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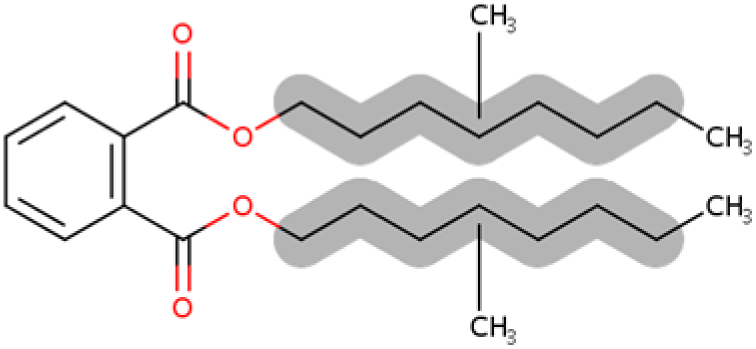
Fate Assessment for Diisononyl Phthalate (DINP)

Technical Support Document for the Risk Evaluation

CASRNs: 28553-12-0 and 68515-48-0



(Representative Structure)

January 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 Abstracts Service Registry Number
DMR	Discharge Monitoring Reports
DOC	Dissolved organic carbon
DRE	Destruction and removal efficiency
dw	Dry weight
DW	Drinking water
DWTP	Drinking water treatment plant
EPA	(U.S.) Environmental Protection Agency (or the Agency)
EPI Suite^{TM}	Estimation Program Interface Suite^{TM}
ESI	Electrospray ionization
FID	Flame ionization detector
FPD	Flame photometric detector
GC	Gas chromatography
HLC	Henry's Law constant
HPLC	High-performance liquid 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
LOD	Limit of detection
LOQ	Limit of quantification
lw	Lipid weight
M	Molarity (mol/L = moles per Liter)
MDL	Method Detection Limit
MRL	Method Reporting Limit
MS	Mass spectrometry
n	Sample size
N/A	Not applicable
ND	Non-detection
NR	Not reported
OECD	Organisation for Economic Co-operation and Development
·OH	Hydroxyl radical
OPE	Organophosphate ester
QSAR	Quantitative structure activity relationship
RSD	Relative standard deviation
SI	Supplemental information
SIM	Selected ion monitoring
SPE	Solid phase extraction
STP	Sewage treatment plant

TMF	Trophic magnification factor
TOC	Total organic carbon
TOF	Time of flight
TRI	Toxics Release Inventory
TSD	Technical support document
UPLC	Ultra-performance liquid chromatography
UV (UV-Vis)	Ultraviolet (visible) light
ww	Wet weight
WWTP	Wastewater treatment plant

SUMMARY

This technical support document (TSD) is for the *Risk Evaluation for Diisononyl Phthalate (DINP)* (U.S. EPA, 2025c). DINP is a common chemical name for the category of chemical substances that includes the following substances: 1,2-benzenedicarboxylic acid, 1,2-isononyl ester (CASRN 28553-12-0) and 1,2-benzenedicarboxylic acid, di-C9-11-branched alkyl esters, and C9-rich (CASRN 68515-48-0). Both CASRNs contain mainly C9 dialkyl phthalate esters. See the risk evaluation for a complete list of all the TSDs for DINP.

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

- is expected to undergo significant direct photolysis and will rapidly degrade in the atmosphere ($t_{1/2}$ = 8.5 hours) (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 via biotic and abiotic processes to form monoisononyl phthalate, isononanol, 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 (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); and
- is likely to be found in and accumulate in indoor dust (Section 5).

As a result of limited studies identified, there is moderate evidence that DINP

- 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);
- has limited bioaccumulation potential in fish in the water column (Section 7);
- may bioaccumulate in benthic organisms exposed to sediment with elevated concentrations of DINP proximal to continual sources of release (Section 7); and
- is expected to be removed in conventional 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

DINP is considered ubiquitous in various environmental media to due to its presence in both point and non-point source discharges from industrial and conventional wastewater treatment effluents, biosolids and sewage sludge, stormwater runoff, and landfill leachate (Net et al., 2015). As an isomeric mixture, the fate and transport properties of DINP can be difficult to classify. However, the following sections of the fate and transport analysis of DINP are present the general fate and transport characteristics of DINP.

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 the parameters used in the current risk evaluation. In assessing the environmental fate and transport of DINP, EPA considered the full range of results from data sources that were rated high-quality. Information on the full extracted data set is available in the *Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport for Diisononyl Phthalate (DINP)* (U.S. EPA, 2025a). Other fate estimates were based on modeling results from Estimation Program Interface (EPI) Suite™ (U.S. EPA, 2012), a predictive tool for physical and chemical properties and environmental fate estimation.

Table 2-1 provides a summary of the selected environmental fate data that EPA considered while assessing the fate of DINP and were updated after publication of *Final Scope of the Risk Evaluation for Di-isononyl Phthalate (DINP); CASRNs 28553-12-0 and 68515-48-0* (U.S. EPA, 2021) with additional information identified through the systematic review process.

Table 2-1. Summary of Environmental Fate Information for DINP

Parameter	Value	Source(s)
Octanol:Water (Log K_{OW})	8.8	ECHA (2016)
Organic Carbon:Water (Log K_{OC})	5.5 (estimated; MCI method); 5.7 (estimated; Kow method	U.S. EPA (2017)
Adsorption Coefficient (Log K_d)	2.97 (suspended particulate matter/water)	Li et al. (2017a)
	3.27 (sediment/water)	Li et al. (2017a)
Octanol:Air (Log K_{OA})	11.9 (estimated)	U.S. EPA (2017)
Air:Water (Log K_{AW})	−2.20 (estimated)	Lu (2009)
	−2.43 (estimated)	Cousins and Mackay (2000)
Aerobic primary biodegradation in water	32–67.8% in 24 hours >90% in 5 days >99% in 28 days	(O'Grady et al., 1985; SRC, 1983; Monsanto, 1978)
Aerobic ready biodegradation in water	57–81% in 28 days	(ECJRC, 2003b)
Aerobic ultimate biodegradation in water	57–84% in 28 days	(HSDB, 2015; Monsanto, 1983)
Aerobic biodegradation in sediment	0.54% in 14 days 1.11% in 28 days	(Johnson et al., 1983)
Anaerobic biodegradation in sediment	0% in 100 days	(Ejlertsson et al., 1996)

Parameter	Value	Source(s)
Aerobic biodegradation in soil	No significant change in concentration after 2 years	(ECJRC, 2003b)
Hydrolysis	152 days at pH 8 and 25 ºC, and 4.2 years at pH 7 and 25 ºC	(U.S. EPA, 2017)
Photolysis	$t_{1/2}$ (air) = 5.36 to 8.5 hours $t_{1/2}$ (water_{pH} =7) = 140 days	(U.S. EPA, 2017; Lertsirisopon et al., 2009; Peterson and Staples, 2003)
Environmental degradation half-lives (selected values for modeling)	5.36 hours (air) 10 days (water) 20 days (soil) 90 days (sediment)	(U.S. EPA, 2017)
WWTP Removal	>93%	(U.S. EPA, 2017)
Aquatic Bioconcentration (BCF)	<3 L/kg ww (rainbow trout; *Oncorhynchus mykiss*) 5.2 L/kg ww (upper trophic Arnot-Gobas estimation)	(U.S. EPA, 2017; EC/HC, 2015a)
Aquatic Bioaccumulation (BAF)	68 (75 ug/kg ww in mussel from field study in Seine estuary, France) 21 L/kg ww (upper trophic Arnot-Gobas estimation)	(U.S. EPA, 2017; ECJRC, 2003b)
Aquatic Food web Magnification Factor (FWMF)	0.46 (Experimental; 18 marine species)	(Mackintosh et al., 2004)
Terrestrial Bioconcentration (BCF)	0.01–0.02 Experimental; earthworms (*Eisenia fetida*)	(ECJRC, 2003b)
Terrestrial Biota-Sediment Accumulation Factor (BSAF)	0.018 OECD Test Guideline 207 (*Eisenia fetida*)	(EC/HC, 2015a)

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 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). A Tier II analysis involves reviewing environmental release information for DINP to determine whether further assessment is warranted for each environmental medium. Environmental release data for DINP was not available from the Toxics Release Inventory (TRI) or Discharge Monitoring Reports (DMRs); however, between 250 and 550 million lb of CASRN 28553-12-0 and between 100 and 1,000 million lb of CASRN 68515-48-0 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. DINP is used as a plasticizer in polyvinyl chloride (PVC) and non-PVC products (U.S. EPA, 2020; EC/HC, 2015a). DINP may be released to the environment during production, distribution, processing in PVC and non-PVC polymers, use of DINP-containing products such as paints and sealants, disposal or recycling, wastewater treatment, and disposal of solid and liquid waste (ECJRC, 2003b).

Environmental release information is also 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 to reduce levels of uncertainties. Based on DINP’s observed and calculated environmental half-lives, partitioning characteristics, and the results of Level III Fugacity modeling (see Figure 4-1 below), DINP 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 DINP partitioning can be found in Section 4.

The following inputs parameters were used for the Level III Fugacity model in EPI Suite™:

- Melting Point = −48.00 °C
- Vapor Pressure = 5.40×10^{-7} mm Hg
- Water Solubility = 6.10×10^{-4} mg/mL
- Log K_{OW} = 8.8
- SMILES: CCCCCCC(C)COC(=O)c1ccccc1C(=O)OCCCCC(C)C(C)C (representative structure)

3 TRANSFORMATION PROCESSES

DINP released to the environment will transform to the monoester form (monoisononyl phthalate) via abiotic processes such as photolysis (direct and indirect) 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, then to phthalic acid, and ultimately biodegradation of phthalic acid to form CO_2 and/or CH_4 (Huang et al., 2013). The monoisononyl phthalate is both more soluble and more bioavailable than DINP. It is also expected to undergo biodegradation more rapidly than the diester form. EPA considered DINP transformation products and degradants qualitatively but due to their lack of persistence we 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 DINP are described in the sections below.

3.1 Biodegradation

DINP can be considered readily biodegradable under most aquatic and terrestrial environments. The EPA extracted and evaluated fourteen data sources containing DINP biodegradation information in water, soil, and sediments under aerobic and anaerobic conditions (Table 3-1). Eight of the sources were classified as overall high-quality, five as overall medium-quality, and one as overall low-quality data sources. DINP is considered an isomeric mixture and certain components of DINP might biodegrade more readily than others (ECJRC, 2003b). DINP's aerobic primary biodegradation in water has reported to be 32 to 67.8 percent in 24 hours (O'Grady et al., 1985; Monsanto, 1978), greater than 90 percent in 5 days (O'Grady et al., 1985), and greater than 99 percent in 28 days (U.S. EPA, 2019) with a half-life of 1.5 to 5.31 days under acclimated conditions (SRC, 1984, 1983) and 7 to 40 days under unacclimated conditions (EC/HC, 2015a). Several studies evaluating the readily biodegradability of phthalate esters in water have reported DINP's half-life of 10.3 days (ExxonMobil, 2010) and 57 to 81 percent DINP removal in 28 days by CO_2 evolution (HSDB, 2015; ECJRC, 2003b). The required 60 percent degradation during the 10-day pass window was met only in two of the four available studies (ECJRC, 2003b). However, DINP in water has been reported to completely biodegrade into its basic elements by 57 to 84 percent after 28 days (based on the available ultimate biodegradation information) (EC/HC, 2015a; HSDB, 2015; Monsanto, 1983; SRC, 1983). In contrast to the rapid biodegradation of DINP in aerobic environments, available information suggests that DINP is expected to have very low biodegradation potential under low oxygen conditions (Ejlertsson et al., 1996) and could remain longer in subsurface sediments and soils (Kickham et al., 2012; ECJRC, 2003b; Johnson et al., 1984, 1983).

Table 3-1. Summary of Biodegradation Information for DINP

Environmental Conditions	Degradation Value	Half-Life (days)	Reference	Overall Data Quality Ranking
Aerobic primary biodegradation in water	32% in 24 hours	ND	(Monsanto, 1978)	High
	94 to 96% in 9 days	1.5 days (average; 1–1.9 days)	(SRC, 1984)	High
	67.8% in 24 hours >90% in 5 days	ND	(O'Grady et al., 1985)	High
	>99% in 28 days	5.31 days	(SRC, 1983)	High
	91 to 100% in 7 days	7–40 days	(EC/HC, 2015a)	Medium
	>95% in 12 days	ND	(HSDB, 2015)	Medium
	90–100% in 5–28 days 68% in 1 day	ND	(U.S. EPA, 2019)	Medium

Environmental Conditions	Degradation Value	Half-Life (days)	Reference	Overall Data Quality Ranking
Aerobic ready biodegradation in water	70.5% in 28 days	ND	(ECJRC, 2003b)	Medium
	57% in 28 days	ND		
	81% in 28 days	ND		
	74% in 28 days	ND	(HSDB, 2015)	Medium
	ND	10.3 days	(ExxonMobil, 2010)	Low
Aerobic ultimate biodegradation in water	61.5% in 28 days	ND	(SRC, 1983)	High
	84% in 28 days	ND	(Monsanto, 1983)	Medium
	57–71% in 28 days	ND	(HSDB, 2015)	Medium
	56.6 % in 29 days	ND	(EC/HC, 2015a)	Medium
	67.5% in 28 days	ND		
	74% in 28 days	ND		
Aerobic biodegradation in sediment	0.54% in 14 days 1.11% in 28 days	ND	(Johnson et al., 1983)	High
	ND	12,000 days	(Kickham et al., 2012)	High
	0.7% at 12 °C 1.2% at 22 °C 2.2% at 28 °C	ND	(Johnson et al., 1984)	High
Anaerobic biodegradation in sediment	0% in 100 days	ND	(Ejlertsson et al., 1996)	High
Aerobic biodegradation in soil	No significant change in concentration after 2 years	ND	(ECJRC, 2003b)	Medium

3.2 Hydrolysis

Traditionally accepted methods of testing for abiotic hydrolysis (OECD Guideline Test 111) are not viable for DINP due to the low aqueous solubility (ECJRC, 2003a). Therefore, hydrolysis rates of DINP are difficult to accurate measure experimentally (ECJRC, 2003a). EPI Suite$^{\text{TM}}$ was utilized to estimate the hydrolysis half-lives of DINP at 152 days at pH 8 and 25 ºC, and 4.2 years at pH 7 and 25 ºC (U.S. EPA, 2017), indicating that hydrolysis is a possible degradation pathway of DINP under more caustic conditions. Lertsirisopon (2009) reported the hydrolysis half-lives of 720 days (pH = 5), 1,200 days (pH = 6), negligible (pH = 7), 1,000 days (pH = 8), and 460 days (pH = 9), at average temperature of 10.8 °C. However, this study received a low data quality ranking in the systematic review process due to poorly documented and variable test conditions.

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

3.3 Photolysis

DINP contains chromophores that absorb light at greater than 290 nm wavelength (NCBI, 2020);

therefore, direct photodegradation is a relevant degradation pathway for DINP released to air. Modeled indirect photodegradation half-lives indicated a slightly more rapid rate of degradation, estimating a half-life of 0.22 days (5.36 hours) (·OH rate constant of 2.39×10^{-11} cm^3/molecule-second and a 12-hour day with 1.5×10^6 ·OH/cm^3) (U.S. EPA, 2017). Similarly, Peterson (2003) reported a calculated DINP photodegradation half-life of 0.35 days (8.5 hours) (·OH rate constant of 2.35×10^{-11} cm^3/molecule-second and 1×10^6 ·OH/cm^3). DINP photodegradation in water is expected to be slower than air due to the typical light attenuation in natural surface water. The aquatic direct photodegradation half-lives of 32, 52, 140, 61, and 36 days were observed 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) (Lertsirisopon et al., 2009).

4 PARTITIONING

Environmental release data for DINP was not available from the TRI or DMRs; therefore, DINP 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, 2017). DINP's physical and chemical properties were taken directly from Section 2.1 of *Physical Chemistry Assessment for Diisononyl Phthalate (DINP)* (U.S. EPA, 2025b).

Environmental release information is useful for fugacity modeling because the emission rates will predict a real-time percent distribution for each medium. An environmental degradation half-life in water of 10 days was selected in this risk evaluation to represent the range of identified primary biodegradation half-life values (Section 3.1) from high- and medium-quality studies to reduce levels of uncertainties. EPA used environmental degradation half-lives of 5.36 hours in air (based on AEROWIN^{TM} predicted values, an atmospheric fate prediction model within EPI Suite^{TM}), 20 days in soil (double the half-life in water), and 90 days in sediment (9 times the half-life in water) as recommended for EPIWIN estimations (U.S. EPA, 2017). Based on DINP's environmental half-lives, partitioning characteristics, and the results of Level III Fugacity modeling, DINP 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 DINP partitioning can be found in Sections 5.1, 5.2, and 5.3.

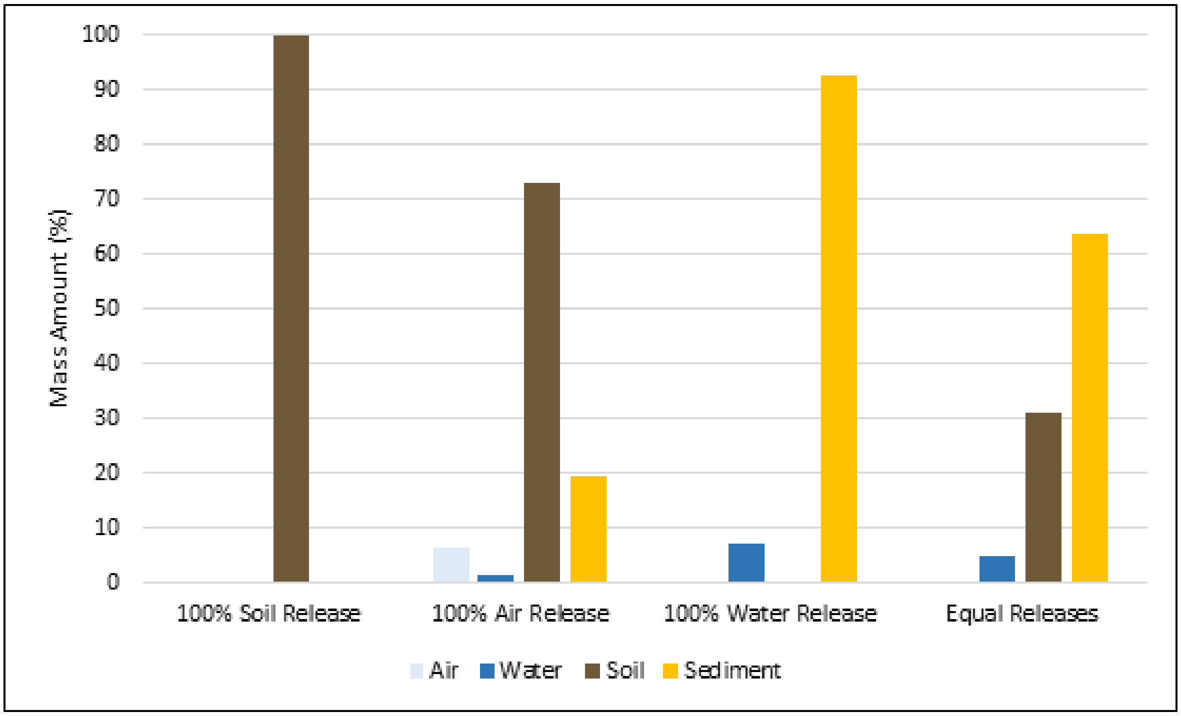

Figure 4-1. EPI Suite^{TM} Level III Fugacity Modeling Graphical Result for DINP

5 MEDIA ASSESSMENTS

DINP has been reported to be present in the atmosphere, aquatic environments, and terrestrial environments. Once in the air, DINP 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, DINP is likely to partition to house dust and airborne particles and is expected to have a longer half-life compared to ambient (outdoor) air. DINP present in surface water is expected to mostly partition to aquatic sediments. DINP is expected to have an aerobic biodegradation half-life between 14 and 28 days. In terrestrial environments, DINP has the potential to be present in soils and groundwater but is likely to be immobile in both media types. In soils, DINP is expected to be deposited via air deposition and land application of biosolids. DINP in soils is expected to have a half-life on the order of days to weeks, have low bioaccumulation potential, and biomagnification potential in terrestrial organisms. DINP is released to groundwater via wastewater effluent and landfill leachates, expected to have a half-life of 14 to 56 days, and not likely to be persistent in most groundwater/subsurface environments.

5.1 Air and Atmosphere

DINP is a liquid at environmental temperatures with a melting point of −48 °C (Haynes, 2014; O'Neil, 2013) and a vapor pressure of 5.40×10^{-7} mmHg at 25 °C (NLM, 2015). Based on its physical and chemical properties and short half-life in the atmosphere ($t_{1/2}$ = 5.36 hours (U.S. EPA, 2017)), DINP was assumed to not be persistent in the air. The AEROWIN^{TM} module in EPI Suite^{TM} estimated that a large fraction of DINP could be sorbed to airborne particles and these particulates might be resistant to atmospheric oxidation. DINP has not been detected in ambient air; however, studies have detected DINP in settled house dust, indoor air samples and in indoor particulate phase air samples (NCBI, 2020; Kubwabo et al., 2013; ECJRC, 2003b).

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). Most of the studies reported DEHP (di-ethylhexyl phthalate) to be the predominant phthalate esters in the environment. Despite limited information on the presence of DINP on the atmosphere, similar trends to those reported for DEHP could be expected based on their similar vapor pressure (ECHA, 2013). Limited studies have reported the presence of particle-bound DINP on indoor and outdoor settings (Gupta and Gadi, 2018; Hasegawa, 2003; Helmig et al., 1990). Once in indoor air, DINP is expected to partition to organic carbon present on indoor airborne particles. In indoor environments, DINP 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 DINP in dust under indoor environments to be higher than outdoors dust and to be associated with the presence of phthalate-containing articles and the proximity to manufacturing facilities (Kubwabo et al., 2013; Wang et al., 2013; Abb et al., 2009). Kubwabo (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 were detected in all the collected dust samples comprising 88 percent of the median total concentration of phthalates in dust. Wang (2013) evaluated the presence of phthalates in dust samples collected from indoor and outdoor settings in two major Chinese cities. The study reported 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 (2009) evaluated the presence of phthalates in indoor dust samples collected from 30 households in Germany. The study revealed the presence of DEHP, DIDP, and DINP in all the collected samples. Samples collected from households containing a high percentage of plastics (>50% plastic content) resulted with higher aggregate concentration of phthalates in dust. The aggregate concentration of DEHP, DIDP, and DINP accounted for 87 percent of the total phthalate concentration in dust.

Similarly, recent studies monitoring the presence of phthalates in dust from U.S. households have revealed DEHP and DINP to de detected in 96 to 100 percent of the collected samples (Hammel et al., 2019; Dodson et al., 2017). Hammel (2019) and Dodson (2017) reported the presence of phthalate esters on indoor air and dust samples collected in U.S. homes. Hammel (2019) reported that DINP accounted for close to 83 percent of the total concentration of phthalates found in indoor dust. Dodson (2017) evaluated the presence of phthalate esters in air samples of U.S. homes before and after occupancy reporting increased presence of DINP after occupancy due to daily anthropogenic activities that might introduce phthalate containing products into indoor settings. Increasing trends could be expected for DINP with its increased use on household’s construction materials or consumer products.

5.2 Aquatic Environments

5.2.1 Surface Water

DINP 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. DINP and other phthalate esters have been detected in surface waters, although at lower frequencies than some other phthalate esters (Wen et al., 2018). The principal properties governing the fate and transport of DINP in surface water are water solubility, organic carbon partitioning coefficients, and volatility. Due to its Henry’s Law constant (9.14×10^{-5} atm·m^3/mol at 25 °C) of DINP, volatilization is not expected to be a significant source of loss of DINP from surface water. A partitioning analysis of DINP released to the environment is described in Section 4 above. The analysis estimates that during releases to surface water bodies, greater than 92 percent of DINP released to surface water will partition to both suspended and benthic sediments.

DINP has a low water solubility of 0.00061 mg/L, but is likely to form a colloidal suspension and may be detected in surface water at higher concentrations (EC/HC, 2015b). Based on DINP’s water solubility and partitioning coefficients, DINP in water will partition to suspended organic material present in the water column. DINP is expected to be readily biodegradable in water (Section 3.1). In addition, total seawater samples concentrations of DINP measured in False Creek ranged from 61 to 135 ng/L, the dissolved fraction concentrations ranged from 29 to 64 ng/L, and the suspended particulate fraction concentration ranged from 14,700 to 50,400 ng/g dry weight (dw) (EC/HC, 2015a; Mackintosh et al., 2006). Concentrations of DINP above the aqueous solubility of 0.00061 mg/L are not uncommon in monitoring studies proximal to releases of DINP to surface water (Wen et al., 2018).

5.2.2 Sediments

Based on the water solubility (0.00061 mg/L) and affinity for sorption to organic matter (log K_{OC} = 5.5 to 5.7), DINP will partition to the organic matter present in soils and sediment when released into the environment. Once in water, DINP is expected to be readily biodegradable and the Level III Fugacity Model in EPI Suite™ (U.S. EPA, 2017) predicts that greater than 92 percent of the DINP will partition to and remain in sediments (Section 4). The available information suggests that DINP could persist longer in subsurface sediments and soils than in water. In terrestrial and aquatic environments, DINP has potential to accumulate in sediments at areas of continuous release, such as a surface water body receiving discharge from a municipal wastewater treatment plant.

Due to the strong sorption to organic carbon, DINP is expected to be found predominantly in sediments near point sources, with a decreasing trend in sediment concentrations downstream. This is consistent with monitoring information for phthalate esters from Sweden and Korea. One study reported the presence of DINP in only one sediment sample near a point source in Sweden that recently have replaced DEHP with DINP in their production processes (Parkman and Remberg, 1995). The presence of DINP has been documented in urban sediments at concentrations ranging between 130 to 3,200 µg/kg total solids with a 62 percent detection frequency (Cousins et al., 2007). In a similar study, Kim (2021) evaluated the presence of plasticizers in sediments from highly industrialized bays of Korea. DINP 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 bays located farther away from industrial activities. The findings of this study suggest industrial activities to be the major contributor of phthalates in sediments within the area.

Monitoring data from the Rhine River and the Neckar River in Germany detected DINP concentrations in sediment samples of 30, 220, 650, and 1,460 ppb and 430, 570, 1,050 ppb (3 sites), respectively (NCBI, 2020). DINP was also detected in sediment from 21 locations in the Netherlands at concentrations up to 6.16 mg/kg dw (ECJRC, 2003b).

5.3 Terrestrial Environments

5.3.1 Soil

DINP is expected to be deposited to soil via two primary routes—application of biosolids and sewage sludge in agricultural applications or sludge drying applications as well as atmospheric deposition. Based on DINP’s Henry’s Law constant of 9.14×10^{-5} atm·m^3/mol at 25 °C and vapor pressure of 5.40×10^{-7} mmHg, DINP is not likely to volatilize from soils.

DINP shows an affinity for sorption to soil and its organic constituents (log $K_{OC} = 5.5–5.7$; log $K_d = 2.55–3.27$ (Li et al., 2017b; Li et al., 2017a; U.S. EPA, 2012)) 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, DINP is expected to have low mobility in soil environments.

Under aerobic conditions, DINP is expected to have a half-life in soil of 20 days. This aerobic biodegradation half-life for soil was estimated by doubling the experimentally derived half-life of DINP in water as very limited soil biodegradation data for DINP identified in the systematic review process (SRC, 1983).

Under anaerobic conditions that might be present in some soil profiles, there is very little evidence to support that DINP appreciably biodegrades (ECJRC, 2003b; Ejlertsson et al., 1996). One study found that 0 percent degradation had occurred under anaerobic conditions after 100 days by CH_4 evolution and no transformation reported based on the concentrations of methane and test substance with gas chromatographic analysis in municipal solid waste samples with an anaerobic microflora inoculum (Ejlertsson et al., 1996). Furthermore, another study reported less than 1 percent DINP degradation in anaerobic sediments after 28 days (Johnson et al., 1984).

In general, DINP 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, but continuous exposure to DINP in soil proximal to points of releases might be possible if the rate of releases exceeds the rate of biodegradation under aerobic conditions. Under anaerobic conditions in soil, DINP is assumed to be persistent and continuous exposure is likely.

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 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.

There is limited information about the presence and biodegradation of DINP in biosolids. As described in Section 6.2, DINP 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 DINP present in wastewater is expected to be accumulated in sewage sludge and discharged into biosolids. Once in biosolids, DINP can be transferred to soil during land applications. It will be strongly sorbed to organic matter on soils and to be more persistent in soil profiles with anaerobic conditions (ECJRC, 2003b). Due to its strong sorption to soils, land-applied DINP is not expected to be bioavailable; thus, exposures to environmental organisms and people are negligible. In addition, based on its water solubility and hydrophobicity, DINP will have low bioaccumulation while 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 DINP 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: (1) an "upper-landfill" zone, with normal environmental temperatures and pressures, where biotic processes are the predominant route of degradation for DINP; and (2) a "lower-landfill" zone where elevated temperatures and pressures exist, and abiotic degradation is the predominant route of degradation for DINP. 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 DINP. 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 completely irrelevant at 70 °C (Huang et al., 2013).

DINP is deposited in landfills continually and in high amounts from the disposal of consumer products containing DINP. However, due to its strong sorption to soils and low water solubility, small concentrations of DINP are likely to be present in landfill leachate. DINP 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 might occur. In lower-landfills, there is some evidence to support that hydrolysis can be the main route of abiotic degradation of phthalate esters (Huang et al., 2013).

Despite the expected persistence of DINP in landfills, it is not expected to be bioavailable and mainly sorbed to organic matter in soils due to the low water solubility of DINP (0.00061 mg/L) and its high sorption to organic carbon (log K_{OC} = 5.5–5.7). Although DINP might be present at small concentrations 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 potential sources of DINP in groundwater, including wastewater effluents and landfill leachates, which are discussed in Sections 5.3.3 and 6.2. Further, in environments where DINP is found in surface water, it can 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 DINP to adsorb to organic matter present in soils and sediments (log K_{OC} = 5.5–5.7) (U.S. EPA, 2012) DINP is expected to have low mobility in soil and groundwater environments. Furthermore, due to the insoluble nature of DINP (0.00061 mg/L), high concentrations of DINP in groundwater are unlikely. In instances where DINP could reasonably be expected to be present in groundwater environments (*e.g.*, proximal to landfills or agricultural land with a history of land applied biosolids), limited persistence is expected based on rates of biodegradation of DINP in aerobic environments. Thus, DINP is not likely to be persistent in groundwater/subsurface environments unless anoxic conditions exist.

6 PERSISTENCE POTENTIAL OF DINP

DINP is not expected to be persistent in the environment; it is expected to degrade rapidly under most environmental conditions, with delayed biodegradation in low-oxygen media. In the atmosphere, DINP is unlikely to remain for long periods of time as it is expected to undergo photolytic degradation through reaction with atmospheric hydroxyl radicals, with estimated half-lives of 5.36 hours. DINP 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). DINP has the potential to remain for longer periods of time in soil and sediments, but due to its inherent hydrophobicity (log K_{OW} = 8.8) and sorption potential (log Koc = 5.5–5.7), DINP is not expected to be bioavailable for uptake. Using the Level III Fugacity model in EPI Suite^{TM} (LEV3EPITM) (Section 4), DINP's overall environmental half-life was estimated to be approximately 34 days (U.S. EPA, 2012). Therefore, DINP 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. DINP is classified as a hazardous substance 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) (U.S. EPA, 1981).

Currently there is no information available on the DRE of DINP. 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 DINP present during incineration processes will be very likely to be released to landfills and the remaining small fraction released to air. Based on its hydrophobicity and sorption potential, DINP released to landfills is expected to partition to waste organic matter. Similarly, DINP released to air is expected to be rapidly react via indirect photochemical processes within hours (U.S. EPA, 2017) and partition to soil and sediments as described in Section 4. DINP 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. 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 DINP in wastewater treatment systems. The EPA selected two high-quality sources reporting the removal of DINP in wastewater treatment systems employing aerobic and anaerobic processes. One study reported 98.0 percent DINP removal efficiencies in a municipal wastewater treatment facility in France, employing a combined decantation and activated sludge tank (Tran et al., 2014). Like other phthalates esters with long carbon chains and high log Kow, DINP was reported to be mainly removed by particle sorption and retained in the sewage sludge. 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 DINP during conventional wastewater treatment by sorption to sludge with the potential of increased removal via rapid aerobic biodegradation processes (U.S. EPA, 2012). In addition, the treatment of wastewater final solids via aerobic digestions processes have been reported to achieve 41.1 to 85.9 percent reduction on DINP concentration from the digestion effluents (Armstrong et al., 2018). In addition, the same study reported anaerobic solids digestion to be not effective in the removal of DINP. In general, the available information suggest that aerobic processes have the potential to help biodegrade DINP from wastewater in agreement with the expected aerobic biodegradation described in Section 3.1.

Overall, DINP has a high log K_{OW} and remains in suspended solids and efficiently removed from wastewater via accumulation in sewage sludge (Tran et al., 2014), partially removed during aerobic solids digestion processes (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 DINP 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, and reservoirs) as well as 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.

Very limited information is available on the removal of DINP in drinking water treatment plants. No data was identified by the EPA for DINP in drinking water. Based on the water solubility and Log K_{OW}, DINP 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 92.7 percent of DINP released to water partitioning to sediments (U.S. EPA, 2012). The available information on the DEHP removal efficiency of flocculants and filtering media suggest that DINP could potentially be partially removed during drinking water treatment by sorption into suspended organic matter. This data source reported 58.7 percent reduction on drinking water DEHP concentration from a conventional drinking water treatment effluent in China and 78 to 86 percent loss of DINP during storage of treated drinking water effluent after 48 hours in Taiwan using chlorine for disinfection prior to distribution (Kong et al., 2017; Yang et al., 2014). These findings suggest that conventional drinking water treatment systems may have the potential to partially remove DINP is present in drinking water sources via sorption to suspended organic matter and filtering media and the use of disinfection technologies.

7 BIOACCUMULATION POTENTIAL OF DINP

The presence of DINP in several marine aquatic species in North America suggests that it might be bioavailable in aquatic environments (Mackintosh et al., 2004). However, based on the very low water solubility and high hydrophobicity, DINP is expected to have low bioaccumulation potential, low biomagnification potential, and low potential for uptake. EPA selected three overall high- and two overall medium-quality data source reporting the aquatic bioconcentration, aquatic bioaccumulation, aquatic food web magnification, terrestrial biota-sediment accumulation, and terrestrial bioconcentration of DINP (Table 7-1). The available data sources discussed below suggest that DINP has both low bioaccumulation potential in aquatic and terrestrial organisms (EC/HC, 2015a; Solbakken et al., 1985; Chemical Manufacturers, 1984) 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 DINP in several marine species. Solbakken (1985) evaluated the bioconcentration of DINP in Arca zebra in a 24-hour exposure study, followed by a 14-day depuration period. The study reported DINP bioconcentration factor (BCF) values of 8.2, 183.8, 13.6, and 9.3 dpm/µL during the 24-hour exposure period on Arca zebra muscle, hepatopancreas, gills, and blood, respectively (Table 7-1). The study reported a 92 to greater than 99 percent decrease on BCF values during the 14-days depuration period. A similar study evaluating the presence of phthalates on estuaries reported a mussel BAF of 68 and DINP content of 75 µg/kg wet weight (ww) (ECJRC, 2003b). A DINP exposure study on rainbow trout reported BCFs lower than 3 L/kg ww and biomagnification factors lower than 0.1 (EC/HC, 2015a). The reported low BCF values suggest that DINP has low potential to bioaccumulate in aquatic organisms. On the other hand, the Chemical Manufacturers Association (1984) reported a higher predicted DINP aquatic BCF of 1,155 using a regression model based on the substance water solubility. Despite of the different range of reported BCF values, an empirical rapid BCF decrease during a 14-days depuration period (Solbakken et al., 1985), an empirical aquatic trophic magnification factor (TMF) of 0.46 (Mackintosh et al., 2004), and a modeled upper trophic BCF of 5.2 L/kg ww and upper trophic BAF of 21 L/kg ww (U.S. EPA, 2017), help support that DINP will 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 DINP in terrestrial environments. Based on DINP's strong sorption organic matter (log K_{OC} 5.5–5.7) (U.S. EPA, 2017) and water solubility (0.00061 mg/L) (Letinski et al., 2002), DINP is not expected to be bioavailable in soils. This is supported by the reported low BCF values of 0.1 to 0.2 on earthworms (*Eisenia foetida*) (ECJRC, 2003b). Therefore, DINP is expected to have low bioaccumulation and biomagnification potential in terrestrial organisms.

Table 7-1. Summary of Bioaccumulation Information for DINP

Endpoint	Value	Details	Reference	Overall Quality Ranking
Aquatic Bioconcentration (BCF)	8.2 at day 0 4.6 at day 1 1.3 at day 4 0.03–0.01 at day 14 (dpm/mg)	Experimental; Muscle BCF; 14-C DINP 24 hours of exposure followed by 14-day depuration period; *Arca* Zebra (mollusk)	(Solbakken et al., 1985)	High
	183.8 at day 0 125.2 at day 1 64.5 at day 4 14.4 at day 14 (dpm/mg)	Experimental; Hepatopancreas BCF; 14-C DINP 24 hours of exposure followed by 14-day depuration period; *Arca* Zebra (mollusk)		
	13.6 at day 0 12.4 at day 1 6.5 at day 4 0.8 at day 14 (dpm/mg)	Experimental; Gills BCF; 14-C DINP 24 hours of exposure followed by 14-day depuration period; *Arca* Zebra (mollusk)		
	9.3 at day 0 5.6 at day 1 4.4 at day 4 0.1 at day 14 (dpm/µL)	Experimental; Blood BCF; 14-C DINP 24 hours of exposure followed by 14-day depuration period; *Arca* Zebra (mollusk)		
	0.46 at day 0 0.45 at day 1 0.26 at day 4 0.13 at day 14 (dpm/mg)	Experimental; BCF; 14-C DINP 24 hours of exposure followed by 14-day depuration period; *Diploria Strigosa* (coral)		
	1,155	Predicted; log BCF = (0.542 × log K_{OW})+0.124; calculated using Kow values that were calculated from water solubility; log Kow = 5.2–0.68 × log (micromolar WS).	(Chemical Manufacturers, 1984)	High
	<3 L/kg ww	Experimental; rainbow trout; *Oncorhynchus mykiss*; elimination rate: 1.16/day; tissue elimination half-life: <1 day; biomagnification factor (BMF): <0.1	(EC/HC, 2015a)	Medium
Aquatic Bioaccumulation (BAF)	68	Experimental; preliminary study; Field study; Mussel; Collected from Seine estuary, France; 75 ug/kg ww in mussel from field study in Seine estuary, France	(ECJRC, 2003b)	Medium

Endpoint	Value	Details	Reference	Overall Quality Ranking
Aquatic Food Web Magnification Factor (FWMF)	0.46	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	Terrestrial BCF; Experimental; earthworms (*Eisenia fetida*); steady state may not have been achieved.; 14 days	(ECJRC, 2003b)	Medium
Terrestrial Biota-Sediment Accumulation Factor (BSAF)	0.018	experimental; other: OECD Test Guideline 207 (Earthworm, acute toxicity; OECD 1984a); earthworm; *Eisenia fetida*;	(EC/HC, 2015a)	Medium

8 OVERALL FATE AND TRANSPORT OF DINP

The inherent physical and chemical properties of DINP govern its environmental fate and transport. Based on DINP's aqueous solubility, slight tendency to volatilize, and strong tendency to adsorb to organic carbon, this phthalate 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 DINP as indicated by its physicochemical 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 away from the point of releases. This also holds true for DINP concentrations in both suspended and benthic sediments. While DINP undergoes relatively rapid aerobic biodegradation, it is persistent in anoxic/anaerobic environments (sediment, landfills), and like other phthalates, is expected to slowly hydrolyze under normal environmental conditions.

If released directly to the atmosphere, DINP 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 DINP might 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 low vapor pressure. DINP released to air may undergo rapid photodegradation and is not expected to be a candidate chemical for long-range transport.

Under indoor settings, air released DINP is both expected to partition to airborne particles at concentrations three times higher than in vapor phase (ECJRC, 2003a) and to have extended lifetime as compared to outdoor settings. The available information suggests that DINP's indoor dust concentrations to be associated with the presence of phthalate-containing articles, 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 DINP-containing products into indoor settings (Dodson et al., 2017).

DINP has a predicted average environmental half-life of 35 days. In situations where aerobic conditions are predominant, DINP is expected to degrade rapidly and be more persistent under anoxic/anaerobic conditions. In some sediments, landfills, and soils, DINP might be persistent as it is resistant to anaerobic biodegradation. In anerobic environments, such as deep landfill zones, hydrolysis is expected the most prevalent process for the degradation of DINP.

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 DINP

- is expected to undergo significant direct photolysis (Section 3.3);
- will partition to organic carbon and particulate matter in air (Sections 4);
- will biodegrade in aerobic surface water, soil, and wastewater treatment processes (Sections 5.2.1, 5.3.1, and 6.2);
- does not biodegrade in anaerobic environments (Section 5.2 and 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 expected to biodegrade under anoxic conditions and may have high persistence in anaerobic soils and sediments (Sections 3.1, 5.2.2, and 5.3.2);
- may show persistence in surface water and sediment proximal to continuous points of release (Sections 3.1, 5.2.2, and 5.3.2); and
- is expected to transform to monoisononyl phthalate, isononanol, and phthalic acid in the environment (Section 3).

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

- is expected to 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 (Section 6.3);
- has limited bioaccumulation potential (Section 7); and
- has shown no significant degradation via hydrolysis under standard environmental conditions but its hydrolysis rate has been 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 found 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.

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

Armstrong, DL; Rice, CP; Ramirez, M; Torrents, A. (2018). Fate of four phthalate plasticizers under various wastewater treatment processes. J Environ Sci Health A Tox Hazard Subst Environ Eng 53: 1075-1082. http://dx.doi.org/10.1080/10934529.2018.1474580

Barron, MG; Albro, PW; Hayton, WL. (1995). Biotransformation of di(2-ethylhexyl)phthalate by rainbow trout [Letter]. Environ Toxicol Chem 14: 873-876. http://dx.doi.org/10.1002/etc.5620140519

Bradlee, CA; Thomas, P. (2003). Aquatic toxicity of phthalate esters. In C Staples (Ed.), The Handbook of Environmental Chemistry, vol 3Q (pp. 263-298). Berlin, Germany: Springer-Verlag. http://dx.doi.org/10.1007/b11469

Chemical Manufacturers, A. (1984). Phthalate esters panel: Summary report: Environmental studies - Phase I. Generation of environmental fate and effects data base on 14 phthalate esters. Washington, DC.

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

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

EC/HC. (2015a). State of the science report: Phthalate substance grouping 1,2-Benzenedicarboxylic acid, diisononyl ester; 1,2-Benzenedicarboxylic acid, di-C8-10-branched alkyl esters, C9-rich (Diisononyl Phthalate; DINP). Chemical Abstracts Service Registry Numbers: 28553-12-0 and 68515-48-0. Gatineau, Quebec. https://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=47F58AA5-1

EC/HC. (2015b). State of the Science Report: Phthalates Substance Grouping: Long-chain Phthalate Esters. 1,2-Benzenedicarboxylic acid, diisodecyl ester (diisodecyl phthalate; DIDP) and 1,2-Benzenedicarboxylic acid, diundecyl ester (diundecyl phthalate; DUP). Chemical Abstracts Service Registry Numbers: 26761-40-0, 68515-49-1; 3648-20-2. Gatineau, Quebec: Environment Canada, Health Canada. https://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=D3FB0F30-1

ECB. (2008). European Union risk assessment report: Bis(2-ethylhexyl)phthalate (DEHP) [Standard]. In 2nd Priority List. (EUR 23384 EN). Luxembourg: Office for Official Publications of the European Communities. https://op.europa.eu/en/publication-detail/-/publication/80eaeafa-5985-4481-9b83-7b5d39241d52

ECHA. (2013). Evaluation of new scientific evidence concerning DINP and DIDP in relation to entry 52 of Annex XVII to REACH Regulation (EC) No 1907/2006. Helsinki, Finland. http://echa.europa.eu/documents/10162/31b4067e-de40-4044-93e8-9c9ff1960715

ECHA. (2016). Committee for Risk Assessment RAC - Annex 1 - Background document to the Opinion proposing harmonised classification and labelling at EU level of 1,2-Benzenedicarboxylic acid, di-C8-10-branched alkylesters, C9- rich; [1] di-“isononyl” phthalate; [2] [DINP] EC Number: 271-090-9 [1] 249-079-5 [2] CAS Number: 68515-48-0 [1] 28553-12-0 [2]. Helsinki, Finland. https://echa.europa.eu/documents/10162/23665416/clh_bd_dinp_7397_en.pdf/28ab9b6c-d3f4-

31f0-8be3-c1a1d4cf61d3

ECJRC. (2003a). European Union risk assessment report, vol 36: 1,2-Benzenedicarboxylic acid, Di-C9-11-Branched alkyl esters, C10-Rich and Di-"isodecyl"phthalate (DIDP). In 2nd Priority List. (EUR 20785 EN). Luxembourg, Belgium: Office for Official Publications of the European Communities. http://publications.jrc.ec.europa.eu/repository/bitstream/JRC25825/EUR%2020785%20EN.pdf

ECJRC. (2003b). European Union risk assessment report: 1,2-Benzenedicarboxylic acid, di-C8-10-branched alkyl esters, C9-rich - and di-"isononyl" phthalate (DINP). In 2nd Priority List, Volume: 35. (EUR 20784 EN). Luxembourg, Belgium: Office for Official Publications of the European Communities. http://bookshop.europa.eu/en/european-union-risk-assessment-report-pbEUNA20784/

Ejlertsson, J; Johansson, E; Karlsson, A; Meyerson, U; Svensson, BH. (1996). Anaerobic degradation of xenobiotics by organisms form municipal solid waste under landfilling conditions. Antonie Van Leeuwenhoek 69: 67-74. http://dx.doi.org/10.1007/BF00641613

ExxonMobil. (2010). Attachment 1a: Endpoint summaries for mammalian and environmental toxicity submitted in the REACH registration dossier for Diisononyl phthalate (DINP) and table of contents for DINP dossier information on the ECHA website and provided to CPSC on a DVD. Bethesda, MD: U.S. Consumer Product Safety Commission. https://www.cpsc.gov/s3fs-public/exxonDINP.pdf

Gobas, FAP; Mackintosh, CE; Webster, G; Ikonomou, M; Parkerton, TF; Robillard, K. (2003). Bioaccumulation of phthalate esters in aquatic food-webs. In CA Staples (Ed.), Handbook of environmental chemistry, vol 3Q (pp. 201-225). Berlin, Germany: Springer Verlag. http://dx.doi.org/10.1007/b11467

Gupta, S; Gadi, R. (2018). Temporal variation of phthalic acid esters (PAEs) in ambient atmosphere of Delhi. Bull Environ Contam Toxicol 101: 153-159. http://dx.doi.org/10.1007/s00128-018-2337-1

Hammel, SC; Levasseur, JL; Hoffman, K; Phillips, AL; Lorenzo, AM; Calafat, AM; Webster, TF; Stapleton, HM. (2019). Children's exposure to phthalates and non-phthalate plasticizers in the home: The TESIE study. Environ Int 132: 105061. http://dx.doi.org/10.1016/j.envint.2019.105061

Hasegawa, A. (2003). Determination of Phthalate Esters in the Atmosphere by LC/MS/MS. Bunseki Kagaku 52: 15-20. http://dx.doi.org/10.2116/bunsekikagaku.52.15

Haynes, WM. (2014). Diisodecyl phthalate. In CRC handbook of chemistry and physics (95 ed.). Boca Raton, FL: CRC Press.

Helmig, D; Bauer, A; Mueller, J; Klein, W. (1990). Analysis of particulate organics in a forest atmosphere by thermodesorption GC/MS. Atmos Environ A 24: 179-184. http://dx.doi.org/10.1016/0960-1686(90)90454-U

HSDB. (2015). Diisononyl phthalate (CASRN: 28553-12-0). Bethesda, MD: National Library of Medicine. https://pubchem.ncbi.nlm.nih.gov/source/hsdb/4491

Huang, J; Nkrumah, PN; Li, Y; Appiah-Sefah, G. (2013). 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

Johnson, BT; Heitkamp, MA; Jones, JR. (1983). Environmental and chemical factors influencing the biodegradation of phthalic acid esters in freshwater sediments with attachments [TSCA Submission]. (EPA/OTS Doc #40-8326268). University of Missouri School of Natural Resources Forestry Program. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0522411.xhtml

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

Kickham, P; Otton, SV; Moore, MM; Ikonomou, MG; Gobas, FAPC. (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, S; Kim, Y; Moon, HB. (2021). Contamination and historical trends of legacy and emerging plasticizers in sediment from highly industrialized bays of Korea. 765: 142751. https://heronet.epa.gov/heronet/index.cfm/reference/download/reference_id/7976686

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

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

Letinski, DJ; Connelly Jr, MJ; Peterson, DR; Parkerton, TF. (2002). Slow-stir water solubility measurements of selected alcohols and diesters. Chemosphere 43: 257-265. http://dx.doi.org/10.1016/S0045-6535(02)00086-3

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

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

Monsanto. (1978). Environmental Sciences Report: Biodegradabilty of plasticizers. (ES-78-SS-7). St. Louis, MO. https://ntrl.ntis.gov/NTRL/dashboard/searchResults.xhtml?searchQuery=OTS0200240

Monsanto. (1983). Ultimate biodegradation screening of selected phthalate esters [TSCA Submission]. (ES-80-SS-33. OTS0206236. 878210838. TSCATS/018418). https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0206236.xhtml

NCBI. (2020). PubChem database: compound summary: diisononyl phthalate. https://pubchem.ncbi.nlm.nih.gov/compound/Diisononyl-phthalate

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

NLM. (2015). PubChem: Hazardous Substance Data Bank: Di-isononyl phthalate, 28553-12-0 [Website]. https://pubchem.ncbi.nlm.nih.gov/compound/590836#source=HSDB

O'Grady, DP; Howard, PH; Werner, AF. (1985). Activated sludge biodegradation of 12 commercial phthalate esters. Appl Environ Microbiol 49: 443-445. http://dx.doi.org/10.1128/AEM.49.2.443-445.1985

O'Neil, MJ. (2013). Diisononyl phthalate. In MJ O'Neil; PE Heckelman; PH Dobbelaar; KJ Roman; CM Kenney; LS Karaffa (Eds.), (15th ed., pp. 517). Cambridge, UK: Royal Society of Chemistry.

Parkman, H; Remberg, M. (1995). Phthalates in Swedish sediments (pp. 27). (IVLB1167). Stockhold, Sweden: Swedish Environmental Research Institute.

Peterson, DR; Staples, CA. (2003). Series Anthropogenic Compounds Degradation of phthalate esters in the environment. In The Handbook of Environmental Chemistry book series. New York, NY: Springer-Verlag. http://dx.doi.org/10.1007/b11464

Solbakken, JE; Knap, AH; Orr, PL. (1985). Uptake and elimination of lindane and a phthalate ester in tropical corals and mussels. Mar Environ Res 16: 103-114.

SRC. (1983). 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. (1984). Activated sludge biodegradation of 12 commercial phthalate esters contract No. PE-17.0-ET-SRC [TSCA Submission]. (SRC-TR-84-643. OTS0508490. 42005 G9-2. 40-8426080. TSCATS/038162). Chemical Manufacturers Association. https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/OTS0508490.xhtml

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. (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?Dockey=10003NOR.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. (2012). Estimation Programs Interface Suite™ for Microsoft® Windows, v 4.11 [Computer Program]. Washington, DC. Retrieved from https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface

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). Manufacturer request for risk evaluation: Diisononyl phthalate (DINP). Washington, DC. https://www.epa.gov/sites/production/files/2019-06/documents/dinp_main_submission_epa_05_23_19.pdf

U.S. EPA. (2020). 2020 CDR data [Database]. Washington, DC: U.S. Environmental Protection Agency, Office of Pollution Prevention and Toxics. Retrieved from https://www.epa.gov/chemical-data-reporting/access-cdr-data

U.S. EPA. (2021). Final scope of the risk evaluation for di-isononyl phthalate (DINP) (1,2-benzene-dicarboxylic acid, 1,2-diisononyl ester, and 1,2-benzenedicarboxylic acid, di-C8-10-branched alkyl esters, C9-rich); CASRNs 28553-12-0 and 68515-48-0 [EPA Report]. (EPA-740-R-21-002). Washington, DC: Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/system/files/documents/2021-08/casrn-28553-12-0-di-isononyl-phthalate-final-scope.pdf

U.S. EPA. (2025a). Data Quality Evaluation and Data Extraction Information for Environmental Fate and Transport for Diisononyl Phthalate (DINP). Washington, DC: Office of Pollution Prevention and Toxics.

U.S. EPA. (2025b). Physical Chemistry Assessment for Diisononyl Phthalate (DINP). Washington, DC: Office of Pollution Prevention and Toxics.

U.S. EPA. (2025c). Risk Evaluation for Diisononyl Phthalate (DINP). Washington, DC: Office of Pollution Prevention and Toxics.

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

Wen, ZD; Huang, XL; Gao, DW; Liu, G; Fang, C; Shang, YX; Du, J; Zhao, Y; Lv, LL; Song, KS. (2018). Phthalate esters in surface water of Songhua River watershed associated with land use types, Northeast China. Environ Sci Pollut Res Int 25: 7688-7698. http://dx.doi.org/10.1007/s11356-017-1119-3

Yang, GCC; Liou, SH; Wang, CL. (2014). The Influences of Storage and Further Purification on Residual Concentrations of Pharmaceuticals and Phthalate Esters in Drinking Water. Water Air Soil Pollut 225: 1-11. http://dx.doi.org/10.1007/s11270-014-1968-z