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Persistent Organic Pollutants Review Committee
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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.

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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-destructive[1]. BZTs are preferred for thermoset plastics, organic substrates, and coatings that function against weathering[2]. 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 coatings[3]. UV-328 is used as a printing ink additive in food contact materials, too[4]. 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 cosmetics[7]. 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 plastics[8]. 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 pesticides[9].

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 regulation[5].

2. Chemical identity

2.1 Names and registry numbers

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

Common	UV-328
IUPAC	2-(2*H*-Benzotriazol-2-yl)-4,6-bis(2-methylbutan-2-yl)phenol
CAS	Phenol, 2-(2*H*-benzotriazol-2-yl)-4,6-*bis*(1,1-dimethylpropyl)-
Synonyms	2-(2*H*-Benzotriazol-2-yl)-4,6-di-*tert*-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

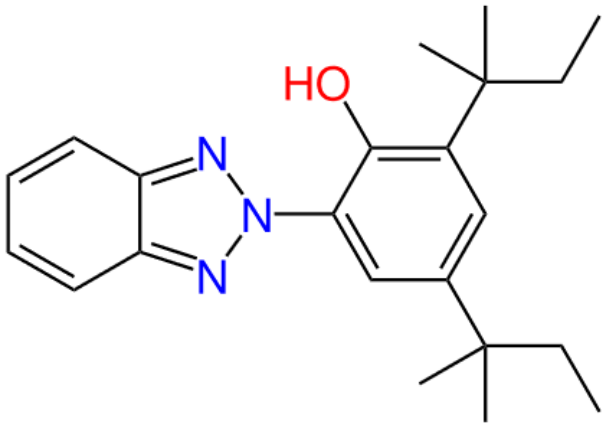


Figure 1. Chemical structure of UV-328.

Table 2. Molecular characteristics of UV-328.

Molecular formula	$C_{22}H_{29}N_3O$
Molecular weight	351.5 g/mol
SMILES code (canonical)	CCC(C)(C)c1cc(c(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.4.10[22].

	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 dossier[10]**
	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/a[10]. ECHA recently listed UV-328 as a high-volume plastic additive used in the EU[12]. 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 2016[13]. 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/a[14].

[†] Results modelled with OPERA[184].

[‡] 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 t[8].

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 2017[15].

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 either[1]. Due to its high logK_{OW} and logK_{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 soils[18]. In another similar study, UV-328 had a DT_{50} of 99–223 days[19].

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 1985[11] and the highest concentration recorded in the sediment core was 74 µg/g in 1976[20]. The concentrations near the surface remained 3–6 µg/g, which corresponds to more recent years. Similar historical concentration trends are described by Hartmann *et al.,* 2005[21] (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 estimations[22]. According to AopWin v1.92[22], the photodegradation half-life in the gas phase is 16.3 h, with an overall reaction rate constant of 15.8×10^{-12} cm^3/(molecule·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 × 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 407-000-3[1] (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, logK_{OW} 3.30[22]). M1 dissipates rapidly, i.e., in a few days, to the sediment[1]. 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 ($DegT_{50}$).

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 gaps[25], 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 $DegT_{50}$ > 180 days. Thus, UV-328 fulfils the persistence criteria.

3.2 Bioaccumulation

13. UV-328 is considered to bioaccumulate, because it has a logK_{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 (logK_{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	$BCF_{lipid\text{-}normalised}$
0.1	940[a]	1.1×10^3
0.01	$620–1.8 \times 10^3$ [a]	$740–2.2 \times 10^3$
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 measured[26–28], see Section 4.2.4. UV-328 has also been detected in foodstuff and human adipose tissue[29]. 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-327[30]. 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 study[30] 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, respectively[27]. 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 pursued[1].

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 BCF[32], 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-328[36]. 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.3.01[22].

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 logK_{OW} identify UV-328 as bioaccumulative according to the Stockholm Convention threshold (logK_{OW} > 5). There are also several experimental values where the BCF is > 5×10^3 L/kg ww. Different estimation models suggest a bioaccumulative potential, too, with BCF and BAF values > 5×10^3 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 logK_{OW} and logK_{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 logK_{OA} > 10 represents a partitioning into atmospheric aerosol particles that is virtually irreversible[37], 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 km[38].

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 logK_{OW} and logK_{OC}), a strong chemical bond between the BZT group and the aromatic ring, and resistant side chains[1]. Under specific environmental conditions, such as in the ocean, UV-328 may be partly in an anionic form or form an intramolecular hydrogen bond[39–41]. 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 logK_{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 Suite[22]: [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)	351.5
[a] logK_{AW}	−10.6
[b] logK_{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

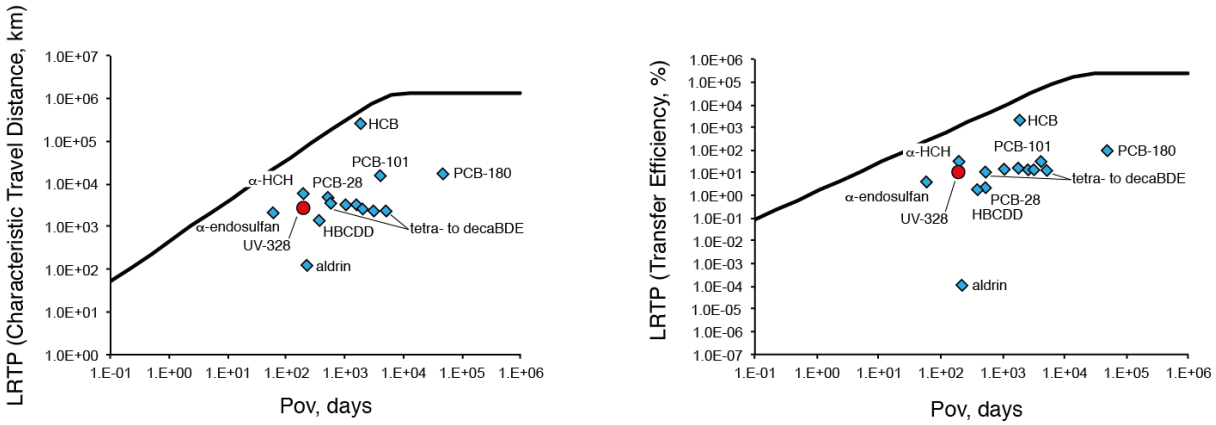


Figure 2. LRTP-P_{OV} plot comparing UV-328 (red dot) and benchmark POPs (blue diamonds) for CTD, TE and P_{OV} (adapted from [42]). The bold black line shows the LRTP of highly volatile substances. The input data (Table 22) and generated values (Table 23) are provided in Section 6.4.

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 air[43]. UV-328 has been detected in biota in Lake Superior, Great Lakes[44], and in the Canadian[45] and Norwegian Arctic[46]. 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 location[46]. 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 biota[28] (e.g. seafood, fish), accumulated in tissues (e.g. liver, muscle)[30,44–46,48,49], or in plastic debris[50] (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 2017[51] and the global annual production of plastics reached 348 Mt in 2017[52]. 79% of plastic waste may be disposed of in landfills or in the environment[51], of which about 8 Mt end up in the ocean every year[53]. 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 Poles[54], particularly in sea ice[53], south of Svalbard, Norway[55], or on the Tibetan Plateau[56]. 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 world[57]. 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 Ocean[60]. 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 aerosols[61]. 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 hydrolysis[64]. Weathering modulates bioavailability, too, because in eroded pellets chemical additives have increased distribution coefficients and slower distribution kinetics[65]. 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 layer[68]. 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 habits[67].

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 year[75]. 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 chemicals[76]. 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 years[77]. 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 matrix[84]. 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 Norway[46]. 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 polymers[82], representing around 40% of its total global production[7]. 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 pathway[85].

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 persistence[86]. 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 transport[87]. Flame retardants leaching was increased in finer particles of acrylonitrile-butadiene-styrene (ABS) polymer[88]. 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 biotransformation[89], 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 matrix[12] 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 logK_{PW}, the loss of additives from plastic particles is slower and is more likely to be limited by the aqueous boundary layer diffusion[68]. Ziccardi *et al.,* 2016 listed no logK_{PW} value for UV-328, but did for DDT and di-2-ethylhexyl phthalate (DEHP) (Table 10). As UV-328's logK_{OW} and molecular weight is in the same range as DDT and DEHP, also its logK_{PW} can be assumed to be in the same range as for DDT and DEHP[92]. 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, leaching[68]. ECHA has released a report discussing parameters to predict the release potential of chemicals from a solid matrix through diffusion or partitioning[85].

Table 10. Comparison of physico-chemical properties between UV-328 and two chemicals with derived logK_{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] logK_{OW}	[b] logK_{OC}	[c] logK_{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 system[94]. When this kind of transfer occurs, longer timeframes of exposure would have to be considered, resulting from cumulative episodes[95]. The leaching of flame retardants included in plastic matrices, e.g. BTBPE (molecular weight 687.6 g/mol, logK_{OW} 9.15[22]) and decabromodiphenyl ethane (molecular weight 971.2 g/mol, logK_{OW} 13.6[22]), into the digestive fluids of birds has already been reported. The leaching proportions were higher in finer sizes of plastic and with increasing logK_{OW}. There was a significant contribution of the ingested plastic to the bioaccumulation of highly hydrophobic flame retardants in the studied birds[88]. A previous study suggested also the transfer of polybrominated diphenyl ethers (PBDEs) (molecular weight 801.4–959.2 g/mol, logK_{OW} 10.3–12.1[22]) 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 plastics[96]. Thus, oily components in the digestive fluid facilitate leaching of hydrophobic plastic additives and their accumulation in adipose and hepatic tissues[97]. PBDEs were also found in the abdominal adipose tissue of other seabirds (*Puffinus tenuirostris*) in the North Pacific Ocean[98]. Importantly, UV-328 has been detected, among other common chemical additives, in PP plastic fragments ingested by seabirds[99]. 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 conditions[100]. 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 Islands[101], one sample of preen gland oil of thick billed murre had 654 ng/g-lipid of UV-328[102]. 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 oil[103].

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 tract[8]. 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 unlikely[10].

3.4.1.1 Acute toxicity

35. In an oral gavage study in rats and mice, no gross organ changes and an oral LD_{50} (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 LD_{50} 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 LD_{50} > 2.0 g/kg bw. These results were in accordance with other studies, too[10].

36. The measured acute inhalation LC_{50} (lethal concentration) in rats was from 0.4–4.1 g/L[105]. A Ciba-Geigy study (similar to OECD TG 403, non-GLP, 1973) in rats generated an LC_{50} > 0.4 mg/L, after single exposure for 4 h. Another study (similar to OECD TG 403, non-GLP, 1977) in rats reported an LC_{50} > 0.13 mg/L air, after 1 h[10]. Measured dermal LD_{50} in rabbits was from 1.1–3.0 g/kg bw[105]. 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 reported[10].

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, mutagenicity[110], 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 receptor[111]. UV-328 showed no relevant estrogenic activity[112]. 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)SAR[113] and ECOSAR[22]. ECOSAR predicts a chronic value (ChV, geometric mean of NOEC (no observed effect concentration) and LOEC (lowest observed effect concentration)) and LC_{50}/EC_{50} < 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 fish[8]. Again, most testing conditions were above water solubility for UV-328.

Table 11. ECOSAR v1.11 results for the BZTs class[22]. 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 EC_{50} 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 EC_{50} > 10 mg/L was reported after 72 h[114]. In microorganisms (activated sludge), the EC_{50} and IC_{50} 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 LC_0/EC_0 (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 EC_{50} > 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 EC_{50} > 10 mg/L after 48 h. After 24 h, under static conditions, the EC_{50} was > 100 mg/L[10]. In yet another study with *D. magna* (OECD TG 202, non-GLP, 1988), after 24 h, values of EC_{50}/EC_{100} > 100 mg/L and EC_0 58 mg/L were reported[116].

45. In fish (*Danio rerio*), a static study (OECD TG 203, non-GLP, 1988), generated a NOEC/LC_{50} > 100 mg/L, after 96 h[117]. Another semi-static study in fish (*Oryzias latipes*, OECD TG 203, GLP, 2007), determined an LC_{50} > 0.08 mg/L, after 96 h. It was a limit test and LC_{50} 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 compartment[8]. 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-328[36].

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 bodies[10]. 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 ground[110]. 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 rates[121].

51. UV-328 may enter soils from the application of wastewater (WW) biosolids, commonly used in enrichment[8]. 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 logK_{OC}. If released into water, UV-328 will adsorb to suspended solids and sediments[110].

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 Bay[123]), sewage treatment plants (0.55–4.7 mg/L in WWTP effluents[123]), river sediment (300 mg/kg in Narragansett Bay[11]), or secondary poisoning (0.65 mg/kg lw in preen gland oil of a thick billed murre from Pribilof Island[99]). 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 crayfish[125]) 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-328[121].

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-328[121].

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 visible[126].

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 level[20]. 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 treatment[120]. 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 contamination[132]. 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 vector[133].

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/L[136]. In Sweden, UV-328 was present

in the tens of µg/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 µg/g dw[43]. In Norway, WWTP effluents had 7–57 ng/L of UV-328[46].

63. In Canada, UV-328 (140 ng/g dw) and UV-234 were the most abundant BZT UV absorbers[133]. 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 Ontario[137]. In Narragansett Bay, a municipal WWTP upstream the old chemical plant had UV-327 and UV-328 in its sludge in µg/g dw level[138]. 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 2011[139]. The EDI via consumption of breast milk was estimated to be 0.36 µg/kg bw/day. The study points out the lack of an established provisional tolerable daily intake (PTDI) value for benzotriazoles[139]. 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 ng/g lw, 16% DF; levels lower than the reference dose)[140]. The reference dose for UV-328 used in this study is 10 µg/kg bw/day[109]. Human adipose tissue from Japan, Republic of Korea, China, Spain, and USA had UV-328 also (up to 35 ng/g lw, 45.2% DF[29]).

65. In a Canadian urban creek, UV-328 was present in 33–57% of the biota sampled, at concentrations of up to 1.3 µg/g lw (crayfish)[125]. In the Pearl River Estuary in China, several BZTs, including UV-328, were present up to 258.9 ng/g lw in marine wildlife[141]. In an earlier study, UV-328 had not been detected in wild aquatic organisms , but it was present in farmed red snapper (0.8 ng/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 ng/g ww in common carp[48]. Similar results had been reported before in marine biota from the USA (Florida) and Canada (Ontario), up to 3.9 ng/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 0.6–20.7 ng/g ww concentration range[143]. In Japanese marine mammal blubber samples collected in 1990, there were maximum concentrations around 70 ng/g lw[127]. In finless porpoises, the mean concentration of UV-328 was 38 ng/g lw, which was about four times higher than in small fish (8.4 ng/g lw). UV-328 concentrations in marine organisms varied among species, and higher concentrations were detected in livers of mullets and hammerhead sharks[30]. UV absorbers were present in all samples from Ariake Sea marine organisms, having UV-328 concentrations of up to 55 ng/g ww[144]. Very high concentrations, up to 460 ng/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) also[144]. 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 (< 0.2–55.0 ng/g ww, 89.3% DF)[27]. UV-328 was predominant in finless porpoises[27]. In the Norwegian fjords, UV-328 was present in biota (up to 19.5 ng/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 detected[147]. UV-328 was present in all analysed biomatrices (moss and periphyton, brown trout) from a Norwegian river in low ng/g levels[148].

66. Foodstuff samples from Japan and the Republic of Korea also contained BZTs. Contamination was ubiquitous, with highest concentrations in seafood (1.7 ng/g ww) and meat (1 ng/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 ng/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 (1–460 ng/g lw)[144]. UV-328 and UV-327 were dominant in higher trophic species[144]. Another study with mussels reported UV-328 frequently with highest concentrations in Hong Kong and the Republic of Korea (around 0.8 µg/g lw). In the USA, UV-328 was detected in few samples of mussels, and showed a maximum around 0.3 µg/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 ng/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 ubiquitous[153]. 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 chemicals[158].

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 range[46]: 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 position[44]. 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.

References

1. ECHA. *Member State Committee Support Document for Identification of 2-(2H-Benzotriazol-2-yl)-4,6-ditertpentylphenol (UV-328) as a Substance of Very High Concern Because of Its PBT/vPvB Properties*. (2014).
2. MatWeb. M Chemical GUARD DOG™ UV328 Benzotriazole UV Absorber.
3. Mayzo, I. BLS® 1328. (2015).
4. Van Bossuyt, M., Van Hoeck, E., Vanhaecke, T., Rogiers, V. & Mertens, B. Printed paper and board food contact materials as a potential source of food contamination. *Regul. Toxicol. Pharmacol.* (2016) doi:10.1016/j.yrtph.2016.06.025.
5. Official Journal of the European Union. Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH). Legislation, Volume 396 of 30. December 2006.
6. US EPA. Toxic Substances Control Act: 15 U.S.C. §§ 2601 et seq.; last amended in 2016. *Public Law* vol. 99 469 (2016).
7. Germany. *Annex XV Report: Proposal for Identification of a Substance of Very High Concern on the Basis of the Criteria Set Out in REACH Article 57*. (2014).
8. Canada. *Screening Assessment Report on Phenol, 2-(2H-Benzotriazol-2-Yl)-4,6-Bis(1,1-Dimethylpropyl)-(BDTP)*. (2016).
9. ECHA. *Estimating the number and types of applications for 11 substances added to the Authorisation List in February 2020*. (2020) doi:10.2823/11134.
10. ECHA. 2-(2H-benzotriazol-2-yl)-4,6-ditertpentylphenol Registration Dossier. *REACH* (2018).
11. Lopez-Avila, V. & Hites, R. A. Organic compounds in an industrial wastewater. Their transport into sediments. *Environ. Sci. Technol.* **14**, 1382–1390 (1980).
12. ECHA. High-volume plastic additives mapped. (2019).
13. SPIN. 2-(2-Hydroxy-3,5-Di-Tert-Amylphenyl)-2H-Benzotriazole. (2018).
14. Brooke, D. & Burns, J. *Environmental prioritisation of low production volume substances under REACH: PBT screening*. (2010).
15. J-Check. 2-(2H-1,2,3-Benzotriazol-2-yl)-4,6-di-tert-pentylphenol. *NITE* (2018).
16. Gao, J., Ellis, L. B. M. & Wackett, L. P. The University of Minnesota Biocatalysis/Biodegradation Database: improving public access. *Nucleic Acids Res.* **38**, D488–D491 (2009).
17. Ciba-Geigy. *Test for Ready Biodegradability of Tinuvin 328 in the Modified Sturm Test, OECD-Guideline No. 301 B*. (1988).
18. Lai, H.-J. *et al.* Occurence and dissipation of benzotriazoles and benzotriazole ultraviolet stabilizers in biosolid-amended soils. *Environ. Toxicol. Chem.* **33**, 761–767 (2014).
19. Lai, H.-J. *et al.* Field dissipation and plant uptake of benzotriazole ultraviolet stabilizers in biosolid-amended soils. *Environ. Sci. Process. Impacts* **16**, 558–566 (2014).
20. Cantwell, M. G. *et al.* Source determination of benzotriazoles in sediment cores from two urban estuaries on the Atlantic Coast of the United States. *Mar. Pollut. Bull.* **101**, 208–218 (2015).
21. Hartmann, P. C., Quinn, J. G., Cairns, R. W. & King, J. W. Depositional history of organic contaminants in Narragansett Bay, Rhode Island, USA. *Mar. Pollut. Bull.* **50**, 388–395 (2005).
22. EPA, U. Estimation Programs Interface Suite for Microsoft® Windows. (2012).
23. Rorije, E., Verbruggen, E. M. J., Hollander, A., Traas, T. P. & Janssen, M. P. M. Identifying potential POP and PBT substances: Development of a new Persistence/Bioaccumulation-score, RIVM Report 601356001/2011. (2011).
24. Strempel, S., Scheringer, M., Ng, C. A. & Hungerbühler, K. Screening for PBT Chemicals among the "Existing" and "New" Chemicals of the EU. *Environ. Sci. Technol.* **46**, 5680–5687 (2012).
25. Brandt, M., Becker, E., Jöhncke, U., Sättler, D. & Schulte, C. A weight-of-evidence approach to assess chemicals: case study on the assessment of persistence of 4,6-substituted phenolic benzotriazoles in the environment. *Environ. Sci. Eur.* **28**, 4 (2016).

26. Kim, J.-W. *et al.* Contamination and bioaccumulation of benzotriazole ultraviolet stabilizers in fish from Manila Bay, the Philippines using an ultra-fast liquid chromatography–tandem mass spectrometry. *Chemosphere* **85**, 751–758 (2011).

27. Nakata, H., Murata, S. & Filatreau, J. Occurrence and Concentrations of Benzotriazole UV Stabilizers in Marine Organisms and Sediments from the Ariake Sea, Japan. *Environ. Sci. Technol.* **43**, 6920–6926 (2009).

28. Nakata, H. *et al.* Asia–Pacific mussel watch for emerging pollutants: Distribution of synthetic musks and benzotriazole UV stabilizers in Asian and US coastal waters. *Mar. Pollut. Bull.* **64**, 2211–2218 (2012).

29. Yanagimoto, H. *et al.* Poster: Occurrence of Benzotriazole UV Stabilizers and Synthetic Musks in Human Adipose Tissues Collected from Japan, South Korea, China, Spain, and the USA. in *32nd SETAC (Society of Environmental Toxicology and Chemistry) North America* 257 (2011).

30. Nakata, H., Shinohara, R., Murata, S. & Watanabe, M. Detection of benzotriazole UV stabilizers in the blubber of marine mammals by gas chromatography-high resolution mass spectrometry (GC-HRMS). *J. Environ. Monit.* **12**, 2088–2092 (2010).

31. Arnot, J. A. & Gobas, F. A Generic QSAR for Assessing the Bioaccumulation Potential of Organic Chemicals in Aquatic Food Webs. *QSAR Comb. Sci.* **22**, 337–345 (2003).

32. Arnot, J. & Gobas, F. A review of bioconcentration factor (BCF) and bioaccumulation factor (BAF) assessments for organic chemicals in aquatic organisms. *Environ. Rev.* **14**, 257–297 (2006).

33. Arnot, J. A., Mackay, D., Parkerton, T. F. & Bonnell, M. A database of fish biotransformation rates for organic chemicals. *Environ. Toxicol. Chem.* **27**, 2263–2270 (2008).

34. Arnot, J. A., Mackay, D. & Bonnell, M. Estimating metabolic biotransformation rates in fish from laboratory data. *Environ. Toxicol. Chem.* **27**, 341–351 (2008).

35. Arnot, J. A. & Gobas, F. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. *Environ. Toxicol. Chem.* **23**, 2343–2355 (2004).

36. Giraudo, M. *et al.* Food-borne exposure of juvenile rainbow trout (Oncorhynchus mykiss) to benzotriazole UV stabilizers alone and in mixture induces specific transcriptional changes. *Environ. Toxicol. Chem.* **n/a**, (2020).

37. Wania, F. Assessing the Potential of Persistent Organic Chemicals for Long-Range Transport and Accumulation in Polar Regions. *Environ. Sci. Technol.* **37**, 1344–1351 (2003).

38. van der Does, M., Knippertz, P., Zschenderlein, P., Giles Harrison, R. & Stuut, J.-B. W. The mysterious long-range transport of giant mineral dust particles. *Sci. Adv.* **4**, eaau2768 (2018).

39. Waiblinger, F. *et al.* Light-Induced Opening of the Intramolecular Hydrogen Bond of UV Absorbers of the 2-(2-Hydroxyphenyl)-1,3,5-triazine and the 2-(2-Hydroxyphenyl)benzotriazole Type. *J. Phys. Chem. A* **104**, 1100–1106 (2000).

40. Stein, M. *et al.* Influence of Polymer Matrixes on the Photophysical Properties of UV Absorbers. *J. Phys. Chem. A* **106**, 2055–2066 (2002).

41. Werner, T. Triplet deactivation in benzotriazole-type ultraviolet stabilizers. *J. Phys. Chem.* **83**, 320–325 (1979).

42. Scheringer, M., MacLeod, M. & Wegmann, F. The OECD POV and LRTP Screening Tool. (2009).

43. Brorström-Lundén, E. *et al. Screening of benzothiazoles, benzenediamines, dicyclohexylamine and benzotriazoles 2009, Report B2023. IVL-Swedish Environmental Research Institute* (2011).

44. Lu, Z. *et al.* Substituted Diphenylamine Antioxidants and Benzotriazole UV Stabilizers in Aquatic Organisms in the Great Lakes of North America: Terrestrial Exposure and Biodilution. *Environ. Sci. Technol.* **52**, 1280–1289 (2018).

45. Lu, Z. *et al.* Occurrence of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in Arctic seabirds and seals. *Sci. Total Environ.* **663**, 950–957 (2019).

46. Schlabach, M. *et al. Screening Programme 2017 – AMAP Assessment Compounds, NILU Project no. O-117085*. (2018) doi:10.13140/RG.2.2.36121.47200.

47. Tashiro, Y. & Kameda, Y. Concentration of organic sun-blocking agents in seawater of beaches and coral reefs of Okinawa Island, Japan. *Mar. Pollut. Bull.* **77**, 333–340 (2013).

48. Lu, Z. *et al.* Substituted diphenylamine antioxidants and benzotriazole UV stabilizers in blood plasma of fish, turtles, birds and dolphins from North America. *Sci. Total Environ.* **647**, 182–190 (2019).

49. Lu, Z., Peart, T. E., Cook, C. J. & De Silva, A. O. Simultaneous determination of substituted diphenylamine antioxidants and benzotriazole ultra violet stabilizers in blood plasma and fish homogenates by ultra high

performance liquid chromatography–electrospray tandem mass spectrometry. *J. Chromatogr. A* **1461**, 51–58 (2016).

50. Nelms, S. E. *et al.* Microplastics in marine mammals stranded around the British coast: ubiquitous but transitory? *Sci. Rep.* **9**, 1075 (2019).

51. Geyer, R., Jambeck, J. R. & Law, K. L. Production, use, and fate of all plastics ever made. *Sci. Adv.* **3**, e1700782 (2017).

52. PlasticsEurope. *Plastics – the Facts 2018*. (2018).

53. Peeken, I. *et al.* Arctic sea ice is an important temporal sink and means of transport for microplastic. *Nat. Commun.* **9**, 1505 (2018).

54. Obbard, R. W. Microplastics in Polar Regions: The role of long range transport. *Curr. Opin. Environ. Sci. Heal.* **1**, 24–29 (2018).

55. Lusher, A. L., Tirelli, V., O'Connor, I. & Officer, R. Microplastics in Arctic polar waters: the first reported values of particles in surface and sub-surface samples. *Sci. Rep.* **5**, 14947 (2015).

56. Zhang, K. *et al.* Microplastic pollution of lakeshore sediments from remote lakes in Tibet plateau, China. *Environ. Pollut.* **219**, 450–455 (2016).

57. Lavers, J. L. & Bond, A. L. Exceptional and rapid accumulation of anthropogenic debris on one of the world's most remote and pristine islands. *Proc. Natl. Acad. Sci.* **114**, 6052 LP – 6055 (2017).

58. Lavers, J. L., Dicks, L., Dicks, M. R. & Finger, A. Significant plastic accumulation on the Cocos (Keeling) Islands, Australia. *Sci. Rep.* **9**, 7102 (2019).

59. Bouwman, H., Evans, S. W., Cole, N., Choong Kwet Yive, N. S. & Kylin, H. The flip-or-flop boutique: Marine debris on the shores of St Brandon's rock, an isolated tropical atoll in the Indian Ocean. *Mar. Environ. Res.* (2016) doi:10.1016/j.marenvres.2015.12.013.

60. Eriksen, M., Thiel, M. & Lebreton, L. Nature of Plastic Marine Pollution in the Subtropical Gyres BT - Hazardous Chemicals Associated with Plastics in the Marine Environment. in (eds. Takada, H. & Karapanagioti, H. K.) 135–162 (Springer International Publishing, 2019). doi:10.1007/698_2016_123.

61. Cheng, W. *et al.* Organophosphorus esters in the oceans and possible relation with ocean gyres. *Environ. Pollut.* **180**, 159–164 (2013).

62. Van Sebille, E., England, M. H. & Froyland, G. Origin, dynamics and evolution of ocean garbage patches from observed surface drifters. *Environ. Res. Lett.* **7**, 6 (2012).

63. Maximenko, N., Hafner, J. & Niiler, P. Pathways of marine debris derived from trajectories of Lagrangian drifters. *Mar. Pollut. Bull.* **65**, 51–62 (2012).

64. Andrady, A. L., Hamid, H. & Torikai, A. Effects of solar UV and climate change on materials. *Photochem. Photobiol. Sci.* **10**, 292–300 (2011).

65. Karapanagioti, H. K. & Klontza, I. Testing phenanthrene distribution properties of virgin plastic pellets and plastic eroded pellets found on Lesvos island beaches (Greece). *Mar. Environ. Res.* **65**, 283–290 (2008).

66. Andrady, A. L. Microplastics in the marine environment. *Mar. Pollut. Bull.* **62**, 1596–1605 (2011).

67. Shim, W. J., Hong, S. H. & Eo, S. Marine Microplastics: Abundance, Distribution, and Composition (Chapter 1). in *Microplastic Contamination in Aquatic Environments - An Emerging Matter of Environmental Urgency* (ed. Zeng, E.) 1–26 (Elsevier, 2018). doi:10.1016/B978-0-12-813747-5.00001-1.

68. Kwon, J.-H., Chang, S., Hong, S. H. & Shim, W. J. Microplastics as a vector of hydrophobic contaminants: Importance of hydrophobic additives. *Integr. Environ. Assess. Manag.* **13**, 494–499 (2017).

69. O'Connor, I. A., Golsteijn, L. & Hendriks, A. J. Review of the partitioning of chemicals into different plastics: Consequences for the risk assessment of marine plastic debris. *Mar. Pollut. Bull.* **113**, 17–24 (2016).

70. Jahnke, A. *et al.* Reducing Uncertainty and Confronting Ignorance about the Possible Impacts of Weathering Plastic in the Marine Environment. *Environ. Sci. Technol. Lett.* **4**, 85–90 (2017).

71. Hong, S. H., Shim, W. J. & Jang, M. Chemicals Associated With Marine Plastic Debris and Microplastics: Analyses and Contaminant Levels (Chapter 9). in *Microplastic Contamination in Aquatic Environments - An Emerging Matter of Environmental Urgency* (ed. Zeng, E.) 271–315 (Elsevier, 2018). doi:10.1016/B978-0-12-813747-5.00009-6.

72. Groh, K. J. *et al.* Overview of known plastic packaging-associated chemicals and their hazards. *Sci. Total Environ.* **651**, 3253–3268 (2019).

73. Bergmann, M., Gutow, L. & Klages, M. *Marine anthropogenic litter*. (Springer, 2015).

74. Lithner, D., Larsson, Å. & Dave, G. Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Sci. Total Environ.* **409**, 3309–3324 (2011).

75. Rydberg, T. *et al.* Emissions of additives from plastics in the societal material stock: a case study for Sweden (Chapter 14). in *Handbook of Environmental Chemistry. Global Risk-Based Management of Chemical Additives I* 253–264 (Springer, 2011).

76. OECD. *Emission Scenario Document On Plastic Additives. Series On Emission Scenario Documents. Number 3*. (2009).

77. OECD. *Complementing Document To The Emission Scenario Document On Plastic Additives: Plastic Additives During The Use Of End Products. Series on Emission Scenario Documents No. 38*. (2019).

78. Rochman, C. M. *et al.* Classify plastic waste as hazardous. *Nature* **494**, 169 (2013).

79. Rochman, C. M. The Role of Plastic Debris as Another Source of Hazardous Chemicals in Lower-Trophic Level Organisms BT - Hazardous Chemicals Associated with Plastics in the Marine Environment. in (eds. Takada, H. & Karapanagioti, H. K.) 281–295 (Springer International Publishing, 2019). doi:10.1007/698_2016_17.

80. Hirai, H. *et al.* Organic micropollutants in marine plastics debris from the open ocean and remote and urban beaches. *Mar. Pollut. Bull.* **62**, 1683–1692 (2011).

81. Al-Odaini, N. A., Shim, W. J., Han, G. M., Jang, M. & Hong, S. H. Enrichment of hexabromocyclododecanes in coastal sediments near aquaculture areas and a wastewater treatment plant in a semi-enclosed bay in South Korea. *Sci. Total Environ.* **505**, 290–298 (2015).

82. Ogata, Y. *et al.* International Pellet Watch: Global monitoring of persistent organic pollutants (POPs) in coastal waters. 1. Initial phase data on PCBs, DDTs, and HCHs. *Mar. Pollut. Bull.* **58**, 1437–1446 (2009).

83. Heskett, M. *et al.* Measurement of persistent organic pollutants (POPs) in plastic resin pellets from remote islands: Toward establishment of background concentrations for International Pellet Watch. *Mar. Pollut. Bull.* **64**, 445–448 (2012).

84. Jang, M. *et al.* Styrofoam Debris as a Source of Hazardous Additives for Marine Organisms. *Environ. Sci. Technol.* **50**, 4951–4960 (2016).

85. ECHA. *Plastic additives initiative - Supplementary Information on Scope and Methods*. (2019).

86. Rani, M. *et al.* Benzotriazole-type ultraviolet stabilizers and antioxidants in plastic marine debris and their new products. *Sci. Total Environ.* **579**, 745–754 (2017).

87. Tanaka, K., Takada, H., Ikenaka, Y., Nakayama, S. M. M. & Ishizuka, M. Occurrence and concentrations of chemical additives in plastic fragments on a beach on the island of Kauai, Hawaii. *Mar. Pollut. Bull.* (2020) doi:10.1016/j.marpolbul.2019.110732.

88. Guo, H., Zheng, X., Ru, S., Luo, X. & Mai, B. The leaching of additive-derived flame retardants (FRs) from plastics in avian digestive fluids: The significant risk of highly lipophilic FRs. *J. Environ. Sci.* **85**, 200–207 (2019).

89. Denghel, H., Leibold, E. & Göen, T. Oxidative phase I metabolism of the UV absorber 2-(2H-benzotriazol-2-yl)-4,6-di-tert-pentylphenol (UV 328) in an in vitro model with human liver microsomes. *Toxicol. Vitr.* (2019) doi:10.1016/j.tiv.2019.06.012.

90. Syberg, K. *et al.* Microplastics: addressing ecological risk through lessons learned. *Environ. Toxicol. Chem.* **34**, 945–953 (2015).

91. Lee, H., Chang, S., Kim, S.-K. & Kwon, J.-H. Fugacity analysis of polycyclic aromatic hydrocarbons between microplastics and seawater. *Ocean Sci. J.* **52**, 43–55 (2017).

92. Ziccardi, L. M., Edgington, A., Hentz, K., Kulacki, K. J. & Kane Driscoll, S. Microplastics as vectors for bioaccumulation of hydrophobic organic chemicals in the marine environment: A state-of-the-science review. *Environ. Toxicol. Chem.* **35**, 1667–1676 (2016).

93. Wang, F., Wang, F. & Zeng, E. Y. Sorption of Toxic Chemicals on Microplastics (Chapter 7). in *Microplastic Contamination in Aquatic Environments - An Emerging Matter of Environmental Urgency* (ed. Zeng, E.) 225–247 (Elsevier, 2018). doi:10.1016/B978-0-12-813747-5.00007-2.

94. Browne, M. A., Dissanayake, A., Galloway, T. S., Lowe, D. M. & Thompson, R. C. Ingested Microscopic Plastic Translocates to the Circulatory System of the Mussel, Mytilus edulis (L.). *Environ. Sci. Technol.* **42**, 5026–5031 (2008).

95. Kershaw, P. J. & Rochman, C. M. Sources, fate and effects of microplastics in the marine environment: part 2 of a global assessment. *Reports Stud. Jt. Gr. Expert. Sci. Asp. Mar. Environ. Prot. eng no. 93* (2015).

96. Tanaka, K. *et al.* Facilitated Leaching of Additive-Derived PBDEs from Plastic by Seabirds' Stomach Oil and Accumulation in Tissues. *Environ. Sci. Technol.* **49**, 11799–11807 (2015).

97. Tanaka, K., Yamashita, R. & Takada, H. Transfer of Hazardous Chemicals from Ingested Plastics to Higher-Trophic-Level Organisms. in (eds. Takada, H. & Karapanagioti, H. K.) 267–280 (Springer International Publishing, 2019). doi:10.1007/698_2018_255.

98. Tanaka, K. *et al.* Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastics. *Mar. Pollut. Bull.* **69**, 219–222 (2013).

99. Tanaka, K., van Franeker, J. A., Deguchi, T. & Takada, H. Piece-by-piece analysis of additives and manufacturing byproducts in plastics ingested by seabirds: Implication for risk of exposure to seabirds. *Mar. Pollut. Bull.* **145**, 36–41 (2019).

100. Tanaka, K. *et al.* In Vivo Accumulation of Plastic-Derived Chemicals into Seabird Tissues. *Curr. Biol.* **30**, (2020).

101. Britannica, E. Pribilof Islands. *Encyclopædia Britannica, inc.* (2016).

102. Takada, H., Tanaka, K., Yamashita, R. & Watanuki, Y. ENVR 139: Transfer of additives from ingested plastics to seabirds and their accumulation in the tissue. in *ACS Spring 2019 National Meeting & Exposition* (2019).

103. Takada, H. Hazardous chemicals in marine plastics and their threat to marine organisms - Keynote. in *Dioxin* (2019).

104. RAC, C. for R. A. *Opinion on the specific target organ toxicity of 2-benzotriazol-2-yl-4,6-di-tert-butylphenol (UV-320) and 2-(2H-benzotriazol-2-yl)-4,6-ditertpentylphenol (UV-328)*. (2013).

105. Ciba-Geigy. *Acute Oral LD50 In The Rat Of TK 10046*. (1978).

106. Til, H., van der Meulen, H., Huismans, J. & de Groot, A. *Short-term (49-day) and sub-chronic (90-day) toxicity studies with 'BY 1137' in rats*. (1968).

107. Ciba-Geigy. *Three months Toxicity Study. Tinuvin 328. Dietary administration – Beagle Dogs*. (1970).

108. Ciba-Geigy. *Final Report TK 10047 - Three-month toxicity study on dogs*. (1981).

109. OECD. *Case Study on the Use of an Integrated Approach to Testing and Assessment for the Repeated-Dose Toxicity of Phenolic Benzotriazoles - ENV/JM/MONO(2017)23*. (2017).

110. (US), N. L. of M. 2-(2H-Benzotriazol-2-yl)-4,6-di-tert-pentylphenol; CASRN: 25973-55-1. *Hazardous Substances Data Bank [Internet]* (2018).

111. Zhuang, S. *et al.* Benzotriazole UV 328 and UV-P showed distinct antiandrogenic activity upon human CYP3A4-mediated biotransformation. *Environ. Pollut.* **220**, 616–624 (2017).

112. Kawamura, Y. *et al.* Estrogenic Activities of UV Stabilizers Used in Food Contact Plastics and Benzophenone Derivatives Tested by the Yeast Two-Hybrid Assay. *J. Heal. Sci.* **49**, 205–212 (2003).

113. Denmark, T. U. of. Danish (Q)SAR Database. *Division of Diet, Disease Prevention and Toxicology, National Food Institute* (2018).

114. Hicks, S. & Gledhill, D. *Acute Toxicity Screen of Tinuvin 328 to Scenedesmus subspicatus*. (1993).

115. Kim, J.-W., Chang, K.-H., Isobe, T. & Tanabe, S. Acute toxicity of benzotriazole ultraviolet stabilizers on freshwater crustacean (Daphnia pulex). *J. Toxicol. Sci.* **36**, 247–251 (2011).

116. Ciba-Geigy. *Test for acute Toxicity of TK 10046 to Daphnia magna, OECD-Guideline No. 202*. (1988).

117. Ciba-Geigy. *Test for acute toxicity of TK 10046 to Zebra Fish, OECD-Guideline No. 203*. (1988).

118. Giraudo, M. *et al.* Transcriptional and cellular effects of benzotriazole UV stabilizers UV-234 and UV-328 in the freshwater invertebrates Chlamydomonas reinhardtii and Daphnia magna. *Environ. Toxicol. Chem.* **36**, 3333–3342 (2017).

119. Brausch, J. M. & Rand, G. M. A review of personal care products in the aquatic environment: Environmental concentrations and toxicity. *Chemosphere* **82**, 1518–1532 (2011).

120. Nakata, H. & Shinohara, R.-I. Concentrations of Benzotriazole UV Stabilizers and Polycyclic Musks in Wastewater Treatment Plant Samples in Japan. *Interdiscip. Stud. Environ. Chem. — Environ. Specim. Bank* **4**, 51–59 (2010).

121. ECHA. 2-(2H-benzotriazol-2-yl)-4,6-ditertpentylphenol Brief Profile. *REACH* (2018).

122. Kim, J.-W. *et al.* Contamination of benzotriazole ultraviolet stabilizers in house dust from the Philippines: Implications on human exposure. *Sci. Total Environ.* **424**, 174–181 (2012).

123. Jungclaus, G., Avila, V. & Hites, R. Organic compounds in an industrial Wastewater: a case study of their environmental impact. *Environ. Sci. Technol.* **12**, 88–96 (1978).

124. Tanaka, K. *et al.* Direct In Vivo Evidence for Accumulation of Plastic Derived Chemicals in Seabird Tissue. *Curr. Biol.* (2019).

125. Lu, Z. *et al.* Distribution, Partitioning and Bioaccumulation of Substituted Diphenylamine Antioxidants and Benzotriazole UV Stabilizers in an Urban Creek in Canada. *Environ. Sci. Technol.* **50**, 9089–9097 (2016).

126. Parajulee, A., Lei, Y. D., Kananathalingam, A., Mitchell, C. P. J. & Wania, F. Investigating the Sources and Transport of Benzotriazole UV Stabilizers during Rainfall and Snowmelt across an Urbanization Gradient. *Environ. Sci. Technol.* **52**, 2595–2602 (2018).

127. Nakata, H. Benzotriazole UV Stabilizer (BUVS) in Human and Wildlife - Is it a POPs? in *4th International Conference on Environmental Health Science - 2011* (2011).

128. Pruell, R. J., Hoffman, E. J. & Quinn, J. G. Total hydrocarbons, polycyclic aromatic hydrocarbons and synthetic organic compounds in the Hard shell clam, Mercenaria mercenaria, purchased at commercial seafood stores. *Mar. Environ. Res.* **11**, 163–181 (1984).

129. Hites, R. A., Jungclaus, G. A., Lopez-Avila, V. & Sheldon, L. S. Potentially Toxic Organic Compounds in Industrial Wastewaters and River Systems: Two Case Studies. *Monit. Toxic Subst.* **94**, 5–63 (1979).

130. Pruell, R. J. & Quinn, J. G. Geochemistry of organic contaminants in Narragansett Bay sediments. *Estuar. Coast. Shelf Sci.* **21**, 295–312 (1985).

131. Reddy, C. M., Quinn, J. G. & King, J. W. Free and Bound Benzotriazoles in Marine and Freshwater Sediments. *Environ. Sci. Technol.* **34**, 973–979 (2000).

132. Peng, X. *et al.* Persistence, temporal and spatial profiles of ultraviolet absorbents and phenolic personal care products in riverine and estuarine sediment of the Pearl River catchment, China. *J. Hazard. Mater.* **323**, 139–146 (2017).

133. Lu, Z., Smyth, S. A., Peart, T. E. & De Silva, A. O. Occurrence and fate of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in various Canadian wastewater treatment processes. *Water Res.* **124**, 158–166 (2017).

134. Montesdeoca-Esponda, S., Álvarez-Raya, C., Torres-Padrón, M. E., Sosa-Ferrera, Z. & Santana-Rodríguez, J. J. Monitoring and environmental risk assessment of benzotriazole UV stabilizers in the sewage and coastal environment of Gran Canaria (Canary Islands, Spain). *J. Environ. Manage.* **233**, 567–575 (2019).

135. García-Guerra, R. B. *et al.* Rapid monitoring of residual UV-stabilizers in seawater samples from beaches using fabric phase sorptive extraction and UHPLC-MS/MS. *Chemosphere* **164**, 201–207 (2016).

136. Carpinteiro, I., Ramil, M., Rodríguez, I. & Nogueira, J. M. F. Combining stir-bar sorptive extraction and large volume injection-gas chromatography-mass spectrometry for the determination of benzotriazole UV stabilizers in wastewater matrices. *J. Sep. Sci.* **35**, 459–467 (2012).

137. De Silva, A., Muir, D. & Smyth, S. *Unpublished monitoring data submitted to Ecological Assessment Division of Environment Canada*. (2014).

138. Oviatt, C. A. *et al.* Fate and effects of sewage sludge in the coastal marine environment: A mesocosm experiment. *Mar. Ecol. Ser.* **41**, 187–203 (1987).

139. Lee, S. *et al.* Synthetic musk compounds and benzotriazole ultraviolet stabilizers in breast milk: Occurrence, time–course variation and infant health risk. *Environ. Res.* **140**, 466–473 (2015).

140. Kim, J.-W. *et al.* Occurrence of benzotriazole ultraviolet stabilizers (BUVSs) in human breast milk from three Asian countries. *Sci. Total Environ.* **655**, 1081–1088 (2019).

141. Peng, X. *et al.* Bioaccumulation and biomagnification of ultraviolet absorbents in marine wildlife of the Pearl River Estuarine, South China Sea. *Environ. Pollut.* **225**, 55–65 (2017).

142. Peng, X., Jin, J., Wang, C., Ou, W. & Tang, C. Multi-target determination of organic ultraviolet absorbents in organism tissues by ultrasonic assisted extraction and ultra-high performance liquid chromatography–tandem mass spectrometry. *J. Chromatogr. A* **1384**, 97–106 (2015).

143. Lu, Z., De Silva, A. O., Peart, T. E., Cook, C. J. & Tetreault, G. R. Tissue Distribution of Substituted Diphenylamine Antioxidants and Benzotriazole Ultraviolet Stabilizers in White Sucker (Catostomus commersonii) from an Urban Creek in Canada. *Environ. Sci. Technol. Lett.* **4**, 433–438 (2017).

144. Nakata, H. *et al.* Occurrence and Concentrations of Persistent Personal Care Products, Organic UV Filters, in the Marine Environment. *Interdiscip. Stud. Environ. Chem. — Environ. Res. Asia* 239–246 (2009).

145. Langford, K. H., Reid, M. J., Fjeld, E., Øxnevad, S. & Thomas, K. V. Environmental occurrence and risk of organic UV filters and stabilizers in multiple matrices in Norway. *Environ. Int.* **80**, 1–7 (2015).

146. Thomas, K. *et al. Screening programme 2013: New bisphenols, organic peroxides, fluorinated siloxanes, organic UV filters and selected PBT substances.* (2014).

147. Wick, A., Jacobs, B., Kunkel, U., Heininger, P. & Ternes, T. A. Benzotriazole UV stabilizers in sediments, suspended particulate matter and fish of German rivers: New insights into occurrence, time trends and persistency. *Environ. Pollut.* **212**, 401–412 (2016).

148. Allan, I., Jenssen, M. T. S. & Braaten, H. F. V. Priority substances and emerging contaminants in selected Norwegian rivers–The River Monitoring Programme 2017. *NIVA-rapport* (2018).

149. Nakata, H. *et al.* Benzotriazole UV Stabilizers in the Environment: Is it a POPs? in *32nd SETAC North America* (2011).

150. Kim, J.-W., Ramaswamy, B. R., Chang, K.-H., Isobe, T. & Tanabe, S. Multiresidue analytical method for the determination of antimicrobials, preservatives, benzotriazole UV stabilizers, flame retardants and plasticizers in fish using ultra high performance liquid chromatography coupled with tandem mass spectrometry. *J. Chromatogr. A* **1218**, 3511–3520 (2011).

151. Luongo, G., Avagyan, R., Hongyu, R. & Östman, C. The washout effect during laundry on benzothiazole, benzotriazole, quinoline, and their derivatives in clothing textiles. *Environ. Sci. Pollut. Res.* **23**, 2537–2548 (2016).

152. Avagyan, R., Luongo, G., Thorsén, G. & Östman, C. Benzothiazole, benzotriazole, and their derivates in clothing textiles—a potential source of environmental pollutants and human exposure. *Environ. Sci. Pollut. Res.* **22**, 5842–5849 (2015).

153. Carpinteiro, I., Abuín, B., Rodríguez, I., Ramil, M. & Cela, R. Pressurized solvent extraction followed by gas chromatography tandem mass spectrometry for the determination of benzotriazole light stabilizers in indoor dust. *J. Chromatogr. A* **1217**, 3729–3735 (2010).

154. Kim, J.-W. *et al.* Analysis of Benzotriazole UV Stabilizers in House Dust Using an UHPLC-MS/MS. *Interdiscip. Stud. Environ. Chem.* **6**, (2012).

155. Zhang, D., Liu, C. & Yang, Y. Determination of UV Absorbers and Light Stabilizers in Food Packing Bags by Magnetic Solid Phase Extraction Followed by High Performance Liquid Chromatography. *Chromatographia* **79**, 45–52 (2016).

156. Gao, Y., Gu, Y. & Wei, Y. Determination of Polymer Additives–Antioxidants and Ultraviolet (UV) Absorbers by High-Performance Liquid Chromatography Coupled with UV Photodiode Array Detection in Food Simulants. *J. Agric. Food Chem.* **59**, 12982–12989 (2011).

157. Chang, L. *et al.* Simultaneous Analysis of Trace Polymer Additives in Plastic Beverage Packaging by Solvent Sublation Followed by High-Performance Liquid Chromatography. *J. Agric. Food Chem.* **61**, 7165–7171 (2013).

158. Rani, M. *et al.* Qualitative Analysis of Additives in Plastic Marine Debris and Its New Products. *Arch. Environ. Contam. Toxicol.* **69**, 352–366 (2015).

159. Becker, L., Scheringer, M., Schenker, U. & Hungerbühler, K. Assessment of the environmental persistence and long-range transport of endosulfan. *Environ. Pollut.* **159**, 1737–1743 (2011).

160. Klasmeier, Jö. *et al.* Application of Multimedia Models for Screening Assessment of Long-Range Transport Potential and Overall Persistence. *Environ. Sci. Technol.* **40**, 53–60 (2006).

161. POPRC. *Risk profile on hexabromocyclododecane*. (2010).

162. Liu, Q., Krüger, H. & Zetzsch, C. Degradation study of the aerosol-borne insecticides Dicofol and DDT in an aerosol smog chamber facility by OH radicals in relation to the POPs convention. *Geophys. Res. Abstr.* **7**, 05760 (2005).

163. Ruan, T. *et al.* Concentrations and Composition Profiles of Benzotriazole UV Stabilizers in Municipal Sewage Sludge in China. *Environ. Sci. Technol.* **46**, 2071–2079 (2012).

164. Wang, X. *et al.* Determination of six benzotriazole ultraviolet filters in water and cosmetic samples by graphene sponge-based solid-phase extraction followed by high-performance liquid chromatography. *Anal. Bioanal. Chem.* **410**, 6955–6962 (2018).

165. Apel, C., Tang, J. & Ebinghaus, R. Environmental occurrence and distribution of organic UV stabilizers and UV filters in the sediment of Chinese Bohai and Yellow Seas. *Environ. Pollut.* **235**, 85–94 (2018).

166. Song, S., Ruan, T., Wang, T., Liu, R. & Jiang, G. Occurrence and removal of benzotriazole ultraviolet stabilizers in a wastewater treatment plant in China. *Environ. Sci. Process. Impacts* **16**, 1076–1082 (2014).

167. Liu, R. *et al.* Determination of nine benzotriazole UV stabilizers in environmental water samples by automated on-line solid phase extraction coupled with high-performance liquid chromatography–tandem mass spectrometry. *Talanta* **120**, 158–166 (2014).

168. Zhao, X. *et al.* Occurrence and fate of benzotriazoles UV filters in a typical residential wastewater treatment plant in Harbin, China. *Environ. Pollut.* **227**, 215–222 (2017).

169. Zhang, Z. *et al.* Determination of Benzotriazole and Benzophenone UV Filters in Sediment and Sewage Sludge. *Environ. Sci. Technol.* **45**, 3909–3916 (2011).

170. Vimalkumar, K. *et al.* Occurrence of triclocarban and benzotriazole ultraviolet stabilizers in water, sediment, and fish from Indian rivers. *Sci. Total Environ.* **625**, 1351–1360 (2018).

171. Kameda, Y., Kimura, K. & Miyazaki, M. Occurrence and profiles of organic sun-blocking agents in surface waters and sediments in Japanese rivers and lakes. *Environ. Pollut.* **159**, 1570–1576 (2011).

172. Rodríguez-Pereiro, I. & Casado-Agrelo, J. *Benzotriazole UV Stabilizers in Soil and Suspended Particulate Matter Samples*. (2012).

173. Schlabach, M. *et al.* Screening program 2018. Volatiles, Gd, BADGE, UV filters, Additives, and Medicines. NILU rapport, 20/2019. *NILU Rapp.* (2019).

174. Montesdeoca-Esponda, S., Sosa-Ferrera, Z., Kabir, A., Furton, K. G. & Santana-Rodríguez, J. J. Fabric phase sorptive extraction followed by UHPLC-MS/MS for the analysis of benzotriazole UV stabilizers in sewage samples. *Anal. Bioanal. Chem.* **407**, 8137–8150 (2015).

175. Montesdeoca-Esponda, S., Sosa-Ferrera, Z. & Santana-Rodríguez, J. J. On-line solid-phase extraction coupled to ultra-performance liquid chromatography with tandem mass spectrometry detection for the determination of benzotriazole UV stabilizers in coastal marine and wastewater samples. *Anal. Bioanal. Chem.* **403**, 867–876 (2012).

176. Montesdeoca-Esponda, S., Sosa-Ferrera, Z. & Santana-Rodríguez, J. J. Microwave-assisted extraction combined with on-line solid phase extraction followed by ultra-high-performance liquid chromatography with tandem mass spectrometric determination of benzotriazole UV stabilizers in marine sediments and sewage sludges. *J. Sep. Sci.* **36**, 781–788 (2012).

177. Casado, J., Rodríguez, I., Carpinteiro, I., Ramil, M. & Cela, R. Gas chromatography quadrupole time-of-flight mass spectrometry determination of benzotriazole ultraviolet stabilizers in sludge samples. *J. Chromatogr. A* **1293**, 126–132 (2013).

178. Carpinteiro, I., Abuín, B., Ramil, M., Rodríguez, I. & Cela, R. Matrix solid-phase dispersion followed by gas chromatography tandem mass spectrometry for the determination of benzotriazole UV absorbers in sediments. *Anal. Bioanal. Chem.* **402**, 519–527 (2012).

179. Carpinteiro, I., Abuín, B., Rodríguez, I., Cela, R. & Ramil, M. Headspace solid-phase microextraction followed by gas chromatography tandem mass spectrometry for the sensitive determination of benzotriazole UV stabilizers in water samples. *Anal. Bioanal. Chem.* **397**, 829–839 (2010).

180. Maceira, A., Borrull, F. & Marcé, R. M. Occurrence of plastic additives in outdoor air particulate matters from two industrial parks of Tarragona, Spain: Human inhalation intake risk assessment. *J. Hazard. Mater.* (2019) doi:10.1016/j.jhazmat.2019.04.014.

181. Apel, C., Joerss, H. & Ebinghaus, R. Environmental occurrence and hazard of organic UV stabilizers and UV filters in the sediment of European North and Baltic Seas. *Chemosphere* **212**, 254–261 (2018).

182. De Silva, A. & Muir, D. Benzotriazole UV Stabilizers and Substituted Diphenylamine Antioxidants: Emerging Organic Pollutants in San Francisco Bay. in *ECWG Meeting, Spring* (2015).

183. Allinson, M., Kameda, Y., Kimura, K. & Allinson, G. Occurrence and assessment of the risk of ultraviolet

filters and light stabilizers in Victorian estuaries. *Environ. Sci. Pollut. Res.* **25**, 12022–12033 (2018).

184. Mansouri, K., Grulke, C. M., Judson, R. S. & Williams, A. J. OPERA models for predicting physicochemical properties and environmental fate endpoints. *J. Cheminform.* **10**, 10 (2018).

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: CO_2 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

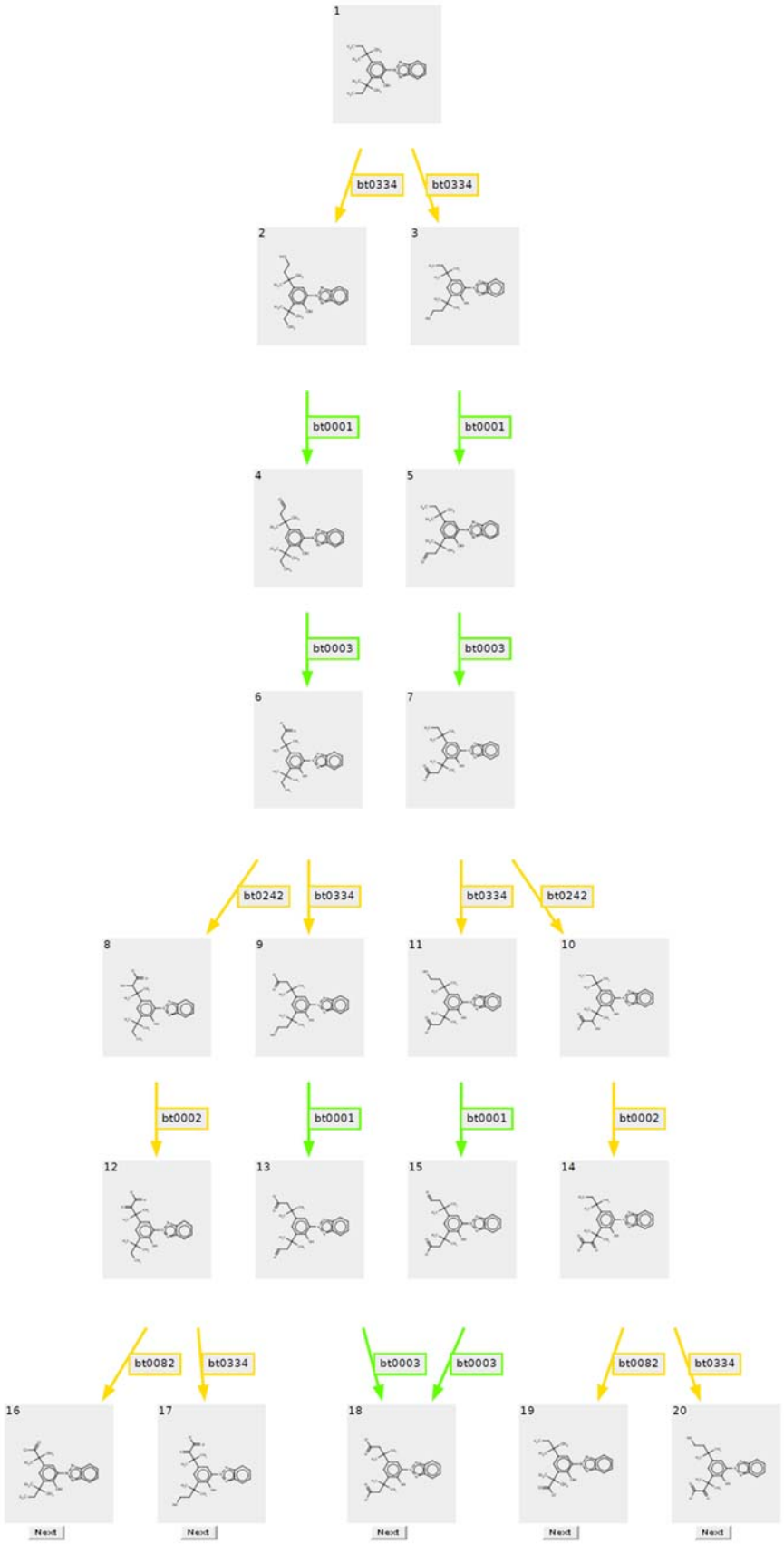


Figure 3. EAWAG-BBD prediction for UV-328 using the SMILES code present on Table 2[16].

6.3. Nakata *et al.* Bioaccumulation Studies

6.3.1 Nakata *et al.*, 2010 Study[30]

Table 14. Concentrations of BZT UV absorbers (ng/g ww) in the blubbers of finless porpoises (FP) collected from the Ariake Sea, Japan[30].

	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 lw[30].

	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 samples[30].

	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 Study[27]

Table 18. Concentrations of BZT UV absorbers (ng/g ww) in tidal flat and shallow water organisms collected from the Ariake Sea, Japan[27].

	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 lw[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 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 samples[27].

	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 Suite[22]: [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
CCl_4[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}
CCl_4	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

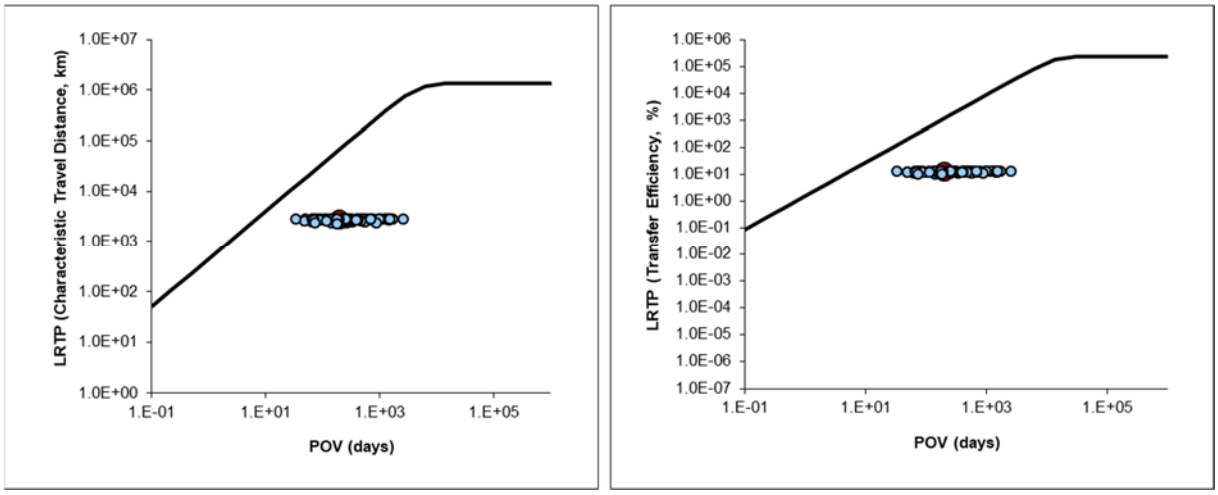


Figure 4. Monte Carlo analysis of the OECD P_{OV} and LRTP Tool results for UV-328 with the same input values as in Table 22. The dispersion factor for each Tool input except the molecular weight is 10.

Alternatively to the input data for UV-328 estimated with EPI Suite presented in Table 22 above, the logK_{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 logK_{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 logK_{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.

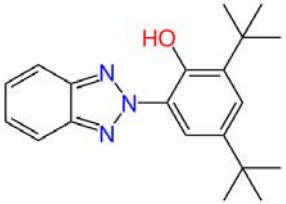
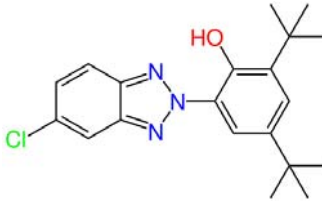
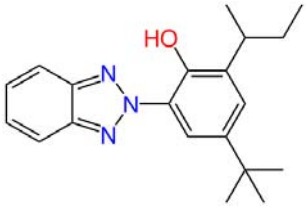
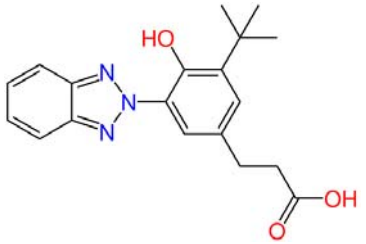
Table 24. Impact of using different logK_{AW} values as input (other input data unchanged).

Input vales for logK_{AW}		OECD Tool generated values		
Method	**logK_{AW}**	**P_{OV} (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 3.435×10^{-12} cm^3/(molecule·s). In contrast, Liu, Krüger and Zetzsch (2005)[162] found a measured value of 5×10^{-13} cm^3/(molecule·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 2.5×10^3 km and 9.2%, respectively, even at a high logK_{AW} (logK_{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*K*OW	6.3	6.9	6.3	3.3
[d] log*K*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=CC3=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 database[113].

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 (µg/L)	*n* = 1			ND
		PCPs (µg/L)	*n* = 5, 80% DF		771	226††, ND
	Beijing[18]	WWTP biosolids (ng/g)		108		< limit of detection (LOD)
	Beijing[156]	food packaging (µg/g)	*n* = 27, 4% DF		6.0	
	Beijing[157]	beverage packaging (µg/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 (µg/g)	*n* = 1		24.8	
		snack packing (µg/g)	*n* = 1		30.5	
	Pearl River Estuary[141]	marine wildlife muscles (ng/g lw): pelagic-neritic, bentho-pelagic 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)	*n* = 27, 93% DF		17.9	0.4††, < LOQ
	Pearl River Estuary[142]	farmed red snapper carcasses (ng/g dw)	*n* = 2, 50% DF		0.8	< LOQ
		wild fishes species carcasses (ng/g dw)	*n* = 11			ND
	Shandong Province[167]	WWTP influent (ng/L)	*n* = 4, 50% DF		2.9	2.6††, ND
		WWTP effluent (ng/L)	*n* = 4, 50% DF		0.6	ND
		river (ng/L)	*n* = 4			ND
	Songhua River, Northeast (mainly residential)[168]	WWTP influent (ng/L)	*n* = 81, 94% DF	9.6	29	0.3
		WWTP A/O sludge (ng/g dw)	*n* = 6, 100% DF	115	163	93.3
		WWTP dewatered sludge (ng/g dw)	*n* = 5, 100% DF	89.3	121	39.6
	Songhua River, Northeast[169]	sediment (ng/g dw)	*n* = 6, 100% DF	3.8	7.1	2.1
		WWTP sludge (ng/g dw)	*n* = 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)	*n* = 59, 30–38% DF		5.2	ND
		sediment, river (ng/g)	*n* = 58, 80–88% DF		4.3	ND
		fish muscle, river (ng/g)	*n* = 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)	*n* = 28, 89% DF		55	0.2††, < 0.15
		sediments (ng/g dw)	*n* = 16, 100% DF		320	2.6
	Ariake Sea[30]	blubber of finless porpoises (ng/g ww)	*n* = 5, 100% DF	29	64	11
	Saitama Prefecture[171]	water from streams (ng/L)	*n* = 2, 50% DF	70		
		WWTP effluents (ng/L)	*n* = 4, 75% DF	62	88	47

Location		Matrix	Others	Average	Max	Min
		water from heavily polluted rivers (ng/L)	*n* = 6, 67% DF	701	4.8×10^3	149
		water from moderately polluted rivers (ng/L)	*n* = 12, 67% DF	152	583	30
		water from background sites (ng/L)	*n* = 5			ND
		sediment from streams (µg/kg)	*n* = 2, 100% DF	102	1.2×10^3	10
		sediment from WWTP effluents (µg/kg)	*n* = 4, 75% DF	13	85	10
		sediment from heavily polluted rivers (µg/kg)	*n* = 6, 100% DF	117	1.7×10^3	21
		sediment from moderately polluted rivers (µg/kg)	*n* = 12, 75% DF	59	213	10
		sediment from background sites (µg/kg)	*n* = 5, 60% DF	58	89	29
	Not described[127]	road dust (ng/g dw)	*n* = 9, 100% DF		40	2.0
		marine mammal blubber (ng/g ww)	*n* = 29, 66% DF		70	
		sediment cores (ng/g dw)	*n* = 2, 100% DF		10	4.0
	Okinawa Island: seawater of beaches and coral reefs[47]	seawater at beach sites (ng/L)	*n* = 23, 61% DF		287	2.8††, ND
		seawater at river and coral reef sites (ng/L)	*n* = 15, 60% DF		263	5.7††, ND
	Mukojima Island[99]	black-footed albatross (µg/g-plastic, PP fragment)	*n* = 194 (plastic fragments), 1% DF		1.4	
	Awashima Island[100,124]	streaked shearwater chicks from semi-field conditions (µg/g-lipid)	*n* = 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, *n* = 17, 82% DF	50	304	ND
		municipal dump, house dust (ng/g)	*n* = 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% DF	34.2	563	1.5[††], ND
	Manila Bay[150]	fish muscle (ng/g lw)	*n* = 5, 100% DF		179	18.4
Republic of Korea	Geoje Island[86]	new plastic (ng/g)	*n* = 27, 100% DF		770	2.7
		marine plastic debris (ng/g)	*n* = 29, 97% DF		1.6×10^3	1.5[††], ND
	Geoje Island[158]	marine plastic debris	qualitative analysis, *n* = 19, 16% DF			
		new plastic	qualitative analysis, *n* = 25, 31% DF			
	Residential (Seoul, Pyeongchon), industrial (Ansan), rural (Jeju)[139]	human breast milk (ng/g lw)	39.7 median, *n* = 208, 98% 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% 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% 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% 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% 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)	*n* = 4, 25% DF		26	ND
	Rivers Rhine, Saale, Saar, Elbe, and Moselle[147]	river sediments (ng/g)	4.6 median, *n* = 8, 100% DF		10	2.0††, ND
		suspended particulate matter (ng/g)	*n* = 5, 100% DF		15	5.0††, ND
		bream liver (ng/g)	*n* = 4, 100% DF		40	1.0††, ND
Denmark	Faroe Islands[99]	northern fulmar (µg/g-plastic, PP fragment)	*n* = 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)	*n* = 5			< 0.5
		common eider eggs (ng/g ww)	*n* = 5, 100% DF		0.3	0.1
		European shag eggs (ng/g ww)	*n* = 5, 60% DF		0.2	< 0.2
		kittiwake eggs (ng/g ww)	*n* = 5, 100% DF		0.3	0.1
		glaucous gull eggs (ng/g ww)	*n* = 5, 60% DF		0.3	< 0.1
		polar bear blood plasma (ng/g ww)	*n* = 10			< 0.3
		mink liver (ng/g ww)	*n* = 5, 100% DF		0.4	0.1
		common gull eggs (ng/g ww)	*n* = 5, 60% DF		0.2	< 0.2
		WWTP effluent (ng/L)	*n* = 6, 100% DF		57	7.0
	River Alna[148]	water (ng/L)	*n* = 2, 100% DF		1.9	1.0
		suspended particulate matter (ng/g dw)	*n* = 2, 100% DF		53	39
		benthic macroinvertebrate (ng/g ww)	*n* = 2			< 1.0
		moss and periphyton (ng/g ww)	*n* = 3, 100% DF		17.7	7.4
		brown trout whole body (ng/g ww)	*n* = 2, 100% DF		0.7	
		brown trout muscle/liver (ng/g ww)	*n* = 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), *n* = 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)	*n* = 15, 7% DF		81	< 5.0
		WWTP sludge (ng/g dw)	*n* = 10			< 5.0
		water leachate (ng/L)	*n* = 6			< 5.0
		particulate leachate (ng/g dw)	*n* = 6			< 5.0
		lake sediment (ng/g dw)	*n* = 10, 50% DF		25	3.0††, < 25
	Oslo area[173]	sewage water (ng/L)	*n* = 7, 100% DF		68	22.0
		surface water (ng/L)	*n* = 9, 100% DF		17	0.8
		sediment (ng/g dw)	*n* = 5, 60% DF		21	1.7††, < 2.5
		common mussel (ng/g ww)	*n* = 5, 20% DF		0.7	< 0.6
		gull egg (ng/g ww)	*n* = 10, 100% DF		60	0.4
		settled floor dust (ng/g)	*n* = 26, 100% DF		1.8×10^3	0.9
		indoor air (ng/m^3)	*n* = 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)	*n* = 12			< LOD
		WWTP effluent (ng/L)	*n* = 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)	*n* = 4, 75% DF		24	20.7††, < LOQ
		WWTP sludge (ng/g)	*n* = 3, 67% DF		12.2	0.9††, < LOD

Location		Matrix	Others	Average	Max	Min
		beach seawater (ng/L)	*n* = 3			< LOD
	Northwest[177]	WWTP sludge (ng/g)	*n* = 8, 88% DF		152	20††, ND
		sediment (ng/g)	*n* = 1, 100% DF		20	
	Not described[153]	indoor dust (ng/g)	*n* = 10, 100% DF	91.0	149	46
	Not described[178]	river and marine sediment (ng/g)	*n* = 6, 100% DF		56	7.9
	Not described[179]	WWTP influent (ng/L)	*n* = 5, 80% DF		19	1.0††, ND
	Tarragona, industrial parks[180]	Constantí: particulate phase of outdoor air (pg/m^3)	*n* = 10, 70% DF	20	43	ND
		Tarragona harbour: particulate phase of outdoor air (pg/m^3)	*n* = 10, 100% DF	14	21	6.5
Sweden	Background (Gårdsjön and Sandsjön) and urban sites (Stockholm and Borås)[43]	storm water (ng/L)	*n* = 6, 75% DF		1.3	0.2††, < 0.1
		surface water (ng/L)	*n* = 6, 100% DF		4.1	1.7
		air (ng/m^3)	*n* = 8			< 0.02
		air deposition (ng/m^2 day)	*n* = 4			< 70
		sediment (µg/kg dw)	*n* = 6, 67% DF		1.3	< 0.7
		fish whole body (µg/kg dw)	*n* = 4			< 0.3
		landfill effluent particles (µg/kg dw)	*n* = 1, 100% DF		3.1	
		landfill effluent (ng/L)	*n* = 3, 100% DF		91	7.0
		WWTP effluent particles (µg/kg dw)	*n* = 1			< 110
		WWTP effluent (ng/L)	*n* = 5, 100% DF		15	6.8
		WWTP sludge (µg/kg dw)	*n* = 8, 50% DF		37	2.4††, ND
		soil (µg/kg dw)	*n* = 4, 25% DF		0.74	2.4††, < 0.4
	Retailers with garments made worldwide[151]	clothing textile samples (ng/g)	*n* = 27, 15% DF		85.3	47.8††, ND
	Stockholm, retailers available in up to 88 countries worldwide[152]	garments (ng/g)	*n* = 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)	*n* = 48, 31–50% DF	0.1	0.9	< MDL
		coastline sediments (ng/g dw)	*n* = 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)	*n* = 3, 100% DF		76	53
		WWTP effluent (ng/L)	*n* = 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)	*n* = 6 (eggs), *n* = 5 (liver)			< 450 (eggs), < 990 (liver)
		northern fulmars (pg/g ww): Prince Leopold Island (eggs, liver), Labrador Sea (liver)	*n* = 5 (eggs), *n* = 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	*n* = 14			< 900
	Not described[133]	WWTP influent (ng/L)	45.1 median, *n* = 34, 97% DF	34.4	126	< LOQ
		WWTP effluent (ng/L)	3.6 median, *n* = 34, 79% DF	2.6	63.1	< LOQ
		WWTP biosolids (ng/g dw)	239 median, *n* = 39, 92% DF	140	824	< LOQ
	Not described[137]	WWTP influent (ng/L)	*n* = 9, 100% DF		107	8.3
		WWTP effluent (ng/L)	*n* = 9, 100% DF		4.0	0.5
		WWTP biosolids (ng/g dw)	*n* = 12, 100% DF		278	39
		surface water (ng/L)	*n* = 32, 37.5% DF		1.5	0.05
		sediment (ng/g dw)	*n* = 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 (µg/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 (µg/g dw)			180	
		WWTP influent (mg/tank/99days)			276	34.4
	Narragansett Bay, Rhode Island[131]	river sediment cores (µg/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) (µg/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)	*n* = 10			< LOQ
		white sucker whole body (pg/g ww)	*n* = 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)	*n* = 4, 100% DF		216	48.4
		sediment (µg/kg dw)	*n* = 4, 75% DF		18.1	15.5