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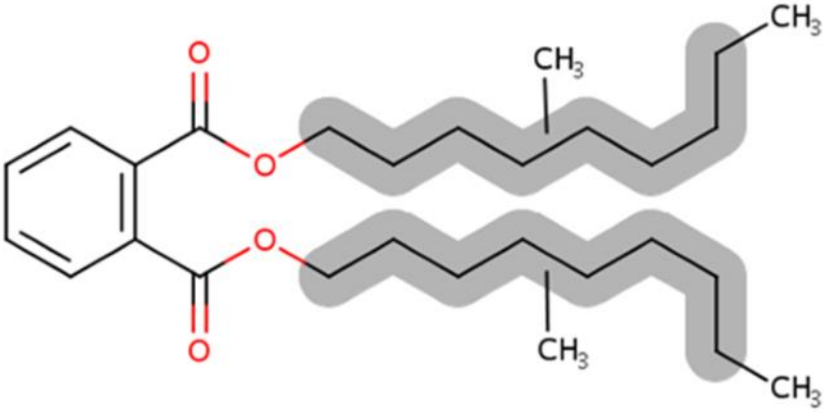
Pollution Prevention

# Fate and Transport Assessment for Diisodecyl Phthalate (DIDP)

## Technical Support Document for the Risk Evaluation

## CASRNs: 26761-40-0 and 68515-49-1



(Representative Structure)

*December 2024*

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# KEY ABBREVIATIONS AND ACRONYMS

| | |
|---|---|
| BAF | Bioaccumulation factor |
| BCF | Bioconcentration factor |
| CASRN | Chemical Abstract Service registry number |
| dw | Dry weight |
| DW | Drinking water |
| DWTP | Drinking water treatment plant |
| EPA | Environmental Protection Agency |
| FID | Flame ionization detector |
| FPD | Flame photometric detector |
| GC | Gas chromatography |
| ISO | International Organization for Standardization |
| $K_{OA}$ | Octanol-air partition coefficient |
| $K_{OC}$ | Organic carbon-water partition coefficient |
| $K_{OW}$ | Octanol-water partition coefficient |
| L/d | Liters per day |
| lw | Lipid weight |
| M | Molarity (mol/L = moles per liter) |
| mL/min | Milliliters per minute |
| mM | Millimolar |
| mg/kg | Milligrams per Kilogram |
| mg/L | Milligrams per Liter |
| $mg/m^3$ | Milligrams per cubic meter |
| MS | Mass Spectrometry |
| n | Sample Size |
| ND | Non-detection |
| ng/L | Nanograms per liter |
| nm | Nanometers |
| NR | Not reported |
| OECD | Organisation for Economic Co-operation and Development |
| ·OH | Hydroxyl radical |
| OPE | Organophosphate ester |
| pg/L | Picograms per liter |
| ppm | parts per million |
| SIM | Selected ion monitoring |
| µg/L or µg/mL | Micrograms per liter or per milliliter |
| UV (UV-Vis) | Ultra violet (visible) light |
| ww | Wet weight |
| WWTP | Wastewater treatment plant |

# SUMMARY

This technical document is in support of the TSCA *Risk Evaluation for Diisodecyl Phthalate (DIDP)* (U.S. EPA, 2024c). DIDP is a common chemical name for the category of chemical substances that includes the following substances: 1,2-benzenedicarboxylic acid, 1,2-diisodecyl ester (CASRN 26761-40-0) and 1,2-benzenedicarboxylic acid, di-C9-11-branched alkyl esters, C10-rich (CASRN 68515-49-1). Both CASRNs contain mainly C10 dialkyl phthalate esters. See the risk evaluation for a complete list of all the technical support documents for DIDP.

In this document, EPA evaluated the reasonably available information to characterize the environmental fate and transport of DIDP, the key points are summarized below. Given the consistent results from numerous high-quality studies, there is robust evidence that DIDP:

- Is expected to undergo significant direct photolysis and will rapidly degrade in the atmosphere ($t_{1/2}$ = 0.32 days) (Section 3.3).
- Is expected to degrade rapidly via direct and indirect photolysis (Section 3.3).
- Is not expected to appreciably hydrolyze under environmental conditions (Section 3.2).
- Is expected to have environmental biodegradation half-life in aerobic environments on the order of days to weeks (Section 3.1).
- Is not expected to be subject to long range transport.
- Is expected to transform in the environment and via biotic and abiotic processes to form monoisodecyl phthalate, isodecanol, and phthalic acid (Section 3).
- Is expected to show strong affinity and sorption potential for organic carbon in soil and sediment (Sections 5.2.2, 5.3.2)
- Will be removed at rates greater than 93 percent in conventional wastewater treatment systems via sorption to sludge (Section 6.2)
- When released to air, will not likely exist in gaseous phase, but will show strong affinity for adsorption to particulate matter (Sections 4 and 5).
- Is likely to be found in, and accumulate in, indoor dust (Section 5).

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

- Is not expected to biodegrade under anoxic conditions and may be persistent in anaerobic soils and sediments (Sections 3.1, 5.2.2, 5.3.2).
- Is not bioaccumulative in fish in the water column (Section 7).
- Is expected to be partially 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 (Section 6.3).

# 1 INTRODUCTION

The following sections of the fate and transport analysis of DIDP present the general fate and transport characteristics of DIDP. Fate in each specific compartment of environmental media (soil, sediment, surface water, groundwater, atmospheric and indoor air) are discussed, as well as the fate endpoints (*e.g.*, biodegradation, transformation, hydrolysis, photolysis, sorption) that contribute to the fate and transport of DIDP within, and through, various environmental media.

# 2 APPROACH AND METHODOLOGY

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 $K_{OC}$)—are parameters used in the current risk evaluation. In assessing the environmental fate and transport of DIDP, EPA considered the full range of results from the available highest quality data sources obtained during systematic review. Information on the full extracted data set is available in the *Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport for Diisodecyl Phthalate (DIDP)* (U.S. EPA, 2024a). Other fate estimates were based on modeling results from EPI Suite™ (U.S. EPA, 2012), a predictive tool for physical and chemical properties and environmental fate estimation.

Table 2-1 provides identified environmental fate data that EPA considered while assessing the fate of DIDP and were updated after publication of *Final Scope of the Risk Evaluation for Di-isodecyl Phthalate (DIDP) CASRN 26761-40-0 / 68515-49-1* (U.S. EPA, 2021) with additional information identified through the systematic review process.

**Table 2-1. Summary of DIDP's Environmental Fate Information**

| Parameter | Value | Source(s) |
|---|---|---|
| Octanol:Water (Log $K_{OW}$) | 10.21 | (U.S. EPA, 2017) |
| Organic Carbon:Water (Log $K_{OC}$) | 5.04–5.78 | (Analytical Bio-Chemistry Labs, 1991) |
| Adsorption Coefficient (Log $K_d$) | 2.22–3.60 | (Mackay et al., 2006; Williams et al., 1995) |
| Octanol:Air (Log $K_{OA}$) | 13.034 (estimated) | (U.S. EPA, 2017) |
| Air:Water (Log $K_{AW}$) | −2.824 (estimated) | (U.S. EPA, 2017) |
| Aerobic primary biodegradation in water | 39% at 9 days,<br>53% at 21 days<br>>99% at 28 days | (ECJRC, 2003a) |
| Aerobic ready biodegradation in water | 88% to >99% at 28 days | (ECJRC, 2003a; SRC, 1983) |
| Aerobic ultimate biodegradation in water | 56.2% at 28 days | (SRC, 1983) |
| Anaerobic biodegradation in sediment | 0% after 100 days by $CH_4$ | (Ejlertsson et al., 1996) |
| Hydrolysis | 125 days at pH 8 and 25 ºC, and 3.4 years at pH 7 and 25 ºC | (U.S. EPA, 2017) |
| Photolysis | $t_{1/2}$ (air) = 4.7 to 7.68 hours | (U.S. EPA, 2017) |
| Environmental degradation half-lives (selected values for modeling) | 7.68 hours (air)<br>10 days (water)<br>20 days (soil)<br>90 days (sediment) | (U.S. EPA, 2017) |
| WWTP removal | >94% | (U.S. EPA, 2017) |

| Parameter | Value | Source(s) |
|---|---|---|
| Aquatic bioconcentration factor (BCF) | <14.4 L/kg wet weight (Experimental; Fish, *Cyprinus carpio*)<br>1.3 L/kg wet weight (upper trophic Arnot-Gobas estimation) | (U.S. EPA, 2017; ECJRC, 2003b) |
| Aquatic bioaccumulation factor (BAF) | 9.9 L/kg wet weight (upper trophic Arnot-Gobas estimation) | (U.S. EPA, 2017) |
| Aquatic food web magnification factor (FWMF) | 0.44<br>(Experimental; 18 marine species) | (Mackintosh et al., 2004) |
| Terrestrial bioconcentration factor (BCF) | 0.01–0.02<br>Experimental; earthworms (*Eisenia fetida*) | (ECJRC, 2003b) |

## 2.1 EPI Suite™ Model Inputs and Settings

The approach described by Mackay et al. (1996) using the Level III Fugacity model in EPI Suite™ (LEV3EPI™) was used in the fate assessment of DIDP. 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, 2012). A Tier II analysis involves reviewing environmental release information for DIDP to determine if a specific media evaluation is needed. Environmental release data for DIDP was not available from the Toxics Release Inventory (TRI) or Discharge Monitoring Reports (DMRs); however, between 50,000 and 500,000 tons of CASRN 26761-40-0 and between 100,000,000 and 1,000,000,000 tons of CASRN 68515-49-1 were produced annually from 2016 to 2019 for use in commercial products, chemical substances or mixtures sold to consumers, or at industrial sites according to production data from the Chemical Data Reporting (CDR) 2020 reporting period. Environmental release information is useful for fugacity modeling because the emission rates will predict a real-time percent mass distribution for each environmental medium. Environmental degradation half-lives were taken from high and medium quality studies that were identified through systematic review. Based on DIDP's observed and calculated environmental half-lives, partitioning characteristics, and the results of Level III Fugacity modeling (see Figure 4-1), DIDP is expected to partition primarily to soil and sediment, regardless of the compartment of the environmental release. The LEV3EPI™ results were consistent with environmental monitoring data. Further discussion of DIDP partitioning can be found in Section 4.

## 2.2 Evidence Integration for Fate and Transport Properties of DIDP

Sources identified in the systematic review process for DIDP were only considered if they received a data quality ranking of medium or high. Data sources were considered as described in the media-specific sections below, and in Section 9, the weight of scientific evidence for DIDP.

# 3 TRANSFORMATION PROCESSES

DIDP will undergo both direct and indirect photolysis to form the monoester form (monoisodecyl phthalate) and isodecanol through carboxylic acid ester hydrolysis (https://qed.epa.gov/cts/about/cts/). DIDP has been shown to biodegrade under aerobic conditions when exposed to *Baccillus* sp. into the monoester (monoisodecyl phthalate), isodecanol, and further to phthalic acid (Park et al., 2009). Biodegradation pathways for the phthalates consist of primary biodegradation from phthalate diesters to phthalate monoesters, then to phthalic acid, and ultimately biodegradation of phthalic acid to form $CO_2$ and/or $CH_4$ (Huang et al., 2013). The monoisodecyl phthalate is both more soluble and more bioavailable than DIDP. It is also expected to undergo biodegradation more rapidly than the diester form. EPA considered DIDP transformation products and degradants qualitatively but due to their lack of persistence, the Agency does do not expect them to substantially contribute to risk. Thus, EPA is not considering them further in this risk evaluation. Both biotic and abiotic routes of degradation for DIDP are described in the sections below.

## 3.1 Biodegradation

DIDP can be considered readily biodegradable under most aquatic and terrestrial environments. The EPA identified three medium and four high quality sources reporting the biodegradation of DIDP in water, activated sludge and sediments under aerobic and anaerobic conditions. DIDP is considered an isomeric mixture, therefore, it can be assumed based of the study results shown in Table 3-1 that certain components of DIDP biodegrade more readily than others (ECJRC, 2003a). When in water under aerobic conditions DIDP has been reported to achieve greater than 99 percent primary biodegradation (ECJRC, 2003a), 88 to greater than 99 percent ready biodegradation (ECJRC, 2003a; SRC, 1983) and 56.2 to 74 percent ultimate biodegradation in 28 days (EC/HC, 2015; SRC, 1983). Similarly, the biodegradation of DIDP in activated sludge under aerobic conditions have been reported to be 90 percent in 9 days, 97 percent in 12 days and a half-life of 0.77 days (O'Grady et al., 1985; SRC, 1984). The available information suggests that DIDP is expected to have very low biodegradation potential under low oxygen conditions and could remain longer in subsurface sediments and soils (ECJRC, 2003a; Ejlertsson et al., 1996). However, due to the predicted overall environmental half-life of 35 days, DIDP is not expected to be persistent in the atmosphere, aquatic, or terrestrial environments.

**Table 3-1. Summary of DIDP's Biodegradation Information**

| Environmental Conditions | Degradation Value | Half-Life (days) | Reference | Overall Data Quality Ranking |
|---|---|---|---|---|
| Aerobic ultimate biodegradation in water | 42% at 21 days,<br>74% at 28 days,<br>74% at 47 days | N.D. | (EC/HC, 2015) | Medium |
| Aerobic ready biodegradability in water | 88% at 28 days ($CO_2$ evolution)<br>42% at 21 days (BOD)<br>67.1% at 28 days (Respirometry) | N.D. | (ECJRC, 2003a) | Medium |
| Aerobic inherent biodegradability in water | 39% at 9 days,<br>53% at 21 days<br>>99% at 28 days<br>(Primary biodegradation) | 9.6 | (ECJRC, 2003a) | Medium |

| Environmental Conditions | Degradation Value | Half-Life (days) | Reference | Overall Data Quality Ranking |
|---|---|---|---|---|
| Aerobic ready biodegradability in water (AS inoculum) | >99% at 28 days | 9.6 | (SRC, 1983) | High |
| Aerobic ultimate biodegradability in water (AS inoculum) | 56.2% at 28 days | <28 | (SRC, 1983) | High |
| Aerobic biodegradation in water | Water: 0% after 20 days in unacclimated microorganisms in water<br>7% after 20 days in the presence of acclimated microorganisms from an industrial treatment plant for petrochemical waste. | N.D. | (Union Carbide, 1974) | Medium |
| Aerobic ready biodegradability in activated sludge | 68% at 24 hours<br>90% at 9 days | N.D. | (O'Grady et al., 1985) | High |
| Aerobic inherent biodegradability in activated sludge | 97% in 12 days | 0.77 | (SRC, 1984) | High |
| Biodegradation Anaerobic in Sediment | 0% after 100 days by $CH_4$ evolution and no transformation reported by GC analysis for methane and test substance concentrations. Municipal solid waste anaerobic microflora inoculum. | N.D. | (Ejlertsson et al., 1996) | High |
| Biodegradation Anaerobic in Sediment | 20 and 50% after 244 and 296 days, respectively, in 2 of 9 samples<br>20, 20 and 40% after 127, 127 and 358 days, respectively, in 3 of 20 samples<br>Very low apparent biodegradation potential of DIDP under anaerobic conditions. | N.D. | (ECJRC, 2003a) | Medium |
| AS = Activated sludge | | | | |

## 3.2 Hydrolysis

Traditionally accepted methods of testing for abiotic hydrolysis of DIDP (OECD Guideline Test 111) are not viable due to the aqueous solubility of DIDP (ECJRC, 2003a). Therefore, hydrolysis rates of DIDP are difficult to accurately measure experimentally (ECJRC, 2003a). EPI Suite$^{TM}$ was utilized to estimate the hydrolysis half-lives of DIDP at 125 days at pH 8 and 25 ºC, and 3.4 years at pH 7 and 25 ºC (U.S. EPA, 2017) indicating that hydrolysis is a possible degradation pathway of DIDP under more caustic conditions. When compared to other degradation pathways, hydrolysis it is not expected to be a significant degradation pathway under typical environmental conditions. However, at higher temperatures, variations from typical environmental pH, and chemical catalysts present in the deeper anoxic zones of landfills may be favorable to the degradation of DIDP via hydrolysis (Huang et al., 2013). This is discussed further in Section 5.3.3.

## 3.3 Photolysis

DIDP contains chromophores that absorb light at wavelengths greater than 290nm (HSDB, 2015), therefore, direct photodegradation is a relevant degradation pathway for DIDP released to air. Furthermore, indirect photodegradation half-lives of 0.32 days (based on ·OH reaction rate constant of $2.6\times10^{-11}$ $cm^3$ /mol·second with $1\times10^6$ $OH/cm^3$) was observed in atmospheric air (Peterson and Staples, 2003). Modelled indirect photodegradation half-lives indicated a slightly more rapid rate of degradation, estimating a half-life of 4.7 hours (·OH rate constant of $2.7\times10^{-11}$ $cm^3$ /molecule-second and a 12-hour day with $1.5\times10^6$ $OH/cm^3$) (U.S. EPA, 2017).

# 4 PARTITIONING

Environmental release data for DIDP was not available from the Toxics Release Inventory (TRI) or Discharge Monitoring Reports (DMRs), therefore DIDP releases to the environment could not be estimated. The approach described by (Mackay et al., 1996) using the Level III Fugacity model in EPI Suite$^{TM}$ (LEV3EPI$^{TM}$) 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, 2012). DIDP's physical and chemical properties were taken directly from Section 2.1 of the *Data Quality Evaluation and Data Extraction Information for Physical and Chemical Properties for Diisodecyl Phthalate (DIDP)* (U.S. EPA, 2024b). Environmental release information is useful for fugacity modeling because the emission rates will predict a real-time percent distribution for each medium. Environmental degradation half-live in water of 10 days (ECJRC, 2003a) was taken from a medium quality study that were identified through systematic review to reduce levels of uncertainties. Environmental degradation half-lives of 0.32 days in air (based on a global average OH radical concentrations of $10^6$ molecule/cm$^3$ in air) (Mackay et al., 2006), 20 days in soil (double the half-life in water), and 90 days in sediment (nine times the half-life in water) as recommended for EPIWIN estimations (U.S. EPA, 2012). Based on DIDP's environmental half-lives, partitioning characteristics, and the results of Level III Fugacity modeling, DIDP is expected to be found predominantly in water, soil, and sediment (Figure 4-1). The LEV3EPI$^{TM}$ results were consistent with environmental monitoring data. Further discussion of DIDP partitioning can be found in Sections 5.1, 5.2, and 5.3.

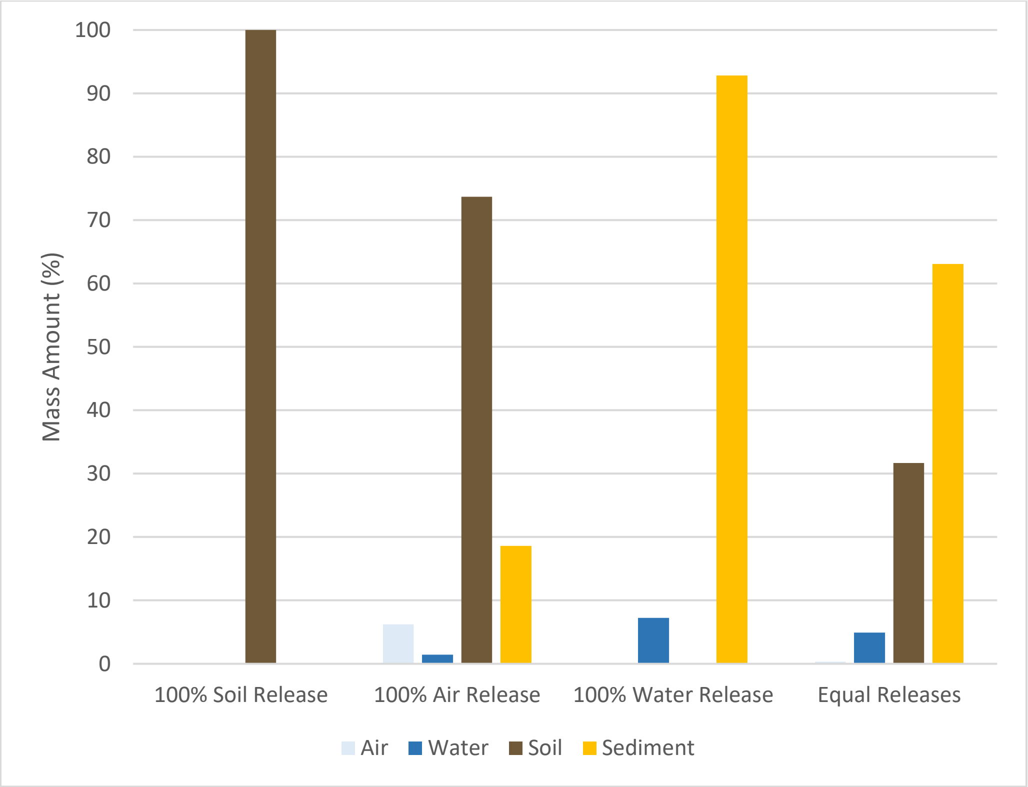

**Figure 4-1. EPI Suite$^{TM}$ Level III Fugacity Modeling Graphical Result for DIDP**

# 5 MEDIA ASSESSMENTS

DIDP has been reported to be present in the atmosphere, aquatic environments, and terrestrial environments. Once in the air, DIDP will be most predominant in the organic matter present in airborne particles and expected to have a short half-life in the atmosphere. Based on the physical and chemical properties, DIDP is very likely to partition to house dust and airborne particles and is expected to have a longer half-life compared to ambient (outdoor) air. DIDP present in surface water is expected to mostly partition to aquatic sediments with an expected biodegradation half-life between 14 and 26 days. In terrestrial environments DIDP has the potential to be present in soils and ground water environments. In soils, DIDP is expected to be released via air deposition and land application of biosolids, it is expected to have a half-life of 28 to 52 days and have low bioaccumulation potential and biomagnification potential in terrestrial organisms. DIDP may enter groundwater via wastewater effluent with an expected half-life of 14 to 56 days, and not likely to be persistent in most groundwater/subsurface environments.

## 5.1 Air and Atmosphere

DIDP is a liquid at environmental temperatures with a melting point of −50°C (Haynes, 2014) and a vapor pressure of $5.28 \times 10^{-7}$ mm Hg at 25°C (NLM, 2020). Based on its physical and chemical properties and short half-life in the atmosphere, $t_{1/2} = 7.6$ hours (Mackay et al., 2006), DIDP was assumed to not be persistent in the air. The AEROWIN$^{TM}$ module in EPI Suite$^{TM}$ estimates that a large fraction (75 to 80 percent) of DIDP could be sorbed to airborne particulates and these particulates may be resistant to atmospheric oxidation. There is very limited available monitoring information on DIDP in ambient air or indoor air; however, studies have detected DIDP in settled house dust (Kubwabo et al., 2013; Wang et al., 2013; Abb et al., 2009).

### 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). There is limited information on the presence of DIDP in the atmosphere and most of the studies reported diethylhexyl phthalate (DEHP) to be the predominant phthalate esters in the environment. Despite the limited information on the presence of DIDP in the atmosphere, similar trends to those reported for DEHP could be expected based on their similar vapor pressure (ECHA, 2013). Once in indoor air, DIDP is expected to partition to organic carbon present on indoor airborne particles at concentrations higher than in vapor phase (ECJRC, 2003a). Under indoor environments, DIDP is expected to be more persistent in indoor air than in ambient (outdoor) air due to the lack of natural chemical removal processes, such as solar photochemical degradation.

The available information suggests that the concentration of DIDP in dust in indoor environments is likely to be higher than outdoor dust, and to be associated with the indoor presence of phthalate containing articles and the proximity to the facilities producing them (Kubwabo et al., 2013; Wang et al., 2013; Abb et al., 2009). Kubwabo et al. (2013) monitored the presence of 17 phthalate compounds in vacuum dust samples collected in 126 urban single-family homes. The study reported that DEHP, DIDP, and DINP (diisononyl phthalate) were detected in all the collected dust samples accounting for 88 percent of the median total concentration of phthalates in dust. Wang et al. (2013) evaluated the presence of phthalates in dust samples collected from indoor and outdoor settings in two major Chinese cities. The study reported that the total phthalates concentration of the collected indoor dust samples were 3.4 to 5.9 times higher than those collected outdoors. The aggregate concentration of DEHP, DINP, and DIDP in indoor dust samples accounted for 91 to 94 percent of the total phthalate's concentration. The study revealed that the aggregate concentration of phthalates was higher in the commercial and industrial areas with heavy production of textiles, costumes, and toys. Abb et al. (2009)

evaluated the presence of phthalates in indoor dust samples collected from 30 households in Germany. The study revealed the presence of DEHP, DINP and DIDP in all the collected samples. Samples collected from households containing consumer products with a high percentage of plastics (greater than 50 percent) resulted with higher aggregate concentration of phthalates in dust. The aggregate concentration of DEHP, DINP and DIDP accounted for 87 percent of the total phthalate concentration in dust.

Similarly, recent studies monitoring the presence of phthalates in dust from USA households have revealed DEHP and DINP to be detected in 96 to 100 percent of the collected samples (Hammel et al., 2019; Dodson et al., 2017). Hammel et al. (2019) and Dodson et al. (2017) reported the presence of phthalate esters on indoor air and dust samples collected in USA homes. Hammel et al. (2019) reported that DEHP and DINP accounted for close to 83 percent of the total concentration of phthalates found in indoor dust. Dodson et al. (2017) evaluated the presence of phthalate esters in air samples of USA homes before and after occupancy reporting 97 percent and 31 percent increase in indoor air concentrations for DEHP and DINP, respectively after occupancy due to daily anthropogenic activities that might introduce phthalate containing products into indoor settings. Similar trends could be expected for DIDP as it might be present in household construction materials or consumer products, where it could potentially result in its increased detection in indoor dust.

## 5.2 Aquatic Environments

### 5.2.1 Surface Water

DIDP is expected to be released to surface water via industrial and municipal wastewater treatment plant effluent, surface water runoff, and, to a lesser degree, atmospheric deposition. DIDP and other phthalate esters have been detected in surface waters worldwide (Wen et al., 2018). The principal properties governing the fate and transport of DIDP in surface water are water solubility, organic carbon partitioning coefficients, and volatility. Due to the Henry's law constant ($2.132 \times 10^{-4}$ atm·$m^3$/mol at 25 °C) of DIDP, volatilization from surface water is not expected to be a significant source of loss of DIDP from surface water. A partitioning analysis of DIDP released to the environment is described in Section 4 above. The analysis estimates that during releases to surface water bodies, >93 percent of DIDP released to surface water will partition to both suspended and benthic sediments. DIDP has a water solubility of 0.00017 mg/L but is likely to form a colloidal suspension and may be detected in surface water at higher concentrations (EC/HC, 2015). Concentrations of DIDP above the aqueous solubility of 0.00017 mg/L are not uncommon in monitoring studies proximal to releases of DIDP to surface water (Wen et al., 2018).

Biodegradation of DIDP in surface water is rapid, but DIDP does not pass a 10-day ready biodegradability test. DIDP is predicted to biodegrade in surface water with a predicted half-life of 14 to 26 days (U.S. EPA, 2012).

### 5.2.2 Sediments

Based on the water solubility (0.00017 mg/L) and affinity to sorb to organic matter (log $K_{OC}$ = 5.04–5.78), DIDP will partition mostly to the organic matter present in soils and sediment when released into the environment. Once in water, the Level III Fugacity Model in EPI Suite™ (U.S. EPA, 2017) predicts that greater than 93 percent of the DIDP will partition to and remain in sediments (Section 4). The available information suggests that DIDP could remain longer in subsurface sediments and soils but is not expected to be persistent in the atmosphere as well as aquatic and terrestrial environments with a predicted overall environmental half-life of 35 days (Section 3.1). Due to the strong sorption to soils, DIDP will be expected to be found predominantly in sediments near point sources, with a decreasing trend in sediment concentrations downstream. This is consistent with monitoring information from Sweden, and Korea.

One study reported the presence of DIDP in one sediment sample near point sources in Sweeden (Parkman and Remberg, 1995). The presence of DIDP has been documented in urban sediments at higher concentrations than DINP and DEHP (Cousins et al., 2007). In a similar study, (Kim et al., 2021) evaluated the presence of plasticizers in sediments from industrialized bays of Korea. DIDP was detected in all surface sediment samples. The study revealed a gradual decreasing trend in the overall concentration of phthalates toward the outer region of the bay, distally from industrial activities. The findings of this study suggest industrial activities to be the major contributor of phthalates in sediments within the area.

## 5.3 Terrestrial Environments

### 5.3.1 Soil

DIDP is expected to be deposited to soil via two primary routes: application of biosolids and sewage sludge in agricultural applications or sludge drying applications; and atmospheric deposition. DIDP has a Henry's Law constant ($2.132\times10^{-4}$ atm·$m^3$/mol at 25 °C) and is not likely volatilize from soils.

DIDP shows an affinity for sorption to soil and its organic constituents, log $K_{OC}$ = 5.04 to 5.78, $K_d$ of $1.66\times10^2$ to $3.97\times10^3$ (Mackay et al., 2006; Williams et al., 1995; Analytical Bio-Chemistry Labs, 1991), and an estimated log $K_{OW}$ of 10.21 (U.S. EPA, 2017). Given that these properties indicate the likelihood of strong sorption to organic carbon present in soil, DIDP is expected to be immobile in soil environments.

Under aerobic conditions, DIDP is expected to have a half-life in soil of 28 to 52 days (SRC, 1983). This aerobic biodegradation half-life for soil was estimated by doubling the experimentally derived half-life of DIDP in water (as recommended for most satisfactory EPIWIN estimations, see Section 4) as no biodegradation data for DIDP in soil was identified in the systematic review process (Mackay et al., 2006). However, under anerobic conditions that may be present in some soil profiles, there is very little evidence to support that DIDP appreciably biodegrades (ECJRC, 2003a; Ejlertsson et al., 1996).

Under anaerobic conditions in soil, DIDP is assumed to be persistent, and continuous exposure is likely. One study found that 0 percent degradation had occurred under anaerobic conditions after 100 days by $CH_4$ evolution and no transformation reported by GC analysis for methane and test substance concentrations using a municipal solid waste anaerobic microflora inoculum (Ejlertsson et al., 1996). Furthermore, another study identified degradation rates of 20 and 50 percent after 244 and 296 days, respectively, in 2 of 9 sample points (ECJRC, 2003a). Another experiment in the same study showed 20 percent degradation in 2 sample points after 127 days and 40 percent degradation in 1 sample point in 358 days. Degradation of DIDP in this study was only observed in 3 of 20 samples. These results show low apparent biodegradation potential of DIDP under anaerobic conditions. There is sufficient evidence to conclude that DIDP is likely not persistent in soil as long as the rate of release does not exceed the rate at which biodegradation can occur, but continuous exposure to DIDP in soil proximal to points of releases may be possible if the rate of releases exceeds the rate of biodegradation under aerobic conditions.

### 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 meet 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 physical and chemical characteristics which indicate low water solubility and high sorption potential are expected to be sorbed to suspended solids and efficiently removed from wastewater via sorption to sewage sludge and biosolids.

There is limited information about the presence and biodegradation of DIDP in biosolids. As described in Section 6.2, DIDP in wastewater has been reported to be mainly removed by particle sorption and retained in the sewage sludge. In general, greater than 93 percent of the DIDP present in wastewater is expected to be accumulated in sewage sludge and discharged into biosolids. Once in biosolids, DIDP is expected to have a biodegradation half-life and could be transferred to soil during land application. As described in Section 3, DIDP is expected to have a half-life of 28 to 52 days in soils (SRC, 1983) and be more persistent in soil profiles with anaerobic conditions (ECJRC, 2003a; Ejlertsson et al., 1996). However, based on the water solubility and hydrophobicity, DIDP will have low bioaccumulation, biomagnification appears to be of minimal concern. Additionally, terrestrial species have been reported to have the capacity to metabolize phthalate substances (Bradlee and Thomas, 2003; Gobas et al., 2003; Barron et al., 1995) and DIDP is expected to have low bioaccumulation potential and biomagnification potential in terrestrial organisms (Section 7).

### 5.3.3 Landfills

For the purpose of this assessment, landfills will be considered to be divided into two zones: an "upper-landfill" zone, with normal environmental temperatures and pressures, where biotic processes are the predominant route of degradation for DIDP, and a "lower-landfill" zone where elevated temperatures and pressures exist, and abiotic degradation is the predominant route of degradation. In the upper-landfill zone where oxygen may 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 DIDP. At temperatures at and above 60 ºC, biotic processes are significantly inhibited, and are likely to be completely irrelevant at 70 ºC (Huang et al., 2013).

DIDP may be deposited in landfills continually and in high amounts from the disposal of consumer products containing DIDP. Thus, small concentrations of DIDP are likely to be present in landfill leachate. DIDP is likely to be persistent in landfills due to the apparent lack of anaerobic biodegradation and unfavorable conditions for biodegradation in lower-landfills. Some aerobic biodegradation in upper-landfills may occur. In lower-landfills, there is some evidence to support that hydrolysis may be the main route of abiotic degradation of phthalate esters (Huang et al., 2013). However, more evidence is needed to support this conclusion.

The persistence of DIDP in landfills is likely to be mitigated by the lack of risk for migration to groundwater due to the water solubility of DIDP (0.00017 mg/L), and its affinity for sorption to organic carbon (log $K_{OC}$: 5.04 to 5.78) prevalent in landfills. Although DIDP is likely to be present in landfill leachate, it is unlikely to migrate to, or be mobile in groundwater proximal to landfills, and would not be expected to be transported distally from landfills via groundwater.

### 5.3.4 Groundwater

There are several likely sources of DIDP in groundwater: wastewater effluents, land applied biosolids and landfill leachates, which are discussed in Sections 5.3.3, 5.3.4, and 6.2. In environments where DIDP 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.

Given the strong affinity of DIDP to adsorb to organic matter present in soils and sediments (log $K_{OC}$ 5.04 to 5.78, and $K_d$ of $1.66 \times 10^2$ to $3.97 \times 10^3$) (U.S. EPA, 2012; Mackay et al., 2006; Williams et al., 1995), DIDP is expected to be immobile in soil and groundwater environments. Furthermore, due to the insoluble nature of DIDP (0.00017 mg/L), migration of DIDP to groundwater from these sources is unlikely. In instances where DIDP 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 DIDP in aerobic environments (half-life ~14 to 26 days in water and ~28 to 56 days in soil) (ECJRC, 2003a), DIDP is not likely to be persistent in groundwater/subsurface environments unless anoxic conditions exist.

# 6 PERSISTENCE POTENTIAL OF DIDP

DIDP is not expected to be persistent in the environment, as it is expected to degrade rapidly under most environmental conditions, with delayed biodegradation in low-oxygen media. In the atmosphere, DIDP is unlikely to remain for long periods of time as its expected to undergo photolytic degradation through reaction with atmospheric hydroxyl radicals, with estimated half-lives of 7.68 hours. DIDP is predicted to hydrolyze slowly at ambient temperature, but it is not expected to persist in aquatic media as it undergoes rapid aerobic biodegradation (Section 5.2.1). DIDP has the potential to remain for longer periods of time in soil and sediments, but due to the inherent hydrophobicity (log $K_{OW}$: 10.21) and sorption potential (log $K_{OC}$: 5.04 to 5.78) DIDP is expected to have limited bioavailability for uptake. Using the Level III Fugacity model in EPI Suite$^{TM}$ (LEV3EPITM) (Section 4), DIDP's overall environmental half-life was estimated to be approximately 35 days (U.S. EPA, 2012). Therefore, DIDP 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. Currently there is no information available on the DRE of DIDP. However, the DEHP annual releases from a Danish waste incineration facility were estimated to be 9 percent to air and 91 percent to municipal land fill (ECB, 2008). These results suggest that during incineration more than 90 percent of DIDP is expected to be released to landfills and the remaining fraction released to air. Based inherent hydrophobicity and sorption potential, DIDP released to landfills is expected to partition into the landfills organic matter. Similarly, DIDP released to air is expected to partition to soil and sediments as described in Section 4. In addition, DIDP in sediments and soils is expected to have limited bioavailability for uptake, and biodegrade within days (Call et al., 2001). Lastly, DIDP in air is not expected to be persistent as it is expected to degrade within hours (Mackay et al., 2006).

## 6.2 Removal in Wastewater Treatment

Wastewater treatment is performed to remove contaminants from wastewater using physical, biological, and chemical processes. A chemical may be removed from a treatment plant via sorption to sludge, stripping to air, biodegradation, or hydrolysis. These removal mechanisms are determined or influenced by a chemical substance's physical-chemical properties and structure. 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. After undergoing primary treatment, the wastewater undergoes a secondary treatment. Secondary treatment processes 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 such as tertiary treatment is utilized to further clean water for additional protection using advanced treatment techniques (*e.g.*, ozonation, chlorination, disinfection).

Limited information is available in the fate and transport of DIDP in wastewater treatment systems. The EPA identified two high quality sources reporting the removal of DIDP in wastewater treatment systems employing aerobic and anaerobic processes. One study reported 98.9 percent DIDP removal efficiencies in a municipal wastewater treatment facility in France, employing a combined decantation and activated sludge tank (Tran et al., 2014). The study reported 83.7 to 86.7 percent of DIDP to be distributed into the wastewater suspended sediments. Phthalates esters with long carbon chains and log $K_{OW}$ greater than 5, such as DIDP, DINP and DEHP, were reported to be mainly retained in the sewage sludge accounting for up to 99% of their removal from wastewater. This finding is supported by STPWIN$^{TM}$, an EPI Suite$^{TM}$ module that estimates chemical removal in sewage treatment plants. The model predicts greater than 93 percent removal of DIDP in wastewater by sorption to sludge (U.S. EPA, 2012).

In addition to the expected removal via sorption, the rapid aerobic biodegradation of DIDP could potentially contribute to additional removal during wastewater treatment. In a recent study exploring the treatment of wastewater final solids, Armstrong (2018) reported up to 65.8 and 85.9 percent decrease in concentrations of DIDP and DINP from aerobic digestion effluents, respectively. The study reported no significant change in the concentration of DIDP and DINP in the final solids. These findings suggest aerobic biodegradation to partially contribute to overall removal of DIDP and DINP from wastewater. In addition, the same study reported anaerobic solid digestion to be not effective in the removal of DIDP (Armstrong et al., 2018). In general, the available information suggest that aerobic biodegradation processes have the potential to partially contribute to the overall removal of DIDP from wastewater.

Overall, DIDP has a predicted log $K_{OW}$ of 10.21 and remains in suspended solids and efficiently removed from wastewater via accumulation in sewage sludge (Tran et al., 2014), partially removed with membrane bio reactor systems under aerobic wastewater treatment (Armstrong et al., 2018), and ineffectively removed under anaerobic solids digestion conditions (Armstrong et al., 2018). Biodegradation and air stripping are not expected to be significant wastewater removal processes. Therefore, greater than 93 percent of the DIDP present in wastewater is expected to be accumulated in sewage sludge and released with biosolids disposal or application, with the remaining fraction sorbed to suspended solids in the wastewater treatment effluent and discharged with surface water (Tran et al., 2014; U.S. EPA, 2012).

## 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 then flows to a treatment plant where it undergoes a series of water treatment steps before being dispersed to homes and communities. In the United States, public water systems often use conventional treatment processes that include coagulation, flocculation, sedimentation, filtration, and disinfection, as required by law and regulations.

Very limited information is available on the removal of DIDP in drinking water treatment plants. No data was identified by the EPA for DIDP in drinking water. Based on the low water solubility and log $K_{OW}$, DIDP in water it is expected to mainly partition to suspended solids present in water. This is supported by the Level III Fugacity model in EPI Suite$^{TM}$ (Section 4) which predicts 92.8 percent of DIDP in water to partition to sediments (U.S. EPA, 2012). The available information suggest that the use of flocculants and filtering media could potentially help remove DIDP during drinking water treatment by sorption into suspended organic matter. Recent studies exploring the presence of phthalate esters in drinking water sources and their removal in a conventional drinking water treatment plant in China have reported to remove DEHP, DIBP, and DBP by 58.7, 47, and 65 percent from the treated drinking water effluent, respectively (Kong et al., 2017). In addition, Yang (2014) documented the change in phthalate esters concentrations during the storage stage of treated drinking water in the presence of residual chlorine resulting in 78 to 86, −2 to 44, and 14 to 21 percent loss of DINP, DEHP, and DBP in 48 hours, respectively. Several studies have reported chlorine species to have the potential to partially reduce the concentration of organic chemicals in water (Lee and von Gunten, 2010). These findings suggest that conventional drinking water treatment systems may have the potential to partially remove DIDP present in drinking water sources depending on via sorption to suspended organic matter and filtering media and the use of disinfection technologies.

# 7 BIOACCUMULATION POTENTIAL OF DIDP

The presence of DIDP in several aquatic species suggest that the substance is bioavailable in aquatic environments (EC/HC, 2015). However, based on the very low water solubility and hydrophobicity, DIDP is expected to have low bioaccumulation potential, biomagnification potential, and low potential for uptake. The EPA identified one low, one medium, and three high quality data sources reporting the aquatic bioconcentration, aquatic food web magnification, and terrestrial bioconcentration of DIDP (Table 7-1). The available data sources discussed below, suggest that DIDP has low bioaccumulation potential in aquatic and terrestrial organisms (ECJRC, 2003b; Brown and Thompson, 1982a, b), and no apparent biomagnification across trophic levels in the aquatic food web (Mackintosh et al., 2004).

Several studies have investigated the aquatic bioconcentration and food web magnification of DIDP in several marine species. Brown (1982a) evaluated the bioconcentration of DIDP in water fleas (*Daphnia magna*) in a 21-day exposure study. The study reported a mean BCF of 116 for DIDP (Table 7-1). Another study investigated the bioconcentration of DIDP in fish (*Cyprinus carpio*) and reported a BCF of less than 14.4, following OECD Guideline 305C (ECJRC, 2003b). The reported low BCF values suggest that DIDP has low potential to bioaccumulate in aquatic organisms. In the other hand, a mean BCF value of 3488 have been reported in mussels (*Mytilus edulis*) during the 28 days continuous exposure of 5 and 50 µg/L DIDP in water (Brown and Thompson, 1982b). However, the study reported a rapid loss of DIDP from mussels at an approximate half-life of 3.5 days and no apparent adverse effects under the tested conditions. In addition, a study exploring the distribution of phthalate esters within a marine food web system reported an empirical aquatic food web magnification factor (FWMF) of 0.44 which indicates that DIDP do not biomagnified in the aquatic system (Mackintosh et al., 2004). Despite the different range of reported BCF values, based on the rapid loss of DIDP after exposure, the empirical aquatic FWMF of 0.44 (Mackintosh et al., 2004), modeled upper trophic BCF of 1.297 and upper trophic BAF of 9.903 (U.S. EPAF, 2017), it is expected that under normal environmental conditions DIDP is expected to have low bioconcentration potential and low biomagnification potential across trophic levels in the aquatic food web.

There is very limited information on the bioconcentration and bioaccumulation of DIDP in terrestrial environments. Based on DIDP's affinity for sorption to organic matter (log $K_{OC}$ 5.04 to 5.78, Kd of $1.66 \times 10^2$ to $3.97 \times 10^3$) (U.S. EPA, 2012; Mackay et al., 2006; Williams et al., 1995) and water solubility (170 ng/L) (Letinski et al., 2002), DIDP has limited bioavailability in soils. This is supported by the reported BCF values of 0.01 to 0.02 on earthworms (*Eisenia foetida*) (ECJRC, 2003a). Therefore, DIDP is expected to have low bioaccumulation potential, biomagnification potential in terrestrial organisms.

**Table 7-1. Summary of DIDP's Bioaccumulation Information**

| Endpoint | Value(s) | Details | Reference | Overall Quality Ranking |
|---|---|---|---|---|
| Aquatic Bioconcentration (BCF) | 100, 147, 128, 90 (116 mean) | Experimental; 14-C DIDP synthesized from 14-C phthalic anhydride (5.20 mCi/g); *Daphnia Magna*; Daphnia were fed daily by addition of algae (*Chlorella vulgaris*) and yeast suspension; pH = 8.3; T = 20 °C; BCF at 3.2, 10, 32 and 100 µg/L, respectively: 100, 147, 128, 90; Scintillation counting also detected radiochemical metabolites that were likely present at a 2:1 Parent compound: metabolite concentration ratio. | (Brown and Thompson, 1982a) | High |
| Aquatic Bioconcentration (BCF) | 3,977 and 2,998 (3,488 mean) | Experimental; 14-C labelled DIDP (4.76 mCi/g) synthesized from 14-C phthalic anhydride; Mussels (Mytilus edulis); T = 15 °C; Continuously fed saltwater; tank with DIDP; BCF at 5.0 µg/L: 3977; BCF at 50 µg/L: 2998; BCF Mean: 3488 | (Brown and Thompson, 1982b) | High |
| Aquatic Bioconcentration (BCF) | <14.4 | Experimental; Fish, *Cyprinus carpio*; BCF <14.4; | (ECJRC, 2003b) | Medium |
| Aquatic Bioconcentration (FWMF) | 0.44 | experimental; 18 marine species, representing four trophic levels; trophic dilution, predominantly absorbed via the diet and depurated at a rate greater than the passive elimination rate via fecal egestion and respiratory ventilation, due to metabolism; FWMF (food web magnification factor) = 0.44 | (Mackintosh et al., 2004) | High |
| Terr. Bioconcentration (BCF) | 0.01–0.02 | Experimental; earthworms; Eisenia Foetida; 14 days; BCF = ca. 0.01 to 0.02 | (ECJRC, 2003a) | Low |
| Bioaccumulation Factor (BAF) | 9.9 | Estimates; Arnot-Gobas method, upper trophic | (U.S. EPA, 2017) | |

# 8 OVERALL FATE AND TRANSPORT OF DIDP

The inherent physical chemical properties of DIDP govern its environmental fate and transport with a predicted average environmental half-life of 35 days (U.S. EPA, 2012). Based on DIDP's aqueous solubility (0.00017 mg/L), Henry's Law Constant ($2.132\times10^{-4}$ atm·$m^3$/mol at 25 °C), and organic carbon partitioning coefficient (Log Koc: 5.04 to 5.78), this chemical substance will be preferentially sorbed into sediments, soils, and suspended solids in wastewater treatment processes. Soil, sediment, and sludge/biosolids are predicted to be the major receiving compartments for DIDP as indicated by physical and chemical and fate properties, partitioning analysis, and 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 releases. This also holds true for DIDP concentrations in both suspended and benthic sediments. While DIDP undergoes aerobic biodegradation, it does not pass a 10-day ready biodegradation test. Furthermore, biodegradation of DIDP is inhibited in anoxic environments (sediment, landfills) and like other phthalates is expected to slowly hydrolyze and be persistent in anaerobic environments.

If released directly to the atmosphere, DIDP 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 DIDP may be elevated proximal to sites of releases. Off gassing from landfills and volatilization from wastewater treatment processes are expected to be negligible releases in terms of ecological or human exposure in the environment due to its vapor pressure, and rapid photodegradation rates. Thus, DIDP is not expected to be a candidate chemical for long range transport.

Under indoor settings, DIDP is expected to partition to airborne particles at concentrations three times higher than in vapor phase (ECJRC, 2003a) and is expected to have extended lifetime compared to outdoor settings. The available information suggests that DIDP's indoor dust concentrations are associated with the presence of phthalate containing articles and the proximity to the facilities producing them (Kubwabo et al., 2013; Wang et al., 2013; Abb et al., 2009) as well as daily anthropogenic activities that might introduce DIDP containing products into indoor settings (Dodson et al., 2017).

DIDP has a predicted average environmental half-life of 35 days. In the environment, DIDP is not expected to remain long in air and water but may stay in soil and sediments for a longer time, depending on specific conditions. However, it may be found in water where there is continuous release of DIDP. It also may find its way into, and remain in, household dust. Under aerobic conditions, DIDP is expected to degrade rapidly. In environments where anoxic conditions persist, such as benthic sediments, landfills, and some soils, DIDP may be persistent. In anerobic environments, such as deep landfill zones, there is some evidence that suggests that DIDP may be degraded by catalyzed hydrolysis.

# 9 WEIGHT OF SCIENTIFIC EVIDENCE CONCLUSIONS FOR FATE AND TRANSPORT

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

Given the consistent results from numerous high-quality studies, there is a robust confidence that DIDP:

- is expected to undergo significant direct photolysis (Section 3.3);
- will partition to organic carbon and particulate matter in air (Sections 4, 5.1.1);
- will biodegrade in aerobic surface water, soil, and wastewater treatment processes (Sections 5.2.1, 5.3.2, 6.2);
- does not biodegrade in anaerobic environments (Sections 5.2, 5.3);
- will be removed after undergoing wastewater treatment and will sorb to sludge at high fractions, with a small fraction being present in effluent (Section 6.2);
- is not bioaccumulative (Section 7);
- is not expected to biodegrade under anoxic conditions and may have high persistence in anaerobic soils and sediments (Sections 3.1, 5.2.2, 5.3.2);
- may show pseudo-persistence in surface water and sediment proximal to continuous points of release (Sections 3.1, 5.2.2, 5.3.2); and
- is expected to transform to monoisodecyl phthalate, isodecanol, and phthalic acid in the environment (Section 3).

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

- is expected to be partially 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 (Section 6.3); and
- showed no significant degradation via hydrolysis under standard environmental conditions but hydrolysis rate was seen to increase with increasing pH and temperature in deep-landfill environments (Section 5.3.3).

Findings that were found to have a robust weight of evidence supporting them had one or more high-quality studies that were largely in agreement with each other findings that were said to have a moderate weight of evidence were based on a mix of high and medium-quality studies that were largely in agreement but varied in sample size and consistence of findings.

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