

**Stockholm Convention
on Persistent Organic
Pollutants**

**Persistent Organic Pollutants Review Committee
Sixteenth meeting**

Geneva (online), 11–16 January 2021
Item 4 (b) of the provisional agenda**

**Technical work: consideration of a proposal for the
inclusion of UV-328 in Annexes A, B and/or C to
the Convention**

**Proposal to list UV-328 in Annex A to the Stockholm
Convention on Persistent Organic Pollutants****Note by the Secretariat****I. Introduction**

1. Switzerland has submitted a proposal to list UV-328 in Annex A to the Convention pursuant to paragraph 1 of Article 8 of the Convention (see annex to the present note). The proposal is being circulated as submitted and has not been formally edited. The Secretariat's verification of whether the proposal contains the information specified in Annex D is set out in document UNEP/POPS/POPRC.16/INF/6/Rev.1.

II. Proposed action

2. The Committee may wish:

- (a) To consider the information provided in the present note;
- (b) To decide whether it is satisfied that the proposal fulfils the requirements of Article 8 of and Annex D to the Convention;
- (c) To develop and agree on, if it decides that the proposal fulfils the requirements referred to in subparagraph 2 (b) above, a workplan for preparing a draft risk profile pursuant to paragraph 6 of Article 8.

* Reissued for technical reasons on 21 October 2020.

** UNEP/POPS/POPRC.16/1.

Annex

Proposal to list UV-328 in Annex A to the Stockholm Convention on Persistent Organic Pollutants

1. Introduction

1. UV-328 is a substituted phenolic benzotriazole (BZT) used as a UV absorber in many products. BZTs absorb the full spectrum of UV light and are mostly used in transparent plastics, coatings, and personal care products (PCPs). Due to their mechanism of action, their uptake of energy from UV light is reversible and non-destructive¹. BZTs are preferred for thermoset plastics, organic substrates, and coatings that function against weathering². UV-328 in particular can be used in many types of plastic polymer matrices, typically in concentrations between 0.1 and 0.5% of mass. However, the final amount can reach 1% of mass in some plastic matrices and 3% of mass in coatings³. UV-328 is used as a printing ink additive in food contact materials, too⁴. Because it is not bound to the polymer, UV-328 can migrate from within the polymer matrix and eventually diffuse out of the matrix and enter the environment.

2. For UV-328, there are currently nine active registrants/suppliers in the European Union (EU) under the REACH regulation (Registration, Evaluation, Authorisation and Restriction of Chemicals)⁵, and five in the United States of America (USA), under the Toxic Substances Control Act (TSCA)⁶. UV-328 is used worldwide in high volumes (tens of thousand tonnes). A large global manufacturer estimates that around 50% of the produced UV-328 is used in coatings, around 40% is used in plastics, rubber and polyurethanes (PUs), and the remaining 10% in cosmetics⁷. In Canada, in 1986, UV-328 was used for industrial purposes only (63% in plastics sector, 37% in paints and coatings). In 2000, the primary use was in automotive and plastics⁸. Based on information recently provided to the European Chemicals Agency (ECHA), UV-328 is used in a number of applications, including as a UV stabiliser in polyolefin and plastic shrink films, outdoor furniture and clear coat automotive finishes and for light stabilization in coatings, ABS resin, epoxy resin, fibre resin, PVC, unsaturated polyesters, polyacrylates and polycarbonates. It is particularly recommended as UV absorber for polyolefins, polyurethanes, PVC, polyacrylate, epoxy and elastomers. Further uses include construction materials, fillers, surface treatment, adhesives, paint/lacquers/varnishes, printing inks, consumer fragrances, fabric/textile/leather products and inert pesticides⁹.

3. Under the European REACH regulation, UV-328 has been identified as a substance of very high concern (SVHC) due to its PBT/vPvB (persistent, bioaccumulative, toxic/very persistent and very bioaccumulative) properties. On these grounds, in February 2020, UV-328 was added to Annex XIV (Authorisation List) of the REACH regulation⁵.

2. Chemical identity

2.1 Names and registry numbers

Table 1. Names and registry numbers of UV-328.

Common	UV-328
IUPAC	2-(2 <i>H</i> -Benzotriazol-2-yl)-4,6-bis(2-methylbutan-2-yl)phenol
CAS	Phenol, 2-(2 <i>H</i> -benzotriazol-2-yl)-4,6- <i>bis</i> (1,1-dimethylpropyl)-
Synonyms	2-(2 <i>H</i> -Benzotriazol-2-yl)-4,6-di- <i>tert</i> -pentylphenol
Commercial	BDTP, BLS 1328, Chiguard 328, Chisorb 328, Cyasorb UV 2337, Eversorb 74, GSTAB 328, Hostavin 3310 P, Kemisorb 74, Lowilite 28, Milestab 328, Seesorb 704, Songsorb 3280, Sumisorb 350, Thasorb UV328, Tin 328, Tinuvin 328, UV 2337, UV 74, Uvinul 3028, Viosorb 591
CAS RN	25973-55-1
EC No.	247-384-8

2.2 Structure

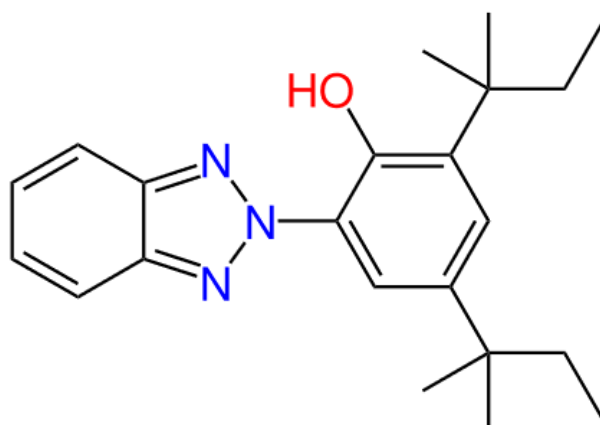


Figure 1. Chemical structure of UV-328.

Table 2. Molecular characteristics of UV-328.

Molecular formula	C ₂₂ H ₂₉ N ₃ O
Molecular weight	351.5 g/mol
SMILES code (canonical)	CCC(C)(C)c1cc(c(c1)n2nc3ccccc3n2)O)C(C)(C)CC
Chemical group	Organic
Chemical sub-group	Benzotriazol (BZT), phenol
Substance type	Mono-constituent
Degree of purity	≥ 80–100% (w/w)

2.3 Physico-chemical properties

Table 3. Physico-chemical properties of UV-328.

	Value	Source
Physical state	Yellow powder (20 °C, 101 kPa)	US EPA (2009), REACH registration dossier ¹⁰
Melting point	80–83 °C	Experimental, US EPA (2001)
	137 °C	Estimated (104–202 °C), US EPA
	202 °C	EPI Suite* (MPBPVP v1.43, Mean or Weighted MP)
Boiling point	Decomposition > 180 °C, before boiling	Experimental, Differential Scanning Calorimetry (DSC, 2013), REACH registration dossier ¹⁰
	> 230 °C	Estimated, Thermogravimetric Analysis (2012), REACH registration dossier ¹⁰
	478 °C	EPI Suite (MPBPVP v1.43, Adapted Stein & Brown method)
Vapour pressure	2.6 × 10 ⁻⁸ Pa (25 °C)	EPI Suite (MPBPVP v1.43, Modified Grain method)
	5.0 × 10 ⁻⁶ Pa (20 °C), 0.1 Pa (100 °C)	Experimental, DSC (1976), REACH registration dossier ¹⁰
Henry's law constant	6.5 × 10 ⁻¹³ atm·m ³ /mol	EPI Suite (HENRYWIN v3.20, Bond Method)

* Results modelled with EPI Suite™ v.4.10²².

	6.2×10^{-8} atm·m ³ /mol	OPERA [†]
pK_a	8.9±0.5 (acid), 0.7±0.3 (basic)	ACD/Labs, Classic Module Report
	10.3±0.8 (acid), -1.0±1.5 (basic)	ACD/Labs, GALAS Module Report
Water solubility	< 1 µg/L (20 °C, pH 6.3–6.4)	Experimental, EU Method A.6, Column Elution Method (2001), REACH registration dossier ¹⁰
	1.3×10^{-5} mg/L	Estimated (4.2×10^{-8} – 3.1×10^{-5} mg/L), US EPA
	0.015 mg/L	EPI Suite (WSKOW v1.42, from logK _{OW})
	0.42 mg/L	EPI Suite (WATERNT v1.01, from fragments)
	0.02 mg/L	Experimental, Dynamic Coupled Column ¹¹
Density	1.1 g/cm ³	Estimated (1.1–1.2 g/cm ³), US EPA
	1.2 g/cm ³ (20 °C)	Experimental, IA 79/1 (Air Comparison Pycnometer, 1976), REACH registration dossier ¹⁰
Air-water partition coefficient, logarithmic (logK_{AW})	-10.6	EPI Suite (KOAWIN v1.10, HenryWin est.)
Octanol-water partition coefficient, logarithmic (logK_{OW})	> 6.5 (23 °C, pH 6.4)	Experimental, OECD TG 117[‡] (2012), REACH registration dossier¹⁰
	7.3 (25 °C)	EPI Suite (KOAWIN v1.10, KowWin v1.68)
Soil adsorption partition coefficient, logarithmic (logK_{OC})	3.6	Estimated, US EPA
	5.2	EPI Suite (KOCWIN v2.00, Kow method), 2011
	5.6 (20 °C)	EPI Suite (KOCWIN v2.00, MCI method), 2011
Octanol-air partition coefficient, logarithmic (logK_{OA})	10.5	OPERA
	17.8	EPI Suite (KOAWIN v1.10, KOAWIN v1.10 estimate)

2.4 Tonnage

4. The OECD designated UV-328 as a high-production-volume chemical (HPVC). In Europe, UV-328 is fully registered under REACH in the tonnage band of 100–1,000 t/a¹⁰. ECHA recently listed UV-328 as a high-volume plastic additive used in the EU¹². According to the Substances in Products in the Nordic Countries (SPIN) database, the total use of UV-328 has been < 10 t/a in the Nordic countries (Denmark, Finland, Norway, and Sweden) since 2006. In Sweden, in 2015, there was a sharp increase up to 244 t, but it decreased to 1 t in 2016¹³. In the UK, UV-328 was part of a list with high priority for further investigation due to its PBT potential and for being in the European market in the range of 10–1,000 t/a¹⁴.

[†] Results modelled with OPERA¹⁸⁴.

[‡] Organisation for Economic Co-operation and Development Test Guidelines (OECD TG). The key for the tests' names is provided in Section 6.1.2.

5. In the USA, in 2011, the reported national aggregate production volume was around 1,000 t; from 2012 to 2016, it was around 450–4,500 t/a. UV-328 is not manufactured in Canada. Nevertheless, in 2000, 100–1,000 t were imported to be used as an UV absorber in automotive and industrial coatings, paints, and plastics. Between 2012 and 2013, the tonnage was 10–100 t⁸.

6. In Japan, UV-328 was manufactured and/or imported in the tonnage band of 1–1,000 t/a from 2012 to 2014, 1,000–2,000 t in 2015, and 1–1,000 t in 2016 and 2017¹⁵.

3. Information on UV-328 in relation to the Persistent Organic Pollutant (POP) screening criteria

3.1 Persistence

7. UV-328 is a persistent substance, as experimental results suggest a very low biodegradation potential^{1,10,15}. The EAWAG Biocatalysis/Biodegradation Database (EAWAG-BBD) prediction is provided in the Appendix (Section 6.2)¹⁶. Abiotic degradation of UV-328 is not expected to be relevant either¹. Due to its high $\log K_{ow}$ and $\log K_{oc}$, UV-328 adsorbs to (or absorbs into) suspended organic matter or sewage sludge, for example. This provides some level of protection from degradation. Hydrolysis (no hydrolysable structural element, low water solubility), oxidation and photo-transformation (UV absorber characteristics) are not expected to be significant either.

8. In a ready biodegradability test, after 28 days, 10 mg/L of UV-328 was 2–8% degraded (activated sludge, OECD TG 301 B, Good Laboratory Practice (GLP) was not applied)¹⁷. In a study with sludge-amended soils monitored over a year, UV-328 had disappearance half-lives (DT_{50}) of 179–218 days. The study has limitations, such as lack of homogeneous sampling, only dissipation monitoring, and a long analysis period over three years. Nevertheless, it is clear that UV-328 is very persistent in soils¹⁸. In another similar study, UV-328 had a DT_{50} of 99–223 days¹⁹.

9. An extensive monitoring data set from Narragansett Bay, USA, reported the presence of UV-327 and UV-328 in sediments decades after their release from manufacturing into the environment was stopped. Sediment cores from near the manufacturing plant were studied. These sediment samples were anaerobic. UV-328 production took place from 1970 to 1985¹¹ and the highest concentration recorded in the sediment core was 74 $\mu\text{g/g}$ in 1976²⁰. The concentrations near the surface remained 3–6 $\mu\text{g/g}$, which corresponds to more recent years. Similar historical concentration trends are described by Hartmann *et al.*, 2005²¹ (see Section 4.2.2).

10. The estimated DT_{50} for UV-328 is < 2 days in water (removal by sedimentation, not by degradation) and > 100 days in sediment, which is supported by BIOWIN v4.10 estimations²². According to AopWin v1.92²², the photodegradation half-life in the gas phase is 16.3 h, with an overall reaction rate constant of $15.8 \times 10^{-12} \text{ cm}^3/(\text{molecule}\cdot\text{s})$. The BIOWIN3 model generates a 74-day half-life in water. A 136-day half-life in soil is derived from this value ($1.85 \times$ half-life in water)^{23,24}.

11. As there are no simulation tests for water or sediment using UV-328, a read across was performed to cover this data gap. The substance M1 (molecular weight 339.4 g/mol, CAS RN 84268-36-0) is structurally very similar to UV-328 (substituents at phenyl group: *n*-propionic acid and *tert*-butyl vs. two *tert*-pentyl groups) and is a major degradation product of the BZT analogue EC 407-000-3¹ (see analogue structures in Section 6.5). M1 is formed in the water phase and is more hydrophilic than UV-328 (water solubility 102.4 mg/L, $\log K_{ow}$ 3.30²²). M1 dissipates rapidly, i.e., in a few days, to the sediment¹. There, it persists with a calculated DT_{50} up to 238 and 248 days, depending on the type of sediment. The different side chain of M1 (a propionic acid substituent is located in position 4 of the phenolic ring) degrades faster than that of UV-328. So, under the (reasonable) assumption that UV-328 and M1 fate properties are similar, M1 results may be expected to be a best-case representative of UV-328's DT_{50} and degradation half-life (DegT₅₀).

Conclusion on persistence:

12. UV-328 is highly hydrophobic, adsorbs and/or absorbs strongly to organic material, and has a low tendency to volatilise. When released to water, it will likely partition to particles and organic matter, suspended or deposited. Experimental and estimated data indicate that UV-328 does not degrade rapidly in water, soil, or sediment. Under a weight-of-evidence (WoE) approach to cover the experimental data gaps²⁵, the read across for its degradation in sediment from a structural analogue (another BZT substance) is also supportive of persistence. In addition, its presence in the environment, decades after releases stopped, indicates DegT₅₀ > 180 days. Thus, UV-328 fulfils the persistence criteria.

3.2 Bioaccumulation

13. UV-328 is considered to bioaccumulate, because it has a $\log K_{ow} > 5$, measured bioconcentration factors (BCFs) and modelled bioaccumulation factors (BAFs) above the bioaccumulation threshold, and low metabolic transformation rates. UV-328 will bioaccumulate in organisms mostly after uptake through diet. It has been detected in fish, several marine mammals, algae, and crustaceans.

14. Bioaccumulation in aquatic organisms was tested in two different studies in 2000 and 2007 (both studies: test species common carp, *Cyprinus carpio*, test protocol OECD TG 305 C)¹⁵. The first test lasted 60 days and no information on the use of a dispersant was given. The BCF was normalised to a 5% lipid content, calculated with the average lipid content of the test's start and end (Tables 4 and 5). The depuration half-life was 16 days (at 0.01 µg/L) and 33 days (at 0.10 µg/L). Additional data show BCF measurements for skin, head, innards, and edible parts. The highest BCFs were found in the following order: innards > head > skin > edibles (Table 6). The lowest BCF values were found for the highest concentrations, which might be linked to UV-328's low water solubility. UV-328 is a highly hydrophobic chemical ($\log K_{ow} > 4.5$) and, if a non-dietary exposure route is applied, UV-328 may not be dissolved completely in water and therefore could be available only partially for the uptake by the aquatic test organism. Thus, the resulting overestimation of the concentration of UV-328 in water could have led to underestimated BCF values^{1,15}.

Table 4. Sixty-days long BCF study: BCF and lipid normalisation (L/kg wet weight (ww)), based on nominal concentrations of the test substance in water. The average lipid content of the test fish was 4.19%^a or 3.26%^b 15,1.

Test concentration (µg/L)	BCF	BCF _{lipid-normalised}
0.1	940 ^a	1.1×10^3
0.01	620– 1.8×10^3 ^a	$740\text{--}2.2 \times 10^3$
0.01	2.4×10^3 ^b	3.7×10^3

Table 5. Sixty-days long BCF study: time evolution of the BCF (L/kg ww), based on nominal concentrations of the test substance in water^{15,1}.

Test concentration (µg/L)	Exposure time (days)				
	12	26	40	50	60
0.1	870	1.1×10^3	990	820	1.0×10^3
	570	1.4×10^3	780	1.0×10^3	1.0×10^3
0.01	620	890	1.5×10^3	1.3×10^3	1.0×10^3
	650	1.3×10^3	1.8×10^3	980	1.7×10^3

Table 6. Sixty-days long BCF study: BCFs in different tissues (L/kg ww), based on nominal concentrations of the test substance in water^{15,1}.

Test concentration (µg/L)	Skin	Head	Innards	Edible
0.1	770	1.4×10^3	2.3×10^3	600
	940	1.6×10^3	3.6×10^3	620
0.01	900	990	1.5×10^4	420
	2.0×10^3	2.3×10^3	3.6×10^4	840
0.01	2.3×10^3	3.7×10^3	1.4×10^4	1.6×10^3
	3.1×10^3	5.8×10^3	1.5×10^4	1.8×10^3

15. In the second study (OECD TG 305 C), the reported maximum BCFs were 5.6×10^3 (non-normalised) or 6.6×10^3 L/kg ww (lipid normalised) and the average lipid normalized BCF was 5.5×10^3 L/kg ww (Table 7). The fish had a lipid content of 4.2% at start. Apart from the slightly higher maximum values, the remaining BCF values were similar to each other and a steady state could be assumed. The average BCF at week eight was 4.59×10^3 L/kg ww for a lipid content of 4.2% and approximately 5.46×10^3 L/kg ww if the lipid content was normalised to 5%.

Table 7. BCF study during eight weeks, BCFs in L/kg ww, based on nominal concentrations of the test substance in water^{15,1}.

Exposure time (weeks)		0.8 µg/L		0.08 µg/L	
2	Non-normalised	1.3×10^3	1.3×10^3	2.3×10^3	2.3×10^3
	Lipid-normalised	1.5×10^3	1.6×10^3	2.7×10^3	2.7×10^3
4	Non-normalised	1.7×10^3	1.1×10^3	3.7×10^3	3.3×10^3
	Lipid-normalised	2.0×10^3	1.3×10^3	4.4×10^3	3.9×10^3
6	Non-normalised	1.7×10^3	2.8×10^3	4.4×10^3	5.6×10^3
	Lipid-normalised	2.0×10^3	3.3×10^3	5.2×10^3	6.6×10^3
8	Non-normalised	2.1×10^3	2.4×10^3	4.4×10^3	4.8×10^3
	Lipid-normalised	2.5×10^3	2.8×10^3	5.2×10^3	5.7×10^3

16. There are numerous findings of UV-328 in aquatic biota in monitoring studies, where concentrations of several hundred ng/g lipid weight (lw) have been measured²⁶⁻²⁸, see Section 4.2.4. UV-328 has also been detected in foodstuff and human adipose tissue²⁹. The occurrence of several BZTs in blubber of five finless porpoise in the Ariake Sea, Japan, was monitored from 1998 to 2009. In the blubbers of the studied finless porpoises, on average, it was found 29 ng/g ww of UV-328 and 14 ng/g ww of UV-327. These values are equivalent to 19 ng/g lw of UV-327 and 38 ng/g lw of UV-328, adjusted to the blubber lipid content of each analysed specimen. When calculating the total load of BZTs, blubber weight was considered to be 28.8% of the whole-body weight. This generated whole-body concentrations of 4.0 ng/g ww of UV-327 and 8.4 ng/g ww of UV-328. Thus, the BAF[§] of UV-327 between water and this marine mammal was estimated to be 3.3×10^4 L/kg ww (4 ng/g ww/0.12 ng/L), which is around one order of magnitude higher than what is reported for small fish (3.2×10^3 L/kg ww = 0.39 ng/g ww/0.12 ng/L). These BAFs for UV-327 were calculated using 0.12 ng/L as a reference value for concentrations found in environmental aquatic samples in Japan. Such a reference environmental value was not given for UV-328, but a comparison can be done. In 2001, the annual production and import tonnage of UV-327 in Japan was 100–1,000 t and 1,000–10,000 t for UV-328. The potentially higher emissions of UV-328 may partly be compensated by its lower fraction remaining in the water phase. Since the mean concentration in this study for UV-328 (8.4 ng/g ww) is two times higher than UV-327 (4.0 ng/g ww), one may assume the BAF for UV-328 will be similar as for UV-327³⁰. If the UV-327 environmental reference value would be used for UV-328, the estimated BAF for UV-328 will be 7.0×10^4 L/kg ww (8.4 ng/g ww/0.12 ng/L). These BAFs can be lipid-normalised to a 5% lipid content. UV-327 and UV-328 will then have BAFs of 8.0×10^3 L/kg lw and 1.6×10^4 L/kg lw, respectively. The detailed values are described on Section 6.3.1. If the same terms of the above finless porpoises study³⁰ are applied to the study of Nakata *et al.*, 2009 with small fishes, lipid-normalised BAFs for UV-327 and UV-328 in small fishes would be 6.7×10^3 L/kg and 4.2×10^3 L/kg, respectively²⁷. See Section 6.3.2 for more details.

17. To comprehend how phenolic BZTs will likely enter the food chain, the potential role of benthic animals needs to be considered. These will filter-feed on suspended matter or ingest sediment particles, where BZTs will possibly be adsorbed strongly. This scenario fits the expected nutritional habits of finless porpoises, which feed on small fish, shrimps, and cephalopods, which, in turn, feed on benthic organisms. Due to the chemical similarity of UV-327 and UV-328 (only difference: *tert*-butyl vs. *tert*-pentyl groups), a read-across can be safely pursued¹.

18. As pointed out earlier, different bioaccumulation values generated from laboratory bioconcentration experiments may not adequately account for the bioaccumulation of substances via diet, which can be a decisive factor for chemicals with $\log K_{OW} > 4$ ³¹. For these substances, the BAF is substantially greater than the BCF³², because BCFs only account for the exposure from water (respiratory) and do not consider uptake from food. Thus, a BAF with metabolism correction is a more appropriate parameter to characterize the bioaccumulation potential. Metabolic transformation in aquatic organisms of high- K_{OW} chemicals is not expected to be significant. UV-328 is predominantly present in its neutral form under environmental conditions (Table 3) and has a slow metabolism rate. In the Canadian assessment of UV-328, the metabolic rate constant (k_M) was calculated as 0.01/day in a 184 g fish. This k_M is considered low when compared to other organic chemicals^{33,34}. Hence, this estimation supports the claim that when UV-328 is consumed by a higher trophic level predator, biomagnification will likely occur due to low metabolism. The BAF of UV-328 is estimated at

[§] The original text reports the estimated bioaccumulation parameter as a BCF. However, this is based on samples collected from the field. Thus, by definition, the estimated values in this study are BAFs.

approximately 8.7×10^4 L/kg ww in mid-trophic level fish, indicating significant BMF in aquatic organisms when considering food intake, according to the AQUAWEB model^{8,35}. An unpublished food web study from Hamilton Harbour, Canada, shows possible trophic magnification of UV-328³⁶. The estimation from EPI Suite (Table 8) also predicts the bioconcentration and bioaccumulation of UV-328 in the marine food web.

Table 8. EPI Suite estimation results for UV-328 obtained with BCFBAF v.3.01²².

BCF (regression-based method)	6.0×10^3 L/kg ww
Biotransformation Half-Life (Fish)	14.3 days
BCF Arnot-Gobas method (upper trophic)	1.1×10^3 L/kg
BCF Arnot-Gobas method (mid trophic)	1.5×10^3 L/kg
BCF Arnot-Gobas method (lower trophic)	1.7×10^3 L/kg
BAF Arnot-Gobas method (upper trophic)	9.3×10^4 L/kg
BAF Arnot-Gobas method (mid trophic)	1.5×10^5 L/kg
BAF Arnot-Gobas method (lower trophic)	2.0×10^5 L/kg

Conclusion on bioaccumulation:

19. Experimental and estimated values of $\log K_{OW}$ identify UV-328 as bioaccumulative according to the Stockholm Convention threshold ($\log K_{OW} > 5$). There are also several experimental values where the BCF is $> 5 \times 10^3$ L/kg ww. Different estimation models suggest a bioaccumulative potential, too, with BCF and BAF values $> 5 \times 10^3$ L/kg. UV-328 has also been detected in the marine food web and there is evidence that it biomagnifies in the food chain. Therefore, UV-328 fulfils the criteria for bioaccumulation.

3.3 Potential for long-range transport

20. UV-328 is not expected to undergo atmospheric long-range transport in the gas phase due to its low vapour pressure, low Henry's law constant, and short estimated half-life in air^{8,22}. However, its high $\log K_{OW}$ and $\log K_{OC}$ values imply that UV-328 will strongly partition into organic matter, including absorption into and adsorption onto aerosol particles in air, as well as to suspended solids in water. The $\log K_{OA} > 10$ represents a partitioning into atmospheric aerosol particles that is virtually irreversible³⁷, meaning that the fraction in the gas phase is extremely small. Once adsorbed to aerosol particles in air, UV-328 will travel with these particles, and will undergo long-range transport with the particles and subsequently be deposited to soil, vegetation, and water. This atmospheric transport of aerosol particles has extensively been described in the scientific literature, e.g. for larger mineral dust particles coming from the Sahara desert passing over the Atlantic Ocean, over a distance up to 3.5×10^3 km³⁸.

21. UV-328 is designed to be photostable and it is, therefore, mostly unaffected by photodegradation and/or oxidation. Hydrolysis is also unlikely due to very low water solubility (high $\log K_{OW}$ and $\log K_{OC}$), a strong chemical bond between the BZT group and the aromatic ring, and resistant side chains¹. Under specific environmental conditions, such as in the ocean, UV-328 may be partly in an anionic form or form an intramolecular hydrogen bond³⁹⁻⁴¹. Charged molecules have a lower affinity for sorption to suspended matter and thus have a lower sedimentation rate, which increases the potential for long-range transport.

3.3.1 OECD POV-LRTP Tool (OECD Tool)

22. With the OECD Tool, a decision support tool for environmental persistence and long-range transport potential (LRTP)⁴², and the input data presented in Table 9, the overall persistence (P_{OV}) of UV-328 is determined as 196 days, its characteristic travel distance (CTD) as 2.8×10^3 km, and its transfer efficiency (TE) as 12.4%. The Monte Carlo analysis and a brief discussion on the $\log K_{AW}$ input values and their impact showing the uncertainty of these results are provided in Figure 4, Section 6.4. These results place UV-328 in a position of typical POP-like substances, see Figure 2. In comparison with acknowledged POPs, such as hexabromocyclododecane (HBCDD) and α -endosulfan, UV-328 has similar P_{OV} , CTD, and TE. For the LRTP metrics, CTD and TE, also the results of, e.g. decaBDE and octaBDE are similar (Figure 2 and Table 23, Section 6.4).

Table 9. OECD Tool input data for UV-328. Values from EPI Suite²²: ^a KOAWIN v1.10 (HenryWin est), ^b KOAWIN v1.10 (KowWin v1.68), ^c AopWin v1.92, ^d BIOWIN3 (BIOWIN v4.10), and ^e calculated $t_{1/2}$ in soil ($1.85 \times$ half-life in water)²³.

Molecular weight (g/mol)	351.5
^a log K_{AW}	-10.6
^b log K_{ow}	7.3
^c $t_{1/2}$ in air (h)	16.3
^d $t_{1/2}$ in water (h)	1.8×10^3
^e $t_{1/2}$ in soil (h)	3.3×10^3

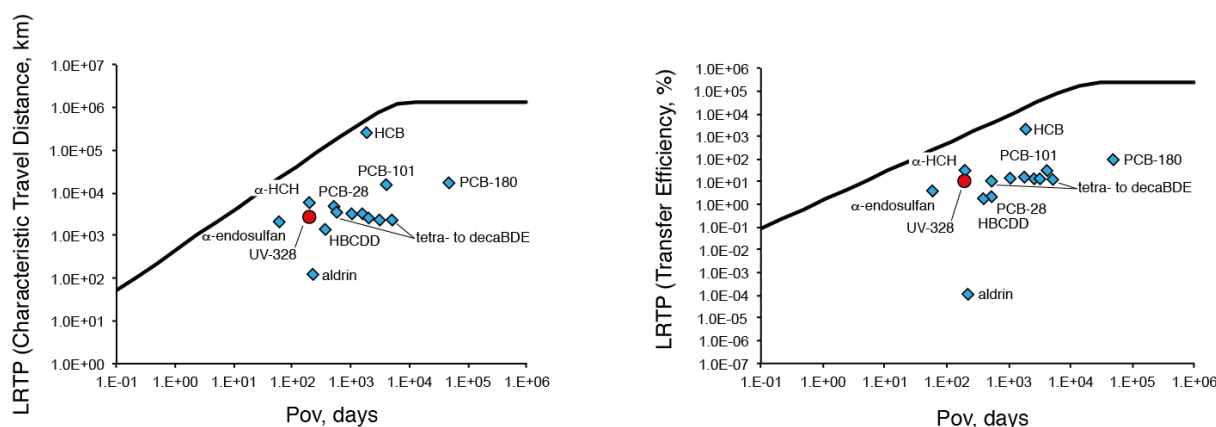


Figure 2. LRTP- P_{OV} plot comparing UV-328 (red dot) and benchmark POPs (blue diamonds) for CTD, TE and P_{OV} (adapted from ⁴²). The bold black line shows the LRTP of highly volatile substances. The input data (Table 22) and generated values (Table 23) are provided in Section 6.4.

3.3.2 Field Data

23. The detection of UV-328 in remote locations has not been extensive. UV-328 is not yet a chemical routinely measured in samples from remote locations and the limited field data are not conclusive. Still, UV-328 has been detected in Swedish background sites in storm water and sediments, but not in the air⁴³. UV-328 has been detected in biota in Lake Superior, Great Lakes⁴⁴, and in the Canadian⁴⁵ and Norwegian Arctic⁴⁶. Around Lake Superior, UV-328 had up to 100% DF in herring gull eggs. The gulls in Lake Superior feed more frequently from terrestrial sources than gulls from other locations in the Great Lakes area, which are mostly piscivores. Thus, the herring gulls from this area end up ingesting small mammals and plastic debris containing UV absorbers more often. In Prince Leopold Island (Nunavut, Canada), UV-328 was detected in one bird liver (11% DF). On Svalbard Island, Norway, UV-328 was not detected in the air, but had a 60–100% DF in Arctic biota, except in polar bears, which were in the most remote sampling location⁴⁶. The absence of detection in polar bears may be caused by the fact that blood plasma, but not adipose tissue, was collected. As UV-328 is hydrophobic, plasma as a hydrophilic body fluid is not expected to be a reservoir of UV-328 in mammalian organisms. A more informative approach would be to sample the polar bears' adipose tissue. UV-328 has also been detected in Pacific Ocean wildlife^{28,47}. More details are provided in Section 4.2.6.

3.3.3 Transport by Environmental Carriers

24. UV-328 is transported with particles to which it is adsorbed or absorbed, such as dust, sediments, migratory animals, or through matrices in which it is included as additive, e.g. polymers. In addition, although in small amounts, it is likely that migratory animals also carry UV-328 to remote locations either through suspended solids or sediment material trapped in their body (e.g., paws, feathers), in the stomach content after feeding off contaminated biota²⁸ (e.g. seafood, fish), accumulated in tissues (e.g. liver, muscle)^{30,44–46,48,49}, or in plastic debris⁵⁰ (e.g. tangled fish nets in bird feet). In the following section, the long-range transport of UV-328 by plastic debris is explored in more detail.

3.3.3.1 Plastic Debris Pathway

25. It is estimated that 8,300 million tonnes (Mt) of virgin plastics have been produced until 2017⁵¹ and the global annual production of plastics reached 348 Mt in 2017⁵². 79% of plastic waste may be disposed of in landfills or in the environment⁵¹, of which about 8 Mt end up in the ocean every year⁵³. This material persists in the marine environment for decades to centuries and parts of it are transported over long distances to remote regions. Nowadays, plastic debris is present throughout the globe, even in remote places. Microplastics (MPs) account for 13% of the global marine plastic debris mass and 92% of the number of global plastic pieces. MPs have been identified in remote regions, such as in the Poles⁵⁴, particularly in sea ice⁵³, south of Svalbard, Norway⁵⁵, or on the Tibetan Plateau⁵⁶. Henderson Island in the Pacific Ocean, 5×10^3 km away from any major source of pollution and 115 km away from the nearest human settlement of around 40 people, shows a very high density of plastic debris, actually the highest in the world⁵⁷. In the Indian Ocean, the Coco (Keelings) Islands and the isolated tropical atoll of Saint Brandon's Rock also show high concentrations of plastic debris and no relevant nearby sources^{58,59}. A significant portion of marine plastic debris is transported to plastic gyres, which can themselves be located in remote locations such as the South Pacific Ocean⁶⁰. These gyres can be chemical contamination hot spots, as it was found to be the case of organophosphorus esters accumulated in ocean gyres that transferred into oceanic aerosols⁶¹. Once debris reaches the central part of the gyre, it mostly remains stationary. However, fractions may leave and travel further, so the gyre itself acts as a reservoir^{62,63}.

26. Plastics are weathered by biodegradation, photo-degradation, thermo-oxidative and thermal degradation, or hydrolysis⁶⁴. Weathering modulates bioavailability, too, because in eroded pellets chemical additives have increased distribution coefficients and slower distribution kinetics⁶⁵. As different polymers have different densities, some float in seawater (PE, PP, expanded polystyrene (EPS), PU etc.), travelling globally via wind and oceanic currents, while others sink into the benthos (e.g. polyvinyl chloride, PVC)^{66,67}. Adsorption and diffusion of organic pollutants differ according to the material's structure and the environmental conditions. Leaching can be explained by diffusion in the plastic matrix and across the interface of plastic and water, and mass transfer within the surrounding boundary layer⁶⁸. The plastic polymer–water partition coefficient ($\log K_{PW}$) is generally linearly proportional to the $\log K_{OW}$ ⁶⁹. High-trophic-level organisms can be exposed via direct or indirect uptake of MPs, depending on feeding habits⁶⁷.

27. Plastics may contribute to the exposure of surrounding environments to chemicals that originate from the plastic matrix (additives) and/or chemicals that are sorbed to them (contaminants already present in the environment)⁷⁰. Substances that are not chemically bound to the polymer matrix diffuse out of the matrix and enter the surrounding environment^{71–74}. According to model calculations, around 2% of plastic additives are emitted to the environment every year⁷⁵. The “*Emission Scenario Document On Plastic Additives*” from the OECD (2009) estimated the emission rates of plastic matrices during lifetime of outdoor service for many types of chemicals⁷⁶. For example, plasticisers, flame retardants, and UV absorbers have an estimated rate of leaching to the environment (water) of 0.16% of mass times their time-of-service in years. Time-of-service can be anywhere from zero to 50 years⁷⁷. The additives emission rate into the atmosphere was 0.05% of mass over lifetime.

28. Around 78% of 126 priority pollutants listed by the US Clean Water Act are associated with plastic debris^{78,79}. Several chemical additives have been detected in plastic debris. Some are known components of plastic matrices, e.g. hexabromocyclododecanes (HBCDDs), some are probably adsorbed from the environment, e.g. dichlorodiphenyltrichloroethane (DDT)^{80–83}. HBCDDs were detected in EPS marine debris. The high HBCDD levels found in mussels inhabiting this debris indicate that transfer of HBCDD occurred from the EPS polymeric matrix⁸⁴. There are reports exploring the link between plastic debris presence in remote locations and the concomitant transport of chemicals with POP-like properties. For instance, the presence of perfluorohexanoic acid (PFHxA, 0.3–1.0 ng/g dry weight (dw), 100% DF), cypermethrin (< 0.3–6.5 ng/g dw, 50% DF), and bisphenol A (BPA, < 20–24 ng/g dw, 50% DF) has been reported in marine plastic debris recovered from northern Norway⁴⁶. Some of these chemicals are plastic additives, e.g. BPA, some are adsorbed from the environment, e.g. cypermethrin.

i. Evidence of UV-328 transport in plastic particles to remote areas

29. One of UV-328's major use is as additive (UV absorber) in many polymers⁸², representing around 40% of its total global production⁷. For this reason, UV-328 is expected to be present in plastic gyres in the world's oceans. Importantly, plastic matrices function as primary sources of UV-328 releases to the surrounding environment, and simultaneously work as a carrier of additives throughout their environmental distribution pathway⁸⁵.

30. After weathering, UV-328 was still frequently detected in debris from plastics products in significant concentrations when compared to fresh material, due to its persistence⁸⁶. At the Kauai Island, Hawaii, close to the North Pacific subtropical gyre, the monitored plastic debris contained a range of additives. UV absorbers, including UV-328, were found in 33% of the larger plastic fragments (1.5–8 cm) and other additives were found in 13% of the smaller fragments (4–7 mm). Such findings show that additives are released to some extent during fragmentation of the original plastic but the release rate is low enough so that certain amounts are still available for long-range marine transport⁸⁷. Flame retardants leaching was increased in finer particles of acrylonitrile-butadiene-styrene (ABS) polymer⁸⁸. Therefore, over time, degradation and fragmentation of the original plastic matrix will likely result in an increased leaching of UV-328 and contamination of the surrounding medium. Considering that UV-328 is highly persistent and bioaccumulative, with low to negligible biotransformation⁸⁹, toxic effect levels could be achieved with time, see Sections 3.4 and 4.

ii. Release potential in remote areas

31. UV-328 is a plastic additive not covalently-bound to the polymer matrix¹² and its diffusion depends mostly on polymer structure and water temperature^{84,90,91}. Depending on chemical properties, the overall leaching of additives is determined by internal diffusion in the plastic or by the aqueous boundary layer diffusion. With increasing $\log K_{PW}$, the loss of additives from plastic particles is slower and is more likely to be limited by the aqueous boundary layer diffusion⁶⁸. Ziccardi *et al.*, 2016 listed no $\log K_{PW}$ value for UV-328, but did for DDT and di-2-ethylhexyl phthalate (DEHP) (Table 10). As UV-328's $\log K_{OW}$ and molecular weight is in the same range as DDT and DEHP, also its $\log K_{PW}$ can be assumed to be in the same range as for DDT and DEHP⁹². Further, these substances are present in freely diffusible form in the plastic matrix, which may well be the case of UV-328, too. Nevertheless, physical and chemical conditions during formulation may limit the freely diffusible fraction of additives, and thus, leaching⁶⁸. ECHA has released a report discussing parameters to predict the release potential of chemicals from a solid matrix through diffusion or partitioning⁸⁵.

Table 10. Comparison of physico-chemical properties between UV-328 and two chemicals with derived $\log K_{PW}$ values. Values from: ^a KOAWIN v1.10 (KowWin v1.68), ^b KOCWIN v2.00 (MCI method), and ^c a review paper (in relation to PE or PVC)⁹³.

	Molecular weight (g/mol)	^a $\log K_{OW}$	^b $\log K_{OC}$	^c $\log K_{PW}$
UV-328	351.5	7.3	5.6	–
DDT	354.5	6.8	5.2	5.6 (PE) 5.0 (PVC)
DEHP	390.6	7.5	5.1	4.1 (PVC)

32. In addition, plastic debris carrying UV-328 may well accumulate in biological tissues and, thus, make its way up the food chain. It has been reported that plastic particles of certain sizes can pass from the digestive tract of mussel (*Mytilus edulis*) into its circulatory system⁹⁴. When this kind of transfer occurs, longer timeframes of exposure would have to be considered, resulting from cumulative episodes⁹⁵. The leaching of flame retardants included in plastic matrices, e.g. BTBPE (molecular weight 687.6 g/mol, $\log K_{OW}$ 9.15²²) and decabromodiphenyl ethane (molecular weight 971.2 g/mol, $\log K_{OW}$ 13.6²²), into the digestive fluids of birds has already been reported. The leaching proportions were higher in finer sizes of plastic and with increasing $\log K_{OW}$. There was a significant contribution of the ingested plastic to the bioaccumulation of highly hydrophobic flame retardants in the studied birds⁸⁸. A previous study suggested also the transfer of polybrominated diphenyl ethers (PBDEs) (molecular weight 801.4–959.2 g/mol, $\log K_{OW}$ 10.3–12.1²²) from ingested plastics to the tissue of seabirds, such as abdominal adipose and liver tissues. Model calculations and biomonitoring data suggested higher exposure through plastic than from prey. The stomach oil of seabirds (diet-derived) acts as an organic solvent and accelerates PBDEs leaching. This study also noted that other organic digestive fluids, e.g. bile, may also facilitate leaching and bioaccumulation of chemicals from ingested plastics⁹⁶. Thus, oily components in the digestive fluid facilitate leaching of hydrophobic plastic additives and their accumulation in adipose and hepatic tissues⁹⁷. PBDEs were also found in the abdominal adipose tissue of other seabirds (*Puffinus tenuirostris*) in the North Pacific Ocean⁹⁸. Importantly, UV-328 has been detected, among other common chemical additives, in PP plastic fragments ingested by seabirds⁹⁹. This study hypothesises high- K_{OW} chemicals can be retained in plastics during fragmentation and transport in the ocean until seabirds ingest them, for instance. The authors calculated that by day 15 to 16, 42% of UV-328 had leached out of the plastic pellets, and by day 32, 60 % had leached out. Hydrophobic additives leached in greater amounts after facilitated

diffusion from the polymer matrix, which may occur due to swelling caused by the stomach oil. Furthermore, accumulation of plastic-derived chemicals, including UV-328, in seabird tissue has been recently demonstrated based on an *in vivo* plastic feeding experiment under semi-field conditions¹⁰⁰. In Japan, streaked shearwater (*Calonectris leucomelas*) chicks were fed with environmentally relevant doses of plastic resin pellets compounded with one flame retardant and four UV absorbers. After sacrifice, all exposed chicks had the administered pellets unaltered in their digestive tract. Along with other chemicals, UV-328 was found in liver, preen gland oil, and adipose tissue of field and semi-field seabirds. The level of UV-328 found in liver samples indicates no relevant metabolization, as the UV-328 profile increased during the collection period, 16 to 32 days, with a maximum exposure ratio of 1.9×10^3 (calculated from the concentrations of additives in tissues in the exposed group by those in the control group)¹⁰⁰. This means the birds' exposure to additives was higher through plastics than from environmental media. The findings in preen gland oil demonstrate the factual transfer of additives from plastics into tissues. In the Pribilof Islands, in the Bering Sea around 500 km west of Alaska and 400 km north of the Aleutian Islands¹⁰¹, one sample of preen gland oil of thick billed murre had 654 ng/g-lipid of UV-328¹⁰². According to a global survey carried out by this same Japanese research group, around 24% of seabirds are estimated to bioaccumulate plastic additives, based on measurements of BDE-209, DBDPE and DEHP in preen gland oil¹⁰³.

Conclusion on the potential for long-range transport:

33. UV-328 does not primarily undergo long-range transport in the gas phase, due to its physico-chemical properties. However, several pieces of information do indicate a relevant LRTP of UV-328. The OECD Tool shows CTD and TE greater than for some already acknowledged POPs. UV-328 is also likely to be transported via water and/or air while adsorbed to suspended particles, based on its high K_{OC} and K_{OA} , respectively. Moreover, UV-328's ubiquitous presence worldwide, from the Pacific Ocean to the Arctic, provides evidence of long-range transport of UV-328 in the environment, including into remote locations. As an additional pathway, UV-328 has been found to be transported with, and subsequently, released from plastic debris as it is used in significant amounts and has physico-chemical properties compatible with diffusion from plastics. The transport with plastic matrices is long-range and transfers UV-328 to remote locations in direct connection with the plastic particles. Uptake of UV-328 contained in plastic particles by seabirds has been demonstrated. Therefore, UV-328 has the potential for long-range environmental transport.

3.4 Adverse effects

3.4.1 Toxicity

34. The Risk Assessment Committee (RAC) of the ECHA and the registrants concluded that UV-328 meets the criteria for STOT RE 2 (specific target organ toxicity – repeat exposure in sub-category 2), according to the Classification, Labelling and Packaging (CLP) Regulation EC 1272/2008^{1,10,104}. This classification is based on sub-acute (49 days) and sub-chronic (90 days) repeated-dose toxicity studies conducted in rats. Repeated oral (gavage) administration of UV-328 caused toxicity in several organs, in particular in the liver. Modelling suggests UV-328 will not be ionised in the small intestine and it is likely that it will be absorbed in the gastrointestinal tract⁸. According to UV-328's hydrophobic properties, liver will be the main metabolism site and metabolites will be excreted mostly via kidneys. Dermal uptake is unlikely¹⁰.

3.4.1.1 Acute toxicity

35. In an oral gavage study in rats and mice, no gross organ changes and an oral LD₅₀ (lethal dose) around 2.3 g/kg body weight (bw) was reported, after single exposure (non-GLP)¹⁰⁵. In a Ciba-Geigy study (similar to OECD TG 401, non-GLP, 1978), the oral LD₅₀ in rats was > 7.75 g/kg bw, after single administration. A study with albino rats (similar to OECD TG 401, non-GLP, 1987), led to an oral LD₅₀ > 2.0 g/kg bw. These results were in accordance with other studies, too¹⁰.

36. The measured acute inhalation LC₅₀ (lethal concentration) in rats was from 0.4–4.1 g/L¹⁰⁵. A Ciba-Geigy study (similar to OECD TG 403, non-GLP, 1973) in rats generated an LC₅₀ > 0.4 mg/L, after single exposure for 4 h. Another study (similar to OECD TG 403, non-GLP, 1977) in rats reported an LC₅₀ > 0.13 mg/L air, after 1 h¹⁰. Measured dermal LD₅₀ in rabbits was from 1.1–3.0 g/kg bw¹⁰⁵. The results are based on a Geigy Ltd. study (similar to OECD TG 402, non-GLP, 1969), after single exposure. No dermal irritation/sensitisation or eye irritation was reported¹⁰.

3.4.1.2 Repeated-dose toxicity

37. In the study of Til *et al.* (1968) male and female rats were fed a diet containing UV-328 for 49 (short-term) and 90 (subchronic) days (test protocol similar to OECD TG 408, non-GLP, 1968)¹⁰⁶. The main organs affected were liver and kidneys. The NOAEL (no observed adverse effect level) was 100 ppm of a UV-328 dose, equivalent to around 22 mg/kg bw/day for rats, in a test range of 100–1,600 mg/kg. Microscopic examination showed liver and kidney changes. Focal necrosis of the liver and tubular nephrosis at the feeding level 52.7–98.7 mg/kg bw/day met the criteria of significant toxicity to human health, at exposure concentrations meeting guidance values for category STOT RE 2 (10 mg/kg bw/day < C ≤ 100 mg/kg bw/day). The calculated LOAEL (lowest adverse effect level) was of 10 mg/kg bw and the NOAEL was < 10 mg/kg bw^{104,106}.

38. Beagle dogs were given UV-328 via diet for 90 days (similar to OECD TG 409, non-GLP) in a concentration range of 15–240 mg/kg bw/day. The main target organs were again liver and kidney. Some animals of the higher-dose groups also had alterations in reproductive organs. The NOEL (no observed effect level) in this study was < 15 mg/kg bw/day and the NOAEL was 30 mg/kg/day. The pathological changes in liver and kidneys observed at lower dose levels did not meet the criteria defined in the CLP regulation. However, the histopathological effects observed in dogs exposed to 60 mg/kg bw/day met these criteria. The changes in the activity of several enzymes in serum and changes observed in protein pattern in serum in animals exposed to > 15 mg/kg support classification as STOT RE^{104,107}. In another dietary study with beagle dogs (similar to OECD TG 409, non-GLP, 1981), 91 days, the NOEL was 31.75 mg/kg bw/day for males and 34.6 mg/kg bw/day for females, and there were no gross or histopathological changes related to treatment^{108,109}.

3.4.1.3 Genotoxicity and reproductive toxicity

39. There are no carcinogenicity studies for UV-328. No genotoxicity, mutagenicity¹¹⁰, reproductive or developmental toxicity have been reported. There are no experimental data regarding toxicity to reproduction.

3.4.1.4 Endocrine and metabolic assessments

40. After metabolic activation by human CYP3A4 enzyme-hydroxylation, a more potent anti-androgenic activity was observed at 0.25 mM for UV-328. Metabolites of UV-328 formed by human CYP3A significantly enhanced the anti-androgenic activity toward the human androgen receptor¹¹¹. UV-328 showed no relevant estrogenic activity¹¹². Both studies are based on two-hybrid yeast bioassays.

3.4.2 Ecotoxicity

41. Ecotoxicity has not been observed in standard tests^{1,15}. Yet, it is predicted by the Danish (Q)SAR¹¹³ and ECOSAR²². ECOSAR predicts a chronic value (ChV, geometric mean of NOEC (no observed effect concentration) and LOEC (lowest observed effect concentration)) and LC₅₀/EC₅₀ < 0.1 mg/L (Table 11)²². A PNEC (predicted no effect concentration) for UV-328 of 1 µg/L in marine water and of 45.1 mg/kg sediment dw in marine sediment are reported in the registration dossier for UV-328.

42. The only experimental toxicity data available are from acute toxicity studies on aquatic organisms reporting no effect at the water saturation level, which, given the poor bioavailability of UV-328 in water, would not be the adequate route to reach the internal effect concentration in the test organisms. The estimated results suggest risk to aquatic organisms in the surrounding Canadian environment is low, as well as the risk to terrestrial wildlife associated with a long-term consumption of UV-328-contaminated fish⁸. Again, most testing conditions were above water solubility for UV-328.

Table 11. ECOSAR v1.11 results for the BZTs class²². Results are in mg/L.

	ChV	LC ₅₀	EC ₅₀
Fish	7.4 × 10 ⁻⁴	0.06 (96 h)	–
Daphnid	1.6 × 10 ⁻³	0.06 (48 h)	–
Green algae	0.02	–	0.04 (96 h)

3.4.2.1 Short-term

43. No mortality or toxic effect have been observed in fish and crustaceans at 10 mg/L. In algae, there was some effect observed at the lowest concentration after 72 h. However, the EC₅₀ was expected to be > 10 mg/L. In a growth-inhibition study with *Pseudokirchneriella subcapitata* (algae), no effect was observed in a limit test after 72 h, generating a NOEC of 0.016 mg/L (OECD TG 201, semi-static, GLP, 2007)^{10,15}. In another algae (*Scenedesmus subspicatus*) growth inhibition after 72 h was found in all concentrations including the lowest (0.1 mg/L), resulting in a NOEC < 0.1 mg/L. In another study (OECD TG 201, non-GLP, 1993), an EC₅₀ > 10 mg/L was reported after 72 h¹¹⁴. In microorganisms (activated sludge), the EC₅₀ and IC₅₀ after 3 h were > 100 mg/L under static conditions (1988, OECD TG 209, non-GLP)¹⁰.

44. An aquatic invertebrate, *Daphnia pulex* at 24 and 48 h, had an LC₀/EC₀ (effective concentration) > 10 mg/L (nominal) (OECD TG 202)¹¹⁵. In a study with *Daphnia magna* (OECD TG 202, GLP, 2007), semi-static, after 48 h there was an EC₅₀ > 83 µg/L. The UV-328 concentration was above water solubility and no adverse effects were observed throughout the test^{10,15}. In another study with *D. magna*, there was an EC₅₀ > 10 mg/L after 48 h. After 24 h, under static conditions, the EC₅₀ was > 100 mg/L¹⁰. In yet another study with *D. magna* (OECD TG 202, non-GLP, 1988), after 24 h, values of EC₅₀/EC₁₀₀ > 100 mg/L and EC₀ 58 mg/L were reported¹¹⁶.

45. In fish (*Danio rerio*), a static study (OECD TG 203, non-GLP, 1988), generated a NOEC/LC₅₀ > 100 mg/L, after 96 h¹¹⁷. Another semi-static study in fish (*Oryzias latipes*, OECD TG 203, GLP, 2007), determined an LC₅₀ > 0.08 mg/L, after 96 h. It was a limit test and LC₅₀ values were calculated to be greater than the highest applied test concentration of UV-328 (0.08 mg/L)^{10,15}.

46. These experimental findings are not conclusive regarding toxicity for aquatic organisms. No reported value of any toxicity endpoint is considered acceptable to calculate a PNEC for a risk assessment for the aquatic compartment⁸. Nonetheless, ECOSAR predicts a ChV < 0.1 mg/L.

3.4.2.2 Long-term

47. Freshwater green algae (*Chlamydomonas reinhardtii*) and a crustacean (*D. magna*) were exposed to 0.01 and 10 mg/L of UV-234, UV-328, and a mixture of the two. *D. magna* growth, reproduction, and gene transcription were not impacted for 21 days. After 96 h, no differences were observed on cellular viability of *C. reinhardtii* either. In the algae, results showed increased reactive oxygen species production in response to UV-328 and lipid peroxidation following exposure to UV-234. Synergistic effects were evident at transcriptional level with two to six times up-regulation of glutathione peroxidase, which suggests potential impact of UV-234 and UV-328 on the antioxidant defence system in *C. reinhardtii*¹¹⁸. More recently, after 28 days of dietary exposure, UV-328 induced ribosomal proteins transcription and down-regulated genes involved in immune responses in juvenile rainbow trout (*Oncorhynchus mykiss*). Genes involved in iron homeostasis were also affected by UV-328³⁶.

Conclusion on toxicity:

48. UV-328 is considered to be toxic for mammals, endangering human health and the environment, as it may cause damage to liver and kidneys through prolonged or repeated oral exposure (STOT RE 2).

4. Statement of the reasons for concern and need for global action

4.1 Exposure routes

49. UV absorbers enter the environment mainly through the following pathways: (i) via wastewater treatment plant (WWTP) effluents and via plastic debris; and (ii) via weathering of outdoor plastics and coatings that have been UV-protected with UV-328, and via PCPs that contain UV-328 as an additive for UV-protection of the skin^{18,115,119}.

50. In industrial uses, a proportion of UV-328 is released to wastewater. According to EPI Suite, the total removal in WWTPs is around 94%. Nakata and Shinohara (2010) have also reported removal rates for UV-328 in effluents above 90%¹²⁰. The remaining fraction is not eliminated in WWTPs by adsorption to sludge and thus released to the receiving water bodies¹⁰. In workers, it may enter the body through inhalation, dermal absorption, or ingestion. The general population may be exposed by dust inhalation, skin contact with articles that contain the substance, or seafood ingestion. UV-328 released into the air will be absorbed in or adsorbed on particles that eventually will settle to the ground¹¹⁰. The same concept can be applied to UV-328 in water. UV-328 can also be released into the environment through in- and outdoor use of long-life materials with low release rates¹²¹.

51. UV-328 may enter soils from the application of wastewater (WW) biosolids, commonly used in enrichment⁸. For example, in the Nordic countries, environmental exposure is possible due to its relevant level of use (UV-328 has a 3–4 use index out of a maximum of 5 in the SPIN database)¹³. Based on house dust concentrations measured in the Philippines, UV-328’s estimated daily intake (EDI) from dust was reported as 0.2–0.8 ng/day for adults, and 0.5–4.6 ng/day for toddlers, which was below guideline values (9.0×10^4 ng/day for adults and 2.2×10^4 ng/day for toddlers). These guideline values were calculated from an estimated reference dose value for UV-328 (chronic NOEL or NOAEL divided by a safety factor of 1.0×10^4)¹²².

52. If released to soil, UV-328 will have a low mobility according to its high $\log K_{OC}$. If released into water, UV-328 will adsorb to suspended solids and sediments¹¹⁰.

4.2 Monitoring data

53. The concentrations reported in this section have a wide range of values. Due to the variability of the data, the most important conclusion to draw from this collection of studies is that they generally indicate the presence of UV-328 throughout the globe and in various matrices.

54. When comparing PNEC values (Table 12) and monitoring data (Section 6.6), there were several data points that were near to or exceeded the PNEC value. Examples are levels in river water (7–85 µg/L in Narragansett Bay¹²³), sewage treatment plants (0.55–4.7 mg/L in WWTP effluents¹²³), river sediment (300 mg/kg in Narragansett Bay¹¹), or secondary poisoning (0.65 mg/kg lw in preen gland oil of a thick billed murre from Pribilof Island⁹⁹). As a comparison, 5 mg/kg lw in liver or 2 mg/kg lw in preen gland oil were found in streaked shearwater (*Calonectris leucomelas*) chicks from a feeding study designed to estimate exposure to chemical additives present in plastic^{100,124}.

55. Regarding the derived no-effect level (DNEL, Table 13), some high UV-328 concentrations in fish (UV-328 at 1.3×10^3 µg/kg present in crayfish¹²⁵) come relatively close to the DNEL for oral exposure for the general population. For a 70 kg adult, a DNEL of 140 µg/kg/day means a daily oral consumption of 9.8 mg/day. Given that the crayfish mentioned above has a UV-328 concentration of 1.3×10^3 µg/kg, this means 107.7 g of crayfish would have to be consumed per kilogram body weight, or a total value of 7.5 kg. One order of magnitude lower is a realistic amount to be ingested orally on one day in certain regions with diets based on seafood.

Table 12. PNEC values for UV-328¹²¹.

Hazard for aquatic organisms	
Freshwater	10 µg/L
Intermittent releases (freshwater)	100 µg/L
Marine water	1 µg/L
Sewage treatment plant	1 mg/L
Sediment (freshwater)	451 mg/kg sediment dw
Sediment (marine water)	45.1 mg/kg sediment dw
Hazard for terrestrial organisms	
Soil	90 mg/kg soil dw
Hazard for predators	
Secondary poisoning	13.2 mg/kg food

Table 13. DNEL values for UV-328¹²¹.

Data for workers	
Inhalation exposure (systemic, long-term)	700 µg/m ³
Dermal exposure (systemic, long-term)	300 µg/kg bw/day
Data for general population	
Inhalation exposure (systemic, long-term)	170 µg/m ³
Dermal exposure (systemic, long-term)	140 µg/kg bw/day
Oral exposure (systemic, long-term)	140 µg/kg bw/day

56. An extensive review of the literature is provided in Section 6.6. In the next paragraphs, only the reports regarding POP-like features of UV-328 are discussed.

4.2.1 Water

57. Several sun-blocking agents were found, including UV-328, in seawater and freshwater from beaches, reefs, and a river in Okinawa Island, Japan. Concentrations at coral reefs were similar to or even higher than those at beaches or in rivers. UV-328 was predominant in seawater samples from beach sites (2.8–287 ng/L)⁴⁷.

58. In Canada, a monitoring study of BZT UV absorbers in streams showed urban streams displaying similar trends in concentrations across runoff events, and UV-328 was predominant (240 ng/g), ten times higher than in rural samples. The report also suggests that relatively high and consistent emissions from plastic debris, rather than episodes of industrial releases, likely led to homogeneous BZT UV absorber profiles in urban and rural backgrounds. Seasonal effects were also visible¹²⁶.

4.2.2 Sediment

59. In Japan, sediment cores showed an increasing temporal trend, with concentrations rising after 1970, in samples from the sedimentation period between 1930 and 1999 (4–10 ng/g dw)¹²⁷.

60. Narragansett Bay, USA, is a case study for UV-327 and UV-328 contamination and there are several studies investigating their presence even decades after their environmental release stopped. These data show UV-328 concentrations up to the µg/g level²⁰. UV-327 and UV-328 were produced in an industrial plant at the Pawtuxet River, which flows into Providence River and reaches the Narragansett Bay. Production of UV-327 was reported between 1963 and 1972, and production of UV-328 from 1970 to 1985^{11,21}. Since then, UV-328 and others have been found in clams and sediment cores^{123,128–130}. Sediment cores taken in 1977–1978 showed concentrations decreased with depth and distance from the discharge point^{11,130}. The decreases were approximately exponential for all compounds. The depth distribution in sediment cores from 1979–1980 was also investigated and showed a historical record of phenolic BZTs input: UV-328 concentration was the highest at the surface, which is related to its more recent production period. More sediment core analyses were carried out in 1989 and 1997. BZTs were again detected in marine and freshwater samples^{21,131}.

4.2.3 WWTPs

61. One Japanese report shows UV-328 as a frequently found UV absorber in WWTP influents (34 ng/L), with removal rates above 90%. Correspondingly, high concentrations were detected in sewage sludge. Concentrations in effluents were generally below 5 ng/L, indicating relatively effective elimination during WW treatment¹²⁰. Another study shows that UV-328 was predominant in river bed sediment (up to 17.9 ng/g dw), and WWTP effluent was the major source of contamination¹³². These results show that WWTPs are contamination sources into the aquatic ecosystem. In addition, in several countries, sewage sludge is used in agriculture and can become a contamination vector¹³³.

62. On the Gran Canary Island, Spain, samples from coastal waters and WWTPs had UV-328 as one of the most frequent BZT UV absorbers. Samples from the most touristic area had higher concentrations (up to 1.8 µg/kg dw)^{134,135}. UV-328 was also detected in urban sewage waters from Portugal and Spain in the concentration range of 21.0–76.0 ng/L¹³⁶. In Sweden, UV-328 was present

in the tens of $\mu\text{g}/\text{kg}$ in in WWTP effluents and sludge. It was also detected in landfill leachates and storm water. In one sample of landfill effluent particles, UV-328 was detected at $3.1 \mu\text{g}/\text{g dw}$ ⁴³. In Norway, WWTP effluents had 7–57 ng/L of UV-328⁴⁶.

63. In Canada, UV-328 (140 ng/g dw) and UV-234 were the most abundant BZT UV absorbers¹³³. In another study, UV-328 and other phenolic BZTs were detected in WWTP influents and effluents, in biosolids, surface water, and sediments at the ng/L and ng/g level. In addition, UV-328 was present in every layer in a sediment core from 1975 to 2013 in Lake Ontario¹³⁷. In Narragansett Bay, a municipal WWTP upstream the old chemical plant had UV-327 and UV-328 in its sludge in $\mu\text{g}/\text{g dw}$ level¹³⁸. In the 1970s, UV-328 was detected in the industrial WWTP effluent, river water, and sediments. The industrial WWTP of the former chemical manufacturing site was inefficient, so high sediment concentrations of UV-327 and UV-328 were found downstream in the ppm range^{123,129}.

4.2.4 Biomatrices

64. UV-328 was the dominant BZT (97.6% DF) in human breast milk in concentrations up to 334 ng/g lw, in the Republic of Korea in 2011¹³⁹. The EDI via consumption of breast milk was estimated to be $0.36 \mu\text{g}/\text{kg bw}/\text{day}$. The study points out the lack of an established provisional tolerable daily intake (PTDI) value for benzotriazoles¹³⁹. Several BZTs have been detected in human breast milk in Japan, Vietnam, and Philippines, too, and UV-328 is among the BZTs detected ($1.2 \text{ ng}/\text{g lw}$, 16% DF; levels lower than the reference dose)¹⁴⁰. The reference dose for UV-328 used in this study is $10 \mu\text{g}/\text{kg bw}/\text{day}$ ¹⁰⁹. Human adipose tissue from Japan, Republic of Korea, China, Spain, and USA had UV-328 also (up to $35 \text{ ng}/\text{g lw}$, 45.2% DF²⁹).

65. In a Canadian urban creek, UV-328 was present in 33–57% of the biota sampled, at concentrations of up to $1.3 \mu\text{g}/\text{g lw}$ (crayfish)¹²⁵. In the Pearl River Estuary in China, several BZTs, including UV-328, were present up to $258.9 \text{ ng}/\text{g lw}$ in marine wildlife¹⁴¹. In an earlier study, UV-328 had not been detected in wild aquatic organisms, but it was present in farmed red snapper ($0.8 \text{ ng}/\text{g dw}$ maximum)¹⁴². UV-328 showed high DF in blood plasma from several species of fish and one bird in samples from the USA (South Carolina) and Canada (Ontario), up to $3.8 \text{ ng}/\text{g ww}$ in common carp⁴⁸. Similar results had been reported before in marine biota from the USA (Florida) and Canada (Ontario), up to $3.9 \text{ ng}/\text{g}$ in white sucker (whole body)⁴⁹. In an urban creek in Canada, fish liver was the major tissue for accumulation of UV absorbers, with UV-328 in the $0.6\text{--}20.7 \text{ ng}/\text{g ww}$ concentration range¹⁴³. In Japanese marine mammal blubber samples collected in 1990, there were maximum concentrations around $70 \text{ ng}/\text{g lw}$ ¹²⁷. In finless porpoises, the mean concentration of UV-328 was $38 \text{ ng}/\text{g lw}$, which was about four times higher than in small fish ($8.4 \text{ ng}/\text{g lw}$). UV-328 concentrations in marine organisms varied among species, and higher concentrations were detected in livers of mullets and hammerhead sharks³⁰. UV absorbers were present in all samples from Ariake Sea marine organisms, having UV-328 concentrations of up to $55 \text{ ng}/\text{g ww}$ ¹⁴⁴. Very high concentrations, up to $460 \text{ ng}/\text{g lw}$ in a tidal flat gastropod, were detected in organisms from the tidal flat and higher trophic species, such as fish or crustaceans, (whole body, liver) also¹⁴⁴. Concentrations of UV-328 in tidal flat organisms were greater than in shallow water species. UV-328 presence in biota was variable and species-specific ($< 0.2\text{--}55.0 \text{ ng}/\text{g ww}$, 89.3% DF)²⁷. UV-328 was predominant in finless porpoises²⁷. In the Norwegian fjords, UV-328 was present in biota (up to $19.5 \text{ ng}/\text{g}$)^{145,146}. In several German rivers, BZTs were detected in bream liver in low ng/g concentrations, and UV-328 had some of the highest concentrations detected¹⁴⁷. UV-328 was present in all analysed biomatrices (moss and periphyton, brown trout) from a Norwegian river in low ng/g levels¹⁴⁸.

66. Foodstuff samples from Japan and the Republic of Korea also contained BZTs. Contamination was ubiquitous, with highest concentrations in seafood ($1.7 \text{ ng}/\text{g ww}$) and meat ($1 \text{ ng}/\text{g ww}$)²⁹. Mussels from the Pacific Ocean (2003–2007) showed widespread distribution of phenolic BZTs, similar to PCBs, DDTs, and PBDEs. These were detected in all samples, especially from the Republic of Korea and Japan; UV-328 was present up to $830 \text{ ng}/\text{g lw}$ ²⁸. Another report showed highest concentrations in lower benthic organisms from the tidal flat area of Ariake Sea, with UV-328 being one of the predominant BZTs ($1\text{--}460 \text{ ng}/\text{g lw}$)¹⁴⁴. UV-328 and UV-327 were dominant in higher trophic species¹⁴⁴. Another study with mussels reported UV-328 frequently with highest concentrations in Hong Kong and the Republic of Korea (around $0.8 \mu\text{g}/\text{g lw}$). In the USA, UV-328 was detected in few samples of mussels, and showed a maximum around $0.3 \mu\text{g}/\text{g lw}$ ¹⁴⁹. At the Manila Bay, in the Philippines, BZT UV absorbers were detected at the ng/g level in almost all fish samples. UV-328 was in 88% of the samples up to $34.2 \text{ ng}/\text{g}$. The distribution profile of the BZTs was different among fish species, which could reflect differences in accumulation and biodegradability of the substances studied in the diverse species^{26,150}.

4.2.5 Other matrices

67. The presence of BZTs in relevant concentrations in textiles, including UV-328 (up to 106 ng/g), demonstrates a potential source of human and environmental exposure^{151,152}. In Spanish indoor dust samples, UV-328 (91 ng/g) was ubiquitous¹⁵³. In the Philippines, BZTs were also present in house dust from residential and municipal dumping areas; UV-328 was present up to 304 ng/g. The EDIs through house dust ingestion were two to four orders of magnitude lower than guideline values. These guideline values were calculated from an estimated reference dose value for UV-328 (chronic NOEL or NOAEL divided by a safety factor of 1.0×10^4). However, the EDI for toddlers was five times higher than for adults^{122,154}. In 2010, UV-328 was detected, too, in road dust samples from a road with significant traffic. These concentrations were correlated with traffic density (2–40 ng/g dw)¹²⁷.

68. UV absorbers and antioxidants are widely used in plastic food and beverage packages and UV absorber BZTs were detected in such Chinese products, including UV-328 (up to 30.5 µg/g)^{155–157}. In a study on weathered plastics, most antioxidants and UV absorbers concentrations were slightly higher in new plastics compared to corresponding debris, which implies potential leaching. In this study, UV-328 was the least frequent contaminant in debris, but was relatively abundant in new plastic in the high ng/g level. The DF in new plastics was 100% and in debris 97%⁸⁶. In plastic debris collected from coastal beaches, UV-328 was one of the predominately detected chemicals¹⁵⁸.

4.2.6 Remote locations

69. In Norway, UV-328 was detected in urban and remote sites. The remote locations were in Svalbard, specifically Ny-Ålesund and northeast of it, which are around 110 to 170 km away, respectively, from Longyearbyen (the largest settlement in Svalbard, population of 2,368). The island of Svalbard is itself almost 1,000 km away from Tromsø in Northern Norway (population of 76,734). In Arctic biota, UV-328 DF depended on the species and concentrations were in the low ng/g range⁴⁶: 100% DF in common eider (eggs), kittiwake (eggs) and mink (liver), 60% DF in European shag (eggs) and glaucous gull (eggs), and 0% DF in polar bear (blood plasma). As explained in Section 3.3.2, blood plasma is not the most relevant matrix for hydrophobic chemicals such as UV-328.

70. In the Great Lakes, USA, UV-328 was detected at concentrations of up to 13 ng/g ww in herring gull eggs, where it was the only BZT UV absorber frequently measured. In some Lake Superior sampling sites, such as Thunder Bay-Pie Island and Marathon (20 km from the closest airport, 480 km away from Minneapolis), UV-328 had a 100% DF in herring gull eggs, which was consistent with higher levels in WWTP influent, effluent, and biosolids. Birds seemed to accumulate more UV-328 than fish, which might be related to trophic position⁴⁴. In the Canadian Arctic, UV-328 was detected in the liver of northern fulmars from Prince Leopold Island (3.8 ng/g ww, 11% DF)⁴⁵.

4.3 Conclusions

71. Based on the presented data, UV-328 meets the screening criteria in Annex D for persistence, bioaccumulation, LRTP, and adverse effects under the Stockholm Convention. Due to its many applications and ongoing use, UV-328 is emitted into the environment from human activities, e.g. manufacturing processes, consumer products, and disposal and management of waste. The presence of UV-328 is an issue in remote locations and the environmental long-range transport of free UV-328 is aggravated by the long-range transport of plastic debris that acts as a continuous source of UV-328 during its circulation in the environment.

72. The environmental distribution of UV-328 has been demonstrated to lead to the global presence of UV-328, endangering human health and the environment. Its presence in the environment decades after its release was stopped indicates high persistence. Detections also include measurements in biota, water, and sediment within the Arctic Circle and in background samples. In humans, UV-328 has been detected in breast milk and adipose tissue. This evidence indicates that UV-328 is bioaccumulative. UV-328 has been detected in marine biota and there are indications that it biomagnifies in the food chain. Pharmacokinetic modelling suggests that UV-328, like other BZTs, will be absorbed in the gastrointestinal tract, metabolised in the liver, and excreted via kidneys. This leads to liver and kidney toxicity.

73. UV-328 does not volatilise to a large extent and is not distributed in the gas phase. However, it is transported while adsorbed to particulate matter, e.g. aerosol particles in the air or suspended solids in water. The OECD Tool shows P_{OV}, CTD and TE similar to those of several POPs already listed under the Stockholm Convention. Importantly, environmental long-range transport of free UV-328 is complemented by simultaneous transport of plastic debris as a carrier from which UV-328 continuously leaches. This leaching will occur not only into the environment, e.g. water, with

subsequent transfer into the sediment, but also from plastic matrices ingested by biota into their tissues, e.g. UV-328 transfer from plastics inside seabirds' stomachs into preen gland oil.

74. In the EU, UV-328 is an acknowledged STOT RE 2 substance. The experimental data on ecotoxicity are limited, but with read-across and modelling data, it is possible to infer probable hazardous effects in aquatic species.

75. UV-328 is included in several national and international studies or listings, where its hazard properties for human health and the environment have already been identified, e.g. CEPA, NITE, TSCA, OSPAR, SIN List, or SPIN. Under REACH, UV-328 is an SVHC (PBT/vPvB) and therefore it was listed in Annex XIV of the REACH regulation in February 2020. In early 2019, UV-328, among others, was listed as a high-volume plastic additive used in EU and prioritised for further assessment.

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Appendix

6.1. Test Guidelines

6.1.1 European Union Methods

EU Method A.6: Water Solubility

6.1.2 Organization for Economic Co-operation and Development (OECD) Test Guidelines (TGs)

OECD TG 117 – Partition Coefficient (*n*-octanol/water)

OECD TG 201 – Alga, Growth Inhibition Test

OECD TG 202 – *Daphnia* sp. Acute Immobilisation Test

OECD TG 203 – Fish, Acute Toxicity Test

OECD TG 209 – Activated Sludge, Respiration Inhibition Test

OECD TG 301 B – Ready Biodegradability: CO₂ Evolution Test

OECD TG 305 C – Bioconcentration: Flow-Through Fish Test

OECD TG 401 – Acute Oral Toxicity

OECD TG 402 – Acute Dermal Toxicity

OECD TG 403 – Acute Inhalation Toxicity

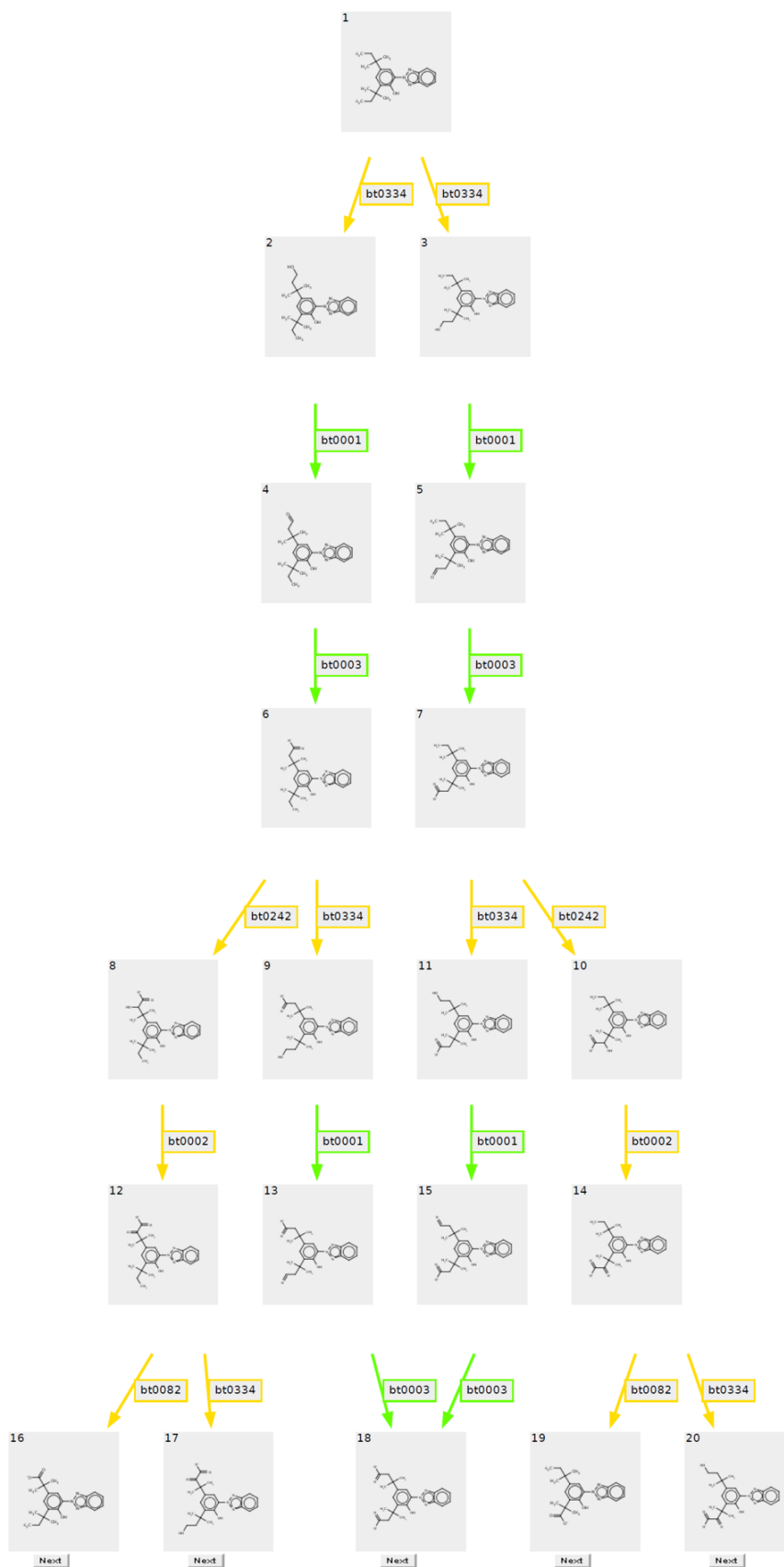
OECD TG 408 – Repeated Dose 90-Day Oral Toxicity in Rodents

OECD TG 409 – Subchronic Oral Toxicity – Non-Rodent: 90-Day study

OECD TG 414 – Prenatal Developmental Toxicity Study

OECD TG 422 – Combined Repeated Dose Toxicity Study with the Reproduction/Developmental Toxicity Screening Test

6.2. EAWAG-BBD Prediction System

Figure 3. EAWAG-BBD prediction for UV-328 using the SMILES code present on Table 2¹⁶.

6.3. Nakata *et al.* Bioaccumulation Studies

6.3.1 Nakata *et al.*, 2010 Study³⁰

Table 14. Concentrations of BZT UV absorbers (ng/g ww) in the blubbers of finless porpoises (FP) collected from the Ariake Sea, Japan³⁰.

	FP-1	FP-2	FP-3	FP-4	FP-5	Average
UV-327	4.5	9.5	6.3	31	18	14
UV-328	20	64	11	34	16	29

Table 15. Concentrations from Table 14 converted into ng/g lw³⁰.

	FP-1	FP-2	FP-3	FP-4	FP-5	Average
Blubber lipid content	81%	83%	87%	59%	91%	-
UV-327	5.6	11.4	7.2	52.5	19.8	19.3
UV-328	24.7	77.1	12.6	57.6	17.6	37.9

Table 16. Concentrations from Table 15 lipid-normalised to a lipid content of 5%³⁰.

	FP-1	FP-2	FP-3	FP-4	FP-5	Average
UV-327	0.3	0.6	0.4	2.6	1.0	1.0
UV-328	1.2	3.9	0.6	2.9	0.9	1.9

Table 17. Concentrations from Table 14 extrapolated to whole body concentrations, considering the mass fraction of blubber 28.8%. BAF for the finless porpoises is calculated ww- and lw-based. The environmental reference value used for both substances was 0.12 ng/L of UV-327 in water samples³⁰.

	UV-327	UV-328
Whole body concentration (ng/g ww)	4.0	8.4
BAF (L/kg ww)	3.3×10^4	7.0×10^4
BAF (L/kg lw)	8.0×10^3	1.6×10^4

6.3.2 Nakata *et al.*, 2009 Study²⁷

Table 18. Concentrations of BZT UV absorbers (ng/g ww) in tidal flat and shallow water organisms collected from the Ariake Sea, Japan²⁷.

	Flathead	Solefish	Right eye flounder	Sandperch	Sweetlips	Average
Lipid content	2.3%	2.0%	3.3%	7.3%	1.4%	-
UV-327	0.34	0.29	0.34	0.51	0.47	0.39
UV-328	0.26	0.29	0.26	0.23	0.19	0.25

Table 19. Concentrations from Table 18 converted into ng/g lw²⁷.

	Flathead	Solefish	Right eye flounder	Sandperch	Sweetlips	Average
UV-327	0.7	0.7	0.5	0.3	1.7	0.8
UV-328	0.6	0.7	0.4	0.2	0.7	0.5

Table 20. Concentrations from Table 18 lipid-normalised to a lipid content of 5%²⁷.

	Flathead	Solefish	Right eye flounder	Sandperch	Sweetlips	Average
UV-327	0.7	0.7	0.5	0.3	1.7	0.8
UV-328	0.6	0.7	0.4	0.2	0.7	0.5

Table 21. BAF for small fishes is calculated ww- and lw-based. The environmental reference value used for both substances was 0.12 ng/L of UV-327 in water samples²⁷.

	UV-327	UV-328
BAF (L/kg ww)	3.3×10^3	2.0×10^3
BAF (L/kg lw)	6.7×10^3	4.2×10^3

6.4. OECD P_{OV} and LRTP Tool

Since the OECD Tool is intended to enable a relative comparison of different chemicals with respect to P_{OV}, CTD and TE, a standardized method for deriving the input data was applied in order to obtain comparable results.

Table 22. OECD Tool input data used to generate Figure 2. Values from EPI Suite²²: ^a KOAWIN v1.10 (HenryWin est), ^b KOAWIN v1.10 (KowWin v1.68), ^c AopWin v1.92, ^d BIOWIN3 (BIOWIN v4.10), and ^e calculated t_{1/2} in soil (1.85 × half-life in water)²³.

	Molecular weight (g/mol)	^a logK _{AW}	^b logK _{OW}	^c t _{1/2} in air (h)	^d t _{1/2} in water (h)	^e t _{1/2} in soil (h)
α-endosulfan ¹⁵⁹	406.9	-3.6	4.9	194.4	520.8	1.0 × 10 ³
α-HCH ¹⁶⁰	290.8	-3.5	3.9	2.9 × 10 ³	3.2 × 10 ³	3.2 × 10 ³
Aldrin ¹⁶⁰	364.9	-1.3	6.6	2.9	2.7 × 10 ³	3.8 × 10 ³
CCl₄ ¹⁶⁰	154.0	0.2	2.8	6.9 × 10 ⁵	5.9 × 10 ³	5.9 × 10 ³
HBCDD ¹⁶¹	641.7	-3.5	5.6	76.8	1.2 × 10 ⁴	1.5 × 10 ³
HCB ¹⁶⁰	284.8	-1.4	5.7	2.2 × 10 ⁴	3.4 × 10 ⁴	3.4 × 10 ⁴
PCB-101 ¹⁶⁰	326.4	-2.0	6.3	885.0	3.1 × 10 ⁴	1.0 × 10 ⁵
PCB-180 ¹⁶⁰	395.3	-2.5	7.2	2.7 × 10 ³	5.5 × 10 ⁴	1.0 × 10 ⁶
PCB-28 ¹⁶⁰	257.5	-1.9	5.7	255.3	5.5 × 10 ³	1.0 × 10 ³
tetraBDE	485.8	-3.1	6.5	264.0	4.6 × 10 ³	9.2 × 10 ³
pentaBDE	564.7	-3.6	6.8	456.0	8.5 × 10 ³	1.9 × 10 ⁴
hexaBDE	643.6	-3.7	7.4	1.1 × 10 ³	1.6 × 10 ⁴	3.1 × 10 ⁴
heptaBDE	722.5	-4.3	7.3	1.5 × 10 ³	1.9 × 10 ⁴	4.2 × 10 ⁴
octaBDE	801.4	-4.4	8.5	2.2 × 10 ³	2.6 × 10 ⁴	5.1 × 10 ⁴
decaBDE	959.2	-4.8	10.0	7.6 × 10 ³	3.8 × 10 ⁴	7.6 × 10 ⁴
UV-328	351.5	-10.6	7.3	16.3	1.8 × 10 ³	3.3 × 10 ³

Table 23. OECD Tool generated values calculated from the input data in Table 22 and plotted in Figure 2.

	P _{OV} (days)	CTD (km)	TE (%)
α-endosulfan	60.4	2.3 × 10 ³	4.6
α-HCH	195	6.0 × 10 ³	31.5
Aldrin	223	125	1.0 × 10 ⁻⁴
CCl₄	2.5 × 10 ⁴	1.2 × 10 ⁶	1.8 × 10 ³
HBCDD	38.0	1.4 × 10 ³	1.7
HCB	1.9 × 10 ³	2.7 × 10 ⁵	2.0 × 10 ³
PCB-101	4.0 × 10 ³	1.6 × 10 ⁴	30.6
PCB-180	4.8 × 10 ⁴	1.7 × 10 ⁴	90.7
PCB-28	540	5.1 × 10 ³	2.2
tetraBDE	552	3.6 × 10 ³	8.8
pentaBDE	1.1 × 10 ³	3.7 × 10 ³	13.7
hexaBDE	1.9 × 10 ³	3.6 × 10 ³	15.7
heptaBDE	2.5 × 10 ³	3.1 × 10 ³	13.6
octaBDE	3.1 × 10 ³	2.9 × 10 ³	12.7
decaBDE	4.6 × 10 ³	2.9 × 10 ³	12.7
UV-328	196	2.8 × 10 ³	12.4

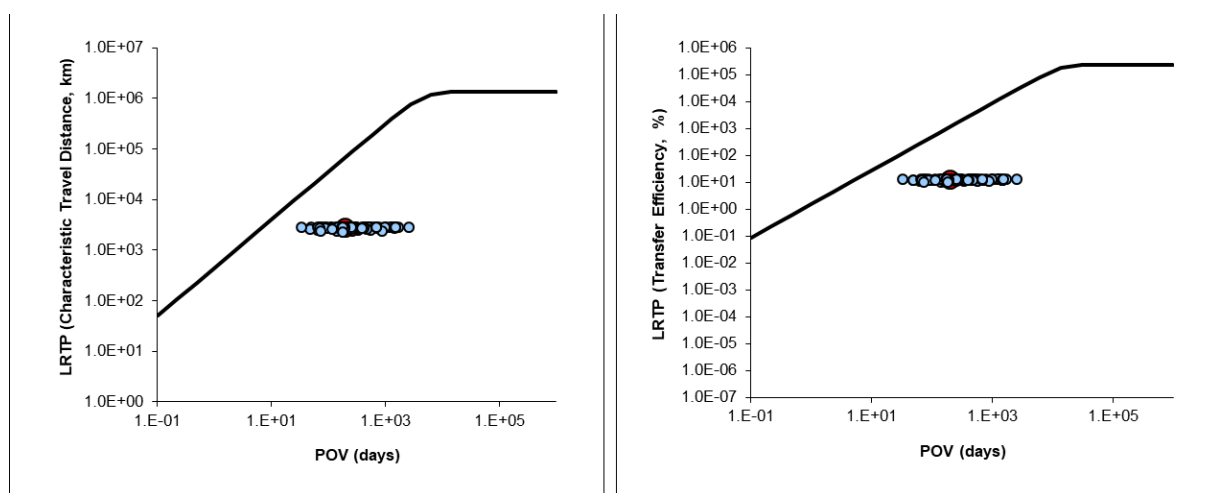


Figure 4. Monte Carlo analysis of the OECD Pov and LRTP Tool results for UV-328 with the same input values as in Table 22. The dispersion factor for each Tool input except the molecular weight is 10.

Alternatively to the input data for UV-328 estimated with EPI Suite presented in Table 22 above, the $\log K_{AW}$ may be obtained from the Henry's law constant estimated by OPERA or calculated from experimental vapour pressure and water solubility (Table 3). As shown in Table 24, the P_{OV} is virtually unaffected of the $\log K_{AW}$ input value, whereas the values for CTD and TE are somewhat lower when the OPERA estimate is used. This finding is remarkable given the fact that the difference between the estimate from EPI Suite and the OPERA estimate is five orders of magnitude. However, further increasing the $\log K_{AW}$ value, such as in the case of the calculated value in Table 24 below, results in lower CTD and TE values. This is because then the K_{OA} , which is $K_{OA} = K_{OW}/K_{AW}$, has a value at which an appreciable fraction of the chemical is present in the gas phase and is degraded in the gas phase in competition to long-range transport.

Table 24. Impact of using different $\log K_{AW}$ values as input (other input data unchanged).

Input vales for $\log K_{AW}$		OECD Tool generated values		
Method	$\log K_{AW}$	Pov (days)	CTD (km)	TE (%)
EPI Suite	-10.6	196	2.8×10^3	12.4
OPERA (Henry's law constant)	-5.6	196	2.5×10^3	9.6
Calculated from exp. vapour pressure and water solubility	-4.5	196	1.2×10^3	2.1

However, the K_{AW} does not influence the CTD in the way shown in Table 24 if one considers that the half-life in air estimated by AopWin is likely too short. AopWin is known to overestimate the reactivity with OH radicals of large molecules. This has been shown, for example for DDT. AopWin v1.92 gives for DDT a 2nd-order rate constant of $3.435 \times 10^{-12} \text{ cm}^3/(\text{molecule}\cdot\text{s})$. In contrast, Liu, Krüger and Zetzsch (2005)¹⁶² found a measured value of $5 \times 10^{-13} \text{ cm}^3/(\text{molecule}\cdot\text{s})$ for DDT, which is by a factor of 7 lower than the value of AopWin. If one assumes a 7 times higher half-life in air also for UV-328, this gives a CTD and a TE for UV-328 of $2.5 \times 10^3 \text{ km}$ and 9.2%, respectively, even at a high $\log K_{AW}$ ($\log K_{AW} = -4.5$).

6.5 Analogues

Table 25. Physico-chemical properties of UV-328 analogues. Values from EPI Suite™ v.4.10: ^a WSKOW v1.42 (from log*K*_{ow}), ^b MPBPVP v1.43 (Modified Grain method), ^c KOAWIN v1.10 (KowWin v1.68), and ^d KOCWIN v2.00 (MCI method)²².

	UV-320	UV-327	UV-350	M1**
CAS RN	3846-71-7	3864-99-1	36437-37-3	84268-36-0
Molecular weight (g/mol)	323.4	357.9	323.4	339.4
^a Water solubility (mg/L)	0.2	0.03	0.1	102.4
^b Vapour pressure (mmHg, 25 °C)	1.1×10^{-9}	2.7×10^{-10}	7.8×10^{-10}	5.2×10^{-12}
^c log <i>K</i> _{ow}	6.3	6.9	6.3	3.3
^d log <i>K</i> _{oc}	5.1	5.3	5.2	3.8

** The estimated properties provided are for the neutral form of M1, based on the SMILES code CC(C)(C)C1=C(C(=CC(=C1)CCC(=O)O)N2N=C3C=CC=N2)O. However, M1 will mostly be in its anionic form in the environment, considering its *pK*_a of 4.7±0.4 estimated by ACD/Labs and available in the Danish (Q)SAR database¹¹³.

6.6. Monitoring Data Studies

6.6.1 Asia

Table 26. UV-328 monitoring data studies summary in Asia.

Location	Matrix	Others	Average	Max	Min	
China	33 cities (mostly in economically developed provinces) ¹⁶³	WWTP sludge (ng/g dw)	20.6 median, 2.5×10^3 maximum, $n = 60$, 97% DF, 18% of total BZTs	57.3	213	3.5 ^{††} , not detected (ND)
	Anning Sewage Plant, Lanzhou, Gansu Province ¹⁶⁴	WW ($\mu\text{g/L}$)	$n = 1$			ND
		PCPs ($\mu\text{g/L}$)	$n = 5$, 80% DF		771	226 ^{††} , ND
	Beijing ¹⁸	WWTP biosolids (ng/g)		108		< limit of detection (LOD)
	Beijing ¹⁵⁶	food packaging ($\mu\text{g/g}$)	$n = 27$, 4% DF		6.0	
	Beijing ¹⁵⁷	beverage packaging ($\mu\text{g/g}$)	$n = 17$, 12% DF		13.9	2.0, < limit of quantification (LOQ), ND
	Bohai and Yellow Seas, Shandong Peninsula ¹⁶⁵	surface sediment (ng/g dw)	$n = 74$	0.05	0.4	< method detection limit (MDL)
	Jinan, Shandong Province (two million local inhabitants) ¹⁶⁶	WWTP effluent (ng/L)	57% DF		2.7	
		WWTP sludge (ng/g dw)	15% DF		508	286
		WWTP influent (ng/L)	12% DF		9.9	
	Kunming, Yunnan Province ¹⁵⁵	milk packing ($\mu\text{g/g}$)	$n = 1$		24.8	
		snack packing ($\mu\text{g/g}$)	$n = 1$		30.5	
Pearl River Estuary ¹⁴¹	marine wildlife muscles (ng/g lw): pelagic-neritic, benthopelagic and demersal fish, pelagic and demersal	$n = 24$, 75% DF		259	0.8 ^{††} , ND	

^{††} Lowest detected value.

Location		Matrix	Others	Average	Max	Min
		cephalopoda, and demersal crustaceans				
	Pearl River Estuary ¹³²	bed sediment (ng/g dw)	<i>n</i> = 27, 93% DF		17.9	0.4 ^{††} , < LOQ
	Pearl River Estuary ¹⁴²	farmed red snapper carcasses (ng/g dw)	<i>n</i> = 2, 50% DF		0.8	< LOQ
		wild fishes species carcasses (ng/g dw)	<i>n</i> = 11			ND
	Shandong Province ¹⁶⁷	WWTP influent (ng/L)	<i>n</i> = 4, 50% DF		2.9	2.6 ^{††} , ND
		WWTP effluent (ng/L)	<i>n</i> = 4, 50% DF		0.6	ND
		river (ng/L)	<i>n</i> = 4			ND
	Songhua River, Northeast (mainly residential) ¹⁶⁸	WWTP influent (ng/L)	<i>n</i> = 81, 94% DF	9.6	29	0.3
		WWTP A/O sludge (ng/g dw)	<i>n</i> = 6, 100% DF	115	163	93.3
		WWTP dewatered sludge (ng/g dw)	<i>n</i> = 5, 100% DF	89.3	121	39.6
	Songhua River, Northeast ¹⁶⁹	sediment (ng/g dw)	<i>n</i> = 6, 100% DF	3.8	7.1	2.1
		WWTP sludge (ng/g dw)	<i>n</i> = 5, 100% DF	1.3×10^3	5.92×10^3	40.6
	India	Rivers Kaveri, Vellar, Thamiraparani of Tamil Nadu ¹⁷⁰	water, river (ng/L)	<i>n</i> = 59, 30–38% DF		5.2
sediment, river (ng/g)			<i>n</i> = 58, 80–88% DF		4.3	ND
fish muscle, river (ng/g)			<i>n</i> = 14, 50–92% DF		6.1	ND
Japan	Five WWTPs, located in an unnamed town (population 680,000) ¹²⁰	WWTP influent (ng/L)		34	52	18
		WWTP effluent (ng/L)		2.6	2.9	2.1
		WWTP sludge (ng/g dw)		510	570	430
	Ariake Sea ²⁷	tidal flat and shallow water organisms (ng/g ww)	<i>n</i> = 28, 89% DF		55	0.2 ^{††} , < 0.15
		sediments (ng/g dw)	<i>n</i> = 16, 100% DF		320	2.6
	Ariake Sea ³⁰	blubber of finless porpoises (ng/g ww)	<i>n</i> = 5, 100% DF	29	64	11
	Saitama Prefecture ¹⁷¹	water from streams (ng/L)	<i>n</i> = 2, 50% DF	70		
WWTP effluents (ng/L)		<i>n</i> = 4, 75% DF	62	88	47	

Location		Matrix	Others	Average	Max	Min
		water from heavily polluted rivers (ng/L)	<i>n</i> = 6, 67% DF	701	4.8×10^3	149
		water from moderately polluted rivers (ng/L)	<i>n</i> = 12, 67% DF	152	583	30
		water from background sites (ng/L)	<i>n</i> = 5			ND
		sediment from streams ($\mu\text{g}/\text{kg}$)	<i>n</i> = 2, 100% DF	102	1.2×10^3	10
		sediment from WWTP effluents ($\mu\text{g}/\text{kg}$)	<i>n</i> = 4, 75% DF	13	85	10
		sediment from heavily polluted rivers ($\mu\text{g}/\text{kg}$)	<i>n</i> = 6, 100% DF	117	1.7×10^3	21
		sediment from moderately polluted rivers ($\mu\text{g}/\text{kg}$)	<i>n</i> = 12, 75% DF	59	213	10
		sediment from background sites ($\mu\text{g}/\text{kg}$)	<i>n</i> = 5, 60% DF	58	89	29
	Not described ¹²⁷	road dust (ng/g dw)	<i>n</i> = 9, 100% DF		40	2.0
		marine mammal blubber (ng/g ww)	<i>n</i> = 29, 66% DF		70	
		sediment cores (ng/g dw)	<i>n</i> = 2, 100% DF		10	4.0
	Okinawa Island: seawater of beaches and coral reefs ⁴⁷	seawater at beach sites (ng/L)	<i>n</i> = 23, 61% DF		287	2.8 ^{††} , ND
		seawater at river and coral reef sites (ng/L)	<i>n</i> = 15, 60% DF		263	5.7 ^{††} , ND
	Mukojima Island ⁹⁹	black-footed albatross ($\mu\text{g}/\text{g}$ -plastic, PP fragment)	<i>n</i> = 194 (plastic fragments), 1% DF		1.4	
	Awashima Island ^{100,124}	streaked shearwater chicks from semi-field conditions ($\mu\text{g}/\text{g}$ -lipid)	<i>n</i> = 21, liver, adipose tissue, preen gland oil		ca. 5	ca. 0.4
Philippines	Malate (residential), Payatas (close to a municipal dumping site) ¹²²	residential, house dust (ng/g)	27.0 median, <i>n</i> = 17, 82% DF	50	304	ND
		municipal dump, house dust (ng/g)	<i>n</i> = 20, 85% DF	18	48	ND

Location		Matrix	Others	Average	Max	Min
	Malate (residential), Payatas (close to a municipal dumping site) ¹⁵⁴	house dust (ng/g)	same values as ¹²²			
	Manila Bay ²⁶	marine fish muscle (ng/g lw): demersal and pelagic fish	$n = 22, 88\% \text{ DF}$	34.2	563	1.5 ^{††} , ND
	Manila Bay ¹⁵⁰	fish muscle (ng/g lw)	$n = 5, 100\% \text{ DF}$		179	18.4
Republic of Korea	Geoje Island ⁸⁶	new plastic (ng/g)	$n = 27, 100\% \text{ DF}$		770	2.7
		marine plastic debris (ng/g)	$n = 29, 97\% \text{ DF}$		1.6×10^3	1.5 ^{††} , ND
	Geoje Island ¹⁵⁸	marine plastic debris	qualitative analysis, $n = 19, 16\% \text{ DF}$			
		new plastic	qualitative analysis, $n = 25, 31\% \text{ DF}$			
	Residential (Seoul, Pyeongchon), industrial (Ansan), rural (Jeju) ¹³⁹	human breast milk (ng/g lw)	39.7 median, $n = 208, 98\% \text{ DF}$		334	< LOQ
Several countries	Cambodia, China, Hong Kong, India, Indonesia, Japan, Republic of Korea, Malaysia, Philippines, USA, Vietnam ²⁸	mussels (ng/g lw)	$n = 68, 65\% \text{ DF}$		830	31.0 ^{††} , ND
	Cambodia, China, Hong Kong, India, Indonesia, Japan, Republic of Korea, Malaysia, Philippines, Vietnam ¹⁴⁴	tidal flat and shallow water organisms (ng/g lw): whole body, liver	$n = 45 (1998-2005), n = 51 (2001-2005)$		460	1.0
	Japan, Republic of Korea, China, India, Spain, Poland, USA ²⁹	adipose tissue (ng/g lw)	$n = 93, 45\% \text{ DF}$		35 (Japan), 20 (Republic of Korea), 6.0 (Spain)	2.0 (USA) ^{††} , ND (Poland)
		foodstuff (ng/g ww): seafood, meat, vegetables, cereals, dairy products	$n = 30, 47\% \text{ DF}$		1.7 (seafood), 1.0 (meat), 0.5 (fruit)	0.2 (vegetables), ND (dairy)
	Japan, Philippines, Vietnam ¹⁴⁰	human breast milk (ng/g lw)	$n = 87, 16\% \text{ DF}$	1.2	42	< MDL

6.6.2 Europe

Table 27. UV-328 monitoring data studies summary in Europe.

Location		Matrix	Others	Average	Max	Min
Germany	River Rhine ¹⁷²	suspended solids (ng/g dw)	<i>n</i> = 4, 25% DF		26	ND
	Rivers Rhine, Saale, Saar, Elbe, and Moselle ¹⁴⁷	river sediments (ng/g)	4.6 median, <i>n</i> = 8, 100% DF		10	2.0 ^{††} , ND
		suspended particulate matter (ng/g)	<i>n</i> = 5, 100% DF		15	5.0 ^{††} , ND
		bream liver (ng/g)	<i>n</i> = 4, 100% DF		40	1.0 ^{††} , ND
Denmark	Faroe Islands ⁹⁹	northern fulmar (µg/g-plastic, PP fragment)	<i>n</i> = 194 (plastic fragments), 1% DF		1.1	
Norway	Arctic (Svalbard, Zeppelin mountain and Kongsfjord area), hot/urban spot (Tromsø area) ⁴⁶	Arctic air (pg/m ³)	<i>n</i> = 5			< 0.5
		common eider eggs (ng/g ww)	<i>n</i> = 5, 100% DF		0.3	0.1
		European shag eggs (ng/g ww)	<i>n</i> = 5, 60% DF		0.2	< 0.2
		kittiwake eggs (ng/g ww)	<i>n</i> = 5, 100% DF		0.3	0.1
		glaucous gull eggs (ng/g ww)	<i>n</i> = 5, 60% DF		0.3	< 0.1
		polar bear blood plasma (ng/g ww)	<i>n</i> = 10			< 0.3
		mink liver (ng/g ww)	<i>n</i> = 5, 100% DF		0.4	0.1
		common gull eggs (ng/g ww)	<i>n</i> = 5, 60% DF		0.2	< 0.2
		WWTP effluent (ng/L)	<i>n</i> = 6, 100% DF		57	7.0
	River Alna ¹⁴⁸	water (ng/L)	<i>n</i> = 2, 100% DF		1.9	1.0
		suspended particulate matter (ng/g dw)	<i>n</i> = 2, 100% DF		53	39
		benthic macroinvertebrate (ng/g ww)	<i>n</i> = 2			< 1.0
		moss and periphyton (ng/g ww)	<i>n</i> = 3, 100% DF		17.7	7.4
		brown trout whole body (ng/g ww)	<i>n</i> = 2, 100% DF		0.7	
brown trout muscle/liver (ng/g ww)		<i>n</i> = 2, 100% DF		0.5	0.4	
	WWTP effluent (ng/L)				< 5.0	

Location	Matrix	Others	Average	Max	Min	
Tromsø/Tomasjord, Oslo/Oslofjord, Ottestad/Lake Mjosa ¹⁴⁵	WWTP sludge (ng/g dw)				< 11	
	landfill leachate (ng/L)				< 5.0	
	sediment (ng/g dw)	12.5 median		25.1	3.2 ^{††} , < 25	
	marine and freshwater biota (ng/g): fish, crustaceans	cod (liver), <i>n</i> = 15, 20% DF; not present in other species		19.5	ND	
	Tromsø/Tomasjord, Oslo/Oslofjord, Ottestad/Lake Mjosa ¹⁴⁶	biota (ng/g ww)	20% DF		19	ND
		WWTP effluent (ng/L)	<i>n</i> = 15, 7% DF		81	< 5.0
		WWTP sludge (ng/g dw)	<i>n</i> = 10			< 5.0
		water leachate (ng/L)	<i>n</i> = 6			< 5.0
		particulate leachate (ng/g dw)	<i>n</i> = 6			< 5.0
		lake sediment (ng/g dw)	<i>n</i> = 10, 50% DF		25	3.0 ^{††} , < 25
	Oslo area ¹⁷³	sewage water (ng/L)	<i>n</i> = 7, 100% DF		68	22.0
		surface water (ng/L)	<i>n</i> = 9, 100% DF		17	0.8
		sediment (ng/g dw)	<i>n</i> = 5, 60% DF		21	1.7 ^{††} , < 2.5
		common mussel (ng/g ww)	<i>n</i> = 5, 20% DF		0.7	< 0.6
		gull egg (ng/g ww)	<i>n</i> = 10, 100% DF		60	0.4
		settled floor dust (ng/g)	<i>n</i> = 26, 100% DF		1.8 × 10 ³	0.9
		indoor air (ng/m ³)	<i>n</i> = 24, 100% DF		5.3	0.1
	Spain	Gran Canary Island ¹⁷⁴	WWTP water (ng/L)		6.0 × 10 ⁴	1.7 × 10 ⁴
		Gran Canary Island ¹⁷⁵	beach seawater (ng/L)	<i>n</i> = 12		< LOD
WWTP effluent (ng/L)			<i>n</i> = 17, 71% DF		13	6.2 ^{††} , < LOD
Five WWTPs in the Gran Canary Island ¹³⁴		WWTP influent (ng/L)			238	22.6
		WWTP effluent (ng/L)			28.4	
		marine sediments (ng/kg dw)			1.8 × 10 ³	347
Gran Canary Island ¹³⁵		seawater from touristic beaches	present, but no values			
Gran Canary Island ¹⁷⁶		marine outfall (ng/g)	<i>n</i> = 4, 75% DF		24	20.7 ^{††} , < LOQ
	WWTP sludge (ng/g)	<i>n</i> = 3, 67% DF		12.2	0.9 ^{††} , < LOD	

Location	Matrix	Others	Average	Max	Min	
		beach seawater (ng/L)	<i>n</i> = 3		< LOD	
	Northwest ¹⁷⁷	WWTP sludge (ng/g)	<i>n</i> = 8, 88% DF		152	20 ^{††} , ND
		sediment (ng/g)	<i>n</i> = 1, 100% DF		20	
	Not described ¹⁵³	indoor dust (ng/g)	<i>n</i> = 10, 100% DF	91.0	149	46
	Not described ¹⁷⁸	river and marine sediment (ng/g)	<i>n</i> = 6, 100% DF		56	7.9
	Not described ¹⁷⁹	WWTP influent (ng/L)	<i>n</i> = 5, 80% DF		19	1.0 ^{††} , ND
	Tarragona, industrial parks ¹⁸⁰	Constantí: particulate phase of outdoor air (pg/m ³)	<i>n</i> = 10, 70% DF	20	43	ND
Tarragona harbour: particulate phase of outdoor air (pg/m ³)		<i>n</i> = 10, 100% DF	14	21	6.5	
Sweden	Background (Gårdsjön and Sandsjön) and urban sites (Stockholm and Borås) ⁴³	storm water (ng/L)	<i>n</i> = 6, 75% DF		1.3	0.2 ^{††} , < 0.1
		surface water (ng/L)	<i>n</i> = 6, 100% DF		4.1	1.7
		air (ng/m ³)	<i>n</i> = 8			< 0.02
		air deposition (ng/m ² day)	<i>n</i> = 4			< 70
		sediment (µg/kg dw)	<i>n</i> = 6, 67% DF		1.3	< 0.7
		fish whole body (µg/kg dw)	<i>n</i> = 4			< 0.3
		landfill effluent particles (µg/kg dw)	<i>n</i> = 1, 100% DF		3.1	
		landfill effluent (ng/L)	<i>n</i> = 3, 100% DF		91	7.0
		WWTP effluent particles (µg/kg dw)	<i>n</i> = 1			< 110
		WWTP effluent (ng/L)	<i>n</i> = 5, 100% DF		15	6.8
		WWTP sludge (µg/kg dw)	<i>n</i> = 8, 50% DF		37	2.4 ^{††} , ND
		soil (µg/kg dw)	<i>n</i> = 4, 25% DF		0.74	2.4 ^{††} , < 0.4
	Retailers with garments made worldwide ¹⁵¹	clothing textile samples (ng/g)	<i>n</i> = 27, 15% DF		85.3	47.8 ^{††} , ND
Stockholm, retailers available in up to 88 countries worldwide ¹⁵²	garments (ng/g)	<i>n</i> = 26, 8% DF		106	8.0 ^{††} , ND	
Several countries	Germany, Norway, Sweden, Netherlands, Poland: North (Skagerrak and Kattegat regions), Baltic (German	sea surface sediments (ng/g dw)	<i>n</i> = 48, 31–50% DF	0.1	0.9	< MDL
		coastline sediments (ng/g dw)	<i>n</i> = 8		0.15	< MDL

Location		Matrix	Others	Average	Max	Min
	Bight and German Baltic Sea) Seas, Rhine-Meuse-Delta and the Oder Lagoon ¹⁸¹					
	Portugal (Lisbon), Spain (Northwest) ¹³⁶	WWTP influent (ng/L)	<i>n</i> = 3, 100% DF		76	53
		WWTP effluent (ng/L)	<i>n</i> = 3, 33% DF		21	ND

6.6.3 North America

Table 28. UV-328 monitoring data studies summary in North America.

Location		Matrix	Others	Average	Max	Min
Canada	Arctic ⁴⁵	black-legged kittiwakes (pg/g ww): Prince Leopold Island (eggs, liver)	<i>n</i> = 6 (eggs), <i>n</i> = 5 (liver)			< 450 (eggs), < 990 (liver)
		northern fulmars (pg/g ww): Prince Leopold Island (eggs, liver), Labrador Sea (liver)	<i>n</i> = 5 (eggs), <i>n</i> = 19 (liver), 11% DF		3.8×10^3 (liver)	< 450 (egg), < 990 (liver)
		seal (liver, pg/g ww): Resolute Bay, Sachs Harbour, Arviat, Lake Melville	<i>n</i> = 14			< 900
	Not described ¹³³	WWTP influent (ng/L)	45.1 median, <i>n</i> = 34, 97% DF	34.4	126	< LOQ
		WWTP effluent (ng/L)	3.6 median, <i>n</i> = 34, 79% DF	2.6	63.1	< LOQ
		WWTP biosolids (ng/g dw)	239 median, <i>n</i> = 39, 92% DF	140	824	< LOQ
	Not described ¹³⁷	WWTP influent (ng/L)	<i>n</i> = 9, 100% DF		107	8.3
		WWTP effluent (ng/L)	<i>n</i> = 9, 100% DF		4.0	0.5
		WWTP biosolids (ng/g dw)	<i>n</i> = 12, 100% DF		278	39
		surface water (ng/L)	<i>n</i> = 32, 37.5% DF		1.5	0.05
sediment (ng/g dw)		<i>n</i> = 19, 100% DF		16	0.3	

Location		Matrix	Others	Average	Max	Min
		sediment core, 1975 to 2013 (ng/g dw)	$n = 16$, 100% DF		77	36
	Southern Ontario, urban creek ¹²⁵	water (ng/L)	$n = 12$			< 0.65
		sediment (ng/g dw)	0.4 median, $n = 12$, 100% DF	0.4	3.0	0.3
		biota whole body (ng/g lw): crayfish, chub, shiner	$n = 55$, 33 – 57% DF		1.3×10^3	< 0.4
	Southern Ontario, urban creek ¹⁴³	fish plasma (ng/g ww)	$n = 14$			ND
		fish liver (ng/g ww)	$n = 17$, 100% DF		20.7	0.6
		fish bile (ng/g ww)	$n = 17$, 0–25% DF		10.2	ND
		fish carcass (ng/g ww)	$n = 18$, 33–75% DF		3.9	ND
	Toronto, watershed ¹²⁶	suspended sediment solids (ng/g)	$n = 168$, 68% DF	240 (urban), 22.0 (rural)	1.2×10^3	0.8 ^{††} , ND
	St. Lawrence River ³⁶	surface water (ng/L)	$n = 8$, 100% DF		3.0	1.2
		Northern pike liver (ng/g lw)	$n = 40$, 40% DF		40.2	< 3.2
USA	Narragansett Bay, Rhode Island ¹¹	WWTP effluent (ppb)	$n = 1$, 100% DF		3.0×10^3	
		river water (ppb)	$n = 25$, > 32% DF		40	0.5
		river sediment (ppm)	$n = 25$, 100% DF		300	0.6
	Narragansett Bay, Rhode Island ²⁰	Narragansett Bay sediment (ng/g)	approximation		7.0×10^4	2.0×10^4
		Salem Sound sediment (ng/g)	approximation		3.5×10^4	1.0×10^3
	Narragansett Bay, Rhode Island ²¹	sediment cores (ng/g dw)	$n = 3$, 100% DF		1.2×10^3	20
	Narragansett Bay, Rhode Island ¹²⁸	clams, industrial pollution background (ng/g ww)	$n = 13$, 46% DF		65	7.0
		clams, unpolluted background (ng/g ww)	$n = 1$, 100% DF		11	
	Narragansett Bay, Rhode Island ¹³⁰	river sediment cores ($\mu\text{g/g dw}$)			7.5	
	Narragansett Bay, Rhode Island ¹²³	WWTP effluent (ppm)			4.7	0.6
		river water (ppm)	$n = 16$		0.01	0.1
sediment (ppm)		$n = 19$		100	1.0	

Location		Matrix	Others	Average	Max	Min
	Narragansett Bay, Rhode Island ¹³⁸	WWTP sludge ($\mu\text{g/g dw}$)			180	
		WWTP influent (mg/tank/99days)			276	34.4
	Narragansett Bay, Rhode Island ¹³¹	river sediment cores ($\mu\text{g/g dw}$)	$n = 2, 100\%$ DF		25	
	Saginaw and Detroit Rivers ¹⁶⁹	sediment (ng/g dw)	$n = 6, 83\%$ DF	116.0	224	0.7
	San Francisco Bay ¹⁸²	water (ng/L)			17	< 1.0
		sediment (ng/g dw)			9.0	< 1.0
	Tern Island, Hawaii ^{100,124}	black-footed albatross (ng/g lw)	$n = 18$		4.8	2.8
Kauai Island, Hawaii ⁸⁷	large plastic fragments (1.5–8 cm) ($\mu\text{g/g-plastic}$)	$n = 23, 0.04\%$ DF		0.2	< LOQ	
Several countries	USA, Canada: Great Lakes (Lake Superior, Lake Huron, Lake Erie, Niagara River, Lake Ontario) ⁴⁴	Granite Island (pg/g ww): herring gull	590 median, $n = 10, 100\%$ DF		9.4×10^3	130
		Agawa Rocks (pg/g ww): herring gull eggs	583 median, $n = 10, 100\%$ DF		3.0×10^3	190
		Chantry Island (pg/g ww): herring gull eggs	307 median, $n = 10, 90\%$ DF		1.1×10^3	< 70
		Middle Island (pg/g ww): herring gull eggs	497 median, $n = 10, 100\%$ DF		1.3×10^4	94
		Port Colborne (pg/g ww): herring gull eggs	226 median, $n = 10, 100\%$ DF		1.7×10^3	73
		Weseloh Rocks (pg/g ww): herring gull eggs	233 median, $n = 6, 83\%$ DF		310	< 70
		Hamilton Harbour (pg/g ww): herring gull eggs	693 median, $n = 10, 100\%$ DF		2.6×10^3	310
		Thunder Bay-Pie Island (pg/g ww): lake trout whole body	$n = 5, 20\%$ DF		570	< 80
		Marathon (pg/g ww): lake trout whole body	$n = 5$			< 80
		Whitefish Bay (pg/g ww): lake trout whole body	$n = 10, 40\%$ DF		6.7×10^3	< 80

Location	Matrix	Others	Average	Max	Min	
	Whitefish Bay (pg/g ww): whole body	pooled samples: deep water sculpin ($n = 35-60$), slimy sculpin ($n = 20$), smelt ($n = 12$), plankton ($n = ?$), mysis ($n = ?$)			< 80	
	Goderich (pg/g ww): lake trout whole body	$n = 5$, 20% DF		4.3×10^3	< 80	
	Dunkirk (pg/g ww): lake trout whole body	$n = 5$, 40% DF		1.4×10^3	< 80	
	Niagara-on-the-Lake (pg/g ww): lake trout whole body	2.2×10^3 median, $n = 5$, 100% DF		6.4×10^3	1.0×10^3	
	Lake Erie western basin (pg/g ww): walleye whole body	$n = 5$			< 80	
	USA (Charleston Harbour, South Carolina), Canada (Hamilton Harbour and Lake Joseph, Ontario), Great Lakes ⁴⁸	blood plasma of lake trout (pg/g ww)	465 median, $n = 4$, 50% DF		816	< 540
		blood plasma of smallmouth bass (pg/g ww)	$n = 3$			< 540
		blood plasma of snapping turtle (pg/g ww)	$n = 10$			< 540
		blood plasma of double-crested cormorants (pg/g ww)	240 median, $n = 20$, 30-60% DF		2.1×10^3	< 540
		blood plasma of gizzard shad (pg/g ww)	762 median, $n = 4$, 50% DF		3.1×10^3	< 540
blood plasma of brown bullhead (pg/g ww)		411 median, $n = 4$, 50% DF		667	< 540	
blood plasma of largemouth bass (pg/g ww)		$n = 4$, 25% DF		1.4×10^3	< 540	
blood plasma of rock bass (pg/g ww)		$n = 4$			< 540	
blood plasma of common carp (pg/g ww)		776 median, $n = 3$, 67% DF		3.8×10^3	< 540	
	dolphin plasma (pg/g ww)	$n = 4$, 50% DF		934	472 ^{††} , < LOQ	

Location		Matrix	Others	Average	Max	Min
	USA (Sarasota Bay, Florida), Canada (St. Lawrence River, Ontario) ⁴⁹	Northern pike plasma (pg/g ww)	<i>n</i> = 10			< LOQ
		white sucker whole body (pg/g ww)	<i>n</i> = 3, 67% DF		3.9 × 10 ³	242 ^{††} , < LOQ

6.6.4 Oceania

Table 29. UV-328 monitoring data studies summary in Oceania.

Location		Matrix	Others	Average	Max	Min
Australia	Port Philip Bay, Victorian estuaries ¹⁸³	water (ng/L)	<i>n</i> = 4, 100% DF		216	48.4
		sediment (µg/kg dw)	<i>n</i> = 4, 75% DF		18.1	15.5