning behavior are further discussed in the media-specific assessment.

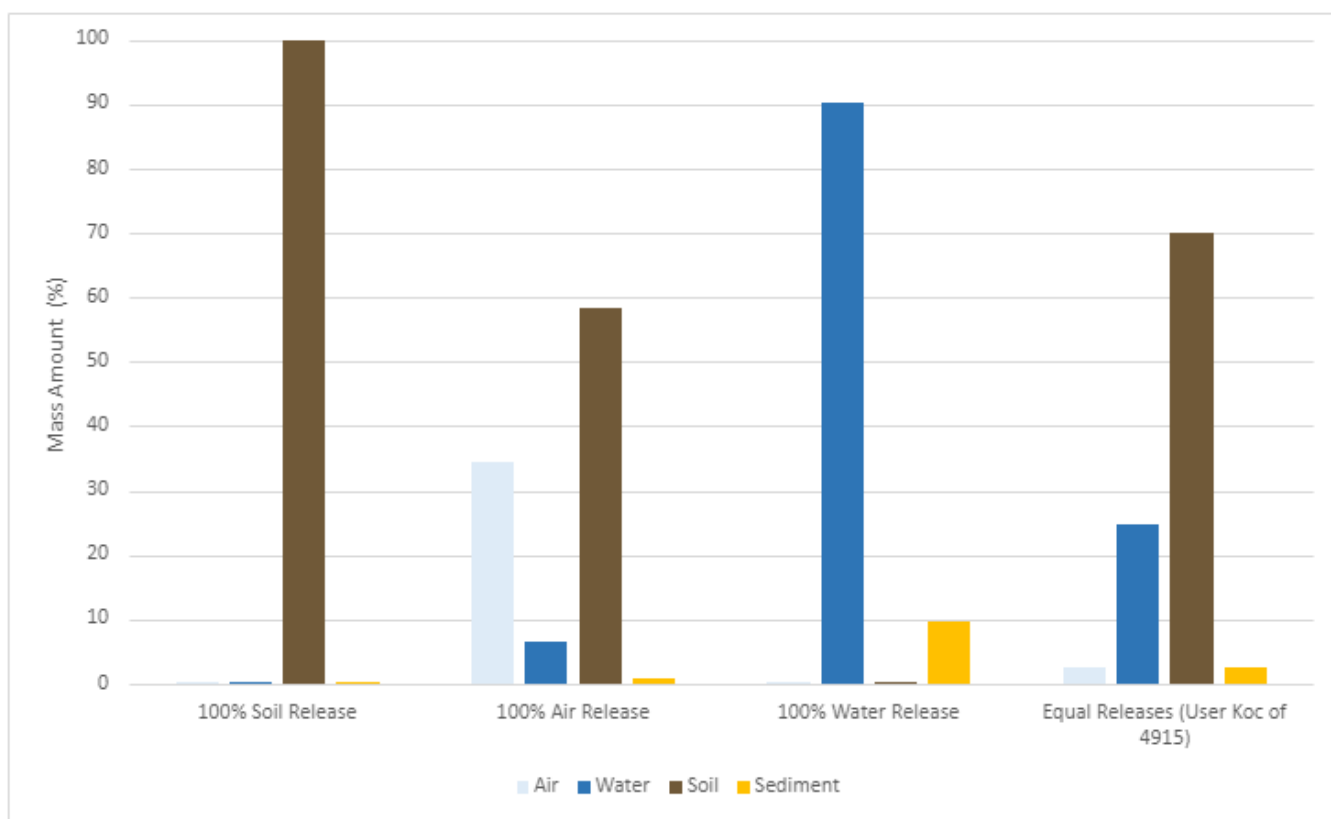


Figure 3-1. EPI Suite™ Level III Fugacity Modeling for DBP

4 TRANSFORMATION PROCESSES

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

4.1 Biodegradation

DBP is expected to be readily biodegradable in most aquatic and terrestrial environments. EPA extracted and evaluated 85 biodegradation studies during systematic review. Twenty-six of these studies were extracted and evaluated as overall high-quality data sources (Table 4-1). For the purposes of the following biodegradation analysis, and due to the large number of available high-quality data sources, EPA focused on studies that received 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 (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 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 rate of 100 percent over 100 hours using an inoculum that was acclimated to DBP over the course of 150 days using a DBP concentration of 100 mg/L ([Jianlong, 2004](#)). A study using a combination inoculum of soil, activated sludge, and raw sewage reported a primary biodegradation rate ranging from 68.3 to greater than 99 percent over 28 days ([SRC, 1983a](#)). Three of the eight identified studies used natural surface water inoculums. Studies reported primary biodegradation rates of 100 percent over 2 days in river water ([Cripe et al., 1987](#)), 100 percent over 14 days in river water ([Fujita et al., 2005](#)), and 100 percent over 14 days in pond water ([Fujita et al., 2005](#)); another reported a half-life of 1.7 to 13 days estuarine and freshwater sites ([Walker et al., 1984](#)).

Two studies evaluating the ultimate biodegradation of DBP in water were also extracted. One of the studies reported rates of 50 to 70 percent, 40 to 60 percent, and 20 to 50 percent when using inoculums of activated sludge, river water, and pond water, respectively ([Fujita et al., 2005](#)). The other reported rates were 47.4 to 74.9 percent, with half-lives of 9.6 to 20.9 days when using a mixture of soil, activated sludge, and raw sewage as the inoculum ([SRC, 1983a](#)). While the biodegradation rate of DBP in water will depend on the microbial community, most of the data on primary and ultimate biodegradation rates suggest an aerobic half-life of less than approximately 30 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.

4.1.2 Biodegradation in Sediment

EPA extracted 23 high-quality studies evaluating biodegradation of DBP in sediment. Out of the 23 studies, the Agency focused on the 9 studies that did not use sediments amended with external inoculums to best represent DBP's biodegradation under natural environmental conditions. One of these

studies reported 70.1 to 84.6 percent biodegradation of DBP in freshwater lake sediment under aerobic conditions ([Johnson et al., 1984](#)). That same study also evaluated the effects of temperature and DBP initial concentration on DBP's aerobic degradation rates. The results showed 70.1 to 72.6 percent 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 °C in 7 days. However, the study reported 16 to 56 percent loss of DBP at 5 to 12 °C. These findings showed that temperature could have a significant effect on biodegradation. Three of the selected data sources reported DBP's biodegradation half-lives of 2.7, 2.9, and 46 days in marine sediments under 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 also reported higher than expected biodegradation half-lives for other phthalates as well ([Kickham et al., 2012](#)). In general, DBP is expected to have a sediment biodegradation half-life of 2.7 to 2.9 days under normal aerobic environmental conditions, but extended half-lives might be possible.

In contrast to aerobic conditions, phthalate esters are expected to have extended half-lives in sediment under anaerobic conditions. Five of the nine extracted studies reported DBP biodegradation information in sediment under anaerobic conditions ([Li et al., 2015](#); [Lertsirisopon et al., 2006](#); [Chang et al., 2005](#); [Kao et al., 2005](#); [Yuan et al., 2002](#)). Chang et. al. (2005) reported 100 percent biodegradation in 28 days at pH 7 and 30 °C in sediments with a nutrient content commonly used to support microbial growth. Kao et. al. (2005) reported 24 percent biodegradation in 30 days at pH 7 and 30 °C in sediment samples with water (not amended with nutrients for microbial growth). Additionally, half-lives in anaerobic sediments have been reported to be 1.2 to 1.6 days in pond sediment ([Lertsirisopon et al., 2006](#)), 3.6 days in submerged marine sediment ([Li et al., 2015](#)), and 5.1 to 12.7 days in river sediment ([Yuan et al., 2002](#)). Overall, the available data show that there is variability in the biodegradation rates and that rates will depend on environmental conditions such as temperature and redox conditions. However, most of the studies evaluated suggest that at ambient temperatures DBP will have a half-life of less than 1 year in both aerobic and anaerobic sediments. 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.

4.1.3 Biodegradation in Soil

EPA extracted eight high-quality studies evaluating biodegradation of DBP under aerobic and anaerobic 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 soils of non-specified types from Broome County, New York ([Russell et al., 1985](#)); 100 percent over 15 days ([Shanker et al., 1985](#)) in an alluvial garden soil; and 66 percent over 30 days in soil taken from a 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., 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 al., 2011](#)); and 17.2 days in loam from a farm ([Zhao et al., 2016](#)). Anaerobic biodegradation rates have been reported to be 97.8 percent over 200 days in a silt loam ([Inman et al., 1984](#)) and 66 percent over 30 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 such as temperature, redox conditions, and pre-exposure of the microbial communities to DBP. However, the study with the slowest biodegradation rate in this evaluation suggests a half-life in soil of approximately 65 days ([Inman et al., 1984](#)). Therefore, for the purposes of this risk evaluation, it is assumed that DBP will have a half-life in soils on the order of weeks to months.

Table 4-1. Summary of DBP Biodegradation Information

Environmental Conditions	Degradation Value	Half-Life (days)	Reference	Overall Data Quality Ranking
Aerobic primary biodegradation in water	100% / 2 days	N/A	Cripe et al. (1987)	High
	>80% / 40 days	N/A	Desai et al. (1990)	High
	100% / 14 days	N/A	Fujita et al. (2005)	High
	100% in acclimated and 0% in unacclimated activated sludge / 100 hours	N/A	Jianlong (2004)	High
	68.3–99% / 28 days	N/A	SRC (1983a)	High
	100% / 7 days	N/A	Tabak et al. (1981)	High
	N/A	1.7–13 days	Walker et al. (1984)	High
	N/A	45.3–47.5 hours	Wang et al. (1997b)	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	Fujita et al. (2005)	High
	47.7–74.9% / 28 days	9.6–20.9 days	SRC (1983a)	High
Aerobic biodegradation in sediment	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
	16% at 5 °C, 56% at 12 °C, 73% at 22 °C, 86% at 28 °C / 7 days	N/A	Johnson et al. (1984)	High
	N/A	46 days in marine inlet sediment	Kickham et al. (2012)	High
	N/A	2.7 days in surface marine sediment	Li et al. (2015)	High
	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
Anaerobic biodegradation in sediment	100% / 28 days	9.4 days	Chang et al. (2005)	High
	24% / 30 days in river sediment	N/A	Kao et al. (2005)	High
	N/A	3.6 days in non-surface layer marine sediment	Li et al. (2015)	High
	N/A	5.1–12.7 days	Yuan et al. (2002)	High

Environmental Conditions	Degradation Value	Half-Life (days)	Reference	Overall Data Quality Ranking
	N/A	1.2–1.6 days in pond sediment	Lertsirisopon et al. (2006)	High
Aerobic biodegradation in soil	N/A	0.338–1.2 days	Cheng et al. (2018)	High
	88–98.6% / 200 days (CO ₂ evolution)	N/A	Inman et al. (1984)	High
	100% / 72 hours	N/A	Russell et al. (1985)	High
	100% / 15 days	N/A	Shanker et al. (1985)	High
	66%/30 days	N/A	Wang et al. (1997a)	High
	N/A	7.8–8.3 days	Xu et al. (2008)	High
	N/A	1.6 days	Yuan et al. (2011)	High
	N/A	17.2 days	Zhao et al. (2016)	High
Anaerobic biodegradation in soil	97.8% / 200 days (CO ₂ evolution)	N/A	Inman et al. (1984)	High
	66% / 30 days	N/A	Shanker et al. (1985)	High

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 half-life was reported to be approximately 22 years ([ATSDR, 1999](#)). Hydrolysis under acidic and alkaline conditions is expected to occur with alkaline hydrolysis being more rapid. Alkaline hydrolysis will yield phthalic acid with the monoester as an intermediate ([Zhang et al., 2019](#); [Huang et al., 2013a](#); [Wolfe et al., 1980](#)). Zhang et al. (2019) evaluated the hydrolysis of DBP in aqueous alkaline solutions (pH 10) at 30 °C. The study reported hydrolysis to be rapid under the tested conditions, reporting a hydrolysis half-life of 45.4 hours. Temperature has also been shown to impact hydrolysis rates with hydrolysis rates increasing with an increase in temperature. The hydrolysis half-life for DBP was reported to be 280.2 hours in neutral solution at a temperature of 80 °C. Wolfe et al. (1980) evaluated the hydrolysis of DBP in aqueous alkaline solutions at 30 °C. The study reported a hydrolysis rate constant of $1.0 \pm 0.05 \times 10^{-2} \text{ M}^{-1} \text{ sec}^{-1}$, which corresponds to half-lives of 22 years at pH 7 and 8 days at pH 10. In addition, EPI Suite™ estimated the hydrolysis half-lives of DBP to be 3.43 years at pH 7 and 25 °C, and 125 days at pH 8 and 25 °C ([U.S. EPA, 2017](#)), indicating that hydrolysis of DBP is more likely under more caustic conditions and supporting DBP's resistance to hydrolysis under standard environmental conditions.

When compared to other degradation pathways, hydrolysis it is not expected to be a significant source of degradation under typical environmental conditions. However, the higher temperatures, variations from typical environmental pH, and chemical catalysts present in the deeper anoxic zones of landfills might be favorable to the degradation of DBP via hydrolysis ([Huang et al., 2013a](#)). This is discussed further in Section 5.3.3.

4.3 Photolysis

DBP contains chromophores that absorb light at greater than 290 nm wavelength ([NLM, 2013](#)); therefore, direct photodegradation is a relevant but minor degradation pathway for DBP released to air. 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 $\cdot\text{OH}$ rate constant of $9.47 \times 10^{-12} \text{ cm}^3/\text{molecule-second}$ and a 12-hour day with $1.5 \times 10^6 \text{ OH/cm}^3$) ([Lei et al., 2018](#)). Similarly, Peterson and Staples ([2003](#)) reported a calculated DBP photodegradation half-life of 1.15 days (≈ 27.6 hours) (calculated from a $\cdot\text{OH}$ rate constant of $9.28 \times 10^{-12} \text{ cm}^3/\text{molecule-second}$ and $1.5 \times 10^6 \text{ OH/cm}^3$). Indirect photodegradation of DBP will yield MBP, phthalic acid, di-butyl 4-hydroxyphthalate (m-OH-DBP), and di-butyl 4-nitrophthalate (m-NO₂-DBP) ([Lei et al., 2018](#)).

DBP photodegradation in water is expected to be 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 DBP aquatic direct photodegradation half-lives of 50, 66, 360, 94 and 57 days at pH 5, 6, 7, 8 and 9, respectively, when exposed to natural sunlight in artificial river water at 0.4 to 27.4 °C (average temperature of 10.8 °C). Peterson and Staples ([2003](#)) also reported a half-life of 3 hours for aqueous photolysis of DBP in natural sunlight when DBP was present in a surface microlayer on the water at mg/L concentrations. The rate was noted to be stimulated by titanium dioxide and hydrogen peroxide. These findings suggest DBP will be susceptible to photochemical decay in air but that photolysis is not expected to be a significant degradation process in surface water.

5 MEDIA ASSESSMENTS

DBP has been reported to be present in the atmosphere, aquatic environments, and terrestrial 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 properties, DBP is likely to partition to house dust and airborne particles in the indoor environment and is expected to have a longer half-life in indoor air as compared to outdoor air. DBP present in surface 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 groundwater but is likely to be immobile in both media types. In soils, DBP is expected to be deposited via air deposition and land application of biosolids. Additionally, DBP in soils is expected to have a half-life on the order of weeks to months as well as to have low bioaccumulation potential and biomagnification potential in terrestrial organisms. DBP will be released to groundwater via infiltration from wastewater effluent and landfill leachates but is not likely to be persistent in groundwater and/or subsurface environments unless anoxic conditions exist.

5.1 Air and Atmosphere

DBP is a liquid at environmental temperatures with a melting point of $-35\text{ }^{\circ}\text{C}$ ([Rumble, 2018b](#); [Haynes, 2014a](#)) and a vapor pressure of 2.01×10^{-5} mmHg at $25\text{ }^{\circ}\text{C}$ ([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](#))), DBP is not expected to be persistent in air. The AEROWIN™ module in EPI Suite™ estimated a log K_{OA} of 8.63, which suggests that a fraction of DBP may be sorbed to airborne particles and that these particulates may be more resistant to atmospheric oxidation. Thus, DBP has the potential to undergo both dry deposition and wet deposition into soils and surface water ([Zeng et al., 2010](#); [Peters et al., 2008](#); [Xie et 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 et al., 2005](#)). A net deposition of DBP from ambient air into the North Sea was also measured ([Xie et al., 2005](#)). Based on DBP's short half-life in the atmosphere, it is not expected to be persistent in atmospheric air under standard environmental conditions.

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/m^3 over the Arctic ([Xie et al., 2007](#)), and 0.4 to 1.8 ng/m^3 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/m^3 in the United States ([Wilson et al., 2003](#); [Wilson et al., 2001](#)) and 0.08 to 15 ng/m^3 in Sweden ([Cousins et al., 2007](#)).

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.

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. ([2001](#)) reported measured samples of indoor air and dust from 10 daycare centers located in North Carolina. DBP was detected in all air and dust samples with a mean concentration of 239 ng/m^3 (108–404 ng/m^3) 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).

EPA also identified several data sources reporting the presence of DBP in indoor air and dust in locations outside of the United States. Das et al. (2014) explored the implications of industrial activities by comparing the presence of phthalates in two different cities from India. That study analyzed indoor air and dust samples from the Jawaharlal Nehru University campus (a city with low industrial activities) and Okhla (a city with high industrial activities related to the use of phthalates), reporting a general tendency of higher detectable concentrations of DBP in air and dust samples collected in Okhla. This finding suggests that higher concentrations of phthalates in air and dust could be expected near facilities with high use and production of phthalates. Wormuth et al. (2006) determined the indoor air and indoor dust 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, respectively. In a study conducted in Sapporo, Japan, DBP was found to range from 79.6 to 740 ng/m³ in air and 1.8 to 1,476 ng/m³ in indoor dust in residential houses (Kanazawa et al., 2010). DBP was found to be the dominating phthalate in a study that analyzed the phthalate content (DBP, butyl benzyl phthalate [BBP], dicyclohexyl phthalate [DCHP], and di-ethylhexyl phthalate [DEHP]) of particulate matter in indoor spaces in Norway (Rakkestad et al., 2007).

5.2 Aquatic Environments

5.2.1 Surface Water

DBP may 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 samples collected from China, Malaysia, and the United Kingdom (UK) at concentrations ranging from 0.541 µg/L to 4.8 µg/L (Zeng et al., 2008a; Tan, 1995; Preston and Al-Omran, 1989).

The principal properties governing the fate and transport of DBP in surface water are water solubility (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 (1.81×10^{–6} atm·m³/mol at 25 °C, Table 2-1), volatilization is not expected to be a significant source of loss of DBP from surface water. A partitioning analysis estimates that about 10 percent of the DBP released to water will partition to sediments and approximately 90 percent will remain in surface water (see Section 3.2.1). However, based on its log K_{OC} (3.14–3.94), DBP in water is expected to partition to suspended particles and sediments. Because DBP is also expected to biodegrade rapidly in most aquatic environments (Section 4.1.1), it is not expected to persist in surface water except at areas 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.

No monitoring data for DBP in surface water was available within the data collected during the systematic review process for the United States. Several non-U.S. studies 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 (1989) explored the presence of phthalates within the River Mersey Estuary in the UK and reported the presence of DBP freely dissolved in water at concentrations ranging from 0.541 to 1.805 µg/L. Tan (1995) reported the presence of DBP in the Klang River in

Malaysia at concentrations of 0.8 to 4.8 µg/L. Zeng et al. (2008a) reported the presence of DBP in the dissolved aqueous phase of urban lakes in Guangzhou City, China, 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, Greece at concentrations of 0.640 to 16 µg/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 K_{OC} 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, LEV3EPI™ predicts 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).

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

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

In another study, Kim et al. (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 at a median concentration of 13.2 ng/g dw. The study revealed a gradual decreasing trend in the overall concentration of phthalates toward the outer region of the bays farther away from industrial activities. The findings of this study suggests that industrial activities are a major contributor of phthalates in sediments within the area. It also suggests that DBP has the potential to accumulate in sediments at areas of continuous release, such as a surface water body receiving discharge from a municipal wastewater treatment plant.

5.3 Terrestrial Environments

5.3.1 Soil

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, DBP is not likely to volatilize significantly from soils.

DBP is expected to show strong affinity for sorption to soil and its organic constituents based on a log K_{oc} of 3.14 to 3.94 ([Xiang et al., 2019](#); [Russell and McDuffie, 1986](#)) and a log K_{ow} of 4.5 ([NLM, 2024](#)). Thus, DBP is expected to have slow migration potential in soil environments. In addition, DBP is expected to biodegrade rapidly in soil with a half-life of weeks to months. In general, DBP is not expected to be persistent in soil as long as the rate of release does not exceed the rate at which biodegradation can occur.

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 150 acre vacant underdeveloped parcel adjacent to the 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 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 ([Liu et al., 2009](#)). 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 µg/g dw, which exceeds the recommended allowable soil concentrations for DBP set by the state of New York (0.081 µg/g). The study attributed the source of the phthalates to wastewater irrigation, sewage sludge application, disposed plastics, and atmospheric deposition ([Zeng et al., 2008b](#)). Similarly, another study found DBP in abundance in Chinese arable soil. Zeng et al. (2009) reported DBP concentrations ranging from 0.206 to 30.1 µg/g dw in soils from roadsides, residential areas, and parks in Guangzhou, China.

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.

As described in Section 6.2, DBP in wastewater has been reported to be mainly removed by particle sorption and retained in the sewage sludge. Based on the STPWINTM module in EPI SuiteTM, about 55 percent of DBP present in wastewater is expected to be accumulated in sewage sludge and discharged into biosolids. The 2008 EPA National Sewage Sludge Survey reported minimum (1.7 ng/g) and maximum (1,260 ng/g) concentrations of DBP in biosolids ([U.S. EPA, 2009](#)). Separately, DBP concentrations ranging from 1.7 to 1,260 ng/g dw were measured in 20 municipal sewage sludge samples from publicly owned treatment works in the United States ([Ikonomou et al., 2012](#)).

Three studies have reported DBP's concentration in sludge in 71 Chinese WWTPs ranging from 0.0004 to 111 µg/g dw ([Zhu et al., 2019](#); [Meng et al., 2014](#)) and 0.58 to 59 µg/g dw in 40 Korean WWTPs ([Lee et al., 2019b](#)). Two U.S. studies reported sludge concentrations ranging from 0.32 to 17 µg/g dw ([Howie, 1991](#)) and 966 µg/L ([ATSDR, 1999](#)). 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 ([ATSDR, 1999](#)).

5.3.3 Landfills

For the purpose of this assessment, landfills are divided into two zones: (1) an “upper-landfill” zone 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 exist and abiotic degradation is the predominant route of degradation for DBP. In the upper-landfill zone where oxygen might still be present in the subsurface, conditions may still be favorable for aerobic biodegradation; however, photolysis and hydrolysis are not considered to be significant sources of degradation in this zone. In the lower-landfill zone, conditions are assumed to be anoxic, and 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 inhibited and are likely to be irrelevant at 70 °C ([Huang et al., 2013a](#)).

DBP is deposited in landfills continually and in high amounts from the disposal of consumer products containing DBP. Some aerobic biodegradation can occur in upper landfills. Similar to other phthalate esters, under anaerobic conditions present in lower-landfills, DBP is likely to be persistent in landfills 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 landfills ([Huang et al., 2013a](#)). Due to the expected persistence of DBP in landfills, it may dissolve into leachate in small amounts based on a water solubility of 11.2 mg/L and may travel slowly to groundwater during infiltration of rainwater based on a log K_{oc} of 3.14 to 3.94. For instance, several data sources have reported the presence of DBP in landfill leachate. These sources have reported a rapid decrease in DBP concentration from core to leachate samples ([Norin and Strömvall, 2004](#); [Jang and Townsend, 2001](#); [Öman and Hynning, 1993](#); [DERS, 1991](#)). These data sources have also reported DBP concentrations ranging from 0.4 to 7.8 mg/kg and 1 to 17 µg/L in landfill core and leachate samples, respectively. The reported rapid decrease in DBP’s concentration aligns with the expectation that DBP is likely to sorb to organic matter in landfill soils.

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.

Despite DBP having a strong affinity to adsorb to organic matter present in soils and sediments (log K_{oc} = 3.14–3.94 ([Xiang et al., 2019](#); [Russell and McDuffie, 1986](#))), DBP partitioning to groundwater remains 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 ([Geraghty & Miller Inc, 1990](#)). In cases where DBP could reasonably be expected to be present in groundwater environments (proximal to landfills or agricultural land with a history of land applied biosolids), limited persistence is expected based on rates of biodegradation of DBP in aerobic and anaerobic environments (Section 4.1), and DBP is not likely to be persistent in groundwater or subsurface environments unless anoxic conditions exist.

6 REMOVAL AND PERSISTENCE POTENTIAL OF DBP

DBP is not expected to be persistent in the environment because 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 and is expected to undergo photolytic degradation through reaction with atmospheric hydroxyl radicals ($\cdot\text{OH}$) and an estimated half-life of 1.15 days. In aquatic environments, DBP is predicted to hydrolyze slowly at ambient temperature but is not expected to persist since it undergoes rapid aerobic biodegradation (Section 5.2.1). In soil and sediments, DBP has the potential to remain for longer periods of time. Due to its rapid biodegradation under most aquatic environments and estimated bioconcentration factor (BCF) of 159.4 L/kg, DBP is expected to have low bioaccumulation potential. Using LEV3EPI™ (Section 3.2.1), DBP's overall environmental half-life was estimated to be approximately 14 days ([U.S. EPA, 2017](#)). Therefore, DBP is not expected to be persistent in the atmosphere or aquatic and terrestrial environments.

6.1 Destruction and Removal Efficiency

Destruction and removal efficiency (DRE) is a percentage that represents the mass of a pollutant removed or destroyed in a thermal incinerator relative to the mass that entered the system. DBP is classified as a hazardous substance (40 CFR 116.4) 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](#)).

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 in Germany, Japan, and the United States at concentrations of 0 to 1,200 $\mu\text{g/kg}$ and 7.66 to 260 $\mu\text{g/m}^3$, respectively ([Jay and Stieglitz, 1995](#); [Nishikawa et al., 1992](#); [Shane et al., 1990](#)). These findings suggest that incineration of DBP-containing waste has the potential 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 suggests that DBP present during incineration 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

Wastewater treatment is performed to remove contaminants from wastewater using physical, biological, and chemical processes. Generally, municipal wastewater treatment facilities apply primary and secondary treatments. During the primary treatment, screens, grit chambers, and settling tanks are used to remove solids from wastewater. Following primary treatment, the wastewater undergoes a secondary level of treatment, which can remove up to 90 percent of the organic matter in wastewater using biological treatment processes such as trickling filters or activated sludge. Sometimes an additional stage of treatment (tertiary treatment) is utilized to further clean water for additional protection using advanced treatment techniques such as ozonation, chlorination, and disinfection ([U.S. EPA, 1988](#)).

EPA selected 12 high-quality data sources reporting the removal of DBP in wastewater treatment systems in the United States and other locations around the world (*i.e.*, China, Denmark, Sweden). These studies included wastewater treatment systems employing both aerobic and anaerobic processes and

reported a range of 38 to greater than 99 percent removal of DBP in WWTPs employing secondary and/or tertiary treatment units such as activated sludge, secondary clarifiers, and sand filtration ([Wu et al., 2017](#); [Huang et al., 2013b](#); [Shao and Ma, 2009](#); [Peterson and Staples, 2003](#)) (Table 6-1). These studies reported that (1) biodegradation accounted for 27 to 58.9 percent of the overall DBP removal ([Shao and Ma, 2009](#); [Peterson and Staples, 2003](#)); (2) the main removal mechanisms are sorption and biodegradation during the primary and secondary treatment, respectively ([Wu et al., 2017](#); [Huang et al., 2013b](#)); and (3) WWTPs employing secondary and tertiary treatment achieve greater than 99 percent removal of DBP ([Wu et al., 2017](#)). The median removal of DBP has been reported to be 68 to 98 percent within 50 WWTPs across the United States ([U.S. EPA, 1982](#)).

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

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

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

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

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

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 treatment steps before being distributed 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.

Limited information is available on the removal of DBP in drinking water treatment plants. A water concentration of 100 ng/L was measured in the city of Philadelphia's drinking water ([Roy F. Weston Inc., 1980](#)). Several available data sources reported concentrations of DBP in drinking water outside the United States (1–1,830 ng/L) ([Ding et al., 2019](#); [Li et al., 2019](#); [Kong et al., 2017](#); [Shan et al., 2016](#); [Das et al., 2014](#); [Shi et al., 2012](#)). Kong et al. (2017) explored the presence and removal of phthalate esters in

a drinking water treatment system in east China employing coagulation, sedimentation, and filtration treatment processes, and reported 64.5 percent removal of DBP from the treated effluent with a drinking water concentration of 17.2 ng/L. Similarly, Shan et al. (2016) explored the removal of phthalate esters in two drinking water treatment plants in east China. The first plant employed coagulation, sedimentation, filtration, and disinfection treatment processes and reported 31 to 48 percent removal of DBP from the treated effluent while the second plant reported 38 to 56 percent removal of DBP from the treated effluent in a drinking water treatment system employing peroxidation, coagulation, combined flocculation and sedimentation, filtration, and disinfection treatment processes. These findings suggest that conventional drinking water treatment systems have the potential to partially remove DBP present in source water via sorption to suspended organic matter and filtering media.

7 BIOACCUMULATION POTENTIAL OF DBP

The presence of DBP in several marine aquatic species in North America suggest that the substance is bioavailable in aquatic environments ([Mackintosh et al., 2004](#)). However, DBP can be considered readily biodegradable in most aquatic environments, and the estimated BCF of 159.4 L/kg ([U.S. EPA, 2017](#)) suggests that it is expected to have low bioaccumulation potential. EPA evaluated 13 high-quality data sources reporting the aquatic bioconcentration, aquatic bioaccumulation, aquatic food web magnification, and terrestrial bioconcentration of DBP (Table 7-1). The available data sources discussed below suggest that DBP has low bioaccumulation potential in aquatic and terrestrial organisms ([Lee et al., 2019a](#); [U.S. EPA, 2017](#); [Teil et al., 2012](#)) with no apparent biomagnification across trophic levels in the aquatic food web ([Mackintosh et al., 2004](#)).

Several overall high-quality data sources have reported the bioconcentration, bioaccumulation, and food web magnification of DBP in aquatic species. One of these data sources reported DBP BCF values of 2.9 to 41.6 in sheepshead minnow, American oyster, and brown shrimp after a 24-hour exposure of DBP (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 0.78 to 7.48 L/kg in fish, respectively ([U.S. EPA, 2017](#); [Adeogun et al., 2015](#); [Chemical Manufacturers Association, 1984](#)). BCF values of 1,500 to 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 loss of DBP during a 7-day depuration period. A monitoring study reported bioaccumulation factor (BAF) values of 110 to 1,247 L/kg dw in crucian carp, skygager, bluegill, and bass samples collected from the Asan Lake in Korea ([Lee et al., 2019a](#)). The highest BAF value reported in Lee et al. (2019a) was 1,247 L/kg dw in crucian carp. This species is a benthic feeder that generally tends to contain higher levels of phthalate esters due to greater interaction with sediments. However, the available overall high-quality data sources containing aquatic biota-sediment accumulation factors (BSAF), reported BSAF values of 0.35 to 11.8 g_{lipid}/g_{OC} for fish, and 130 g_{lipid}/g_{OC} for oysters ([Adeogun et al., 2015](#); [Teil et al., 2012](#); [Huang et al., 2008](#); [McFall et al., 1985](#)). In addition, the available data sources reported aquatic trophic magnification factor (TMF) values of 0.70 to 0.81 ([Kim et al., 2016](#); [Mackintosh et al., 2004](#)). Despite the differences in reported biomonitoring values, DBP is expected to have low bioconcentration potential and low biomagnification potential across trophic levels in the aquatic food web but potentially result in higher uptake by benthic organisms.

There is very limited information on the bioconcentration and bioaccumulation of DBP in terrestrial environments. EPA extracted and evaluated nine high-quality data sources containing DBP terrestrial plant concentration factors (PCFs) and BSAF information for plants and earthworms, respectively (Table 7-1). Based on DBP's expected strong affinity to organic matter and rapid biodegradation (on the order of weeks to months in soil), DBP is expected to have limited bioavailability in soils. This is supported by the reported low BSAF values of 0.242 to 0.460 in earthworms (*Eisenia foetida*) ([Ji and Deng, 2016](#); [Hu et al., 2005](#)). Similarly, low PCF values have been reported in the range of 0.02 to 9.60 for rice, fruits, vegetables, wheat and maize, pond weed, and wetland 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 biomagnification potential in terrestrial organisms.

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

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

Endpoint	Value(s)	Details	Reference(s)	Overall Quality Ranking
Aquatic bioconcentration factor (BCF)	159.4 L/kg (fish)	Estimated steady-state BCF; Arnot-Gobas method, fish upper trophic level	U.S. EPA (2017)	High
	0.78–7.48 L/kg (fish)	Experimental monitoring sample collection in Nigeria. Tested organisms: <i>Tilapia zillii</i> , <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 vollehovenii</i>)	Adeogun et al. (2015)	High
	11.7 (minnow), 21.1–41.6 (oyster), 2.9–30.6 (shrimp)	Experimental laboratory exposure. Organism type: (small fish) sheepshead minnow, <i>Cyprinodon variegatus</i> ; American oyster, <i>Crassostrea virginica</i> ; brown shrimp, <i>Penaeus aztecus</i>	Wofford et al. (1981)	High
	500–6,600 (aquatic invertebrates)	Experimental laboratory exposure; BCF of 3,500–6,600 (midge larvae, 1–7 days); 2,200–5,000 (water flea, 1–7 days); 1,700–6,500 (scud, 1–7 days); 500–1,900 (mayfly insects, 1–7 days); 1,000–2,700 (Damselfly insects, 1–7 days); and 1,500–5,000 (glass shrimp, 1–3 days) 75% DBP loss during 7-day depuration	Mayer Jr et al. (1973)	High
	525 (fish)	Predicted fish BCF calculated from actual K_{ow} determinations: $\log BCF = (0.542 \times \log K_{ow}) + 0.124$	Chemical Manufacturers Association (1984)	High
Aquatic bioaccumulation factor (BAF)	110–1,247 L/kg dw (fish)	Experimental, monitoring study lakes in Korea; average concentrations in fish: 3.3–37.4 µg/kg dw; log BAF: 2.0–3.1 L/kg dw. Organism type: crucian carp, skygager, bluegill, and bass	Lee et al. (2019a)	High

Endpoint	Value(s)	Details	Reference(s)	Overall Quality Ranking
Aquatic biota-sediment accumulation factor (BSAF)	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>Crassostrea virginica</i>	McFall et al. (1985)	High
	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)	Teil et al. (2012)	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</i> , <i>Liza subviridis</i> , <i>Acanthopagrus schlegeli</i> , <i>Zacco platypus</i> and <i>Acrossocheilus paradoxus</i>	Huang et al. (2008)	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	Kim et al. (2016)	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
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.	Hu et al. (2005) ; Ji and Deng (2016)	High

Endpoint	Value(s)	Details	Reference(s)	Overall Quality Ranking
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)	Cai et al. (2017)	High
	0.16–0.19 (radish)	PCF Value: 0.19 (shoot), 0.16 (root)	Cai et al. (2008)	High
	1.38–9.60 (pondweed)	PCF Value: 4.43–8.04 L/kg; Study length: 10 days; root bioconcentration: 9.60 ± 0.8 (control; lower conc. in found sediment) 1.75 ± 0.2 (spiked; higher conc. found in sediment); stems and leaves bioconcentration: 7.40 ± 0.5 (control; lower conc. in sed) 1.38 ± 0.1 (spiked; higher conc. found in sediment)	Chi and Gao (2015) ; Wang (2014)	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)	Li et al. (2018)	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	Sun et al. (2015)	High
	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

8 OVERALL FATE AND TRANSPORT OF DBP

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

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 can 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.

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 (1) the presence of phthalate-containing articles, (2) proximity to the releasing facilities ([Wang et al., 2013](#); [Abb et al., 2009](#)), and (3) 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. It 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 landfill zones, hydrolysis is expected to be the most prevalent process for the degradation of DBP.

9 WEIGHT OF SCIENTIFIC EVIDENCE CONCLUSIONS FOR DBP 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 ($\cdot\text{OH}$) 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 7);
- 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 4).

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).

REFERENCES

- Abb, M; Heinrich, T; Sorkau, E; Lorenz, W. (2009). Phthalates in house dust. *Environ Int* 35: 965-970. <http://dx.doi.org/10.1016/j.envint.2009.04.007>
- Adeogun, AO; Ibor, OR; Omiwole, RA; Hassan, T; Adegbola, RA; Adewuyi, GO; Arukwe, A. (2015). Occurrence, species, and organ differences in bioaccumulation patterns of phthalate esters in municipal domestic water supply lakes in Ibadan, Nigeria. *J Toxicol Environ Health A* 78: 761-777. <http://dx.doi.org/10.1080/15287394.2015.1030487>
- Atlas, E; Giam, CS. (1981). Global transport of organic pollutants: Ambient concentrations in the remote marine atmosphere. *Science* 211: 163-165. <http://dx.doi.org/10.1126/science.211.4478.163>
- ATSDR. (1999). Toxicological profile for di-n-butyl phthalate (update): Draft for public comment [ATSDR Tox Profile]. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service. <https://search.proquest.com/docview/14522785?accountid=171501>
- Cadogan, D; Howick, C. (2000). Plasticizers. In *Kirk-Othmer Encyclopedia of Chemical Technology*. New York, NY: John Wiley & Sons. <http://dx.doi.org/10.1002/0471238961.1612011903010415.a01>
- Cai, Q; Mo, C; Wu, Q; Zeng, Q. (2008). Polycyclic aromatic hydrocarbons and phthalic acid esters in the soil-radish (*Raphanus sativus*) system with sewage sludge and compost application. *Bioresour Technol* 99: 1830-1836. <http://dx.doi.org/10.1016/j.biortech.2007.03.035>
- Cai, QY; Xiao, PY; Zhao, HM; Lü, H; Zeng, QY; Li, YW; Li, H; Xiang, L; Mo, CH. (2017). Variation in accumulation and translocation of di-n-butyl phthalate (DBP) among rice (*Oryza sativa* L.) genotypes and selection of cultivars for low DBP exposure. *Environ Sci Pollut Res Int* 24: 7298-7309. <http://dx.doi.org/10.1007/s11356-017-8365-2>
- Chang, BV; Liao, CS; Yuan, SY. (2005). Anaerobic degradation of diethyl phthalate, di-n-butyl phthalate, and di-(2-ethylhexyl) phthalate from river sediment in Taiwan. *Chemosphere* 58: 1601-1607. <http://dx.doi.org/10.1016/j.chemosphere.2004.11.031>
- Chemical Manufacturers Association. (1984). Phthalate esters panel: Summary report: Environmental studies - Phase I. Generation of environmental fate and effects data base on 14 phthalate esters. Washington, DC.
- Cheng, J; Liu, Y; Wan, Q; Yuan, L; Yu, X. (2018). Degradation of dibutyl phthalate in two contrasting agricultural soils and its long-term effects on soil microbial community. *Sci Total Environ* 640-641: 821-829. <http://dx.doi.org/10.1016/j.scitotenv.2018.05.336>
- Cheng, Z; Liu, JB; Gao, M; Shi, GZ; Fu, XJ; Cai, P; Lv, YF; Guo, ZB; Shan, CQ; Yang, ZB; Xu, XX; Xian, JR; Yang, YX; Li, KB; Nie, XP. (2019). Occurrence and distribution of phthalate esters in freshwater aquaculture fish ponds in Pearl River Delta, China. *Environ Pollut* 245: 883-888. <http://dx.doi.org/10.1016/j.envpol.2018.11.085>
- Chi, J; Gao, J. (2015). Effects of *Potamogeton crispus* L.-bacteria interactions on the removal of phthalate acid esters from surface water. *Chemosphere* 119: 59-64. <http://dx.doi.org/10.1016/j.chemosphere.2014.05.058>
- Cousins, AP; Remberger, M; Kaj, L; Ekheden, Y; Dusan, B; Brorstroem-Lunden, E. (2007). Results from the Swedish National Screening Programme 2006. Subreport 1: Phthalates (pp. 39). (B1750). Stockholm, SE: Swedish Environmental Research Institute. <http://www3.ivl.se/rapporter/pdf/B1750.pdf>
- Cousins, I; Mackay, D. (2000). Correlating the physical-chemical properties of phthalate esters using the 'three solubility' approach. *Chemosphere* 41: 1389-1399. [http://dx.doi.org/10.1016/S0045-6535\(00\)00005-9](http://dx.doi.org/10.1016/S0045-6535(00)00005-9)
- Cripe, CR; Walker, WW; Pritchard, PH; Bourquin, AW. (1987). A shake-flask test for estimation of biodegradability of toxic organic substances in the aquatic environment. *Ecotoxicol Environ Saf* 14: 239-251. [http://dx.doi.org/10.1016/0147-6513\(87\)90067-4](http://dx.doi.org/10.1016/0147-6513(87)90067-4)

- Das, MT; Ghosh, P; Thakur, IS. (2014). Intake estimates of phthalate esters for South Delhi population based on exposure media assessment. *Environ Pollut* 189: 118-125. <http://dx.doi.org/10.1016/j.envpol.2014.02.021>
- Defoe, DL; Holcombe, GW; Hammermeister, DE; Biesinger, KE. (1990). Solubility and toxicity of eight phthalate esters to four aquatic organisms. *Environ Toxicol Chem* 9: 623-636.
- Dempsey, CR. (1993). A comparison of organic emissions from hazardous waste incinerators versus the 1990 toxics release inventory air releases. *J Air Waste Manag Assoc* 43: 1374-1379. <http://dx.doi.org/10.1080/1073161X.1993.10467212>
- DERS. (1991). Environmental site assessment report for Vista Chemical Company Facility Aberdeen, Mississippi with cover letter [TSCA Submission]. (DERS Project No. 400762; EPA/OTS FYI-OTS-0591-0808). Houston, TX: Vista Chemical Company. <https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0000808.xhtml>
- Desai, S; Govind, R; Tabak, H. (1990). Determination of monod kinetics of toxic compounds by respirometry for structure biodegradability relationships. In WD Tedder; FG Pohland (Eds.), *Emerging Technologies in Hazardous Waste Management* (pp. 142-156). Washington, DC: American Chemical Society. <http://dx.doi.org/10.1021/bk-1990-0422.ch009>
- Ding, M; Kang, Q; Zhang, S; Zhao, F; Mu, D; Zhang, H; Yang, M; Hu, J. (2019). Contribution of phthalates and phthalate monoesters from drinking water to daily intakes for the general population. *Chemosphere* 229: 125-131. <http://dx.doi.org/10.1016/j.chemosphere.2019.05.023>
- Dodson, RE; Udesky, JO; Colton, MD; Mccauley, M; Camann, DE; Yau, AY; Adamkiewicz, G; Rudel, RA. (2017). Chemical exposures in recently renovated low-income housing: Influence of building materials and occupant activities. *Environ Int* 109: 114-127. <http://dx.doi.org/10.1016/j.envint.2017.07.007>
- DOE. (2016). Table 1: Chemicals of concern and associated chemical information. PACs. Washington, D.C.
- EC/HC. (1994). Canadian environmental protection act priority substances list assessment report: Dibutyl phthalate. Ottawa, Ontario: Environment Canada, Health Canada. https://www.canada.ca/content/dam/hc-sc/migration/hc-sc/ewh-semt/alt_formats/hecs-sesc/pdf/pubs/contaminants/psl1-lsp1/phthalate_dibutyl_phthalate/butyl_phthalate-eng.pdf
- EC/HC. (2017). Draft screening assessment: Phthalate substance grouping. Ottawa, Ontario: Government of Canada, Environment Canada, Health Canada. <http://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=516A504A-1>
- ECHA. (2012). Committee for Risk Assessment (RAC) Committee for Socio-economic Analysis (SEAC): Background document to the Opinion on the Annex XV dossier proposing restrictions on four phthalates: Annexes. Helsinki, Finland. https://echa.europa.eu/documents/10162/13641/rest_four_phthalates_axvreport_annex_en.pdf/92a98820-0a66-4a2c-fabe-84bf64e45af4
- Elsevier. (2019). Reaxys: physical-chemical property data for dibutyl phthalate. CAS Registry Number: 84-74-2. Available online
- ERM. (1988). Hydrogeological investigation at the Union Carbide solvents and materials coating plant with cover letter dated 070688 [TSCA Submission]. (EPA/OTS Doc #86-880000319). Bound Brook, NJ: Union Carbide Corporation. <https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0514200.xhtml>
- Federal Register. (1981). Federal Register: January 23, 1981, Part 4. Incinerator Standards for Owners and Operators of Hazardous Waste Management Facilities; Interim Final Rule and Proposed Rule. (OSWFR81027). <http://nepis.epa.gov/exe/ZyPURL.cgi?Dockkey=10003NOR.txt>
- Fujita, M; Ike, M; Ishigaki, T; Sei, K; Jeong, JS; Makihiro, N; Lertsirisopon, R. (2005). Biodegradation of Three Phthalic Acid Esters by Microorganisms from Aquatic Environment. *Nihon Mizushori Seibutsu Gakkaishi* 41: 193-201. <http://dx.doi.org/10.2521/jswtb.41.193>

- Gani, KM; Kazmi, AA. (2016). Comparative assessment of phthalate removal and risk in biological wastewater treatment systems of developing countries and small communities. *Sci Total Environ* 569-570: 661-671. <http://dx.doi.org/10.1016/j.scitotenv.2016.06.182>
- Gavala, HN; Atriste-Mondragon, F; Iranpour, R; Ahring, BK. (2003). Biodegradation of phthalate esters during the mesophilic anaerobic digestion of sludge. *Chemosphere* 52: 673-682. [http://dx.doi.org/10.1016/S0045-6535\(03\)00126-7](http://dx.doi.org/10.1016/S0045-6535(03)00126-7)
- Geraghty & Miller Inc. (1990). Phase II - Site investigation: Borden site Carson, California (volume I) with attached appendices and cover letter dated 032790 [TSCA Submission]. (EPA/OTS Doc #86-900000344). Philadelphia, PA: Rohm and Haas Company. <https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0522908.xhtml>
- Grigoriadou, A; Schwarzbauer, J; Georgakopoulos, A. (2008). Molecular indicators for pollution source identification in marine and terrestrial water of the industrial area of Kavala City, North Greece. *Environ Pollut* 151: 231-242. <http://dx.doi.org/10.1016/j.envpol.2007.01.053>
- Hamilton, DJ. (1980). Gas chromatographic measurement of volatility of herbicide esters. *J Chromatogr* 195: 75-83. [http://dx.doi.org/10.1016/s0021-9673\(00\)81544-7](http://dx.doi.org/10.1016/s0021-9673(00)81544-7)
- Haynes, WM. (2014a). Diisodecyl phthalate. In *CRC handbook of chemistry and physics* (95 ed.). Boca Raton, FL: CRC Press.
- Haynes, WM. (2014b). Tris(2-chloroethyl) phosphate. In *WM Haynes (Ed.), CRC handbook of chemistry and physics* (95th ed., pp. 3-542). Boca Raton, FL: CRC Press.
- Howard, PH; Banerjee, S; Robillard, KH. (1985). Measurement of water solubilities octanol-water partition coefficients and vapor pressures of commercial phthalate esters. *Environ Toxicol Chem* 4: 653-662. <http://dx.doi.org/10.1002/etc.5620040509>
- Howie, B. (1991). Effects of dried wastewater-treatment sludge application on ground-water quality in South Dade County, Florida. *Howie, B.* <http://dx.doi.org/10.3133/wri914135>
- Hu, XY; Wen, B; Zhang, S; Shan, XQ. (2005). Bioavailability of phthalate congeners to earthworms (*Eisenia fetida*) in artificially contaminated soils. *Ecotoxicol Environ Saf* 62: 26-34. <http://dx.doi.org/10.1016/j.ecoenv.2005.02.012>
- Huang, J; Nkrumah, PN; Li, Y; Appiah-Sefah, G. (2013a). Chemical behavior of phthalates under abiotic conditions in landfills [Review]. *Rev Environ Contam Toxicol* 224: 39-52. http://dx.doi.org/10.1007/978-1-4614-5882-1_2
- Huang, PC; Tien, CJ; Sun, YM; Hsieh, CY; Lee, CC. (2008). Occurrence of phthalates in sediment and biota: Relationship to aquatic factors and the biota-sediment accumulation factor. *Chemosphere* 73: 539-544. <http://dx.doi.org/10.1016/j.chemosphere.2008.06.019>
- Huang, R; Wang, Z; Liu, G; Luo, Q. (2013b). Removal efficiency of environmental endocrine disrupting chemicals pollutants-phthalate esters in northern WWTP. *Adv Mater Res* 807-809: 694-698. <http://dx.doi.org/10.4028/www.scientific.net/AMR.807-809.694>
- Ikonomou, MG; Kelly, BC; Blair, JD; Gobas, FA. (2012). An interlaboratory comparison study for the determination of dialkyl phthalate esters in environmental and biological samples. *Environ Toxicol Chem* 31: 1948-1956. <http://dx.doi.org/10.1002/etc.1912>
- Inman, JC; Strachan, SD; Sommers, LE; Nelson, DW. (1984). The decomposition of phthalate esters in soil. *J Environ Sci Health B* 19: 245-257. <http://dx.doi.org/10.1080/03601238409372429>
- Ishak, H; Stephan, J; Karam, R; Goutaudier, C; Mokbel, I; Saliba, C; Saab, J. (2016). Aqueous solubility, vapor pressure and octanol-water partition coefficient of two phthalate isomers dibutyl phthalate and di-isobutyl phthalate contaminants of recycled food packages. *Fluid Phase Equilibria* 427: 362-370. <http://dx.doi.org/10.1016/j.fluid.2016.07.018>
- Jang, YC; Townsend, TG. (2001). Occurrence of organic pollutants in recovered soil fines from construction and demolition waste. *Waste Manag* 21: 703-715. [http://dx.doi.org/10.1016/S0956-053X\(01\)00007-1](http://dx.doi.org/10.1016/S0956-053X(01)00007-1)

- Jay, K; Stieglitz, L. (1995). Identification and quantification of volatile organic components in emissions of waste incineration plants. *Chemosphere* 30: 1249-1260. [http://dx.doi.org/10.1016/0045-6535\(95\)00021-Y](http://dx.doi.org/10.1016/0045-6535(95)00021-Y)
- Ji, LL; Deng, L, iP. (2016). Influence of carbon nanotubes on dibutyl phthalate bioaccumulation from contaminated soils by earthworms. In *Energy, Environmental & Sustainable Ecosystem Development*. Singapore: World Scientific. http://dx.doi.org/10.1142/9789814723008_0043
- Jianlong, W. (2004). Effect of di-n-butyl phthalate (DBP) on activated sludge. *Process Biochemistry* 39: 1831-1836. <http://dx.doi.org/10.1016/j.procbio.2003.08.004>
- Johnson, BT; Heitkamp, MA; Jones, JR. (1984). Environmental and chemical factors influencing the biodegradation of phthalic-acid esters in freshwater sediments. *Environ Pollut Ser B* 8: 101-118. [http://dx.doi.org/10.1016/0143-148X\(84\)90021-1](http://dx.doi.org/10.1016/0143-148X(84)90021-1)
- Kanazawa, A; Saito, I; Araki, A; Takeda, M; Ma, M; Saijo, Y; Kishi, R. (2010). Association between indoor exposure to semi-volatile organic compounds and building-related symptoms among the occupants of residential dwellings. *Indoor Air* 20: 72-84. <http://dx.doi.org/10.1111/j.1600-0668.2009.00629.x>
- Kao, PH; Lee, FY; Hseu, ZY. (2005). Sorption and biodegradation of phthalic acid esters in freshwater sediments. *J Environ Sci Health A Tox Hazard Subst Environ Eng* 40: 103-115. <http://dx.doi.org/10.1081/ESE-200033605>
- Kickham, P; Otton, SV; Moore, MM; Ikonomou, MG; Gobas, FAP, C. (2012). Relationship between biodegradation and sorption of phthalate esters and their metabolites in natural sediments. *Environ Toxicol Chem* 31: 1730-1737. <http://dx.doi.org/10.1002/etc.1903>
- Kim, J; Gobas, FA; Arnot, JA; Powell, DE; Seston, RM; Woodburn, KB. (2016). Evaluating the roles of biotransformation, spatial concentration differences, organism home range, and field sampling design on trophic magnification factors. *Sci Total Environ* 551-552: 438-451. <http://dx.doi.org/10.1016/j.scitotenv.2016.02.013>
- Kim, S; Kim, Y; Moon, HB. (2021). Contamination and historical trends of legacy and emerging plasticizers in sediment from highly industrialized bays of Korea. *Sci Total Environ* 765: 142751. <http://dx.doi.org/10.1016/j.scitotenv.2020.142751>
- Kong, YL; Shen, JM; Chen, ZL; Kang, J; Li, TP; Wu, XF; Kong, XZ; Fan, LT. (2017). Profiles and risk assessment of phthalate acid esters (PAEs) in drinking water sources and treatment plants, East China. *Environ Sci Pollut Res Int* 24: 23646-23657. <http://dx.doi.org/10.1007/s11356-017-9783-x>
- Kubwabo, C; Rasmussen, PE; Fan, X; Kosarac, I; Wu, F; Zidek, A; Kuchta, SL. (2013). Analysis of selected phthalates in Canadian indoor dust collected using a household vacuum and a standardized sampling techniques. *Indoor Air* 23: 506-514. <http://dx.doi.org/10.1111/ina.12048>
- Lee, YM; Lee, JE; Choe, W; Kim, T; Lee, JY; Kho, Y; Choi, K; Zoh, KD. (2019a). Distribution of phthalate esters in air, water, sediments, and fish in the Asan Lake of Korea. *Environ Int* 126: 635-643. <http://dx.doi.org/10.1016/j.envint.2019.02.059>
- Lee, YS; Lee, S; Lim, JE; Moon, HB. (2019b). Occurrence and emission of phthalates and non-phthalate plasticizers in sludge from wastewater treatment plants in Korea. *Sci Total Environ* 692: 354-360. <http://dx.doi.org/10.1016/j.scitotenv.2019.07.301>
- Lei, Y; Zhu, C; Lu, J; Zhu, Y; Zhang, Q; Chen, T; Xiong, H. (2018). Photochemical oxidation of di-n-butyl phthalate in atmospheric hydrometeors by hydroxyl radicals from nitrous acid. *Environ Sci Pollut Res Int* 25: 31091-31100. <http://dx.doi.org/10.1007/s11356-018-3091-y>
- Lertsirisopon, R; Soda, S; Sei, K; Ike, M. (2009). Abiotic degradation of four phthalic acid esters in aqueous phase under natural sunlight irradiation. *J Environ Sci* 21: 285-290. [http://dx.doi.org/10.1016/S1001-0742\(08\)62265-2](http://dx.doi.org/10.1016/S1001-0742(08)62265-2)
- Lertsirisopon, R; Soda, S; Sei, K; Ike, M; Fujita, M. (2006). Biodegradability of four phthalic acid esters under anaerobic condition assessed using natural sediment. *J Environ Sci* 18: 793-796.

- Li, J; Zhao, H; Xia, W; Zhou, Y; Xu, S; Cai, Z. (2019). Nine phthalate metabolites in human urine for the comparison of health risk between population groups with different water consumptions. *Sci Total Environ* 649: 1532-1540. <http://dx.doi.org/10.1016/j.scitotenv.2018.08.294>
- Li, R; Liang, J; Duan, H; Gong, Z. (2017a). Spatial distribution and seasonal variation of phthalate esters in the Jiulong River estuary, Southeast China. *Mar Pollut Bull* 122: 38-46. <http://dx.doi.org/10.1016/j.marpolbul.2017.05.062>
- Li, R; Liang, J; Gong, Z; Zhang, N; Duan, H. (2017b). Occurrence, spatial distribution, historical trend and ecological risk of phthalate esters in the Jiulong River, Southeast China [Supplemental Data]. *Sci Total Environ* 580: 388-397. <http://dx.doi.org/10.1016/j.scitotenv.2016.11.190>
- Li, Y, an; Huang, G; Gu, H, ua; Huang, Q; Lou, C; Zhang, L, ei; Liu, H. (2018). Assessing the Risk of Phthalate Ester (PAE) Contamination in Soils and Crops Irrigated with Treated Sewage Effluent. *Water* 10: 999. <http://dx.doi.org/10.3390/w10080999>
- Li, Y; Gao, J; Meng, F; Chi, J. (2015). Enhanced biodegradation of phthalate acid esters in marine sediments by benthic diatom *Cylindrotheca closterium*. *Sci Total Environ* 508: 251-257. <http://dx.doi.org/10.1016/j.scitotenv.2014.12.002>
- Liu, WL; Shen, CF; Zhang, Z; Zhang, CB. (2009). Distribution of phthalate esters in soil of e-waste recycling sites from Taizhou city in China. *Bull Environ Contam Toxicol* 82: 665-667. <http://dx.doi.org/10.1007/s00128-009-9699-3>
- Lu, C. (2009). Prediction of environmental properties in water-soil-air systems for phthalates. *Bull Environ Contam Toxicol* 83: 168-173. <http://dx.doi.org/10.1007/s00128-009-9728-2>
- Mackay, D; Di Guardo, A; Paterson, S; Cowan, CE. (1996). Evaluating the environmental fate of a variety of types of chemicals using the EQC model. *Environ Toxicol Chem* 15: 1627-1637. <http://dx.doi.org/10.1002/etc.5620150929>
- Mackintosh, CE; Maldonado, J; Hongwu, J; Hoover, N; Chong, A; Ikonomou, MG; Gobas, FA. (2004). Distribution of phthalate esters in a marine aquatic food web: Comparison to polychlorinated biphenyls. *Environ Sci Technol* 38: 2011-2020. <http://dx.doi.org/10.1021/es034745r>
- Mackintosh, CE; Maldonado, JA; Ikonomou, MG; Gobas, FA. (2006). Sorption of phthalate esters and PCBs in a marine ecosystem. *Environ Sci Technol* 40: 3481-3488. <http://dx.doi.org/10.1021/es0519637>
- Mayer Jr, F; Sanders, HO; Walsh, DF. (1973). Toxicity, residue dynamics, and reproductive effects of phthalate esters in aquatic invertebrates. *Environ Res* 6: 84-90. [http://dx.doi.org/10.1016/0013-9351\(73\)90020-0](http://dx.doi.org/10.1016/0013-9351(73)90020-0)
- McFall, JA; Antoine, S. R.; Deleon, IR. (1985). Base-neutral extractable organic pollutants in biota and sediments from Lake Pontchartrain. *Chemosphere* 14: 1561-1569. [http://dx.doi.org/10.1016/0045-6535\(85\)90011-6](http://dx.doi.org/10.1016/0045-6535(85)90011-6)
- Meng, XZ; Wang, Y; Xiang, N; Chen, L; Liu, Z; Wu, B; Dai, X; Zhang, YH; Xie, Z; Ebinghaus, R. (2014). Flow of sewage sludge-borne phthalate esters (PAEs) from human release to human intake: implication for risk assessment of sludge applied to soil. *Sci Total Environ* 476-477: 242-249. <http://dx.doi.org/10.1016/j.scitotenv.2014.01.007>
- Mueller, M; Klein, W. (1992). Comparative evaluation of methods predicting water solubility for organic compounds. *Chemosphere* 25: 769-782. [http://dx.doi.org/10.1016/0045-6535\(92\)90067-2](http://dx.doi.org/10.1016/0045-6535(92)90067-2)
- NCBI. (2020). PubChem Compound Summary for CID 3026 Dibutyl phthalate. Center for Biotechnology Information.
- Net, S; Sempéré, R; Delmont, A; Paluselli, A; Ouddane, B. (2015). Occurrence, fate, behavior and ecotoxicological state of phthalates in different environmental matrices [Review]. *Environ Sci Technol* 49: 4019-4035. <http://dx.doi.org/10.1021/es505233b>

- NIOSH. (1976). Occupational health guideline for dibutylphthalate. U.S. Department of Health and Human Services, Public Health Service, Centers for Disease Control, National Institute for Occupational Safety and Health.
- NIOSH. (2007). NIOSH pocket guide to chemical hazards. (DHHS Publication No. (NIOSH) 2005-149; CBRNIAC-CB-112149 ISBN 9780160727511). Cincinnati, OH. <http://www.cdc.gov/niosh/docs/2005-149/>
- Nishikawa, H; Katami, T; Takahara, Y; Sumida, H; Yasuhara, A. (1992). Emission of organic compounds by combustion of waste plastics involving vinyl chloride polymer. *Chemosphere* 25: 1953-1960. [http://dx.doi.org/10.1016/0045-6535\(92\)90034-O](http://dx.doi.org/10.1016/0045-6535(92)90034-O)
- NIST. (2022). NIST Chemistry WebBook: Dibutyl phthalate (84-74-2), Standard Reference Database No. 69. Washington, DC: US Sec Commerce. <https://webbook.nist.gov/cgi/cbook.cgi?ID=84-74-2&Units=SI&cTG=on&cIR=on&cTC=on&cTZ=on&cTP=on&cMS=on&cTR=on&cUV=on&cIE=on&cGC=on&cIC=on&cES=on&cDI=on&cSO=on>
- NITE. (2019). Japan CHEMicals Collaborative Knowledge database (J-CHECK), CASRN: 84-74-2. Available online at https://www.nite.go.jp/chem/jcheck/detail.action?cno=84-74-2&mno=3-1303&request_locale=en (accessed 2022-02-04 00:00:00+00:00).
- NLM. (2013). PubChem: Hazardous Substance Data Bank: Diisobutyl phthalate, 84-69-5. Available online at <https://pubchem.ncbi.nlm.nih.gov/compound/6782#source=HSD>
- NLM. (2024). PubChem: Hazardous substance data bank: Dibutyl phthalate, 84-74-2. Available online at <https://pubchem.ncbi.nlm.nih.gov/compound/3026>
- Norin, M; Strömvall, AM. (2004). Leaching of organic contaminants from storage of reclaimed asphalt pavement. *Environ Technol* 25: 323-340. <http://dx.doi.org/10.1080/09593330409355466>
- O'Neil, MJ. (2013). Dibutyl phthalate. In *The Merck index* (15th ed.). Cambridge, UK: Royal Society of Chemistry.
- Öman, C; Hynning, PÅ. (1993). Identification of organic compounds in municipal landfill leachates. *Environ Pollut* 80: 265-271. [http://dx.doi.org/10.1016/0269-7491\(93\)90047-R](http://dx.doi.org/10.1016/0269-7491(93)90047-R)
- Park, C; Sheehan, RJ. (2000). Phthalic acids and other benzenepolycarboxylic acids. In *Kirk-Othmer encyclopedia of chemical toxicology*. New York: John Wiley & Sons. <http://dx.doi.org/10.1002/0471238961.1608200816011811>
- Parkerton, TF; Staples, CA. (2003). An assessment of the potential environmental risks posed by phthalates in soil and sediment. In *CA Staples (Ed.), Phthalate esters* (pp. 317-349). Berlin, Germany: Springer. <http://dx.doi.org/10.1007/b11471>
- Peng, X; Li, X. (2012). Compound-specific isotope analysis for aerobic biodegradation of phthalate acid esters. *Talanta* 97: 445-449. <http://dx.doi.org/10.1016/j.talanta.2012.04.060>
- Peters, RJB; Beeltje, H; van Delft, RJ. (2008). Xeno-estrogenic compounds in precipitation. *J Environ Monit* 10: 760-769. <http://dx.doi.org/10.1039/b805983g>
- Peterson, DR; Staples, CA. (2003). Degradation of phthalate esters in the environment. In *Series Anthropogenic Compounds*. New York, NY: Springer-Verlag. <http://dx.doi.org/10.1007/b11464>
- Preston, MR; Al-Omran, LA. (1989). Phthalate ester speciation in estuarine water, suspended particulates and sediments. *Environ Pollut* 62: 183-194. [http://dx.doi.org/10.1016/0269-7491\(89\)90186-3](http://dx.doi.org/10.1016/0269-7491(89)90186-3)
- Rakkestad, KE; Dye, CJ; Yttri, KE; Holme, JA; Hongslo, JK; Schwarze, PE; Becher, R. (2007). Phthalate levels in Norwegian indoor air related to particle size fraction. *J Environ Monit* 9: 1419-1425. <http://dx.doi.org/10.1039/b709947>
- Roslev, P; Vorkamp, K; Aarup, J; Frederiksen, K; Nielsen, PH. (2007). Degradation of phthalate esters in an activated sludge wastewater treatment plant. *Water Res* 41: 969-976. <http://dx.doi.org/10.1016/j.watres.2006.11.04>
- Roy F. Weston Inc. (1980). Characterization and fate of the discharge of priority pollutants from the Rohm and Haas Philadelphia plant into the Delaware low level collector of the Philadelphia

- sewer [TSCA Submission]. (RH-28; W.O.#0053-14-01; OTS0205979. 878212294. TSCATS/016780). Philadelphia, PA: Rohm and Haas Company.
<https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0205979.xhtml>
- Rudel, RA; Brody, JG; Spengler, JD; Vallarino, J; Geno, PW; Sun, G; Yau, A. (2001). Identification of selected hormonally active agents and animal mammary carcinogens in commercial and residential air and dust samples. *J Air Waste Manag Assoc* 51: 499-513.
<http://dx.doi.org/10.1080/10473289.2001.10464292>
- Rumble, JR. (2018a). Aqueous solubility and Henry's law constants of organic compounds. In *CRC Handbook of Chemistry and Physics* (99 ed.). Boca Raton, FL: CRC Press. Taylor & Francis Group. https://hbcpc.chemnetbase.com/faces/documents/05_32/05_32_0001.xhtml
- Rumble, JR. (2018b). Dibutyl phthalate. In *CRC handbook of chemistry and physics* (99 ed.). Boca Raton, FL: CRC Press.
- Rumble, JR. (2018c). Flammability of chemical substances. In *CRC Handbook of Chemistry and Physics* (99 ed.). Boca Raton, FL: CRC Press. Taylor & Francis Group.
- Rumble, JR. (2018d). Viscosity of liquids. In *CRC Handbook of Chemistry and Physics* (99 ed.). Boca Raton, FL: CRC Press. Taylor & Francis Group.
https://hbcpc.chemnetbase.com/faces/documents/06_37/06_37_0001.xhtml
- Rumble, JR. (2021). *CRC Handbook of Chemistry and Physics Online* (Version 102nd Edition (Internet Version 2021)) [Database]. Boca Raton, FL: CRC Press/Taylor & Francis. Retrieved from https://hbcpc.chemnetbase.com/faces/documents/02_07/02_07_0001.xhtml
- Russell, DJ; McDuffie, B. (1986). Chemodynamic properties of phthalate esters partitioning and soil migration. *Chemosphere* 15: 1003-1022. [http://dx.doi.org/10.1016/0045-6535\(86\)90553-9](http://dx.doi.org/10.1016/0045-6535(86)90553-9)
- Russell, DJ; McDuffie, B; Fineberg, S. (1985). The effect of biodegradation on the determination of some chemodynamic properties of phthalate esters. *J Environ Sci Health A Environ Sci Eng* 20: 927-941. <http://dx.doi.org/10.1080/10934528509375268>
- Saini, G; Pant, S; Singh, SO; Kazmi, AA; Alam, T. (2016). A comparative study of occurrence and fate of endocrine disruptors: Diethyl phthalate and dibutyl phthalate in ASP- and SBR-based wastewater treatment plants. *Environ Monit Assess* 188: 609. <http://dx.doi.org/10.1007/s10661-016-5617-4>
- Salaudeen, T; Okoh, O; Agunbiade, F; Okoh, A. (2018a). Fate and impact of phthalates in activated sludge treated municipal wastewater on the water bodies in the Eastern Cape, South Africa. *Chemosphere* 203: 336-344. <http://dx.doi.org/10.1016/j.chemosphere.2018.03.176>
- Salaudeen, T; Okoh, O; Agunbiade, F; Okoh, A. (2018b). Phthalates removal efficiency in different wastewater treatment technology in the Eastern Cape, South Africa. *Environ Monit Assess* 190: 299. <http://dx.doi.org/10.1007/s10661-018-6665-8>
- Shan, XM; Wang, BS; Lu, BB; Shen, DH. (2016). [Investigation of pollution of phthalate esters and bisphenols in source water and drinking water in Hefei City, China]. *Huanjing yu Zhiye Yixue* 33: 350-355. <http://dx.doi.org/10.13213/j.cnki.jeom.2016.15419>
- Shane, BS; Henry, CB; Hotchkiss, JH; Klausner, KA; Gutenmann, WH; Lisk, DJ. (1990). Organic toxicants and mutagens in ashes from eighteen municipal refuse incinerators. *Arch Environ Contam Toxicol* 19: 665-673. <http://dx.doi.org/10.1007/BF01183982>
- Shanker, R; Ramakrishna, C; Seth, PK. (1985). Degradation of some phthalic-acid esters in soil. *Environ Pollut Ser A* 39: 1-7. [http://dx.doi.org/10.1016/0143-1471\(85\)90057-1](http://dx.doi.org/10.1016/0143-1471(85)90057-1)
- Shao, XL; Ma, J. (2009). Fate and mass balance of 13 kinds of endocrine disrupting chemicals in a sewage treatment plant. In *2009 3rd International Conference on Bioinformatics and Biomedical Engineering*, Vols 1-11. Piscataway, NJ: Institute of Electrical and Electronics Engineers.
<http://dx.doi.org/10.1109/ICBBE.2009.5162850>

- [Shi, W; Hu, X; Zhang, F; Hu, G; Hao, Y; Zhang, X; Liu, H; Wei, S; Wang, X; Giesy, JP; Yu, H.](#) (2012). Occurrence of thyroid hormone activities in drinking water from eastern China: Contributions of phthalate esters. *Environ Sci Technol* 46: 1811-1818. <http://dx.doi.org/10.1021/es202625r>
- [SRC.](#) (1983a). Exhibit I shake flask biodegradation of 14 commercial phthalate esters [TSCA Submission]. (SRC L1543-05. OTS0508481. 42005 G5-2. 40-8326129. TSCATS/038111). Chemical Manufacturers Association. <https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0508481.xhtml>
- [SRC.](#) (1983b). Measurement of the water solubilities of phthalate esters (final report) [TSCA Submission]. (EPA/OTS Doc #40-8326142). Chemical Manufacturers Association. <https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0508401.xhtml>
- [SRC.](#) (1984). Final report measurement of octanol-water partition coefficients of phthalate esters [TSCA Submission]. (SRC-TR-84-642. OTS0508491. 42005 G9-3. 40-8426081. TSCATS/038164). Chemical Manufacturers Association. <https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0508491.xhtml>
- [Sun, J; Wu, X; Gan, J.](#) (2015). Uptake and metabolism of phthalate esters by edible plants. *Environ Sci Technol* 49: 8471-8478. <http://dx.doi.org/10.1021/acs.est.5b01233>
- [Tabak, HH; Quave, SA; Mashni, CI; Barth, EF.](#) (1981). Biodegradability studies with organic priority pollutant compounds. *J Water Pollut Control Fed* 53: 1503-1518.
- [Tan, GH.](#) (1995). Residue levels of phthalate esters in water and sediment samples from the klang river basin. *Bull Environ Contam Toxicol* 54: 171-176. <http://dx.doi.org/10.1007/bf00197427>
- [Tang, J; An, T; Li, G; Wei, C.](#) (2017). Spatial distributions, source apportionment and ecological risk of SVOCs in water and sediment from Xijiang River, Pearl River Delta. *Environ Geochem Health* 40: 1853-1865. <http://dx.doi.org/10.1007/s10653-017-9929-2>
- [Teil, MJ; Tlili, K; Blanchard, M; Chevreuil, M; Alliot, F; Labadie, P.](#) (2012). Occurrence of Polybrominated Diphenyl Ethers, Polychlorinated Biphenyls, and Phthalates in Freshwater Fish From the Orge River (Ile-de France). *Arch Environ Contam Toxicol* 63: 101-113. <http://dx.doi.org/10.1007/s00244-011-9746-z>
- [Tomei, MC; Mosca Angelucci, D; Mascolo, G; Kunkel, U.](#) (2019). Post-aerobic treatment to enhance the removal of conventional and emerging micropollutants in the digestion of waste sludge. *Waste Manag* 96: 36-46. <http://dx.doi.org/10.1016/j.wasman.2019.07.013>
- [Tran, BC; Teil, MJ; Blanchard, M; Alliot, F; Chevreuil, M.](#) (2014). BPA and phthalate fate in a sewage network and an elementary river of France. Influence of hydroclimatic conditions. *Chemosphere* 119C: 43-51. <http://dx.doi.org/10.1016/j.chemosphere.2014.04.036>
- [U.S. EPA.](#) (1982). Fate of priority pollutants in publicly owned treatment works, Volume i. (EPA 440/1-82/303). Washington, DC: Effluent Guidelines Division. <http://nepis.epa.gov/exe/ZyPURL.cgi?Dockey=000012HL.txt>
- [U.S. EPA.](#) (1993). Standards for the use or disposal of sewage sludge: Final rules [EPA Report]. (EPA 822/Z-93-001). Washington, DC.
- [U.S. EPA.](#) (2006). Data quality assessment: Statistical methods for practitioners [EPA Report]. (EPA/240/B-06/003; EPA QA/G-9S). Washington, DC. <https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=900B0D00.txt>
- [U.S. EPA.](#) (2017). Estimation Programs Interface Suite™ v.4.11. Washington, DC: U.S. Environmental Protection Agency, Office of Pollution Prevention Toxics. Retrieved from <https://www.epa.gov/tsca-screening-tools/download-epi-suitetm-estimation-program-interface-v411>
- [U.S. EPA.](#) (2019). Chemistry Dashboard Information for Dibutyl Phthalate. 84-74-2. <https://comptox.epa.gov/dashboard/dsstoxdb/results?search=DTXSID2021781>
- [U.S. EPA.](#) (2020). Final scope of the risk evaluation for dibutyl phthalate (1,2-benzenedicarboxylic acid, 1,2-dibutyl ester); CASRN 84-74-2 [EPA Report]. (EPA-740-R-20-016). Washington, DC:

Office of Chemical Safety and Pollution Prevention.

https://www.epa.gov/sites/default/files/2020-09/documents/casrn_84-74-2_dibutyl_phthalate_final_scope_0.pdf

- U.S. EPA. (2021). Draft systematic review protocol supporting TSCA risk evaluations for chemical substances, Version 1.0: A generic TSCA systematic review protocol with chemical-specific methodologies. (EPA Document #EPA-D-20-031). Washington, DC: Office of Chemical Safety and Pollution Prevention. <https://www.regulations.gov/document/EPA-HQ-OPPT-2021-0414-0005>
- U.S. EPA. (2025a). Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport for Dibutyl Phthalate (DBP). Washington, DC: Office of Pollution Prevention and Toxics.
- U.S. EPA. (2025b). Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties for Dibutyl Phthalate (DBP). Washington, DC: Office of Pollution Prevention and Toxics.
- U.S. EPA. (2025c). Risk Evaluation for Dibutyl Phthalate (DBP). Washington, DC: Office of Pollution Prevention and Toxics.
- U.S. EPA. (2025d). Systematic Review Protocol for Dibutyl Phthalate (DBP). Washington, DC: Office of Pollution Prevention and Toxics.
- Union Carbide. (1974). Environmental impact analysis product biodegradability testing [TSCA Submission]. (Project No. 910F44. File 19751. OTS0206066. 878212060. TSCATS/017096). Union Carbide Corp.
<https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0206066.xhtml>
- Verbruggen, EM; Klammer, HJC; Villerius, L; Brinkman, UAT; Hermens, JL. (1999). Gradient elution reversed-phase high-performance liquid chromatography for fractionation of complex mixtures of organic micropollutants according to hydrophobicity using isocratic retention parameters. *J Chromatogr A* 835: 19-27. [http://dx.doi.org/10.1016/S0021-9673\(97\)90065-0](http://dx.doi.org/10.1016/S0021-9673(97)90065-0)
- Walker, WW; Cripe, CR; Pritchard, PH; Bourquin, AW. (1984). Dibutylphthalate degradation in estuarine and fresh-water sites. *Chemosphere* 13: 1283-1294. [http://dx.doi.org/10.1016/0045-6535\(84\)90044-4](http://dx.doi.org/10.1016/0045-6535(84)90044-4)
- Wang, A. (2014). Effect of spiked phthalic acid esters on dissipation efficiency of *Potamogeton crispus* L. in the rhizosphere of surface sediments from the Haihe River, China. *Journal of Soils and Sediments* 14: 243-250. <http://dx.doi.org/10.1007/s11368-013-0794-z>
- Wang, A; Chi, J, ie. (2012). Phthalic acid esters in the rhizosphere sediments of emergent plants from two shallow lakes. *Journal of Soils and Sediments* 12: 1189. <http://dx.doi.org/10.1007/s11368-012-0541-x>
- Wang, J; Liu, P; Shi, H; Qian, Y. (1997a). Biodegradation of phthalic acid ester in soil by indigenous and introduced microorganisms. *Chemosphere* 35: 1747-1754. [http://dx.doi.org/10.1016/S0045-6535\(97\)00255-5](http://dx.doi.org/10.1016/S0045-6535(97)00255-5)
- Wang, JL; Liu, P; Shi, HC; Yi, QA. (1997b). Kinetics of phthalic acid ester degradation by acclimated activated sludge. *Process Biochemistry* 32: 567-571. [http://dx.doi.org/10.1016/S0032-9592\(97\)00015-0](http://dx.doi.org/10.1016/S0032-9592(97)00015-0)
- Wang, LM; Richert, R. (2007). Glass transition dynamics and boiling temperatures of molecular liquids and their isomers. *J Phys Chem B* 111: 3201-3207. <http://dx.doi.org/10.1021/jp068825>
- Wang, W; Wu, FY; Huang, MJ; Kang, Y; Cheung, KC; Wong, MH. (2013). Size fraction effect on phthalate esters accumulation, bioaccessibility and in vitro cytotoxicity of indoor/outdoor dust, and risk assessment of human exposure. *J Hazard Mater* 261: 753-762.
<http://dx.doi.org/10.1016/j.jhazmat.2013.04.039>
- WHO. (1997). Environmental health criteria 189. Di-n-butyl phthalate [WHO EHC] (pp. GENEVA). (BIOSIS/98/08376 ISBN 9241571896). Geneva, Switzerland: United Nations Environmental

Programme, International Labour Organization.

<http://www.inchem.org/documents/ehc/ehc/ehc189.htm>

Wilson, NK; Chuang, JC; Lyu, C. (2001). Levels of persistent organic pollutants in several child day care centers. *J Expo Anal Environ Epidemiol* 11: 449-458.

<http://dx.doi.org/10.1038/sj.jea.7500190>

Wilson, NK; Chuang, JC; Lyu, C; Menton, R; Morgan, MK. (2003). Aggregate exposures of nine preschool children to persistent organic pollutants at day care and at home. *J Expo Anal Environ Epidemiol* 13: 187-202. <http://dx.doi.org/10.1038/sj.jea.7500270>

Wofford, HW; Wilsey, CD; Neff, GS; Giam, CS; Neff, JM. (1981). Bioaccumulation and metabolism of phthalate esters by oysters, brown shrimp, and sheepshead minnows. *Ecotoxicol Environ Saf* 5: 202-210. [http://dx.doi.org/10.1016/0147-6513\(81\)90035-x](http://dx.doi.org/10.1016/0147-6513(81)90035-x)

Wolfe, NL; Steen, WC; Burns, LA. (1980). Phthalate ester hydrolysis: Linear free energy relationships. *Chemosphere* 9: 403-408. [http://dx.doi.org/10.1016/0045-6535\(80\)90023-5](http://dx.doi.org/10.1016/0045-6535(80)90023-5)

Wormuth, M; Scheringer, M; Vollenweider, M; Hungerbuhler, K. (2006). What are the sources of exposure to eight frequently used phthalic acid esters in Europeans? *Risk Anal* 26: 803-824. <http://dx.doi.org/10.1111/j.1539-6924.2006.00770.x>

Wu, J; Ma, T; Zhou, Z; Yu, N, a; He, Z; Li, B; Shi, Y; Ma, D. (2019). Occurrence and fate of phthalate esters in wastewater treatment plants in Qingdao, China. *Hum Ecol Risk Assess* 25: 1547-1563. <http://dx.doi.org/10.1080/10807039.2018.1471341>

Wu, Q; Lam, JCW; Kwok, KY; Tsui, MMP; Lam, PKS. (2017). Occurrence and fate of endogenous steroid hormones, alkylphenol ethoxylates, bisphenol A and phthalates in municipal sewage treatment systems. *J Environ Sci* 61: 49-58. <http://dx.doi.org/10.1016/j.jes.2017.02.021>

Xiang, L; Wang, XD; Chen, XH; Mo, CH; Li, YW; Li, H; Cai, QY; Zhou, DM; Wong, MH; Li, QX. (2019). Sorption Mechanism, Kinetics, and Isotherms of Di- n-butyl Phthalate to Different Soil Particle-Size Fractions. *J Agric Food Chem* 67: 4734-4745. <http://dx.doi.org/10.1021/acs.jafc.8b06357>

Xie, Z; Ebinghaus, R; Temme, C; Caba, A; Ruck, W. (2005). Atmospheric concentrations and air-sea exchanges of phthalates in the North Sea (German Bight). *Atmos Environ* 39: 3209-3219. <http://dx.doi.org/10.1016/j.atmosenv.2005.02.021>

Xie, Z; Ebinghaus, R; Temme, C; Lohmann, R; Caba, A; Ruck, W. (2007). Occurrence and air-sea exchange of phthalates in the Arctic. *Environ Sci Technol* 41: 4555-4560. <http://dx.doi.org/10.1021/es0630240>

Xu, G; Li, F; Wang, Q. (2008). Occurrence and degradation characteristics of dibutyl phthalate (DBP) and di-(2-ethylhexyl) phthalate (DEHP) in typical agricultural soils of China. *Sci Total Environ* 393: 333-340. <http://dx.doi.org/10.1016/j.scitotenv.2008.01.001>

Yuan, S; Huang, I; Chang, B. (2010). Biodegradation of dibutyl phthalate and di-(2-ethylhexyl) phthalate and microbial community changes in mangrove sediment. *J Hazard Mater* 184: 826-831. <http://dx.doi.org/10.1016/j.jhazmat.2010.08.116>

Yuan, SY; Lin, YY; Chang, BV. (2011). Biodegradation of phthalate esters in polluted soil by using organic amendment. *J Environ Sci Health B* 46: 419-425. <http://dx.doi.org/10.1080/03601234.2011.572512>

Yuan, SY; Liu, C; Liao, CS; Chang, BV. (2002). Occurrence and microbial degradation of phthalate esters in Taiwan river sediments. *Chemosphere* 49: 1295-1299. [http://dx.doi.org/10.1016/s0045-6535\(02\)00495-2](http://dx.doi.org/10.1016/s0045-6535(02)00495-2)

Zeng, F; Cui, K; Xie, Z; Liu, M; Li, Y; Lin, Y; Zeng, Z; Li, F. (2008a). Occurrence of phthalate esters in water and sediment of urban lakes in a subtropical city, Guangzhou, South China. *Environ Int* 34: 372-380. <http://dx.doi.org/10.1016/j.envint.2007.09.002>

- Zeng, F; Cui, K; Xie, Z; Wu, L; Liu, M; Sun, G; Lin, Y; Luo, D; Zeng, Z. (2008b). Phthalate esters (PAEs): Emerging organic contaminants in agricultural soils in peri-urban areas around Guangzhou, China. *Environ Pollut* 156: 425-434. <http://dx.doi.org/10.1016/j.envpol.2008.01.045>
- Zeng, F; Cui, K; Xie, Z; Wu, L; Luo, D; Chen, L; Lin, Y; Liu, M; Sun, G. (2009). Distribution of phthalate esters in urban soils of subtropical city, Guangzhou, China. *J Hazard Mater* 164: 1171-1178. <http://dx.doi.org/10.1016/j.jhazmat.2008.09.029>
- Zeng, F; Lin, Y; Cui, K; Wen, J; Ma, Y; Chen, H; Zhu, F; Ma, Z; Zeng, Z. (2010). Atmospheric deposition of phthalate esters in a subtropical city. *Atmos Environ* 44: 834-840. <http://dx.doi.org/10.1016/j.atmosenv.2009.11.029>
- Zhang, D; Wu, L; Yao, J; Vogt, C; Richnow, HH. (2019). Carbon and hydrogen isotopic fractionation during abiotic hydrolysis and aerobic biodegradation of phthalate esters. *Sci Total Environ* 660: 559-566. <http://dx.doi.org/10.1016/j.scitotenv.2019.01.003>
- Zhao, H; Du, H; Feng, N; Xiang, L, ei; Li, Y; Li, H, ui; Cai, QY; Mo, C. (2016). Biodegradation of di-n-butylphthalate and phthalic acid by a novel *Providencia* sp 2D and its stimulation in a compost-amended soil. *Biol Fertil Soils* 52: 65-76. <http://dx.doi.org/10.1007/s00374-015-1054-8>
- Zhu, Q; Jia, J; Zhang, K; Zhang, H; Liao, C. (2019). Spatial distribution and mass loading of phthalate esters in wastewater treatment plants in China: An assessment of human exposure. *Sci Total Environ* 656: 862-869. <http://dx.doi.org/10.1016/j.scitotenv.2018.11.458>