Dechlorane Plus

Revised draft risk profile

9 April 2021

Amendments to the text are highlighted in yellow.

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Executive summary

1. At its fifteenth meeting, the Persistent Organic Pollutants Review Committee concluded that Dechlorane Plus (DP, CAS No. 13560-89-9) and its *syn*-isomer (CAS No. 135821-03-3) and *anti*-isomer (CAS No. 135821-74-8) fulfilled the screening criteria in Annex D of the Convention (Decision POPRC-15/2), and decided to establish an intersessional working group to review the proposal further and to prepare a draft risk profile in accordance with Annex E to the Convention. Parties and observers were invited to submit information specified in Annex E, before 2 December 2019. This risk profile presents and considers data on Dechlorane plus and it *syn*- and *anti*-isomers, but for reasons of simplicity denoted as "Dechlorane Plus".

2. The “Dechlorane Plus”™ technical mixture is a commercially available polychlorinated flame retardant that has been in use since the 1960s. It is used as an additive flame retardant in electrical wire and cable coatings, plastic roofing materials, connectors in TV and computer monitors, and as a non-plasticizing flame retardant in polymeric systems, such as nylon and polypropylene plastic.

3. The technical DP mixture contains two stereoisomers, *syn*-DP and *anti*-DP, that are present in ratios of about 1:3 or 25 % *syn*-DP and 75 % *anti*-DP. Commercially available DP mixtures may also contain DP monoadducts, mono-dechlorinated DP and other substances as impurities. DP and its isomers are not known to be unintentionally produced.

4. DP production by Hooker Chemicals and Plastics Corporation (now the Occidental Chemical Company (OxyChem)) in Niagara Falls, New York started in the 1960s and ceased in 2016. DP has been considered as a high production volume chemical in the United States with an estimated annual production of 450-4500 tonnes from 1986. Production at the Anpon Electrochemical Co. manufacturing plant in China started in 2003 and the annual production is reported to be 300–1000 tonnes per year.

5. DP is released to the environment during production, processing and use, as well as from waste disposal and recycling activities. Releases from use include industrial and professional uses as well as releases from consumer products. DP entering the enviroment is expected to predominantly reside in soil and/or sediment, depending on the compartment of release, but DP remaining in air and in water has the potential for dispersion and may undergo environmental transport by particle binding. DP has been detected globally in many locations, ranging from production and recycling sites to urban, rural and remote areas. It has been detected in air, water, sediment, ice, soil, wastewater, sludge, biosolids, landfill leachate, indoor and outdoor dust, wildlife and humans.

6. On a global scale, the high DP concentrations were detected close to known production sites or electronic waste (e-waste) treatment facilities. Monitoring studies document the long-range transport of DP to remote regions via the atmosphere, ocean currents and possibly also via migratory birds. Modelling studies suggests DP has transport and persistence properties similar to listed POPs. DP has been detected in different environmental matrices and biota in the Arctic, Antarctic, the Tibetan Plateau, a mountain region in Asia and in the remote Xilingol Prairie in Inner Mongolia, China. The atmospheric half-life of DP is in one study calculated to ~14 hours, i.e. below the criterion of two days set in Annex D (d) (iii) of the Convention. Another study report estimated half-lives in air of 0.468 and 160.12 days for atmospheric oxidation and ozone reaction, respectively, using the AOPWIN 2010 model (12-hr day). However, modelled half-lives in air are largely based on gas phase reactions and do not consider possibly longer half-lives following sorption to particles, which is presumed to be the primary mode of transport for DP due to its low vapour pressure and a high log octanol-air partitioning coefficient. Available monitoring data confirms this assumption and shows that DP is found predominantly in the particulate phase both in air and water. Long-range transport of DP is thus likely mediated by particle binding. Sorption to particles may slow down reaction rates, increase the actual half-lives in air and facilitate the long-range transport of DP.

7. DP is expected to be persistent in water, soil and sediment, and is also considered to be persistent in air under natural conditions as a result of its binding to air particles and limited photodegradation. It has very low water solubility and a high octanol-water partition coefficient. It is expected to bind to organic matter in soil and sediments and is therefore probably not easily bioavailable for microorganisms. Furthermore, DP has no functional groups that are susceptible for hydrolysis and is not expected to hydrolyse. Modelling predicts that aerobic biodegradation of DP would be very slow and similar to analogue chemicals (mirex, chlordane, heptachlor, dieldrin/endrin, aldrin and endosulfan) that are already listed under the Stockholm Convention. DP is detected in deep soil and sediment cores, indicating its persistence. Monitoring data show limited degradation in soil and sediments over time and support the conclusion that DP is persistent. This is confirmed in a recent laboratory study which shows that half-life of DP in soil greater than 180 days.

8. Bioconcentration factors above 5000 have recently been reported for the DP isomers in a fish study. Depuration half-lives corresponding to a bioconcentration factor above 5000 L/kg have also been reported for DP, as well as biomagnification and trophic magnification factors above 1 in a variety of food chains and locations, including an Antarctic food chain. DP has been identified as a very persistent and very bioaccumulative substance (vPvB) in the European Union.

9. A Canadian screening assessment concludes that there is risk of harm to the environment from DP, and that DP meets the critiera under paragraph 64(a) of Canadian Environmental Protection Act, 199 (CEPA) as it is entering or

may enter the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity.

10. Exposure of the general population to DP takes place by consumption of food and drinking water, inhalation of indoor and ambient air, as well as respiratory and oral uptake of dust containing DP. DP has been detected in human blood and breast milk in many regions of the word. Moreover, the human foetus may be exposed to DP via the umbilical cord blood, and breast milk may be an important source of exposure for infants. The highest DP levels have been observed in occupationally exposed workers and residents living near production facilities and e-waste recycling sites in China.

11. Available animal studies suggest low concern for acute toxicity via oral, inhalative or dermal routes of exposure in humans and other organisms, but the toxicity data on DP is limited and chronic toxicity studies are not available. Toxicity studies with DP report effects such as oxidative damage, indications for neurodevelopmental toxicity and potential for endocrine disruption. Oxidative stress has been observed in marine macroalgae, fish, marine bivalves, earthworm, birds, and mice. In a marine macroalgae exposed to DP, photosynthesis was reduced at low concentrations. In carp embryo, DP increased mortality rate and morphological deformities, reduced hatching rate, delayed hatching time and decreased body length. In addition, neurodevelopmental toxicity was observed in fish embryos/larvae. DP has been shown to activate peroxisome proliferator activator receptor-gamma and inhibit insulin signalling in adipocytes, furthermore, at low doses, DP promoted adipose tissue dysfunction and glucose intolerance in mice. A few studies indicate potential effects to the sex- and thyroid hormone pathway in humans. DP has also been reported to cross the blood-brain barrier and to be maternally transferred to offspring in several species. A study in zebrafish embryos indicate synergistic effects on neurotoxicity parameters and elevated bioaccumulation of both compounds as a synergistic effect of co-exposure to DP and the 3-methyl phenanthrene.

12. DP is widely detected in the global environment, including in remote regions. It is transported to locations far from production sites and places of use. Available scientific data show that DP is persistent and bioaccumulative. Although data are limited and chronic toxicity studies are lacking, there are studies showing that DP may have adverse effects on the environment and that it can be potentially toxic to mammals and humans. Based on evidence for persistence, bioaccumulation and adverse effects of DP observed in some organisms and its widespread occurrence in the global environment including at remote regions, [it is concluded that DP is likely, as a result of its long-range environmental transport, to lead to significant adverse human health and/or environmental effects such that global action is warranted].

1. Introduction

13. In May 2019, Norway submitted a proposal to include the chlorinated flame retardant (FR) Dechlorane Plus (CAS No. 13560-89-9) and its *syn*- (CAS No. 135821-03-3) and *anti*- (CAS No. 135821-74-8) isomers in Annexes A, B and/or C to the Stockholm Convention on Persistent Organic Pollutants (POPs). The proposal (UNEP/POPS/POPRC.15/3) was submitted in accordance with Article 8 of the Convention and reviewed by the Persistent Organic Pollutants Review Committee (POPRC) at its fifteenth meeting in October 2019.

14. The commercial mixture "Dechlorane Plus"™ is an additive FR that has been in use since the 1960s (Shen et al., 2011). For Dechlorane Plus the acronyms DP or DDC-CO are commonly used. In this risk profile, DP is used.

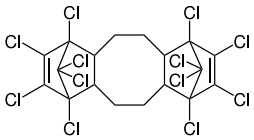
1.1 Chemical Identity

15. The technical mixture Dechlorane Plus™ (CAS No. 13560-89-9) is a commercially available formulation that contains two stereoisomers, *syn*-DP (CAS No. 135821-03-3) and *anti*-DP (CAS No. 135821-74-8) in a ratio of about 1:3 or 25 % *syn*-DP and 75 % *anti*-DP (Sverko et al., 2011). According to the North American manufacturer OxyChem, their commercial product contains approximately 35% *syn*-DP and 65% *anti*-DP (OxyChem, 2013). The fraction of *syn*-DP (f_{syn}) values for commercial DP from the Anpon Electrochemical Co and OxyChem manufacturers are in the range 0.20-0.41 (Wang et al., 2010a).

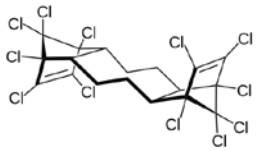
16. The structural formula of DP and its two isomers is shown in Figure 1 below. The chemical identity, and the modelled and experimental physico-chemical properties of DP and its two isomers are listed in Tables 1 and 2, below.

Figure 1. Structural formula of DP and its two isomers

Dechlorane Plus
(CAS No. 13560-89-9)



Anti- (or *exo*) Dechlorane Plus
(CAS No. 1358 21-74-8)



Syn- (or *endo*) Dechlorane Plus
(CAS No. 135821-03-3)

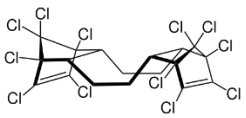


Table 1. The chemical identity of Dechlorane Plus, with its *syn*- and *anti*-isomers.

CAS number:	13560-89-9 (Dechlorane Plus™), 135821-03-3 (*syn*-DP), 135821-74-8 (*anti*-DP)
IUPAC name:	1,6,7,8,9,14,15,16,17,17,18,18-dodecachloropentacyclo-[12.2.1.1^{6,9}.0^{2,13}.0^{5,10}] octadeca-7,15-diene (Dechlorane Plus™), (1S,2S,5R,6R,9S,10S,13R,14R)-1,6,7,8,9,14,15,16,17,17,18,18-dodecachloropentacyclo[12.2.1.1^{6,9}.0^{2,13}.0^{5,10}]octadeca-7,15-diene (*syn*-DP), (1R,2R,5R,6R,9S,10S,13S,14S)-1,6,7,8,9,14,15,16,17,17,18,18-dodecachloropentacyclo[12.2.1.1^{6,9}.0^{2,13}.0^{5,10}]octadeca-7,15-diene (*anti*-DP)
EC number:	236-948-9
EC name:	1,6,7,8,9,14,15,16,17,17,18,18-Dodecachloropentacyclo[12.2.1.16,9.02,13.05,10] octadeca-7,15-diene
Molecular formula:	$C_{18}H_{12}Cl_{12}$
Molecular weight:	653.73 g/mol
Synonyms:	Bis(hexachlorocyclopentadieno)cyclooctane; 1,2,3,4,7,8,9,10,13,13,14,14-Dodecachloro-1,4,4a,5,6,6a,7,10,10a,11,12,12a-dodecahydro-1,4:7,10-dimethanodibenzo[*a*,*e*]cyclooctene; 1,4:7,10-Dimethanodibenzo[a,e]cyclooctene 1,2,3,4,7,8,9,10,13,13,14,14-dodecachloro-1,4,4a,5,6,6a,7,10,10a,11,12,12a-dodecahydro-; Dodecachlorodimethanodibenzocyclooctane; Dodecachlorododecahydrodimethanodibenzocyclooctane; Dodecachlorododecahydrodimethanodibenzocyclooctene; DDCDiMeDiBzcOb
Trade names	Dechlorane Plus; Dechlorane Plus 25 (Dech Plus); Dechlorane Plus 35 (Dech Plus-2); Dechlorane Plus 515; Dechlorane 605; Dechlorane Plus 1000; Dechlorane Plus 2520; Dechlorane A; DP; Escapeflam DK-15 (China); PyroVex SG (grade 515, 25 and 35)

Table 2. Overview of selected physicochemical properties of Dechlorane Plus, with its *syn*- and *anti*-isomers

Property	Value	Reference
Physical state at 20 °C and at 101.3 kPa	Solid white powder	ECHA, 2017d
Melting/freezing point	340-382 °C 350 °C	ECHA, 2017d OxyChem, 2004b
Vapour pressure	0.006 mm Hg ≙ 0.8 Pa (at 200 °C)	OxyChem, 2004b
Water solubility*	<1.67 ng/L (20 – 25 °C) 0.044 – 249 µg/L (insoluble)	ECHA, 2017d OxyChem, 2004b
n-Octanol/water partition coefficient, K_{ow} (log value)	9.3	OxyChem, 2004b
n-Octanol-air partition coefficient K_{oa} (log value)	12.26	OxyChem, 2004b
Sediment/water partition coefficient K_p (log value)	6.65	OxyChem, 2004b
Air-water partition coefficient K_{aw} (log value)	The following log K_{AW} values are estimated at 25 °C -3.2 (from measured water solubility and estimated vapour pressure) 0.44 (from measured water solubility and vapour pressure) -2.8 (from EPIWIN **predicted water solubility using log K_{OW} of 9 and vapour pressure) -3.5 (from HENRYWIN***v.3.20, predicted from structure using Bond Method).	ECHA, 2017d

*There is some uncertainty in the precise value for water solubility (Chou et al. (1979a). However, all available measurements and predictions agree that the substance is very poorly water soluble.

The following modelling programs are individual models in EPI Suite™:
** Estimation Program Interface Suite for Windows
*** Calculates the Henry's Law constant (air/water partition coefficient) using both the group contribution and the bond contribution methods. Since the group contribution method is not applicable to DP due to fragments missing in the database, only the result of the bond contribution method is shown.

17. DP is produced by the Diels-Alder reaction of two moles of hexachlorocyclopentadiene with one mole of 1,5-cyclooctadiene. DP monoadduct (DPMA) may be formed through partial reactions during the synthesis of DP through the diadduct Diels–Alder process and can therefore be present as an impurity in commercial DP (Sverko et al., 2010). DP compounds with lower number of chlorine atoms such as mono-dechlorinated DP (DP-1Cl or DP-Cl_{11}) may also occur as impurities in the commercial substance (Li et al. 2013b).

18. Substances already listed in the Stockholm Convention such as aldrin and chlordecone contain a single chlorinated norbonene ring. The chlorinated norbonene ring is also present in the structure of DP. DP contains two chlorinated norbonene rings attached to a cyclo-octane ring. Consequently, it is a much larger molecule than these listed POPs. Physico-chemical properties of structures containing the chlorinated norbonene ring listed in Stockholm Convention, dechlorane-related substances and dechlorinated DP are described in Table 1 and 2 in the UNEP/POPS/POPRC.17/INF/xx and it can be seen that water solubility and log Kow values of DP are significantly different to these Listed chemicals. Quantitative structure-activity relationship (QSAR) modelling (Sverko et al., 2011; Feo et al., 2012) and testing done by the manufacturer (EHSI, 2004) indicate that DP has characteristics typical of POPs (EHSI, 2004; Sverko et al., 2011; Feo et al., 2012).

1.2 Conclusion of the POPs Review Committee regarding Annex D information

19. At its fifteenth meeting, the POPs Review Committee evaluated the proposal by Norway to list DP (CAS No. 13560-89-9) and its *syn*-isomer (CAS No. 135821-03-3) and *anti*-isomer (CAS No. 135821-74-8) under the Stockholm Convention on Persistent Organic Pollutants and concluded that DP and its *syn*-isomer and *anti*-isomer met the screening criteria specified in Annex D (UNEP/POPS/POPRC.15/7, Decision POPRC-15/4). The Committee decided to review the proposal further and to prepare a draft risk profile in accordance with Annex E to the Convention.

1.3 Data sources

20. The draft risk profile is based on the following data sources:

(a) The nomination report submitted by Norway (UNEP/POPS/POPRC.15/3);

(b) Annex E information and comments by Parties and Observers received in response to the invitation for comments on the draft risk profile in 2020 (Annex E, 2019; Annex E, 2020; Annex E 2021): Australia, Belarus, Canada, Colombia, Costa Rica, Egypt, Germany, Hungary, Monaco, Mexico, Netherlands, New-Zealand, Romania, Republic of Korea, Thailand, The Russian Federation State of Palestine, Qatar, International Pollutants Elimination Network (IPEN) and Alaska Community Action on Toxics (ACAT), the Society of Indian Automobile Manufacturers (SIAM) and the Aerospace Industries Association of Canada (AIAC);

(c) Reports and other grey literature as well as information from peer-reviewed scientific journals;

(d) Documents supporting the identification of DP as a Substance of Very High Concern (SVHC) under the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) regulation in the European Union (EU) (ECHA, 2017b,c);

(e) Screening assessment report and risk management approach from Canada (Canada, 2019a,b).

1.4 Status of the chemical under international conventions

21. DP is currently not known to be included in any international Conventions; however regulatory processes has been initiated in several countries.

22. DP is listed on Canada's Domestic Substances List (DSL) (ECCC, 2019). A final screening assessment of DP was published by Environment and Climate Change Canada (ECCC) and Health Canada in spring 2019 (Canada, 2019a). The assessment concludes that DP meets the criteria for toxicity to the environment, as it is entering or may enter the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity. The proposed regulatory approach is to amend the *Prohibition of Certain Toxic Substances Regulations, 2012* to prohibit the manufacture, import, use, sale or offer for sale of DP and all products containing the substance (Canada, 2019b).

23. In the EU, based on an Annex XV dossier and a Risk Management Options Analysis prepared by the United Kingdom, DP (including its *syn*- and *anti*-isomers) was identified as a SVHC and added to the REACH Candidate List in January 2018 as it is considered to be very persistent and very bioaccumulative (ECHA, 2017a).

24. In Norway, DP was added to the list of priority substances in January 2019 with a national goal to phase out the use by 2020 (Norwegian Environment Agency, 2019b).

25. Certain restrictions, approval, reporting and/or notification requirements for production, imports and/or use are also in place in the United States (US), New Zealand and Thailand, while Egypt reports reports that DP is banned by Decree No. 55 year 1996 of the Minister of Trade (see Table 3 of UNEP/POPS/POPRC.17/INF/xx).

2. Summary information relevant to the risk profile

2.1 Sources

2.1.1 Production, trade, stockpiles

26. DP has been manufactured by Hooker Chemicals and Plastics Corporation (now Occidental Chemical Company (OxyChem)) in Niagara Falls, New York in the US since the 1960s (Shen et al., 2011). DP has been classified as a high production volume chemical (>500 tonnes) in the US (Sverko et al., 2011), and the annual production was estimated to be 450–5000 tonnes since 1986 (Hoh et al., 2006). According to ECHA (2020a) production at the Oxychem facility ceased in mid-2016, but this has not been formally confirmed by the US EPA (Annex E info IPEN, 2021). Manufacturing in China by Anpon Electrochemical Co. in Huai'an has been reported to be 300–1000 tonnes per year since 2003 (Wang et al., 2010a). The resulting global annual production of DP is thus 750–6000 tonnes with an average of 1980 t per year (Hansen et al. 2020). Ren et al. (2008) indicated that the global annual production was approximately 5000 tonnes. Apon Electrochemical Company Ltd. was recently aquired by ADAMA which now appaers to be the sole global manufacturer (ADAMA, 2019).

27. DP is available for sale from a number of vendors globally (PubChem, 2021, see also Annex E, Peru). In the EU, DP has been registered under the REACH Regulation by two registrants (one for each manufacturer) and is manufactured in and / or imported to the European Economic Area, at ≥ 10 to < 100 tonnes per annum (ECHA, 2021). The major registrant since 2013, Occidental Chemical Belgium BVBA, representing the manufacturer OxyChem US, submitted a notification in December 2017 of "cease of manufacture or import" under REACH and no longer produces or imports DP to the EU (ECHA, 2020b). The Dutch company ADAMA Agriculture BV has, since 2016, been the registrant for the sole importer/distributer Velsicol, who represents Anpon Electrochemical Co. The maximum volume imported in the EU was 300-400 tonnes/year. Sweden registered use of 11 tonnes in 2005 and 5 tonnes of DP in 2006 (Kaj et al., 2010). Hungary reports that DP is used by two Hungarian companies at volumes below 2 tonnes/ year (Annex E, 2021).

28. In Canada, based on surveys conducted under section 71 of CEPA for the years 2011 and 2016, between 1000 and 10 000 kg of DP, including DP in some products, was imported by a few companies (Canada, 2019a,b). Based on information gathered from the Statistical Survey of Chemicals under the Chemicals Control Act, no DP was manufactured in the Republic of Korea since 2010. No DP was imported in 2010, but use was around 40 tonnes. In 2016, 11 tonnes were imported, and 17 kg was used. DP has not been exported since 2010, when 0.9 tonnes of DP was exported. In 2018, 0.63 tonnes of DP were imported, and 0.39 tonnes of DP were used (Annex E information, Republic of Korea).

2.1.2 Uses

29. DP is currently marketed as an alternative/replacement for commercial decabromodiphenyl ether (c-decaBDE) and as a substitute for mirex used as a FR (Hoh et al., 2006). It is used as a FR in electronic wiring and cables, automobiles, plastic roofing materials, and hard plastic connectors in televisions and computer monitors, wire coatings, and furniture (Zhang et al., 2015, Annex E, 2019, 2020). It is also used extensively in the aerospace industry as a FR and in other applications (AIAC, 2020). DP is often used as an additive to various formulations and can be found in many aircraft components such as cabin interior panels, ducting, engines and other components. In a recent study from China, DP was detected in various building materials (Hou et al., 2018). Samples of wallpaper (non-woven, PVC, paper), latex paint, boards (e.g. laminated floor, fiber board, solid wood), glue, sealant, PVC line pipes and foam (such as sound absorbing foam and expanded polystyrene panels) contained DP in the range non-detect to 5.1 ng/g (Hou et al., 2018). The Society of Indian Automobile Manufacturers (SIAM), on behalf of their members, state that dechlorane plus is used as an alternative to c-decaBDE in automobile manufacturing (SIAM, 2020). In Canada, DP is reported to be used as a FR in automobile manufacturing and DP is known or expected to be used as a FR in wiring and cable jacketing electronics, roofing materials, and hard plastic connectors (Canada, 2019b). Annex E information from the Netherlands confirms use in automobiles in the powertrain, cooling, chassis and bodywork parts (Annex E, 2019).

30. DP is used in many polymeric systems as a FR. These systems are typically either thermoplastics or thermosets. Examples of thermoplastics that may contain DP include nylon (Weil and Levchik 2009; KEMI, 2019), polyester (KEMI, 2019), acrylonitrile butadiene styrene (ABS), natural rubber, polybutylene terephthalate (PBT), polypropylene, and styrene butadiene rubber (SBR) block co-polymer (OxyChem, 2007). DP may be used in thermosets such as epoxy and polyester resins, polyurethane foam, polyethylene, ethylene propylene diene monomer rubber, polyurethane rubber, silicon rubber, and neoprene (OxyChem, 2007). The amount of DP in these materials ranges from 8% in PBT to up to 40% in silicon rubber (OxyChem, 2007).

31. According to one manufacturer (OxyChem, 2007), DP is manufactured for use solely by industrial customers. However, EU data indicates that DP is commonly used in the manufacturing of both professional and consumer products. More specifically, according to REACH registration data, uses of DP in the EU/ EEA region include uses in formulations or re-packing, at industrial sites and by professional users. DP is also contained in some articles used by consumers (ECHA, 2017a; 2019b). Under REACH, DP is registered for industrial use in the following product categories; polymer preparations and compounds, semiconductors, adhesives and sealants. Registered end uses from formulation and packaging include: 1) Adhesives, sealants, 2) Polymer preparations and compounds and 3) Semiconductors. Registered end uses at industrial sites and by professional users include: 1) Formulation [mixing] of preparations and/or re-packaging (excluding alloys), 2) Manufacture of plastics products, including compounding and conversion and 3) Manufacture of computer, electronic and optical products, electrical equipment. In the service life of articles at consumer use stage use in the following article categories have been indicated in the registration dossier: 1) vehicles, 2) machinery, mechanical appliances, 3) electrical/electronic article, 4) fabrics, textiles and apparel and 5) plastic articles. In addition, use in glued articles in aircraft parts are indicated (ECHA, 2020). A minor use of DP is in the development of fireworks (ECHA, 2019), but this is not a registered use in the EU/EEA.

32. In 2011, a 220-tonne sample of representatively composed e-waste was processed in a Swiss recycling facility with the different output streams being analysed. The highest DP mass flows were in fine-grained plastics and in plastics from consumer products such as monitors and laptop casings (Taverna et al., 2017).

33. DP was detected in plastic casings from two TVs and one computer, but not in circuit boards from similar equipment collected from a recycling facility in China (Li et al., 2019c). In addition, DP was detected in recycled ABS from the Guangdong Province in China (Cao et al., 2020).

34. PyroVex® SG is an additive FR that, according to the supplier, contains > 99% DP (Velsicol Chemicals LLC, 2020). PyroVex® SG is typically mixed into plastics and other materials and can be used alone or be formulated with complementary chemistries to meet performance requirements. It is marketed for use in flame retarded wire and cable, and as a FR for electrical connectors used in computers and other electronic devices. PyroVex® SG grade 515, 25 and 35 is the same chemical mixture only with different average particle size.

35. During a public EU consultation on the ninth draft recommendation for inclusion of DP in Annex XIV of REACH, held from September to 5 December 2018, a number of companies identified applications in which they use

DP as an additive FR. Identified applications were; in the aviation and automotive industries, in polymeric compounds for cable insulation and minor use within the development of fireworks (ECHA, 2017e).

2.1.3 Releases and emissions to the environment

36. DP and its isomers are not known to be unintentionally produced. The occurrence of DP and its isomers in the environment is a result of human activities.

37. A number of activities such as production, use, recycling and waste handling, as well as landfill leachate and run-off and wastewater treatment, can lead to releases of DP to the environment (Wang et al., 2016; Wang and Kelly, 2017). DP from industrial sources is expected to be released primarily to wastewater (a pathway to surface water and the soil environment). DP may also to some extent be released from commercial- and consumer products to the atmosphere. DP is likely highly removed by adsorption to biosolids in wastewater treatment systems (CECBP 2008 as cited in Canada, 2019a).

38. DP has been detected in sludge (e.g. de la Torre et al., 2011; Barón et al., 2014a; Norwegian Environment Agency, 2018b; 2019a; Ji et al., 2018) and biosolids (Davis et al., 2012) from wastewater treatment plants (WWTPs). In addition, sludge from WWTPs contaminated with DP can be used as soil enrichment (Wu et al., 2017) or fertilizer (Dai, 2011 reviewed in Ji et al., 2018). Furthermore, *syn*- and *anti*-DP was detected in the particulate fraction of storm water in Oslo, Norway (Norwegian Environment Agency, 2018b; 2019a).

39. In samples from Canadian WWTPs investigated for chlorinated FRs the detection frequency of DP was greater than 90% in both wastewater and biosolids samples, while median removal efficiencies were between 51% and 66% for total DP (Shanmuganathan et al., 2018). In a conventional WWTP in Shanghai DP levels were detected in wastewater with a mean concentration of 0.46 ng/L (range: 0.05 to 1.40 ng/L). Sludge contained DP in the range of 1.1 to 2.0 ng/g. The fraction of *anti*-DP (f_{anti}) was consistently higher than that of *syn*-DP (f_{syn}). Average f_{syn} values ranged from 0.16 to 0.33 and was in the range of two commercial DP mixtures. The annual release of DP via sewage sludge from all WWTPs in Shanghai were estimated to be 164.8 g (Xiang et al., 2014).

40. DP is used as a FR in electrical and electronic equipment and recycling of e-waste is reported to be a source of DP release to the environment (Wang et al., 2016). Very high levels of DP in soil (3327 ng/g) were detected at a recycling site in China suggesting that e-waste recycling activities serve as an important source of DP emissions in China (Yu et al., 2010). In a Swiss study, the overall DP concentration in e-waste was 33±11 mg/kg. This translates into an annual DP mass flow of 2.3±0.9 tonnes/year in all Swiss e-waste generated in 2011 (Taverna et al., 2017).

41. Indoor dust is the sink and carrier of many pollutants including DP released from consumer products. Indoor dust is also considered to be the main source of pollutants in WWTPs receiving only domestic wastewater (Katsoyiannis and Samara, 2004; Xiang et al., 2013). DP has been detected in indoor dust in the range 0.08 to 124 000 ng/g (Newton et al., 2015; Wong et al., 2017, Cao et al., 2014, Lee et al., 2020). For additional information on releases and emissions from manufacture, WWTPs, e-waste recycling, landfills etc., see Table 4 and for indoor dust and air Table 12 in UNEP/POPS/POPRC.16/INF/14.

42. Based on production, usage and disposal data Hansen et al. (2020) prepared two global atmospheric emission scenarios with a detailed geographical distribution. The total DP emission to air was estimated to be 0.02 t/year and 3.2 t/year in a low and high emission scenario, respectively, reflecting the uncertainties in production volumes and emission factors. Furthermore, 0.41-21.8 tonnes year were predicted to be released to water, and 0-0.004 tonnes/year direct to soil. The validity of the emission estimates was tested by implementing them in the Danish Eulerian Hemispheric Model, an advanced chemistry-transport model, and by evaluating the predicted atmospheric concentrations with all available Arctic measurements from peer-reviewed studies. The high emission estimate was found to be the most probable as the predicted concentration range for the high emission scenario was found to be in line with the measured range, whereas the predicted concentrations for the low emission estimate was more than a factor of 100 lower than the measurements. The estimates reported in this study were based on a global annual production of 75-6,000 tonnes/year (average assumed to be 1,980 tonnes/year) and with two manufacturing plants (one in the United States and one in China).

2.2. Environmental fate

43. The environmental distribution and fate of DP is discussed in Canada (2019a) and ECHA (2017d). In the assessment by Canada (2019a), level III fugacity modelling using the updated EQC model (v1.0, 2012) was applied to describe the fate of DP entering the environment from industrial sources and from commercial and consumer products. The results show that DP is expected to predominantly reside in soil and/or sediment, depending on the compartment of releases.

44. The Canadian assessment further details that the very low water solubility (2.85×10^{-7} mg/L), low vapour pressure (6.57×10^{-11} Pa at 25°C) and very high partition coefficients (log Kow of 8.78, estimated log Koc of 6.65) suggest that DP released into the environment will be less likely to partition into and/or remain in air and water, moving instead to the sediments and soil. If released to air, a small fraction (less than 1%) of DP is expected to remain in air in

gas phase, with most of the substance depositing to soil and water with further partitioning to sediment. The high partition coefficients also indicate that DP released into surface water from wastewater is expected to adsorb to the organic fraction of suspended solids and sediments, with less than 4% remaining in water. However, the small mass of DP that remaining in air and in water has the potential for dispersion and some transport (e.g., particle transport). On the basis of its high log Koc, once in the sediment, DP is not expected to be mobile and may remain in this compartment with little degradation. When DP is released to soil (i.e., through biosolids application to agricultural lands), the majority of the mass fraction is expected to become adsorbed to soil (99.9%) on the basis of its high estimated log Koc and hydrophobic nature. On the basis of its low vapour pressure, DP is not expected to evaporate (volatilize) from dry soil surfaces and is therefore likely to remain in soil. In addition, low degradation is expected in soil; therefore, DP is likely to remain in this compartment, with loss processes driven by soil burial or surface runoff. The results of Level III fugacity modelling support the expectation that DP predominantly resides in soil or sediment, depending on the compartment of release.

2.2.1 Persistence

45. DP is chemically stable in various environmental compartments with minimal or no abiotic degradation (reviewed in Wang et al., 2016, ECHA, 2017d, Canada, 2019a). Due to a very low water solubility and high log K_{ow}, DP is expected to bind to organic carbon in soil and sediments (Wang et al., 2016), reducing the bioavailability of DP for microorganisms and hence the potential for biodegradation. There are no measured half-life data for degradation of DP in surface water, sediment or soil (ECHA, 2017d). However, physical-chemical properties of DP were predicted using three different models (EPISuite, SPARC and Absolv) and the estimated half-lives in water, soil and sediment were predicted to be 180, 3650 and 1621 days, respectively (Zhang et al., 2016).

46. DP does not contain any functional groups that are susceptible to hydrolysis and hydrolysis is not expected to be a relevant degradation process (Canada, 2019a; ECHA, 2017b). Data on photodegradation of DP in air (e.g. Sverko et al., 2008; Wang et al., 2011; Li et al., 2013b; Wang et al., 2013b; Tao et al., 2015),water (Chou et al., 1979b) and soil (only modelled data), are reviewed in ECHA (2017d) and Canada (2019a). Most of the experimental data are from controlled lab studies that cannot be directly related to natural conditions due e.g. to the use of solvents and differences in light intensities.

47. The studies on photo-degradation suggest that *anti*-DP might be more photodegradable in air than *syn*-DP. Furthermore, DP is considered to be relatively photo-stable in air under natural conditions although photodegradation and formation of lower chlorinated dechloranes and other degradation products are reported in some of the studies. Under natural conditions sorption of DP to airborne particles is expected, which would lower the photolysis rate and result in a longer half-life in air (Canada, 2019a). Furthermore, the study of Wang et al. (2013b) indicates that DP should be photo-stable in the environment and only undergo limited degradation undernatural light conditions at the terrestrial surface. Wang et al. (2013b), who investigated the photodegradation of DP *(anti*-DP, *syn*-DP, and commercial DP) in n-nonane by irradiation using a xenon lamp, observed rapid photodegradation by dechlorination under 200–750 nm light. During the first 5 minutes >50% was degraded, and after 30 minutes 99% had been degraded. The degradation was much slower in 280–750 nm light, and only 20% loss was observed after 4 hours. The quantum yields of dechlorination by-products at 200–280 nm (UV-C) were about 2–3 orders of magnitude higher than at 280–320 nm, and no yields were detected in the 320–750 nm range (Wang et al., 2013b). Hence, both in visible light (>400 nm) and UV-A light (320–400 nm), DP exhibited negligible degradation in n-nonane, while UV-C (200–280 nm) played a significant role in the photodegradation of DP and might provide a potentially effective approach to eliminate DP. As the majority of UV exposure in the environment is UV-A light with a wavelength in the 315-400 nm range, these findings indicate that DP should be photo-stable in the environment and only undergo limited degradation under natural light conditions at the terrestrial surface. A sequential degradation process where one to four chlorines are removed from DP and exchanged with hydrogen was observed and suggested that the main overall reaction was photoreduction, leading to photodechlorination of DP (Wang et al., 2013b). Dechlorination was also observed by Li et al. (2013b). In this study, three photolytic degradation experiments were performed by exposing solutions of *anti*-DP, *syn*-DP, and commercial DP to UV light. In addition to *anti*- and *syn*-DP-Cl_{11}, at least two unknown products were identified in all samples following exposure, and in the test substance (Li et al., 2013b).

48. According to ECHA (2017d) photolysis is unlikely to be a significant fate process in natural waters, since light is attenuated with increasing water depth and shading. Radical reactions may also be inhibited by humic substances. They conclude that available information suggests that phototransformation in water is a potential but insignificant removal process for DP. The only controlled laboratory study on photo-degradation in water is by Chou et al. (1979b) who measured the phototransformation of DP in an aqueous solution with 5 % acetonitrile using a mercury lamp emitting wavelengths >290 nm for 168 hours. In Canada (2019a) a half-life for DP of >24 years is reported for this study, while ECHA (2017d) report that the rate constant for phototransformation in water was calculated as $<6 \times 10^{-4}$/h, equivalent to a half-life of >48 days (Chou et al., 1979b). According to ECHA (2017d) the findings from this study are not directly representative of natural conditions as the test solution included a solvent and the initial concentration of DP was 1 ppm (1 mg/L), which is significantly greater than the reported solubility of DP in pure water. For

phototransformation in soil ECHA (2017d) reports that no data were reported by the registrant, and that similar to water, this is unlikely to be a significant removal pathway.

49. Generally, model predictions support experimental findings that aerobic and anaerobic biodegradation of DP is very limited and that DP is expected to be persistent in water, soil, and sediment (Canada, 2019a). Predictions using the BIOWIN v4.10 model (US EPA, 2012), Catalogic (2012) and TOPKAT (2004) all indicate that biodegradation of DP will be very slow (Canada, 2019a; ECHA, 2017d). ECHA (2017d) concludes, based on the estimated data from the BIOWIN v4.10 model and associated uncertainties, that "DP is unlikely to be biodegradable". The BIOWIN results obtained for DP are the same as a number of hexachloro-norbornene-containing analogue chemicals and indicate that DP is as persistent as analogue chemicals such as mirex, chlordane, heptachlor, dieldrin/endrin, aldrin and endosulfan that are already listed as POPs under the Stockholm Convention. Furthermore, the two possible microbial degradation pathways predicted for DP are the same as for the POP analogues. According to ECHA (2017d) it is unlikely that metabolic rates for these pathways will be more rapid for DP, which is significantly less water soluble than the already listed POPs. The validity of the BIOWIN v4.10 model is discussed in detail in ECHA (2017d). Despite some uncertainties BIOWIN v4.10 is generally considered as a valid model for DP.

50. In line with these modelling results laboratory tests discussed in Canada (2019a) have shown that DP is not likely to biodegrade under aerobic conditions; an activated sludge biodegradation test (modified MITI OECD 301C) reported 0.6% biodegradation in 2 weeks (US EPA, 2011; see also Japan J-CHECK, 2020a) and a 21-day test using wastewater biosolids found 0% biodegradation (US EPA, 2009).

51. Monitoring studies provide additional evidence that DP is persistent and that biodegradation in sediment is limited. DP was found to be persistent in suspended sediment with a half-life of approximately 17 years (Sverko et al., 2008). The study of Qiu et al. (2007) suggests that DP can still be present over thirty years after initial deposition. In this study, DP was measured in a sediment core from central Lake Ontario, Canada, and detected in a layer corresponding to around 1980. A linear trend ($r_2 = 0.739$) of increasing f_{anti} values with time was observed, from an average of 0.76 in surficial (recent) sediments to >0.90 in the deeper layers corresponding to around 1980, suggesting that the *anti*-DP could be more persistent than the *syn*- isomer in sediment. However, the variation of f_{anti} in commercial batches over this time period is not known, and the findings could also reflect the isomerisation of *syn*-DP to *anti*-DP. Similar to the findings of Qiu et al. (2007) other studies on DP and f_{anti} values in freshwater (Wang et al., 2010) and marine sediment (Fang et al., 2014), respectively report possible stereoselective depletion of *syn*-DP in freshwater sediment and an enrichment of *anti*-DP. The enrichment of *anti*-DP is explained as likely being due to preferential biodegradation of the *syn*- isomer in the sediment.

52. Some data on the degradation of DP in soil are also available. In a controlled laboratory study that was not considered in the Canadian or the EU assessment (Canada 2019a; ECHA, 2017d), degradation of DP in top layer agricultural soil with a pH of 7.1 and consisting of 64.2% silt, 25.6% sand, 10.2% clay, and 3.12% organic matter, was investigated by spiking the soil with DP in acetone to obtain initial DP concentrations of 0.1, 1, and 10 mg/kg. After 260 days of incubation at 25 ^{o}C 4.2-8.2% of the initial DP had degraded (Cheng et al., 2019). During the same time the labile fraction of DP decreased from 21.25% to 8.2%, 23.8% to 11.7%, and 25.2% to 16.3% in 0.1, 1, and 10 μg/g DP soil, respectively, indicating that the bioavailability of DP decreased under the influence of aging. Model simulations developed to investigate the transformation indicated that the transformation rates were inconstant and statistically different from one another over time. The half-lives of DP in soils spiked with 0.1, 1, and 10 μg/g DP were extrapolated to 1325, 1909, and 2948 days, respectively. As the authors considered that very little bound-residue DP is formed at the start of the aging process, the loss of DP to bound residues was neglected, and the half-lives was simplified to and calculated as the time required for the combined concentration of labile and stable-adsorbed fractions to reduce to half its initial value (Cheng et al., 2019). It has also been reported that *syn*-DP varied little with depth down to 100 cm in soil sampled close to the Chinese manufacturing facility (Wang et al., 2010a), possibly suggesting that isomer-specific degradation or preferential adsorption does not play a significant role in soil. The DP concentration was highest in surface soil (0-5 cm; 837 μg/kg dry weight (dw)), decreasing to 9.16 μg/kg dw at a depth of 60-70 cm and 3.84 μg/kg dw at a depth of 90-100 cm. The f_{anti} value varied little with soil depth but ranged from 0.75 in the surficial layer to 0.67 in the deepest layer (90-100 cm). The study authors make conflicting statements about their finding. On the one hand, they say their findings imply a stereoselective depletion of the *syn*- isomer in soil in comparison with the measured f_{anti} value of the Chinese commercial product, at 0.60. On the other hand, they also state that isomer specific microbial degradation or preferential adsorption does not play a significant role in soil.

53. DP is expected to be persistent in water, soil, and sediment. Based on its binding to particles in air and available laboratory studies and modelling data DP is also considered to be persistent in air under natural conditions. Modelling studies of degradation potential and microbial metabolic pathways suggests that biodegradation of DP is likely to be very slow, and that the probability that DP will degrade any faster than analogue substances listed under the Stockholm Convention is low. Monitoring data show limited degradation in soil and sediments over time and support the conclusion that DP is persistent. The persistency of DP in soil was also recently confirmed in a controlled laboratory study.

2.2.2 Bioaccumulation and toxicokinetics

54. The log K_{ow} for DP is reported to be 9.3 (OxyChem, 2004b). The high log K_{ow} and the very low water solubility (Table 2) indicates that DP is very hydrophobic and partitions to organic matter. This is further supported by the relatively high log sediment-water partition coefficient (log K_p) of 6.65. These properties make it difficult to perform aqueous laboratory studies as it is difficult to maintain stable exposure levels in water. Therefore, aqueous exposure is expected to be of limited importance in terms of bioaccumulation potential. However, significant uptake has been shown to occur in fish in a number of field studies (Guo et al., 2017; Malak et al., 2018; Kurt-Karakus et al., 2019).

55. Although dietary exposure is expected to be more relevant for a substance with physiochemical properties as DP,one fish bioaccumulation study using aqueous exposure reported bioconcentration factor (BCF) values exceeding the bioconcentration criteria set by the Stockholm Convention (BCF >5000 L/kg) (Wang et al., 2019). In this study, carp were exposed continuously for 32 days to DP (range 0.14 to 0.24 ng/L) through water, followed by 32 days depuration. Dosing was carried out using four packages, each containing 1g commercial DP powder (≥ 99% purity) within filter paper enclosed in a filter screen and placed in a hollow steel ball in the bottom of the aquaria. The authors estimated that a steady state water concentration was reached within 3 days, and the equilibrium would subsequently be maintained. Maximum concentrations in muscle for *syn*- and *anti*-isomer were observed at day 32 with the concentrations of 440± 28 ng/g wet weight (ww) and 830 ± 44 ng/g ww, respectively. Equilibrium was reached and the BCF was calculated based on wet weight concentrations in muscle and ratio of uptake and elimination constants. The reported BCF values were 5700 and 9300 L/kg for *syn*- and *anti*-DP, respectively and the depuration half-life was 6.3 and 7.2 days for *syn*- and *anti*-DP, respectively (Wang et al., 2019). The information provided by J-Check English version, on a bioaccumulation test conducted in 1974 with carp exposed to water concentrations of DP at 2.7 and 0.27 μg/L for 8 weeks shows a BCF of 87-121 L/kg (Japan J-Check, 2020b). The concentrations were significantly in excess of the water solubility of the substance. As a consequence, the recorded values may be an underestimate of the true BCF values. However, as the form of exposure may have been a mix of oral and aqeous, this cannot be confirmed. No information on levels observed in fish was provided.

56. Depuration half-life is an important measure of bioaccumulation. A depuration half-life around 8-10 days is indicative of a lipid-normalised and growth-corrected BCF above 5 000 L/kg according to the analysis in Environment Agency (EA), UK (2012). In a dietary bioaccumulation laboratory test using juvenile rainbow trout, the depuration half-life (whole body minus liver) for DP was reported to be 30-40 days for the *anti*-isomer and 50-70 days for the *syn*-isomer, which is highly indicative of a very bioaccumulative substance (Tomy et al., 2008). Neither isomer reached steady state after 49 days of exposure (Tomy et al., 2008). Results from an aquatic food web study from Antarctica (Na et al., 2017) observed, in line with Tomy et al. (2008), higher bioaccumulation of the *anti*-DP isomer. However, a laboratory study in carp showed tissue-specific variations (Tang et al., 2018). Furthermore, a comparison of modelled studies (Larisch and Goss, 2018) and measured oral uptake in fish (Tomy et al., 2008) showed good agreement, indicating that bioaccumulation of super hydrophobic chemicals like DP in fish via food can be predicted. Uptake of DP is slow but will eventually result in substantial bioconcentration as the substance is not metabolised in fish (Larisch and Goss, 2018). Depuration half-life observed in Wang et al., (2019) for carp muscle was lower than what was observed for trout whole body minus liver in Tomy et al., (2008) and could be due to different fish species and experimental setups.

57. Bioaccumulation of *anti*-DP was observed in juvenile marine green macroalgae *U. pertusa*, following a 21-day exposure to 10^{-8} mol DP/L (equal to ~6.53 μg/L) and 21-day depuration (Gong et al., 2018). Accumulated DP levels were 127 and 206 ng/g ww on day 7 and 21, respectively, and the half-life was 1.46 and 14.5 days for *syn*- and *anti*-DP, respectively.

58. In rats exposed to commercial DP 25 by gavage for 90 days at different doses (0, 1, 10, and 100 mg/kg/d), DP preferentially accumulated in liver rather than muscle. At high doses (10 and 100 mg/kg/d) *syn*-DP was predominant isomer in tissues. The elimination half-life for *syn*-DP was about 179 days in liver, 44 days in muscle and 24 days in serum, and for *anti*-DP, 54 days in muscle and 25 days in serum. Depuration time for *anti*-DP for the liver was not calculated due to a non-significant increase in liver after depuration (Li et al., 2013b).

59. Several field studies report biomagnification factors (BMF) and trophic magnification factors (TMF) for DP in various food webs. Factors, such as, temperature, time of sampling, reproduction status, migration, age and tissue versus whole body calculations may affect the calculation of TMF (Borgå et al., 2012; Franklin, 2016). DP biomagnifies in various food webs, both from freshwater, marine waters and terrestric food webs, as BMFs (Tomy et al., 2007; Wu et al., 2018; Sun et al., 2015; 2017, Chen et al., 2020) and TMFs (Sun et al., 2015; Kurt-Karakus et al., 2019; Na et al., 2017) are reported to be greater than 1 in several studies, including in one out of two studies from Antarctica (Na et al., 2017; Kim et al., 2021).For praire animals in the remote Xilingol Prairie in Inner Mongolia in China calculated BMFs for ectotherms and endotherms were in the range 0.146–88.0 and 0.866–17.2, respectively (Chen et al., 2020). Ectotherms were found to selectively enrich *syn*-DP, and stereoselective enrichment increased moving up the food web. The highest DP BMF was for lizards (88.0), which eat insects (Chen et al., 2020). Sample numbers were low in this study and therefore there is some uncertainty in the derived values. Bioaccumulation and maternal transfer have been observed in amphibians collected from a highly contaminated site in South China (Wu et al., 2018). Amphibians occupy an important trophic position in the food web between aquatic organisms and terrestrial biota. Based on the known

predator-prey relationship between frog and insects, BMFs above 1 (1.8 – 2.7) for DP were reported in this study (Wu et al., 2018). In a study of biomagnification of DP in a freshwater reservoir in the vicinity of an electronic recycling facility in South China, the TMFs of the DP isomers were found to be 2-3 times greater than those of PBDE congeners and comparable to or lower than those of the highly recalcitrant PCB congeners in the same food web (Wu et al. 2010). In an aquatic food web study from China, TMFs were reported in the range 1.9 – 3.1. In this study, the biomagnification potential (TMF >1) was also reported for DPMA, *anti*-Cl_{11}-DP and *anti*-Cl_{10}-DP (Wang et al., 2015), and this should also be taken into account when evaluating bioaccumulation for DP. For more information on these substances, see section 1.1 Chemical identity. Three terrestrial studies are available showing BMFs above 1 (Yu et al., 2013; She et al., 2013; Wu et al., 2018). Furthermore, in study investigating levels of DP and other flame-retardants in livers of redlisted and endangered birds in Korea, Jin et al. (2016) found a significant positive relationship (r^2= 0.22 and p=0.031) between the concentrations of DP and trophic position ($\delta^{15}N$) in three residential and carnivorous predatory species, the Eurasian eagle owls (*B. bubo*), common kestrels (*F. tinnunculus*), and collared scops owls (*O. lempiji*), indicating the potential for biomagnification of DP in these birds. For further details on the bioaccumulation studies, see Table 5 in UNEP/POPS/POPRC.17/INF/xx.

60. The isomer composition of DP in environmental samples can be different from the technical products because of their biota isomer-selective uptake or elimination, bioaccumulation, and stereospecific photodegradation (Wang et al., 2015; and reviewed in Wang et al., 2016). Trophic levels (Peng et al., 2014; Tomy et al., 2007; Wang et al., 2015), exposure concentrations (Li et al., 2013a,b; Mo et al., 2013; Peng et al., 2015), life cyclic stages (Klosterhaus et al., 2012; Sühring et al., 2014), the type of wildlife and tissue (Peng et al., 2014; Zhang et al., 2011; Zheng et al., 2014a) and stereoselective excretion (Li et al., 2019) are the main factors for the stereoselective bioaccumulation of DP in organisms.

61. Studies in birds indicate that uptake, distribution and elimination kinetics of DP and its isomers are complex. Chickens bred in the vicinity of an e-waste recycling facility exposed to DP through the environment from sand and food showed preferential accumulation of the *anti*-DP isomer (Zheng et al., 2014a). Lipid content of the tissue were the main factor in the tissue distribution, although the degree of blood perfusion and the tissue function were also important factors. Furthermore, the study revealed tissue specific accumulation of *anti*-DP, with higher f_{anti} levels in brain, fat and liver (0.65-0.64) compared to (0.54-0.59) for other tissues. The study also indicate that *anti*-Cl_{11}-DP was absorbed through the diet rather than metabolised from DP *in vivo* based on the similar ratios for *anti*-Cl_{11}-DP to *anti*-DP in soil and chicken (Zheng et al., 2014a). This information is supported by the fact that *syn*-Cl_{11}-DP and *anti*-Cl_{11}-DP were detected in commercial DP-25, hence, it can be inferred that these chemicals originated from commercial products (Li et al., 2013b).

62. There are some studies on biotransformation of DP (Tomy et al., 2007; 2008; Ren et al., 2009; Sverko et al., 2010). DP has been shown not to metabolise easily in biota (Tomy et al., 2008; Xian et al., 2011) and the results from Tomy et al. suggest that enzyme-induced metabolism of DP in fish might be low, if it does occur. No hydroxylated or sulfonated metabolites of DP was observed in fish liver extracts or in human serum (Tomy et al., 2007; Ren et al., 2009). Degradation products of DP such as decachloropentacyclooctadiene (DP-Cl_{10}) or undecachloropentacyclooctadiene (DP-Cl_{11}) have been detected in bird eggs (Guerra et al., 2011; Muñoz-Arnanz et al., 2011, 2012; Zheng et al., 2014a), and rat (Li et al., 2013) but some studies suggest they are formed through biotic or abiotic processes prior to uptake or even through impurities in the commercial product (Sverko et al., 2008, 2010; Tomy et al., 2008; Zheng et al., 2010, 2014b; Li et al., 2013). Two additional dechlorination analogues were also detected in both quails and the test substance (Li et al., 2013a,b). In a study where eggs from Japanese quail were injected in the yolk sac with DP, no biotransformation of DP was shown (Briels et al., 2018). This was also observed in chicken eggs at day 18, however, at pipping the mass of *syn*- and *anti*-DP in neonate chicks (including the remaining yolks) declined significantly by 5.9% and 15%, respectively, indicating some metabolic activity during the later developmental stage (Li et al, 2019).

63. DP was first detected in archived fish (walleye) from Lake Erie sampled from 1980-2000, which suggested that DP was taken up by the fish (Hoh et al., 2006). Available monitoring studies show that DP is widely dispersed in the environment (reviewed in Canada, 2019a; ECHA 2017b). Global evidence reveals that uptake of DP can occur in various biota samples such as aquatic and terrestrial food webs as well as the human body (reviewed in Wang et al., 2016). Field monitoring data suggest that DP is bioavailable and can exceed levels in biota that are of concern based on critical body burden considerations related to baseline narcosis (ECHA 2017b).

64. DP was detected in human milk samples collected in two Canadian cities (Siddique et al., 2012), in human serum (Brasseur et al., 2014; Yan et al., 2012: Ren et al., 2009; Chen et al., 2015) cord serum (Ben et al., 2014) and adipose tissue (Pan et al., 2020; Yin et al., 2020) as well as in human hair (Chen et al., 2015). Furthermore, very high DP concentrations in blood and hair from workers of a manufacturing facility have been reported (Zhang et al., 2013). See Tables 10 and 13 in UNEP/POPS/POPRC.17/INF/XX for further details on detections in biota and human samples.

65. In summary, a long depuration half-life, which is indicative of a BCF above 5000 L/kg, has been reported for DP. The log K_{ow} for DP is greater than 5. Although aqueous exposure is expected to be of limited importance in terms of bioaccumulation potential of DP a BCF above 5000 L/kg have been reported for DP isomers in a fish study. In

addition, BMFs and TMFs > 1 have been reported for several organisms and food webs. Many studies have revealed the presence of DP in wildlife and humans. Field monitoring data suggests that DP is bioavailable and can achieve high body burdens (ECHA, 2017b). These lines of evidence support the conclusion that DP is bioaccumulative. Additionally, DP was recently identified as a very bioaccumulative (vB) substance in the EU (ECHA, 2017a).

2.2.3 Potential for long-range transport

66. Due to its high hydrophobicity and low vapour pressure (see Section 1.1), DP in the atmosphere adsorbs to airborne particles (Sverko et al., 2011; AMAP, 2017, Canada, 2019a). Monitoring studies have reported mean fractions of particle-bound DP in air as high as 99 % (Hoh et al., 2006). Measurements in seawater confirm the propensity of DP to adsorb to particles. In the East-Greenland Sea, particle-bound DP accounted for, on average, 97% of *syn*-DP and 80% of *anti*-DP. In Atlantic seawater, the particle bound fraction was on average 58% for *syn*-DP and 75% for *anti*-DP (Möller et al., 2010).

67. The modelled half-life (QSAR estimate) of DP in air is in one study reported to be 13.68 hours (Sverko et al., 2011), i.e. below the criterion of two days set in Annex D (d) (iii) of the Convention. The Canadian assessment reports estimated half-lives in air of 0.468 and 160.12 days for atmospheric oxidation and ozone reaction, respectively, using the AOPWIN 2010 model (12-hr day) (Canada, 2019a). However, air modelling that is based on gas phase reactions may underestimate the half-life value because they to not take into account the particle binding which is expected for DP. Particle binding will lower the photolysis rate and result in a longer half-life in air (reviewed in Sverko et al., 2011; Canada, 2019a). As shown already for other chemicals with similar physico-chemical properties such as decaBDE (Brevik et al., 2006; UNEP/POPS/POPRC.10/10/Add.2.; POPRC-10/2), sorption to particles may slow down reaction rates, increase the actual half-life in air and facilitate long-range environmental transport (LRET) of DP on particles (Sverko et al., 2011; Canada, 2019a). As described in ECHA (2017d), the LRET of substances with low vapour pressure that adsorb strongly to particulates in the air is likely to be governed by the fate of the particulates to which they bind and can undergo LRET to remote regions when atmospheric conditions permit (e.g. during dry periods).

68. Modelling studies performed with the OECD P_{OV} and LRTP Screening Tool, a software tool for screening chemicals for persistence and LRET potential (Wegmann et al., 2009), and comparing DP to DP analogues and benchmark chemicals including α-HCH, HCB, PCB-101, PCB-180, PCB-28, HBCD, atrazine, *p*-cresol, chlordene plus, Dec 602, Dec 603 Dec 604, suggest that DP has transport and persistence properties within the range for listed POPs (Sverko et al., 2011), and that DP may be deposited to some degree in remote regions (Canada, 2019a). However, the results generated using the OECD P_{OV} and LRTP Screening Tool are associated with uncertainty, largely because most of the input parameters used in the model are estimated (ECHA, 2017d; Sverko et al., 2011).

69. DP has been detected in many different environmental matrices and biota in remote regions; in Arctic air, snow, soil, sediment, water and biota (reviewed in AMAP, 2017; Vorkamp et al., 2019a,b; Canada, 2019a); in Antarctic air, soil, lichens, mosses, limpet, amphipods, cod, icefish, kelp gulls, penguins, skuas, southern elephant seals and Antarctic fur seals (Möller et al., 2010; Gao et al., 2018; Kim et al., 2015; 2018; 2021; Aznar-Alemany et al., 2019); in air, soil and lichen at the Tibetan Plateau, a remote mountain region in Asia (Yang et al., 2016a; Liu et al., 2018); and in air, water, soil, plant and animal muscle and hair in the remote Xilingol Prairie in Inner Mongolia, China (Chen et al., 2020).

70. In Arctic biota, most detections have been made in species in the marine food web (polar bear, ringed seal, harp seal, hooded seal, beluga whale, killer whale, narwhal, glaucous gull, common gull, black guillemot eggs, common eider, European shag, kittiwake, mink, and cod (*anti*-DP only)). Detections also include terrestrial and freshwater species, i.e. reindeer dung, peregrine falcon eggs and landlocked Arctic char (reviewed in AMAP 2017; Canada, 2019a; see also Norwegian Environment Agency, 2018a; Dreyer et al., 2019; Letcher et al., 2018; Green 2019; Verrault et al., 2018; Vorkamp et al., 2015; 2018; 2019a; Houde et al., 2017; Simond et al., 2017; Schlabach et al., 2011).

71. As discussed in further detail in section 2.3.1 below, while concentrations in remote regions are generally low, they are not always lower than DP levels in source regions (see Table 6. UNEP/POPS/POPRC.17/INF/xx). Reported detection frequencies vary from non-detect to 100% detection. Collectively, the available monitoring data suggests that DP can be transported over large geographical distances to remote areas where it deposits, transfers to the receiving environment and is taken up in biota (reviewed in Sverko et al., 2011; AMAP 2017 and Canada 2019a; see also Möller et al., 2010; 2011; 2012; Na et al., 2015; Yang et al., 2013; Wang et al., 2010b).

72. Möller et al. (2010) were the first to publish environmental monitoring data demonstrating the potential for DP to undergo LRET. In this study, marine boundary layer air and surface seawater samples were collected during a sea expedition from the East-Greenland Sea via the Northern and Southern Atlantics toward Antarctica. DP concentrations ranged from 0.05 to 4.2 pg/m^3 in the atmosphere and from non-detect to 1.3 pg/L in seawater. The results indicate that transport may occur both via air and seawater and show preferential partitioning to particles in both air and seawater (Möller et al., 2010). In the East-Greenland Sea, the authors observed decreasing concentrations of DP in air with increasing latitude. The authors suggest that the finding reflect stereoselective depletion of *anti*-DP likely caused by UV degradation during LRET. The trend of declining levels with increasing latitude was not reflected in the seawater samples. In the study, Western Europe was identified as a source region for DP in the marine environment. The analysis

of air mass back trajectories showed a mix of oceanic, Arctic and continental air masses. The higher levels of DP in seawater observed at some northern sites were probably due to water masses originating from the Atlantic and Arctic Oceans, in combination with freshwater inputs from melting land ice.

73. On a sea expedition from the East China Sea to the Arctic (Möller et al., 2011), DP concentrations ranged from 0.01 to1.4 pg/m^3 in air and from 0.006 to 0.4 pg/L in seawater. The Asian continent was identified as a source of DP to this marine environment, and the air-seawater exchange indicated deposition of DP from air to water. DP concentrations in air in the Chukchi/Bering Sea region were generally low compared to the source region. Decreasing DP concentrations with increasing latitude were also observed in this study; however, concentrations were near detection limits.

74. In yet another study of DP in marine boundary layer air by the same authors, atmospheric samples were collected during a sampling expedition from the East Indian Archipelago toward the Indian Ocean and further to the Southern Ocean (Möller et al., 2012). *Syn*- and *anti*-DP were detected in all marine atmospheric samples at concentrations from 0.26 to 11 pg/m^3. Measured concentrations were in the range of previously reported DP concentrations in the marine atmosphere, with the highest concentrations detected in the East Indian Archipelago (see Table 6 and 7, UNEP/POPS/POPRC.17/INF/xx).

75. As discussed by Vorkamp et al. (2019a), the highest DP levels in Arctic air to date were detected on Greenland in 2012. Reported mean DP concentrations in Greenland air in 2012, 2014 and the period 2014-2016 were 6.7, 0.64 and 4.2 pg/m^3, respectively (Vorkamp et al., 2015; 2019a). DP was also positively detected in atmospheric samples from other Arctic sites: Alert in the Canadian Arctic (Xiao et al., 2012), Longyearbyen in the Norwegian Arctic (Salamova et al., 2014), and Little Fox Lake in the Canadian Sub-Arctic (Yu et al. 2015). Mean DP concentrations reported for these sites were ~0.75, ~1.2 and ~0.25 pg/m^3, respectively. DP was also detected in air from Pallas in Northern-Finland at a mean concentration of 0.039 pg/m^3 (Haglund et al., 2016).

76. LRET of DP via air is also documented by studies from other remote regions than the Arctic. The studies by Möller et al. (2010; 2012) demonstrate ongoing long-range atmospheric transport of DP to the Antarctic and further show that the DP levels in Antarctic air are comparable to those of other FRs such as polybrominated diphenyls (PBDEs) and polybrominated biphenyls (PBBs). These findings are supported by detections of DP in Antarctic biota as further described in section 2.3.1 Environmental levels and trends (see Gao et al., 2018; Kim et al., 2015; 2018; 2021; Aznar-Alemany et al., 2019). DP also undergoes LRET to remote mountain areas. In a study on lichen from the southeast Tibetan Plateau, DP concentrations appeared to decrease with increasing altitude (Yang et al., 2013). According to the authors, the distribution pattern for DP observed in this study, combined with knowledge about the monthly average surface wind vector field, provides evidence that DP transport into the area is mainly driven by the Indian monsoon systems and can be attributed to LRET. Yang et al. (2013) further note that the lower DP levels observed at higher altitudes indicate that DP transported into the area via the atmosphere is cold trapped by the mountains.

77. The Global Atmospheric Passive Sampling (GAPS) network has been deploying passive air samplers quarterly since 2005. A retrospective analysis of GAPS samples from 2005 and 2006 covering 67 different sampling sites indicates that DP is globally distributed in air with total DP levels ranging from non-detect to 30 pg/m^3. Elevated levels were reported for source areas; The median value at urban sites (~0.9 pg/m^3) was approximately 10 times greater than at background, rural, and agricultural sites (Shuster et al., 2020).

78. DP has been detected in muscle, guano, eggs, plasma, preen oil and feathers in different species of birds around the world (Gauthier and Letcher 2009; Guerra et al., 2011; Munoz-Arnanz et al., 2011; 2012; Baron et al., 2014b; 2015; Jin et al. 2016; Vorkamp et al., 2018; Løseth et al., 2019; Briels et al., 2019; Mo et al., 2019; Desjardins et al., 2019; Zhu et al., 2020), including in migratory birds and eggs of migratory birds (e.g. Zhu et al., 2020 Vorkamp et al., 2018;). Vorkamp et al. (2018) detected DP in eggs of Greenland peregrine falcons which migrate from the Caribbean and South America to the Arctic for the summer. The DP detected in these eggs could reflect exposure experienced at the wintering and breeding grounds, as well as during migration.

79. Birds have previously been identified as biovectors for the transport and deposition of POPs to ecosystems in remote regions through deposition of guano, feather loss and decaying carcasses (Evenset et al., 2007; Blais et al., 2005) and may represent an additional transport pathway for DP to remote regions. In Ellasjøen at Svalbard, seabird guano was, based on measurements of PCB, found to account for approximately 14% of the contaminant inventory of the lake catchment area, approximately 80% of the contaminant inventory of the lake itself and was suggested to be approximately thirty times more efficient as a contaminant transport pathway compared to atmospheric LRET (Evenset et al., 2007).

80. In a study by Desjardins et al. (2019), deposition of guano by urban-adapted ring-billed gulls from a colony on Deslauriers Island near the city of Montreal, Canada were postulated to increase exposure locally (Desjardins et al., 2019). The total amount of halogenated FRs deposited by the entire ring-billed gull colony (64 980 gulls, both sexes combined) through guano was estimated to be 1 g during a 28-day period. DP was the predominant halogenated FR detected and *anti*- and *syn*-DP accounted for 2-3% and 1-2% of the total concentration halogenated FRs, respectively.

The authors propose that the migratory ring-billed gulls from this urban-adapted colony could be an underestimated biovector of halogenated FRs to nearby ecosystems.

81. As discussed, Möller et al. (2010) postulated that the fraction of *anti*-DP in environmental samples (i.e. f_{anti} = *anti*/(*syn* + *anti*) based on concentrations) decreases with increasing distance from the source as a result of more rapid degradation of the *anti*-DP isomer than the *syn*-DP isomer in UV-light. However, these findings may also reflect isomerisation of the *anti*- to *syn*-DP (Sverko et al., 2011). Available monitoring studies provide some evidence of changes in isomer ratios with increasing distance from the source (ECHA, 2017b), but observations from the Arctic are generally inconclusive in this regard (AMAP, 2017). More specifically, DP isomer ratios in Arctic air have been reported in some studies to be in the same range as found in commercial mixtures (Xiao et al., 2012; Salamova et al., 2014; Vorkamp et al., 2015), but have in other studies (Möller et al., 2010; 2011; Carlsson et al., 2018; Na et al., 2015) been reported to differ from the commercial mixtures. Findings from Antarctica are consistent with a change in DP isomer ratio during LRET. In a study from the Fildes Peninsula in Antarctica, Gao et al. (2018) estimated the mean f_{anti} values in soil and lichen to 0.37 and 0.24, respectively. These f_{anti} values are lower than those of commercial products (f_{anti} = 0.64–0.80) and the authors conclude that the findings confirms that long-range atmospheric transport is a main source of DP, and that the DP burdens could be driven by a stereoselective enrichment of *syn*-DP during atmospheric transport as postulated by Möller et al. (2010). However, as the concentration of DP in soil and lichen differed between the sampling sites, the authors also suggest that the DP levels may have been influenced by anthropogenic and animal activities in the area (Gao et al., 2018).

2.3 Exposure

2.3.1 Environmental levels and trends

82. DP is globally distributed and is detected worldwide in many different environmental matrices and biota, and at different types of locations spanning from production sites and recycling facilities to urban, rural and remote areas (Sverko et al., 2011; Wang et al., 2016; Schuster et al, 2020; Zafar et al., 2020). Elevated DP levels are reported in source areas (e.g. Schuster et al, 2020,) and are higher in urban centres compared to rural, agricultural and remote sites (Syed et al., 2020; Schuster et al. 2020). As described also in section 2.2.3 above, Schuster et al. (2020) report that the median value at urban sites (~0.9 pg/m^3) was approximately 10 times greater than at background, rural, and agricultural sites.

83. DP levels in remote regions and in ocean water and air are generally lower than levels reported in source regions near production sites and urban centres, although high levels in some instances also have been observed at remote sites (reviewed in Sverko et al., 2011; Canada, 2019a). As further detailed below, due to limited data, the temporal and spatial trends for DP are equivocal.

84. Several DP monitoring studies have been carried out in the vicinity of DP manufacturers and e-waste recycling sites, including in the US, China and Pakistan (Sverko et al., 2011; Ji et al., 2018; Iqbal et al., 2017). High DP levels have been reported in various matrices from these areas (Wang et al., 2016), with reported concentrations up to several ng/m^3 close to known production or e-waste recycling sites (Sverko et al., 2011). The soil concentration of DP near the Anpon Electrochemical Co. production facility in China was found to be 0.83-1200 ng/g dw but decreased by an order of magnitude within 7.5 km (Wang et al., 2010a). Other studies have reported average soil concentrations of 1490 ng/g and 35.6 ng/g for this site (Wang et al., 2010c; Zhang et al., 2015). Atmospheric DP concentrations near the DP production plant in, China were found to be as high as 7737 to 26,734 pg/m^3 (Wang et al., 2010a). In North America, DP has also been detected in air and precipitation at Sturgeon Point and in Niagara River suspended sediment samples, near the OxyChem manufacturing facility in the US (Shen et al., 2011; Olukunle et al., 2018). Total DP levels in air at Sturgeon Point, were reported to be 2.70 pg/m^3 by Olukunle et al. (2018) and 3.1 pg/m^3 by (Liu et al., 2016), while Hoh et al. (2006) in one instance measured concentrations up tp 490 pg/m^3. Total DP levels in precipitation from the same site were 323 pg/m^3 and 890 pg/m^3, as reported by Olukunle et al. (2018) and by Salamova and Hites (2011), respectively. Examination of sediments collected in this area between 1980 and 2006 showed a declining concentration trend for DP (Shen et al., 2011). A declining concentration trend for DP was also reported previously for Niagara River suspended sediments (Sverko et al., 2008) and are consistent with declines observed in a sediment core collected near the river mouth on the Niagara River bar (Sverko et al., 2010). The suspended sediment trends observed by Shen et al. (2011) were also reflected in open lake sediment cores, which generally showed a decline in DP concentrations since 1980 in the Niagara basin. Similar findings were reported for the Mississauga basin (Shen et al., 2010). It is not clear if the decline in DP in Niagara River suspended sediment samples represent changes in production volumes at the Oxychem manufacturing site or whether it reflects emission controls adopted over the past 20 years at the site. Reported DP levels in surface sediments in Lake Ontario (2.230×10^3 -5.860×10^5 pg/g dw) and Lake Erie (61.00– 8.620×10^3 pg/g dw) in North-America, down-stream of the Oxychem manufacturing plant, are at the high end of reported concentrations globally along with levels in surface sediments from some locations in China including the Huai'an area where there is also a DP manufacturing plant (1.860 × 10^3–8.000×10^3 pg/g dw), the Dongjiang River (80.00–1.940 × 10^4), Qiantang River (<9000-1.100x10^3 pg/g dw) and mangrove sediments of the Pearl River Estuary (13.00-1.504 × 10^3 pg/g dw) (Sverko et al., 2008; Wang et al., 2010; He et al., 2014; Sun et al., 2013; 2016, as cited in Niu et al. 2020).

Surface sediments in source areas are likely important reservoirs of DP in the environment (Niu et al. 2020), but monitoring data also provide evidence for uptake in biota and elevated levels in aquatic species in these areas (reviewed in Zafar et al. 2020, see also Table 10 in UNEP/POPS/POPRC.17/INF/xx). For example, DP levels in mucle tissues of commercial fish species from one of Chinas most polluted rivers, the Liaohe River ranged from below detection limit to 470 pg/g lw (Ren et al. 2013).

85. A very high DP concentration (3327 ng/g) was found in soil at an e-waste recycling site in Qingyuan, China. DP levels fell dramatically with increasing distance away from the recycling site (Yu et al., 2010). High levels (average 3398 ng/g) of DP were also reported in soil samples from areas with informal e-waste recycling activities in Karachi, Pakistan (Iqbal et al. (2017). The high DP levels in soil reported at recycling sites by Iqbal et al. (2017) and Yu et al. (2010) are comparable with levels reported from other e-waste recycling sites in Asia (South/ Southeast China) (Li et al., 2018; Ge et al., 2020) and much higher than the DP levels in soil collected in close vicinity to the DP manufacturing facility in Huai'an City, China (Wang et al., 2010a,c; Zhang et al. 2015). In the recycling town Guiyu, in southeastern China total DP levels in soil were 3.8-2100 ng/g, and DP was also found in sediment (1100 - 7200 ng/g dw), road dust (14 – 1100 ng/g dw) and $PM_{2.5}$ particles (180 pg/m^3 *syn*- DP and 170 pg/m^3 *anti*-DP) (Li et al., 2018). Elevated DP levels in air (15 – 85 ng/m^3) were also reported in the e-waste area of Karachi, and are higher than reported levels found in air close to the DP manufacturing site (7.7 – 26.7 ng/m^3) in China (Iqbal et al., 2017; Wang et al., 2010a; see also review by Zafar et al., 2020). Elevated levels have also been reported in biota near e-waste sites. DP concentrations from 19 to 9630 ng/g lw have been reported in aquatic species including water snake and mud carp from an e-waste recycling workshop of South China (Wu et al., 2010). In another study from e-waste recycling site at South Longtang Town China, two species, northern snakehead and Crucian carp were studied for maternal transfer and sex-related accumulation of DP. The concentrations of hepatic DP for northern snakehead and Crucian carp were 260–1920 ng/g lw and 340 to 1670 ng/g lw, respectively (Wu et al. 2013).

86. According to Iqbal et al. (2017) elevated levels of DP at informal recycling sites may originate from emission associated open-air burning-, acid stripping-, manual dismantling-, and shredding of cables, televisions, plastic roofing and connectors Iqbal et al. (2017). Iqbal et al. (2017) furthermore suggest that sites with informal e-waste recycling may have higher environmental impact than manufacturing sites and urban environments, stressing the importance of chemical managing strategies across the entire life cycle of electronics (Iqbal et al., 2017). However, Ge et al. (2020) found elevated concentration levels of DP (range of total DP was 21.8 – 18 000 ng/g and the median levels of *syn*- and *anti*-isomers was 211 ng/g and 504 ng/g, respectively) in soil samples from within a newly (2015) established recycling facility that uses environmentally friendly technology. The study compared soil samples from the recycling facility with samples from the surrounding area (range of total DP was 1.76 – 4 050 ng/g and the median levels of syn- and anti-isomers was 2.81 ng/g and 8.52 ng/g respectively) which had lower levels of DP in soil. DP has also been detected wildlife living at or in close vicinity of e-waste sites (e.g. Li et al., 2014; Wu et al., 2013; 2018, Peng et al. 2019). Peng et al. (2019) found that birds (kingfishers) feeding close to e-waste recycling sites in South China had 10- to 1 000-times higher levels of total DP concentrations than kingfishers from non-e-waste sites.

87. DP levels in urban areas and urban industrialized areas are reported in several studies and from several global regions. In Pakistan atmospheric concentrations of legacy and emerging flame retardants were monitored in eight major cities across the country (Karachi, Lahore, Faisalabad, Rawalpindi, Multan, Quetta, Peshawar and Gilgit) using a high volume-active air sampler (gas + particle phase). *Anti*- and *syn*-DP were detected in all cities with concentrations ranging from 0.60–12.0 pg/m^3 (3.08 pg/m^3) and 0.80–42.3 pg/m^3 (10.8 pg/m^3), respectively. Sum DP concentations ranged from 1.4–54.2 pg/m^3 with an average of 13.8 pg/m^3 (Syed et al., 2020). Ealier studies from Pakistan have reported sum DP concentrations in urban air of 88.0, 0.41 and 2.48 pg/m^3 in the vicinity to River Ravi, River Chenab and Indus Basin, respectively (Syed et al., 2013; Ali et al., 2015; Mahmood et al., 2015). In the study of Ren et al. (2008), DP was measured in air across 97 Chinese urban and rural sites. DP was detected in 51 of these sites, with a mean air concentration in urban centers of 15.6 ±15.1 pg/m^3. The levels in urban centers were approximately 5 times greater than those measured in rural areas 3.5±5.6 pg/m^3 and are comparable to the levels reported in urban centres in Pakistan by Syed et al. (2020). A more recent study from Chinese cities reported average levels of 2.04 pg/m^3 and 1.5 pg/m^3 for *syn*- and *anti*-DP, respectively (Liu et al., 2016b). A study from the Great Lakes, US, showed concentrations of DP ranging from 0.31–4.80 pg/m^3 at an urban site in Chicago (Hoh et al., 2006)). In a study from Canada, urban-feeding ring-billed gulls breeding on Deslauriers Island in an industrialized section of the St. Lawrence River, downstream from Montreal were found to contain high levels of DP. The DP hepatic mean level was 230 ng/g lw. *Anti*- and *syn*-DP were detected in 100% and 93% of the investigated livers, respectively (Gentes et al., 2012). In a study of redlisted and endangered migratory and residential birds in Korea, Jin et al. (2016) found that more than half of the bird samples (65%) contained detectable concentrations of DP. Sum DP concentrations ranged from below the limit of quantification to 4734 ng/g lw with a mean concentration of 172 ng/g lw. Overall, sumDP levels were according to the authors similar to sum PBDE concentrations. DP levels were higher in residential predators than in migratory predators and non-predatory birds, but the difference was not significant. There was also evidence for potential biomagnification in birds (see details in Section 2.2.2).

88. In a study on halogenated FRs in eels from five Latvian lakes, nine dechlorane-related compounds (i.e. mirex, *syn*-DP, *anti*-DP, Dec 602, Dec 603, Dec 604, hexachlorocyclopentadienyldibromocyclooctane, Cl_{10}DP and Cl_{11}DP) were analyzed. The mean total concentration of dechlorane-related compounds was 0.62 ng/g lw and the geographical

distribution was nearly uniform among the five lakes (Zacs et al. 2018). The concentrations of *syn*- and *anti*-DP ranged from non-detect to 0.45 ng/g lw, and from non-detect to 0.89 ng/g lw, respectively, with average concentrations of 0.14 ng/g lw, and 0.24 ng/g lw, respectively. The contribution of *anti*-DP to the total concentration of dechlorane-related compounds was approximately 33%, while *syn*-DP accounted for approximately 10% on average. The composition of mixtures containing *syn*- and *anti*- DP showed an f_{anti} value that was close to the composition of the OxyChem® DP commercial product, indicating the release of DP from disposal of consumer products as the probable source. In another study from Baltic region, de Wit et al. (2020) widely detected DP in Baltic biota, with high levels in some birds (particularly eider).In this study a suite of chemicals of emerging concern, including DP, were analysed in blue mussel, viviparous eelpout, Atlantic herring, grey seal, harbor seal, harbor porpoise, common eider, common guillemot and white-tailed eagle from the Baltic Proper, sampled between 2006 and 2016. Both DP isomers were found in all species. Mean reported *syn*- and *anti*-DP levels were highest in common eider liver (16 and 52 ng/g lw, respectively). Mean reported *syn*- and *anti*-DP levels were also high in grey seal blubber (6.0 and 26 ng/g lw, respectively) and white-tailed eagle (3.7 and 7.8 ng/g lw, respectively). Lower means were found in herring muscle (0.035 and 0.070 *syn*- and *anti*-DP), harbor seal blubber (046 and 0.092 *syn*- and *anti*-DP), harbor porpoise (0.04 and 0.074 *syn*- and *anti*-DP), eider and guillemot eggs (0.10 and 0.21, 0.13 and 0.40 ng/g lw for *syn*- and KDP, respectively). The lowest levels were found in blue mussel and viviparous eelpout with reported *syn*- and *anti*-DP means of <0.18-<0.21 and <0.14 and 0.16.

89. As described above and in section 2.2.3, DP is detected in remote areas globally. See also Tables 6-9 in UNEP/POPS/POPRC.17/INF/xx. A review of emerging Arctic contaminants conducted by AMAP concluded that atmospheric concentrations of DP in the Arctic were comparable to those of PBDEs (AMAP, 2017). While DP concentrations in remote regions are generally low, they are not always lower than in source regions. For example, in the study investigating halogenated FRs in air and soil from Mt. Gongga on the eastern Tibetan Plateau, DP levels ranging from below detection limit to 11.5 pg/m^3 and 48.3 pg/g in air and soil, respectively, were reported (Liu et al., 2018). The Tibetan Plateau is the world's highest elevation plateau and is considered as one of the most remote and isolated areas on earth, and ideal for the study of LRET of POPs (Yang et al., 2016a). According to the authors (Liu et al., 2018), the concentrations of DP in air and soil on the Tibetan Plateau were higher than in the Great Lakes region (0.14–4.0 pg/m^3) (Sverko et al., 2011), but within the range of air samples from China (not detected to 66 pg/m^3) (Ren et al., 2008), and much lower than those observed near a production facility in China (7737–26,734 pg/m^3) (Wang et al., 2010a). In another study on lichen from the southeast Tibetan Plateau, DP was detected in 89% of the samples (Yang et al., 2016a). Concentrations ranged from 27 to 843 pg/g, with an average concentration of 167 pg/g. The concentrations reported were comparable, but slightly higher than in tree bark from Shenzen, Hangzhou and Tanjin in China (Qiu and Hites, 2008) and much lower than in tree barks from South Korea and New York in the US (Qiu and Hites, 2008) but significantly higher than in moss from Ny-Ålesund at Svalbard (Na et al., 2015).

90. In Antarctic soil, lichen and moss samples, DP was detected in all samples (100%) (Gao et al., 2018; Kim et al., 2018). The DP concentrations in Antarctic soil were between one and several orders of magnitude lower than the levels reported in surface soil samples collected from an industrial region (0.0336–4.65 ng/g) and an e-waste recycling area (nd–47.4 ng/g) in South China (Yu et al., 2010). On the other hand, DP levels in both lichen and moss from Antarctica were higher than previously measured in mosses sampled in Ny-Ålesund in Arctic Norway in 2012 (Na et al., 2015). The average concentrations of DP in lichen from Antarctica were similar to those in reeds in northeastern China (0.63 ng/g wet weight (ww)) (Wang et al., 2012) and in lichen in the southeast Tibetan Plateau (167 pg/g) (Yang et a., 2016a) but lower than in tree bark from areas in the northeastern US (0.03–115 ng/g) and South Korea (1.4 ng/g) (Qiu and Hites, 2008).

91. In a Norwegian screening assessment of emerging Arctic contaminants, DP was detected in all species monitored. The detection frequency within the species tested was 100%, except in common eiders where it was 80% (Norwegian Environment Agency, 2018a). S*yn*- and *anti*-DP concentrations in organisms from Svalbard were in the range from non-detect to 6.9 ng/g ww with reported average values of 0.04 and 0.07 ng/g ww for common eider (eggs), 0.01 and 0.03 ng/g ww for kittiwakes (eggs), 0.24 and 1.8 ng/g ww for glaucous gull (eggs), 0.63 and 4.5 ng/g ww for polar bear (blood). Moreover, DP was also detected in European shag (eggs) from Røst, in mink (liver) from Sommarøy and in common gulls from Tromsø. Reported *syn*-and *anti*-DP levels in these species were 0.22 and 1.4 ng/g ww in European shag, 0.21 and 1.2 ng/g ww in mink, and 0.1 and 0.63 ng/g ww in common gulls, respectively. In another study conducted for the Norwegian Environment Agency, DP levels were below the limit of quantification in cod livers from Arctic sites at Svalbard and Kjøfjord in the Outer Varangerfjord and the Tromsø harbour area (Green et al., 2019). In contrast, reported *syn*- and *anti*-DP in cod livers from the Bergen harbour area, a non-Arctic site on the west coast of Norway, were 0.178 and 0.203 µg/kg ww. *Syn*- and *anti*-DP levels in cod livers from the Inner-Oslofjord in Southern Norway were 0.135 and 0.231 µg/kg ww, respectively.

92. In the previously mentioned report by Vorkamp et al. (2019a) on DP in air and biota from Greenland, *anti*-DP isomer was detected in 92% of all air samples from 2014 to 2016 and in 46% of the air samples collected in 2014. As already indicated above in section 2.2.3, the mean reported concentrations in air in this study were 4.2 pg/m^3 and 0.64 pg/m^3 for the period 2014 to 2016 and for 2014, respectively. Except for glaucous gull, DP levels were close to or below limits of quantification in biota samples. Neither of the two DP isomers were found in ringed seal samples from Thule. In ringed seals from East-Greenland, Arctic char and narwhals, only the *anti*-DP isomer was present. *Syn*- and *anti*-DP were both detected in 100% of all glaucous gull samples from Thule at mean concentrations of 0.22

and 0.89 ng/g lw (0.012 and 0.049 ng/g ww), respectively. Detection frequencies for *syn*- and *anti*-DP in glaucous gull samples from East-Greenland were 87.5 and 100%, with mean concentrations of 0.24 and 0.93 ng/g lw for *syn*- and *anti*-DP. A geographical comparison based on samples of glaucous gull and ringed seal showed relatively uniform DP levels. The DP levels measured in this study were generally comparable to concentrations reported for some of the same species (e.g. ringed seals, glaucous gull and Arctic char) from other locations in the Arctic. However, DP concentrations in glaucous gull liver samples from Greenland reported in this study were a factor of 5-10 lower than in corresponding samples from the Canadian Arctic (Verreault et al., 2018). In the study from the Canadian Arctic, *syn*- and *anti*-DP were detected in 65% and 59% of male liver samples from 2012, respectively (Verreault et al., 2018). In females, the detection frequency of both isomers was 21%. Mean concentrations were only calculated for male birds and were 0.18 and 0.16 ng/g ww for *syn*- and *anti*-DP, respectively.

93. DP was also previously detected in air and biota samples from Greenland along with other FRs (Vorkamp et al., 2015). Mean *syn*- and *anti*-DP concentrations in air were 2.3 and 5.2 pg/m^3. While the detection frequency of both isomers in air was 46%, *anti*- and *syn*-DP was detected in 95% and 100% of biota samples, respectively. In biota, ringed seals from East-Greenland had the highest DP concentrations with reported mean blubber concentrations of 0.096 and 0.42 ng/g ww for *syn*- and *anti*-DP, respectively, but were not statistically different from West-Greenland ringed seal samples. Reported mean concentrations for *syn*- and *anti*-DP in other species were; 0.14 and 0.67 ng/g ww in black guillemot eggs, 0.023 and 0.11 ng/g ww in glaucous gull liver, 0.019 and 0.071 ng/g ww in ringed seal blubber from West-Greenland and 0.021 and 0.055 ng/g ww in polar bear adipose tissue.

94. In another study, covering multiple locations in the Canadian Arctic, DP was only detected sporadically in ringed seals (Houde et al., 2017). Detectable levels of *syn*- and *anti*-DP ranged between 0.04 and 0.41 ng/g lw and 0.04-6.3 ng/g lw, respectively.

95. Minke whales and beluga whales from the St. Lawrence Estuary as well as beluga from the Canadian Arctic have also been studied (Simond et al., 2017). In 2013, the mean DP concentration for these three populations was 0.31 (minke whales, St. Lawrence Estuary), 0.44 (beluga, St. Lawrence Estuary) and 1.28 ng/g lw (beluga, Arctic). The highest concentration of 1.28 ng/g lw in the Arctic beluga population was different from the generally observed contaminant distribution, which usually showed higher concentrations at more southern locations, and could not be explained. Concentrations in beluga from St. Lawrence estuary and the Canadian Arctic were roughly the same order of magnitude as observed in two killer whales from Greenland with quantifiable levels of *anti*-DP (Vorkamp et al., 2019).

96. Research on four avian species on King George Island showed that DP was present in Gentoo penguin, Adelie penguin, South Polar skua and Brown skua (Kim et al., 2015). Reported concentrations of DP were 0.250−0.329 ng/g lw in the penguin tissues and 2.12−11.1 ng/g-lw in the skua tissues. Dec 603 and 604 were not detected, whereas Dec 602 was detected in the highest concentrations in all samples, followed by *anti*-DP and *syn*-DP. In another Antarctic study, Gao et al. (2018), reported mean DP concentrations of 0.233 ± 0.089, 0.201 ± 0.086 and 0.15 ± 0.147 ng/g dw in soil from three different locations (i.e. a coastal area, an inland area and Ardely Island) on the Fildes Peninsula. Reported levels in lichen from the same locations were 0.449 ± 0.213, 0.337 ± 0.172 and 1.513 ± 1.376 pg/g dw, respectively. DP was detected in all samples. Kim et al. (2018) reported the presence of DP in lichen and mosses sampled from 16 different sampling sites spread across the South-Shetland Islands, King George Island and Anvers Island in Antarctica. Similar to Gao et al. (2018) and based on geographical differences in DP levels and f_{anti} values, the authors propose that LRET, human activities, melting glacier water and biological activities, e.g. from penguins, are possible sources of DP in the area (Kim et al., 2018). In yet another study by Kim et al, DP was detected in limpet (0.398–11.1 ng/g lw), antarctic cods (0.0–0.548 ng/g lw), amphipods (0.227–8.71 ng/g lw), antarctic icefish (2.50 ng/g lw), gentoo penguins (0.009–11.8 ng/g lw), chinstrap penguins (0.0–3.00 ng/g lw), south polar skua (0.731 ng/g lipid), and kelp gull (0.028 ng/g lw) collected from the Barton Peninsula and Maxwell Bay, King George Island (Kim et al., 2021). Compared with the results from King George Island in 2008/09 (Kim et al., 2015), the DP levels in the penguin samples had slightly increased, although the sample size of the previous study was small. Aznar-Alemany et al. (2019) investigated DP and other dechloranes in dead seals from the South Shetland Islands on the Antarctic Peninsula. Only Dec 602 and *anti*-DP were found. *Anti*-DP was detected in adipose tissue from southern elephant seals and in Antarctic fur seals at average concentrations of 0.60 ng/g lw and 0.16±0.16 ng/g lw, respectively, but were below the limit of quantification in muscle, brain and fur. Dec 602 appeared in higher concentrations than *anti*-DP, especially in adipose tissue (Aznar-Alemany et al., 2019).

97. Several studies have detected 1,3- or 1,5-DPMA in environmental samples such as sediment and fish (e.g. Sverko et al., 2010; Tomy et al., 2013; Sühring et al., 2014; Wang et al., 2015; Wolschke et al., 2015; Rjabova et al., 2016). It is possible that failure to use a non-destructive clean-up procedure during sample preparation could lead to under-reporting of this substance (Rjabova et al., 2016). In some cases, the concentrations of DPMA-isomers were greater than the total DP concentration in the same samples, suggesting the possibilities of greater bioaccumulation of 1,3-DPMA due to its smaller molecular size (See Table 11 in UNEP/POPS/POPRC.17/INF/xx). 1,3-DPMA (but not 1,5-DPMA) was detected in lake trout at an average concentration of 34 ± 43 µg/kg lw, which was around ten-fold higher than concentrations of DP in the same samples reported by Tomy et al. (2007). In peregrine falcon eggs harvested in Spain and Canada, DPMA was detected in 24 out of 25 eggs, with concentrations ranging from 1.7 to 469 ng/g lw and 1.2 to 1660 ng/g lw in peregrine falcon eggs from Spain and Canada, respectively (Guerra et al., 2011). The

corresponding DP levels was in the range 0.3 to 3.6 and 7.5 to 209 ng/g lw, respectively. DPMA has also been detected in tissues (except blood) of brown skua from King George Island, Antarctic indicating potential LRET. The detection frequencies were only 21%, 11%, and 7.1% for DPMA, Dec-604 and *anti*-DP with the highest levels of 136, 165, and 8.9 pg/g dw, respectively (Wolschke et al., 2015).

98. Spatial trends of DP in remote regions were reported in five separate studies from the Arctic (Möller et al., 2010; 2011; Vorkamp et al., 2015; 2019a,b). As discussed in section 2.4.4 above, Möller et al. (2010;2011) observed decreasing concentrations of DP in air, but not in water, with increasing latitude. Biota data from Greenland show either fairly uniform levels or no spatial trends; Vorkamp et al. (2015) found that DP levels were not statistically different in ringed seal samples from East- and West-Greenland, while the Vorkamp et al. (2019a) found a fairly uniform occurrence of DP in glaucous gull and ringed seal samples from different locations.

99. Temporal trend data for DP are also equivocal. Air samples (gas and particle phase) from Harbin, China indicated significant and increasing trends for both *syn*- and *anti*-DP in the period 2008 to 2013 (Li et al., 2016). Reported doubling times were 2.3 and 1.8 years, respectively. The time it took for *syn*- and *anti*-DP concentrations to double was significantly different, but the doubling times of both DP isomers were shorter than reported for 2-ethylhexyl 2,3,4,5-tetrabromobenzoate and bis(2-ethylhexyl) tetrabromophthalate, suggesting a more rapid increase for DP than the other FRs in the study. Air parcel backward trajectories and source contribution suggested a strong input from local sources (Li et al., 2016).

100. In a study from North America, DP concentrations in air (gas and particle phase) increased from 2005 to 2013, with doubling times of 4–6 years at three sites (Chicago, Sleeping Bear Dunes and Eagle Harbor) in the US Great Lakes area. At the two sites closest to a known production site for DP, Cleveland and Sturgeon Point, levels were stable and unchanged in the same time period (Liu et al., 2016a). A study from Canadian sites in the Great Lake Basin (Burnt Island, Egbert and Point Petre) reported trends in levels of *syn*- and *anti*-DP in air from 2005 to 2014 at two of the sites, Burnt Island and Point Petre (Shunthirasingham et al., 2018). The concentration of *anti*-DP in air was observed to decrease over time, with reported half-lives of 3.4 years at Burnt Island and 13 years at Point Petre. The declines leveled off after 2010. *Syn*-DP levels increased over time at Burnt Island with a doubling time of 7.6 years but was similar to *anti*-DP found to decrease at Point Petre with a half-life of 6.6 years. In yet another study from the Great Lakes area, Olukule et al. (2018) report that *syn*-, *anti*-, or total DPconcentrations in the air (vapor + particle phase) and in precipitation have not changed over the period 2005-2015 at four sites near the Great Lakes; Sturgeon Point, Point Petre, Cleveland and Chicago. Salamova and Hites (2011) assessed the temporal trends of DP at US Great Lakes sites over the time period 2005–2009 and observed an overall (vapor, particle, and precipitation phases combined) doubling time of 9.5 years for *anti*-DP and no significant trends for *syn*-DP and total DP. The increasing levels of DP observed in Chicago, Sleeping Bear Dunes and Eagle Harbor by Liu et al. (2016a) and for *syn*-DP at Burnt Island by Shunthirasingham et al. (2018) are according to the authors consistent with the results presented by Salamova and Hites (2011). The results reported by Liu et al. (2016a), suggest a more rapid doubling time in DP levels in the atmosphere around the Great Lakes from 2005-2013 than reported previously by Salamova and Hites (2011) for the period 2005-2009. However, although the monitoring sites in the Great Lakes area are in the same region, they are not identical.

101. When compared to the global data, levels in air in Chicago and at Sturgeon Point are at the higher end of the range and are similar to the average DP level (3.5 pg/m^3) from Taihu, China (Qiu et al., 2010). The increasing levels of DP observed in the Great Lakes region which are reported in some studies (Li et al., 2016; Liu et al., 2016a; Olunkunle et al., 2018; Salamova and Hites, 2011; Shunthirasingham et al., 2018), and in a typical urban city in China (Li et al., 2016) may indicate that DP is being increasingly used. Using a multiple linear regression model of DP concentrations to isolate the variabilities due to sampling date and population near the sampling site, Olukunle et al. (2018) showed that DP concentrations in precipitation, gas and particle phase air samples from the Great Lakes were changing as a function of sampling date. On the other hand, and as described earlier, decreases of DP were observed in suspended sediments, a sediment core, and lake trout samples of Lake Ontario, following peaks in the 1980s (Shen et al., 2011). The reasons for the different time trends in air, precipitation, sediment and fish in the Great Lakes area are not clear, but as indicated by Vorkamp et al. (2018) it is possible that the different matrices studied (and also the study sites) reflect different emission sources.

102. There is also some information on time trends in biota. A recent study by Vorkamp et al. (2018), the only study available on temporal trends of DP in the Arctic, reported DP concentrations in the range 0.984 to 37.9 ng/g lw during the period 1986 to 2014 in peregrine falcon eggs from South Greenland. A non-significant increasing trend (annual change 3.3%) was observed (Vorkamp et al., 2018). In belugas from the St. Lawrence Estuary, DP increased from 1997 to about 2000 and decreased subsequently, possibly with a second peak around 2010 (Simond et al., 2017). No significant change in DP concentrations was found in eggs of white storks or black kites collected in 1999, 2003, 2011 and 2013 from the Doñana Natural Space in Spain (Báron et al., 2015). In a study investigating FRs in Indo-Pacific humpback dolphins and finless porpoises from the South China Sea, Zhu et al. (2014) found a positive temporal trend in the ratio of DP to BDE-209 between 2003 and 2011 in dolphin samples and between 2003 and 2012 in porpoise samples. The finding was by the authors attributed to an increasing use of PBDE alternatives in the South China region following the restriction of the production and use of PBDE commercial mixtures. A study reporting levels and trends of different FRs in the German and polar environments from 2005 to 2015/2016 found declining trends for DP in some

of the investigated biological matrices from Germany (i.e. tree leaves, tree shoots, herring gull eggs) and increasing trends in bream (filet) and blue mussels (soft body) from Germany (Dreyer et al., 2019). In the same period, a small declining trend was also observed in riverine suspended matter. However, the standard deviation in this sample set was quite large. As noticed by the authors, herring gulls feed in the marine environment, e.g. on blue mussels, but may also have a terrestrial diet. Thus, one possibility is that the seemingly contrasting time trends reported by Dreyer et al. (2019) reflect differences in DP emission and deposition between terrestrial and aquatic environments.

103. Rauert et al. (2018) report air monitoring data from 48 global sites across all five UN regions that were collected using polyurethane foam passive air samplers. The study generally showed low detection for all FRs at background sites. PBDEs showed higher mean concentrations and detection frequencies than novel FRs (Rauert et al., 2018). *Syn*-DP was below the detection limit at all 48 sites and *anti*-DP at all sites except Paris, France, where high levels of DP were reported in 2014 (19 pg/m^3 in the first and fourth of the year, 7.5 pg/m^3 in the second quarter two and 116 pg/m^3 in third quarter). When compared to monitoring data from 2005 and 2008 to 2009, respectively, Paris had elevated concentrations of novel FRs and reduced PBDE levels. *Anti*-DP and the FRs β-tetrabromoethylcyclohexane and bis(2-ethyl-1-hexyl) tetrabromophthalate were the dominating novel FRs. Rauert et al. (2018) propose that "the increasing concentrations of the replacement FRs and the reduced levels of the PBDEs may be indicating the shift away from the use of PBDEs to alternative FRs in this urban area".

2.3.2 Human exposure

104. Humans may be exposed to DP from indoor dust, food, indoor and ambient air, water, soil, sediment and breast milk. DP has been detected in indoor air and domestic dust samples from several countries indicating exposure from products and manufactured items containing DP (Shoeib et al., 2012; Johnson et al., 2013; Wong et al., 2017; See Table 12 in UNEP/POPS/POPRC.17/INF/xx). Higher levels of DP in air from urban compared to rural environments have been observed (Ren et al., 2008; Chen et al., 2011). A regular distribution pattern of DP in dust with particle size was observed with stronger enrichment on fine particles than coarse particles, which indicates potential human exposure (Cao et al., 2014). As reported in section 2.1.3 above, in indoor dust in Beijing, China, highest DP levels were found in pooled samples from classrooms (231 ng/g), which was higher than in house dust from Guanzhou, China (mean 18.9 ng/g) (Cao et al., 2014). In indoor dust from different microenvironments (i.e. home, workplace, car) in Greater Cairo, Egypt, *syn*-and *anti*- DP were detected in 100% of car dust samples reaching 5 and 6 ng/g, respectively (Hassan and Shoeib, 2015). In the Republic of Korea, DP was detected in almost all samples of indoor dust from homes (0.30 – 530 ng/g; median 9.1 ng/g), offices (3.7 – 100 ng/g; median 35 ng/g), and daycare centers (6.2 – 56 ng/g; median 12/ng/g) (Lee et al., 2020). The estimated daily intake (EDI) via dust ingestion for toddlers and adults in the worst scenario (95th percentile) were 0.20 and 0.02 ng/kg bw/day for ΣDP (Lee et al., 2020). Air sampling in 15 homes in India found DP levels ranging from 0.2 to 5.43 pg/m^3 (median 2.81 pg/m^3) and 0.52–62.7 pg/m^3 (median 1.62 pg/m^3) for urban and suburban sites, respectively (Yadav et al., 2020). Cequier et al. (2015) found no correlation between DP levels in indoor air, dust or serum levels for 46 Norwegian women. However, both levels in air, dust and serum were low in these samples. Mean DP levels in serum were 2.6 ng/g lw and dust samples from the corresponding homes had mean *syn*- and *anti*-DP levels of 9 and 19 ng/g, respectively while the air concentrations were even lower (Cequier et al., 2014). High levels of DP were detected in ventilation dust from aircraft cabins produced between 1986 and 2008 with a range of 31 to 9600 ng/g for *anti*-DP (Allen et al., 2013).

105. Ghelli et al. (2021) present a new review of DP and related compounds in food, including fish and seafood, dairy products, egg and egg products and meat and meat products, and in animal fat and vegetable oil and other foods. From the analysis of the various articles, it emerges that DP quantified with values that would indicate slightly higher contamination levels in fish and seafood category. In food products from Belgium, DP was found in 5% of the 1289 samples (firm cheese, quail eggs and pork, mean DP levels were 339, 637 and 331 pg/g ww, respectively) (Poma et al., 2018). In 175 samples of 35 different food products from the retail market in the Republic of Korea, the mean DP was in the range from ND to170 pg/g ww, and *syn*- and *anti*-DP were detected in 83.4 and 79.4% of the food products, respectively (Kim et al., 2014). Daily intake of DP was estimated to be 11.2 × 10^3 pg/day, which was 3 orders of magnitude higher than for other dechloranes (Dec 602, 603, 604 and Mirex). The highest level of DP was found in bovine liver DP 170 pg/g ww followed by spanish mackerel, oysters and dried anchovy with 135, 81 and 78 pg/g ww, respectively. However, in this study, DP was detected in all food products except onion. This result shows the wide-spread exposure of DP in the environment of Korea (Kim et al., 2014). In a study from Lebanon of 58 food samples representing fatty food groups, the estimated daily dietary intake for the adult population had lower and upper boundaries of 1344 and 1718 pg/day for DP (Malak et al., 2019). In a market basket study of 123 food samples from Japan, DP was detected in Groups III (sugar and confectionary), V (legumes and their products e.g. soy bean, tofu, fried bean curd, natto, chickpea), X (fish, shellfish and their products), and XI (meat and eggs) at concentrations of 3.3, 2.8, 1.9, and 1.5 pg /g ww, respectively (Kakimoto et al., 2014). The atmospheric concentration of DP in Osaka city was 11 pg/m^3. The estimated daily intake of DP for an average Asian adult via inhalation and diet was (750 pg/day) approximately one percent of that estimated for ΣPBDE (62 ng/ day) (Kakimoto et al., 2014). DP was frequently detected in a study of US and Chinese baby food samples, including infant formula, cereal, and puree. The highest levels were found at 4,000 pg/g and 430 pg/g, in a baby formula sample from China and a cereal sample from the U.S., respectively. Median concentrations of total DP in U.S. baby food samples were 16 pg/g fresh weight in formula

(n=12); 7.8 pg/g fresh weight in cereal (n=15); and 7.3 pg/g fresh weight in puree (n=8). Median concentrations of total DP in Chinese baby food samples were 20 pg/g fresh weight in formula (n=9); 5.9 pg/g fresh weight in cereal (n=12); and 5.8 pg/g fresh weight in puree (n=6) (Liu et al., 2014.)

106. In a study comparing human exposure at a production facility and an e-waste recycling facility in Southern China, high levels of DP were found in food products produced in proximity to the e-waste recycling site and production facility, where vegetables contained from 305 ng/g ww (paikchoi cabbage) to 2720 ng/g ww (green onion) DP (Wang et al., 2013a). Grains (common wheat, maize and soybeans) had levels of 1370, 760 and 498 ng/g ww, respectively, and common carp and other local fish species had levels from 56.8 to 1110 ng/g ww (Wang et al., 2013a). Under the assumptions that consumed food was produced locally estimated dietary intake dose accounted for more than 99% and 93% of the daily intake to people from the production site and e-waste recycling site, respectively (Wang et al., 2013a). For the production site, the estimated dietary exposure dose was 1.1×10^{-2} mg/kg/d for workers, residents and children and the estimated dietary exposure dose was lower for the e-waste recycling site at 3.0×10^{-5} mg/kg/d (Wang et al., 2013a). Total estimated exposure dose (dietary, dermal, inhalation) in people from the e-waste recycling site was 0.03×10^{-3} mg/kg/d, which is 300 times lower than at the manufacturing site (Wang et al., 2013a). In another study from an e-waste recycling area in China, free-range chicken eggs harvested in 2010, 2013 and 2016 contained high levels of FRs, including DP. No temporal trend was observed for DP levels. The highest levels of DP were observed in eggs from 2016, mean and SD was 5413 ± 1978 ng/g lw (Huang et al., 2018).

107. DP has been detected in human serum from several countries. In a study from France mean Σ5 PBDEs (BDE-47, -99, -100, -153 and -154) levels (4.32 ± 2.99 ng/g lw) were in the range typical of Western Europe levels, but lower than the mean Σ5 dechlorane compounds (Dechlorane 603>DP >Mirex>Dechlorane 602 > Chlordene Plus) levels (6.24 ± 4.16 ng/g lw) (Brasseur et al., 2014). Levels of DP observed were lower in serum from Canada, France, Korea and Norway compared to occupationally exposed people in China with median of 2.39, 1.20, 0.73, 1.3 and 42.6 ng/g lw, respectively (Zhou et al., 2014; Brasseur et al., 2014; Kim et al., 2016; Cequier et al., 2015; Ren et al., 2009). In a study from China analyzing pooled human serum samples from different age groups, including 305 subjects from the south coast of Laizhou Bay, Shandong Province, China, there was no relationship between DP levels and age. However, the younger volunteers, age 20 to 29 years, had the highest serum concentrations with mean DP around 7 ng/g lw compared to mean DP for all groups 3.6 ng/g lw (Wang et al., 2014). A time trend of Dechlorane compounds was investigated in seven pooled serum samples from Korean adults to cover the period from 2006 to 2013. No distinct trend was observed for Dechlorane compounds (Kim et al., 2016). In plasma samples of German young adults from the environmental specimen bank collected from 1995 to 2017, no significant trend was observed for DP (Fromme et al., 2020).

108. High levels of DP have been observed from occupational exposure at a DP manufacturing plant compared to nearby areas in eastern China. The DP concentrations ranged from 89.8 to 2958 ng/g lw in whole blood and 4.08 to 2159 ng/g dw in hair. For the workers directly engaged in the DP manufacturing process, their DP levels were significantly higher than in most individuals of the two control groups (Zhang et al., 2013). In another study, DP serum levels in 70 occupationally exposed workers from an e-waste recycling workshops and control group of 3 residents of an urban area in South China were measured. The DP levels were significantly higher in the workers (22–2200 ng/g with median of 150 ng/g lw) than in the urban residents (2.7–91 ng/g with median of 4.6 ng/g lw). The DP concentrations in females were found to be associated with their age but this relationship was not found for males (Yan et al., 2012). Hair samples from female workers at e-waste dismantling facilities in South China contained median and mean concentration of 23.5 ng/g and 61.3 ng/g DP respectively (Qiao et al.,2019). A study of 15 occupationally exposed workers at e-waste recycling sites in Bangladesh found a median DP exposure of 2.3 ng/h/silicone wristband, approximately 130 times higher than non-occupationally exposed US residents (Wang et al., 2020b).

109. In humans, DP has been detected in cord serum (Ben et al., 2014) and in human milk (Siddique et al., 2012) showing transfer to offspring at different developmental stages. Maternal transfer of bioaccumulative substances *in utero* represents a potential risk to embryonic development and may represent the largest source of FRs input to offspring during the first few years of life. In 450 human milk samples from Norway, the Netherlands and Slovakia, DP had detection frequencies in the range of 3 to 9% and 20 to 26% for *syn*- and *anti*-DP, respectively. The concentrations of *syn*-DP ranged between 0.05 and 2.85 ng/g lw and of *anti*-DP between 0.004 and 1.60 ng/g lw (Čechová et al., 2017). Higher detection frequency was observed for 87 human milk samples from Canada (76 and 91% for *syn*- and *anti*-DP, respectively), with mean *syn*- and *anti*-DP concentrations of 0.27 and 0.7 ng/g lw (Siddique et al., 2012). In another Canadian study of 105 milk and 102 maternal serum samples, *syn*- and *anti*-DP detection frequencies were 40 and 50% for milk samples and 77 and 78% for maternal serum samples, respectively (Zhou et al., 2014). However, concentrations of DP in milk samples were lower than in the previous study from Canada (i.e. Siddique et al., 2012). Total DP in maternal serum was 2.37 ng/g lw, which was lower than previously found in two Chinese studies (Ben et al., 2013; Ren et al., 2009), where the median value of total DP in maternal serum samples of mothers who were not exposed to e-waste recycling activities was 13.7 ng /g lw in one study (Ren et al., 2009), and 4.0 ng/ g lw in the other study (Ben et al., 2013). The total DP levels in milk samples were also higher in the Chinese study with a median of 2.19 ng/g lw (Ben et al., 2013). *Syn*-DP and *anti*-DP were detected in all milk (n = 44) and serum samples (n = 45) (Ben et al., 2013). The Cl_{11}-DP detection frequency in breast milk and serum was 45% and 84%, respectively, and DP levels in blood and milk from residents who had been living in Taizhou for >20 years (R(20) group) were significantly higher

than those who had lived in that city for <3 years (R(3) group) (Ben et al., 2013). For overview see Table 13 in UNEP/POPS/POPRC.17/INF/xx.

110. In another publication from the same study of 72 residents of the e-waste recycling area of Wenling, China, DP was detected in placenta and cord serum indicating prenatal exposure of DP across the placenta (Ben et al., 2014). In contrast to human milk samples which have been reported to have a f_{anti} ratio close to that of commercial DP mixtures (0.6–0.8) (Zhou et al., 2014; Siddique et al., 2012; Ben et al., 2013), the concentration ratio between the cord serum and maternal serum was estimated to be 0.45 for *syn*- and 0.35 for *anti*-DP, suggesting the placenta partially limited DP transfer with a greater extent for *anti*-DP (Ben et al., 2014).

2.4 Hazard assessment for endpoint of concern

111. Available studies show that DP has the potential to elicit physiological changes or toxic effects in different organisms. Oxidative stress has been observed after exposure to DP in fish (Chen et al., 2017; Hang et al., 2013; Kang et al., 2016), marine bivalves (Barón et al., 2016, Gagne et al., 2017), earthworm (Zhang et al., 2014; Yang et al., 2016b), birds (Li et al., 2013a) and mice (Wu et al., 2012). The organism may be able to handle oxidative stress/damage by activating/ upregulating antioxidant defence and repair mechanisms. However, reactive oxygen species are important signalling molecules that could trigger unwanted effects by activating signalling cascades regulating cell death and cell survival and by causing damage to DNA, protein, telomeres, cells, tissues etc. Oxidative stress is linked to aging and disease. A positive association was observed between DNA-damage and morphological deformities in carp embryoes exposed to DP (Li et al., 2020a). As further described below, reported effects include oxidative stress to most tested organisms, neurodevelopmental toxicity in zebrafish and histochemical alterations in juvenile carp brain (Chen et al., 2017; Li et al., 2019b). In addition, potential effects to the sex and thyroid hormone pathway in zebrafish (Kang et al., 2016) and humans (Ben et al., 2014; Guo et al. 2018; 2019). Alteration in gene expression indicating immune modulating effects in carp has been observed (Li et al., 2019b). In mammalian and human cells, DP has been shown to activate peroxisome proliferator-activated receptor gamma and inhibit insulin signalling *in vitro*. DP has been shown to impair liver function in mice (Wu et al., 2012;2013) and at low doses alter insulin secretion in a non-monotonic way and promote adipose tissue dysfunction (Peshdary et al., 2019; 2020). However, it should be mentioned that some of the test protocols have not been validated and that the database on toxicity is limited, the relevance of the concentrations used could be questioned for some studies and long-term studies are largely lacking.

2.4.1 Toxicity to aquatic organisms

112. DP has very low water solubility (< 44 ng/L, see Table 2) and will partition to particles, sediment and biota in the aquatic environment. Zhen et al., (2018) showed that total DP levels in natural water could be higher than just the dissolved concentration due to DP adsorbed to particulate matter, e.g. in Laizhou Bay water, median DP was 11.7 ng/L for sum filtered and particulate, the range of DP in filtered and particulate water samples were from not detected (nd) to 10 ng/L and 0.890 to 346 ng/L, respectively (Zhen et al., 2018). For DP, results from most available empirical aquatic toxicity studies are associated with some uncertainty, mainly because test concentrations exceeded the DP water solubility limit. However, all studies use solvent as a vehicle which increase the solubility of DP. Chen et al., (2019) observed a measured water concentration of 55 µg/L with a nominal concentration of 60 µg/L using DMSO as solvent. Detailed information of conditions in the aquatic toxicity studies and measured concentrations in water are provided in Table 14 and 10, respectively, in UNEP/POPS/POPRC.17/INF/xx.

113. Chronic exposure studies for benthic and sediment dwelling organisms are lacking. Recognizing the lack of effects data for DP in important compartments, such as soil and sediment, the use of analogue data can be considered as an approach to addressing key gaps in the hazard profile. In selecting analogues, factors such as structure, reactivity, metabolism and bioavailability can be considered. The Canadian screening assessment used toxicity data on the DP analogue chlordane, due to its structural and functional similarities to DP (OECD QSAR Toolbox, 2016), to fill the data-gap as a worst-case-scenario for sediment dwelling organisms and estimated a predicted no effect concentration (PNEC) of 0.0129 mg/kg dw for DP (Canada, 2019a).

114. Two studies on aquatic algae have investigated effects on photosynthetic and antioxidant endpoints at low doses of DP (Gong et al., 2013; 2018). The freshwater algae *P. subcapitata* was exposed to nominal DP concentrations of 13.51, 135.1 and 1 351 ng/L for up to 72 h (Gong et al., 2013). No significant effects were observed on esterase activity or reactive oxygen species (ROS) levels at 72 h, although transient increases were observed at earlier timepoints (Gong et al., 2013). At the highest dose (135.1 and 1 351 ng/L), average chlorophyll a content was significantly increased at 48 and 72 h (Gong et al., 2013). No measurement of growth was made, so the significance of the chlorophyll a content increase is not known. The authors concluded that DP showed low toxicity and had a marginal effect on *P. subcapitata*. In juvenile marine green macroalgae *U. pertusa*, the uptake of DP affected physiological responses of photosynthesis and caused oxidative stress (Gong et al., 2018). Following a 21-day exposure to 10^{-8} mol DP/L and 21-day depuration, *anti*-DP was prone to bioaccumulate. Physiological effects were observed after 14 days exposure to DP doses of 10^{-8} to 10^{-6} mol/L (equal to ~6.53 to 653 µg/L). Antioxidant enzyme activities (superoxide dismutase (SOD) and catalase), oxidative damage to lipids (malondialdehyde, MDA) and chlorophyll fluorescence parameters were affected in a dose

and time dependent manner after 1, 7 and 14 days of DP exposure. The effect on photosynthesis and SOD were significant different from control at all concentrations tested, while effects on catalase and MDA were significant for the two highest doses. The authors concluded that DP affects photosynthesis in the marine algae, as observed by a low rate of light energy utilization, and that this effect may be ascribed to oxidative damage caused by the uptake of DP (Gong et al., 2018). Marine micro and macroalgae constitute an important part of the marine food web as being primary producers and food source for a number of species. In addition, macroalgal forests provide habitat for many species. Growth was not measured in the test, so the significance of the observed effects is not known. No acute toxicity was observed in the freshwater protozoan, *T. thermophila*, but significant increase in DNA-damage was observed at DP concentrations from 300 to 1500 µg/L after 30 minutes exposure (Dou et al., 2015).

115. Mediterranean mussels were exposed to DP concentrations of 0, 5.6, 56, and 100 µg/L for 6 days by spiked agarose. Concentrations measured in water immediately after dosing were 0, 0.4 ± 0.3, 0.3 ± 0.2 and 0.7 ± 0.5 µg/L, and measured levels in mussel after 23 h exposure were 4.7±3.1, 8.8±2.1 and 21±9.1 µg/mussel, respectively (Barón et al., 2016). DNA strand-breaks were observed in hemocytes at all doses. No clear dose-response was observed, although all doses were significantly different from the negative control; 13, 23 and 18%, respectively. Increased micronuclei formation was observed only for highest dose (100 µg/L) tested (Barón et al., 2016). In a more long-term study, fasting blue mussels were exposed to nominal concentrations of 0, 0.001, 0.01, 0.1 and 1.0 µg DP/L for 29 days, which resulted in ΣDP tissue concentrations of 0, 0.12, 0.98, 7.26 and 57.8 ng/g ww, respectively. Following DP exposure, no histopathological lesions were found in gonads, and no change in hemocyte DNA strand breakage, phagocytosis rate, and viability was observed. Gills were identified as the most responsive tissue. Lipid peroxidation in gills was found to increase by 82% and 67% compared to control at the 0.01 and 1.0 µg DP/L dose, respectively (only significant for 0.01 µg/L), while cyclooxygenase activity (COX) was significantly decreased by 44% compared to control at the 1 µg/L dose (Gagné et al., 2017). The effect on COX could have an impact on the control of spawning process and thus reproduction in blue mussels (Matsutani and Nomura, 1987). These two studies indicate that DP exposure cause oxidative stress at low µg/L concentrations.

116. Acute toxicity fish studies reported (HPV summaries and referred to by ECHA (2017b) were not considered reliable. However, no mortality was observed in a bioaccumulation study of the freshwater fish species Lepomis macrochirus exposed to 0.1 mg/L DP (using acetone as solvent at 0.001%) for 30 days (ECHA 2017b), implying a NOEC > 0.102 mg/L, 30 d.

117. Four studies with embryo/larval zebrafish with waterborne DP exposure have been identified (Hang et al., 2013; Noyes et al., 2015; Chen et al., 2017; Kang et al., 2016). No significant adverse effects on hatchability, survival or malformation were seen in the Noyes et al. (2015), Chen et al. (2017) and Kang et al. (2016) studies following exposure to DP at a nominal concentration of up to 4.18 mg/L, when zebrafish embryos were exposed in the blastula or early gastrula phase (4-6 hpf). Kang et al. (2016) exposed zebrafish embryos to nominal DP concentrations of 0.4, 0.8 or 1.6 mg/L from 4 to 144 hours post fertilization (hpf). Measured water concentrations of DP were 140, 248, 267 µg/L DP at 0 h. In contrast, Hang et al. (2013), observed significant time- and dose-dependent increases in malformations (spine side curve, cardiac edema, tail deformation) in zebrafish embryos exposed to DP at nominal concentrations of 0.037 to 0.37 mg/L from 8 hpf (late gastrula phase) up to 7 days (168 hpf). All solutions were changed once a day and 20 embryos were used to evaluate daily for mortality and malformations. However, it was not possible to judge the reliability of the data due to limited study documentation.

118. In a study of 300 carp embryos exposed from 3 to 120 hpf to DP nominal concentrations of 30, 60, and 120 µg/L (using 0.01% DMSO as solvent) significant time- and dose-dependent increase in mortality rate, delayed hatching time, reduced hatching rate and decreased body length were observed. At 120 hpf significant dose-dependent increase in DNA damage compared to control was observed for all doses (Li et al. 2020a). In the 120 µg/L group, the mortality rate increased to more than 50 % between 48 and 72 hpf and continued to increase to between 70 and 80% at 120 hpf. From 96 hpf mortality rate was approximatly 30% for 60 µg/L group (in control it was below 10%). Morphological deformities were significantly increased compare to control at 120 hpf at all doses (Li et al. 2020a). DNA damage measured as % tail DNA at 24 hpf increased significantly from below 10 to approximatly 30, 45 and 55 % respectively for solvent control, 30, 60 and 120 µg/L group, respectively. A positive association was observed between DNA-damage and morphological deformities. Observations were supported by alteration in gene-expression for key morphology genes and antioxidant markers (Li et al., 2020a).

119. Short-term exposure studies with embryo/larval zebrafish suggest that DP can induce oxidative stress and neurobehavioral changes (Hang et al., 2013; Noyes et al., 2015; Chen et al., 2017). Noyes et al. (2015) observed a significant hyperactive response to dark-stimuli-activation compared to controls at 120 hpf after exposure to nominal concentration of 6.4 µM DP (4.18 mg/L). The level of response was much lower upon exposure to DP than phosphate ester and chlorinated phosphate ester flame retardants in the same test system. Chen et al., (2017) observed significant increase in spontaneous movements at 24 hpf for all nominal doses (15, 30 and 60 µg/L), dose- and time-reduced distances of swimming after touch-induced stimuli, significant decreased free-swimming activity and reduced swimming speed during each dark and light period. These neurobehavioral changes may be linked to axonal and muscular lesions. DP significantly inhibited primary motor neuron axonal growth and induced cell apoptosis and lesions

in muscle fibres of the tail region of larvae in a dose dependent manner at 96 hpf. Axonal growth-related gene-expression (α1-tubulin and gap43) was significantly increased in the highest dose (60 μg/L), oxidative stress markers such as ROS and MDA as well as messenger ribonucleic acid (mRNA) levels of apoptosis related genes were increased at the two highest doses (30 μg/L and 60 μg/L) (Chen et al., 2017).

120. Co-exposure of zebrafish embryos to nominal concentrations of DP (60 μg/L) and/or 3-methyl phenanthren (3-MP) (5 or 20 μg/L) at 6 to 120 hpf resulted in elevated bioaccumulation of both compounds and synergistic effects on neurobehavioral abnormalities, axonal growth reduction, apoptotic markers in muscle and brain Ca^{2+} homeostasis (Chen et al., 2019). ROS levels in whole fish were significantly elevated compared to control from all exposures at 96 hpf, but no significant alteration was observed for the co-exposure. As in the previous study (Chen et al., 2017), a significant increase in spontaneous movement occurred at 24 hpf with 60 μg/L DP, but co-exposure with 5 or 20 μg/L 3-MP gave a significant reduction compared to control. DP and 3-MP co-exposure showed a synergistic or additive effect in reduction of swimming distances after touch-induced stimuli and free-swimming activity. Furthermore, synergistic or additive effect were observed for reduction in ventral axonal length growth, increased number of apoptotic cells in the tail region and induction of axon related gene expression. Brain Ca^{2+} homeostasis was investigated at 96 hpf, and a significant synergistic increase was observed in brain intracellular Ca^{2+} levels, and a synergistic reduction in Ca^{2+} ATPase activity was observed after co-exposure. Furthermore, a synergistic effect was observed in reduced expression of genes related to Ca^{2+} homeostasis. Relative to DP exposure alone, co-exposure with low and high-dose 3-MP increased the accumulation of DP by 14 and 82%, respectively. The larval body burden of DP (60 μg/L) was 583.2 ± 33.5 ng/g ww while co-exposure with 5 and 20 μg/L 3-MP increased DP levels to 665 ± 33.5 and 1061 ± 85.7 ng/g ww, respectively. Relative to 3-MP alone, co-exposure with DP (60 μg/L) increased 3-MP levels by 45 and 47% for the 3 and 20 μg/L 3-MP groups, respectively (Chen et al., 2019). Elevated bioaccumulation resulting from mixture exposure may represent a significant contribution of the synergistic effects observed in combined chemical exposure.

121. Studies also show that DP can cross the blood-brain barrier in fish (Zhang et al., 2011), and *anti*-DP had a high persistence in the brain compared to the liver (Zhang et al., 2011). Stress responses and histopathology changes of the brain and liver were observed in juvenile common carp (n= 30 per group) exposed to nominal concentrations of 0, 30, 60 or 120 μg/L DP for 1, 15 or 30 days (Li et al., 2019b). Tissue morphology (cell abnormality rate) was significantly changed in the groups exposed for 60 and 120 μg/L for 15 and 30 days. In liver, unclear contours, vacuolisation and nuclear lysis were observed, and in the brain, abnormalities were observed in the granular layers, nuclear cell structures, disordered arrangement, microthrombic red blood cells, increased numbers of glia cells and nodulations. Liver SOD, glutathione (GSH) and MDA activity levels decreased with increasing dose and exposure time while the opposite pattern was observed in the brain (Li et al., 2019b). DP also altered expression of immune regulating genes in the brain and liver. CYP1B1 gene expression was significantly upregulated at all timepoints and concentrations in both the liver and brain. In addition, CYP2B and CYP3A1 and the apoptosis related factors bax and bcl-2 were altered in liver. Both the pro-inflamatoric cytokines IL-6 and IL-1β and the anti-inflamatoric cytokine IL-10 were significantly increased compared to control at all doses at 30 days in liver, while in the brain, IL-6 was significantly upregulated with the two highest concentrations, IL-1β with highest concentration only, and IL-10 with 30 μg/L (Li et al., 2019b). This is the first study indicating potential immune modulating effects of DP. These results indicate that DP exposure perturbs metabolism in the liver and brain, inhibits antioxidant enzyme activity, increases lipidperoxidation, promotes inflammation and induces cell apoptosis in juvenile carp (Li et al., 2019b).

122. In another study in juvenile common carp exposed to nominal concentrations of 0, 30, 60 or 120 μg/L DPfor 4 weeks, DP was shown to negatively affect intestinal barrier, increase the relative aboundance of pathogenic bacteria and change the balance of intestinal flora (Li et al., 2020b). mRNA levels of genes involved in tight binding protein genes in the gut of common carp was significant reduced by all exposure concentrations and histomorphology indicated that DP evidently shortened the intestinal folds and damaged the intestinal epithelium layer (Li et al., 2020b). Effects on growth or mortality as a result of DP exposure were not reported for these two studies (Li et al., 2019b or 2020b), and doses were chosen based on results from Chen et al., (2017) and Kang et al.,(2016) where no developmental abnormalities were observed for doses up to 267 μg/L.

123. In juvenile Chinese sturgeon treated with DP at doses of 1, 10, and 100 mg/kg ww for 14 days via a single intraperitoneal dose, liver proteomics indicate that DP had effects on the generalized stress response, small G-protein signal cascades, Ca^{2+} signalling pathway and metabolic process, and may induce apoptosis in the liver (Liang et al., 2014).

124. DP exposure induces oxidative stress in adult zebrafish tissues (Kang et al., 2016; Hang et al., 2013). In zebrafish orally exposed to spiked feed with DP dosed 0.25, 2.5 and 7.5 mg/g bw per day for 7, 14 and 28 days, the highest dose induced apoptosis in intestines, and a significant dose-dependent increase of SOD activity was observed from day 1 to 28 (Hang et al., 2013). Measured concentrations in fish tissue at day 7 were 618.4, 762.8 and 1823.4 ng/g, respectively. The proteomic profile in the liver and brain was significantly altered and identified proteins were related to DNA damage, protein synthesis, immune response, cell apoptosis and cytoskeleton (Hang et al., 2013). In zebrafish

receiving doses of 0, 0.3, 1, or 3 μg/g ww of DP by gavage on day 0 and 2, measured tissue concentrations were 5, 30, 44 and 420 ng/g ww, respectively, on day 6. An increase of catalase activity was observed in the liver, indicating oxidative stress response to the liver on day 6 (Kang et al., 2016). In addition, some indication of endocrine disruption potential of DP was indicated. Transcriptional responses of both thyroid and sex hormone related genes in the brain were altered, suggesting possible thyroid and sex hormone disrupting potentials of DP (Kang et al., 2016). On day 6, significant induction of CYP19b (brain type of aromatase) was observed at all doses at body residual levels that were environmentally relevant. In addition, the brain estrogen receptor, alpha mRNA, was significant elevated at the lowest dose (0.3 μg/g) only. Plasma thyroxine (T4) concentrations was dose-dependent increased but not statistically significant different from control. However, observation was supported by the up-regulation of corticotropin releasing hormone and TSH-β genes in brain at the highest dose (3 μg/g ww), which are both involved in the fine-tuning of the thyroid hormone-pathways in non-mammalian vertebrates (De Groef et al., 2006).

125. Maternal uptake and transfer of DP has been seen in several species of fish (e.g. Wu et al., 2013; Sühring et al., 2015; Peng et al., 2012; Zhang et al., 2011; Zeng et al., 2014) and in frogs (Wu et al., 2018). DP and other FRs were found in developing embryos of female sharks, demonstrating maternal transfer *in utero* (Marler et al., 2018).

2.4.2 Toxicity in terrestrial organisms

126. Oxidative stress responses and neurotoxicity have been observed in earthworms (Zhang et al., 2014; Yang et al., 2016b). No acute toxicity or significant change in body weight were observed in earthworms exposed to DP at nominal doses of 0.1, 0.5, 6.25 and 12.5 mg/kg for 28 days (Yang et al., 2016) or up to 50 mg/kg for 14 days (Zhang et al., 2014). However, oxidative stress was indicated by alteration in markers such as SOD, MDA, glutathione, glutathione-peroxidase, catalase and 8-hydroxy-2-deoxyguanosine (8OHdG) in tissue as well as tail DNA in comet assays of isolated coelomocytes. Furthermore, acetylcholinesterase (AChE) and cellulase activity of earthworms was significantly reduced even by the low dose indicating potential neurotoxic effects in earthworms (Yang et al., 2016). The overall 28-d no-effect concentration (NOEC) for these biomarker responses is <0.1 mg/kg (ECHA, 2017b). The linkage between these biomarkers and adverse effects is not established. No information regarding long term adverse effects in earthworms is available.

127. No acute toxicity has been observed for DP in birds, but studies indicate that DP is bioavailable and transferred to eggs. In paired samples of eggs and plasma from bald eagles from Canada, DP was more abundant in eggs compared to plasma with geometric mean of 0.28 and 0.43 ng/g ww in eggs from inland and Great Lakes, respectively, versus plasma levels of 0.02 ng/g ww (Guo et al., 2018).

128. Crump et al. (2011) studied concentration-dependent effects of DP using *in vivo* and *in ovo* toxicity approaches in domestic chicken embryonic hepatocytes and chicken embryos. DP was injected into eggs prior to incubation and no overt toxic effects were observed up to the maximum dose of 3 μM in hepatocytes, and up to the highest nominal dose of 500 ng/g/egg for pipping success of the young.

129. Li et al. (2013a) studied the effects of DP on male common quails continuously exposed to commercial DP by gavage for 90 days at dose concentrations ranging from 1 to 100 mg/kg bw/d. Mortality, body and liver weight was not altered in any of the exposure groups. Liver enzyme activity and oxidative stress were measured. The authors reported DP effects on some measures of enzyme activity (e.g., significant decrease of 7-pentoxyresorufin-O-demethylase (PROD) in all exposed groups relative to the control, significant increase in alkoxyresorufin O-dealkylase (ERND) and the antioxidant enzyme catalase in the high-dose exposed groups relative to control). Furthermore, the study found DP was more prone to accumulate in liver (vs. serum, muscle), and *syn*-DP accumulated (vs. *anti*-DP) in the two high-exposure groups.

130. The Canadian screening assessment used toxicity data from Mirex, also based on its structural and functional similarities to DP, as a worst-case scenario to fill the data-gap for terrestrial (plant) organisms, with estimated DP predicted no effect concentrations (PNEC) of 0.075 mg/kg dw (Canada, 2019a).

2.4.3 Human toxicity

131. Available assessments and laboratory studies with mammals suggests that, DP is not carcinogenic, mutagenic or toxic to reproduction (as reviewed in ECHA, 2017b,c; Canada 2019a), although the data are limited. In their alternative assessments for decaBDE, in reference to DP, US EPA 2014 stated "There is potential for carcinogenicity based on analogy to chlordane and decaBDE, the latter for expression of adverse effects in longer term studies." (US EPA, 2014). Other biomarkers of exposure and/or potential adverse effects in mammals have been reported such as oxidative stress, liver impairment and endocrine effects as described below.

132. In terms of *in vitro* genotoxicity, results from Ames assays conducted using *Salmonella typhimurium* strains (TA98, TA100, TA1535, TA 1537 and TA 1538) were negative in the presence or absence of metabolic activation (S9) (Mortelmans and Tanaka 1980, as described in OxyChem 2004b). Result from an *in vitro* mouse lymphoma assay was also negative in the presence and absence of S9 (Jotz and Mitchel 1980, as described in OxyChem 2004b). One *in vivo*

genotoxicity study was identified. Mice were orally administered 0, 500, 2000 or 5000 mg/kg-bw/day of DP via gavage for 10 days (Wu et al. 2012). Liver samples were collected for a comet assay and the genotoxicity result was negative.

133. Acute toxicity studies in experimental animals suggest low concern for acute toxicity via the oral, inhalation and dermal routes of exposure. No adverse health effects were observed in any of the identified repeated-dose oral toxicity studies, testing dose levels up to 5000 mg/kg-bw/day for 10 or 28 days (as summarized in ECHA, 2017b,c; Canada 2019a; OxyChem 2004b and further described below). However, there are some data gaps, for example, there are no long-term studies exceeding 90 days, which might be important given the apparently slow uptake and long elimination half-life of the substance. Therefore, in an alternative assessment of decaBDE, the potential for systemic effects of DP for repeated dose studies was rated as moderate based on the high potential for bioaccumulation and potential for expression of adverse effects in longer term studies by analogy to decaBDE (US EPA, 2014). The dosing vehicles might also limit exposure (e.g. due to the presence of undissolved micro-crystals), such that the high doses might not truly reflect the degree of exposure of the organisms (ECHA, 2017b, c). The 5000 mg/kg bw dose was described as particularly viscous (Brock et al., 2010), and is an extremely high dose (5 times the limit dose of the OECD test guideline).

134. Some of the earlier studies provided in Oxychem (2004b) indicate some liver and ovary effects. However, reliability of these studies is uncertain as discussed in ECHA (2017c) and indicated in the the HPV summaries. The description of the studies is also limited. In a 13-week oral feeding study in rats, absolute and relative liver weight was significantly increased in males only, at 500 mg/kg, but no associated histopathology was observed (Oscarson, 1975, as described in Oxychem (2004b)). In a repeated-dose inhalation toxicity study, rats were exposed to 0, 640 or 1524 mg DP as dust/m^3, 6 h per day, 5 days per week for 28 days (Bishop 1975). Significant increase in absolute and relative liver weight was observed at both doses. Corresponding hepatocytomegaly (swelling of liver cells with signs of cytotoxicity and necrosis) of centrolobulær hepatocytes were observed for male rats at both doses and in some females at the higher dose. Effects on lung tissues were observed as increased number of macrophages in alveoli, and significant increase in absolute lung weight was observed for both doses in females and highest dose for males. A lowest-observed-adverse-effect concentration (LOAEC) of 640 mg/m^3 was identified (Bishop 1975, as described in Oxychem 2004b). In a repeat-dermal study, male and female rabbits were dermally exposed to DP in 3% aqueous methylcellulose at 0, 500 or 2000 mg/kg bw/d, for 5 days per week, for 4 weeks on 20% of the total body surface (Trzyna 1975). None to minimal erythema occurred at the application site. In females, significant dose-related decrease in absolute and relative (to body and brain weight) ovary weight was observed at both doses, and a decrease in absolute and relative liver weight was observed at the highest dose. However, no histopathology effects were observed in any of the organs (Trzyna 1975, as described in OxyChem 2004b).

135. In a combined repeated-dose/reproductive/developmental toxicity screening test, conducted according to OECD guideline 422, Crl:CD (SD) rats were exposed to DP in corn oil at 0, 750, 1500 or 5000 mg/kg-bw/day by oral gavage (Brock et al. 2010). Animals (10/sex/dose) were treated for 28 days. No treatment-related effects were observed on clinical signs of toxicity, body weights, food consumption, neurobehavioral and functional observational battery evaluations. No effects were observed on haematology, urinalysis, coagulation or clinical chemistry parameters. No dose-response related changes in organ weights (heart, liver, testes, ovaries, and thyroid/parathyroid glands) were observed. Mortalities and lesions were observed across all dose groups including controls, which were linked to gavage administration errors as indicated by gavage injury and administration of test substance in the thoracic/pericardial cavity. As a consequence, it is not clear whether dosing was correct. The 5000 mg/kg vehicle in corn oil was described as particularly viscous. The authors identified a no-observable-effect level (NOEL) of 5000 mg/kg-bw/day. However, some effects were observed but considered not treatment relevant by the authors due to lack of dose-response relationship, which could be due to gavage-related errors. A significant increase in absolute and relative ovary weight was observed in the 1500 mg/kg group only, and a significant increase in absolute and relative thyroid/parathyroid weight was observed in females in the 750 mg/kg group only. A non-significant increase in the number of mean early resorption sites (1.8) and postimplantation loss index (13.90%) in the 1500 mg DP/kg bw group compared to control (0.8 and 5.39%, respectively) were observed. Furthermore, pup body weight was significantly increased at doses ≥1500 mg/kg bw. The mean pup weight was increased by 3, 15 and 13% compared to control for the 750, 1500 and 5000 mg/kg groups, however these results were considered unrelated to treatment by the study authors. This increased pup weight may be a result of DP's ability to activate the peroxisome proliferator-activated receptor PPAR-γ as described in the paragraph below (Peshdary et al., 2019) as activation of this signal pathway has been linked to pregnancy/fetal outcomes by strengthening of uterus function (Hewitt et al., 2006; Díaz et al., 2012;), but also effects on the thyroid axis.

136. Wu et al. (2012; 2013) report liver impairments in mice at high-dose exposure. In Wu et al., (2013) mice were fed one dose DP in corn-oil mixed diet at 1, 5 or 25 g/kg bw. After two weeks, histopatology showed inflammatory infiltration, hepatocellular swelling in the mouse liver. Inflamatory infiltration was observed at all DP doses with peak at 5 g/kg bw. Pretreatment with 25 g/kg bw DP gave inflammatory infiltration and hepatocellular svelling (Wu et al., 2013). SOD and CAT activity was significantly reduced by all DP doses, and GSH levels was significantly reduced by highest dose only (Wu et al, 2013). In Wu et al., (2012) following a 10-day oral gavage exposure, oxidative stress and altered metabolism was observed in male mouse livers at all doses (500, 2000 or 5000 mg/kg-

bw per day). Relative liver weight was significantly increased in the 2000 mg/kg group only. Oxidative stress to the liver was shown by significant increase in SOD activity and the oxidative DNA-damage marker 8OHdG at all doses, as well as increased catalase (CAT) activity at 2000 mg/kg. However, no increase in DNA strand-breaks was observed by comet assay. Metabolite analysis indicates that DP alters hepatic carbohydrate-, lipid and fat-, nucleotide- and amino acid metabolism. 18 metabolites were significantly altered with fold change ≥ ±1.5 fold compared to control. Microarray profile of gene expression alterations supported the observations from metabolite analysis and further showed significant alterations in genes involved in signaltransduction, endocrine pathways (insulin-, adipocytokine- and PPAR signalling), signal molecules, Gap junction, p53 signaling- and immunesystem pathways (Wu et al., 2012). Interestingly, Dechlorane 602, which has structural similarities with DP, has been shown to alter immune response by dysregulating T-helper-1 and -2 balance in mice and altering immune and transmitter-related metabolism after exposure at low dose (10 μg/kg bw for 7 days) (Feng et al., 2016; Tao et al., 2018). Furthermore, a medical study by Zhou et al. (2021) found that Dechlorane 602 exacerbates mite-induced airway inflammation in model mice and investigates some of the biochemical pathways involved. Such detailed studies om possible immunomodulating effects have yet to be performed for DP.

137. In a 90-day oral study with rats exposed to DP at 0, 1, 10 or 100 mg/kg/d, DP preferentially accumulated in liver with *syn*-DP as the dominating stereoisomer (Li et al., 2013b). Clinical serum parameters such as alanine aminotransferase, alkaline phosphatase and total bile acids were reduced significantly at highest dose, and serum glucose significantly increased. A non-significant increase in thyroid stimulating hormone (TSH) were also observed but other serum thyroids were unaltered. In the liver, gene-expression of several key enzymes were altered such as n-acetyltransferase, sulfotransferases and CYP2B1 (Li et al., 2013b). However, no significant changes in absolute body or liver weight or liver histopathology were observed. It should be noted that background contaminations gave 100% detection of DP in the control tissues with increasing levels during the experiment (at day 90, levels in serum was 45 ng/ml, muscle and liver levels were 200 and 3100 ng/g lw, respectively).

138. In mechanistic *in vitro* studies, DP was shown to inhibit insulin signalling in differentiated murine and human preadipocytes after 48 h exposure to 10, 100 or 1000 nM DP in a dose dependent manner, significant at 1000 nM (Peshdary et al., 2020). Peshdary et al., (2019) showed that DP activated the peroxisome proliferator-activated receptor (PPAR-γ) and induced adipogenesis (observed as lipid accumulation and upregulation of adipogenesis mRNA and protein markers) of both murine and human preadipocytes. These effects were only observed at the highest dose tested (10 μM) but not at the lower doses tested in that study and the study of Peshdary et al., (2020). Some DP-mediated adipogenic endpoints were independent of PPAR-γ activation, suggesting that other potential modes of actions of DP may be involved (Peshdary et al., 2019).

139. Recently two studies in rodents indicate that low and environmental relevant doses of DP may alter adipocytes and gut microbiota in ways that are associated with increased risk of metabolic disorder and development of type 2 diabetes (Peshdary et al., 2020; Zhang et al., 2020). In mice, exposure to DP (10, 100 or 1000 μg/kg) for 28 days by gavage increased serum insulin and amylase levels (at 10 and 100 μg/kg) and altered lipid tissue. At 1000 μg/kg, serum lipase was significantly decreased after 28 days. DP promoted white adipocyte tissue hypertrophy and dysfunction, which was characterized by increased adipocyte size and elevated serum levels of leptin and plasminogen activator inhibitor-1 as well as potential local adipose tissue inflammation. In epidydimal and inguinal adipose tissue, mRNA expression levels of adipogenic genes were significantly decreased. Furthermore, DP exposure induced “whitening” of brown adipose tissue in the control diet group observed as a dose dependent decrease in mRNA expression of uncoupling protein 1 (Peshdary et al., 2020). Only in the high-fat diet group DP promoted glucose intolerance in mice independent of weight gain in a U-shaped dose-response evident from day 7 (Peshdary et al., 2020). In another study, rats were fed 5 mg DP/kg/d through pregnancy and/or lactation. In both offspring and dams, species diversity and richnhess of gut microbiota was reduced and their production of metabolites of short-chain fatty acids were significantly increased in all DP exposure groups (Zhang et al., 2020).

2.4.4 Human epidemiology

140. Epidemiology studies indicate some possible associations between serum DP levels and effects on thyroid- and sex-hormones. .Presented epidemiological investigations followed the case-control design, that have a lower level of evidence compared to cohort studies.The study design and covariates do also differ between the studies, and in some studies levels of DP are highly correlated with other serum flame retardants and thus weakening the results observed for DP. Informations on DP levels, group size and other relevant information is provided in Table 16, UNEP/POPS/POPRC.17/INF/xx. An association between serum thyroid hormones (TH) and DP levels was reported in human mother-infant pairs near an e-waste recycling area in China (Ben et al., 2014). DP concentration in maternal sera from the 20-year residents' group was 2 to 3-fold higher than the 3-year residents' group. *Syn*- and *anti*-DP were detected in placenta and umbilical cord serum samples, indicating that they could translocate from maternal to foetal tissues. Levels of TSH were significantly lower in maternal serum in the group that had lived in the area for 20 years (n= 44), than for those whose mothers had been residents to the area for 3 years or less (n=22), (p=0.046), but correlation analysis showed insignificant association between THs and DP levels in the 20-year residents' group. When concentrations of PBDEs were used as a control variable, the DP concentrations were positively associated with total

triiodothyronine concentrations (TT3) in sera from mothers who had lived in the area for over 20 years (r = 0.37 and p = 0.020 for *syn*-DP, r = 0.360 and p = 0.024 for *anti*-DP). According to the study authors the association between serum DP levels and levels of the thyroid hormone TT3 suggested that DP may have some effects on thyroid hormone in humans (Ben et al., 2014).

141. Another study compared serum levels of adults living in an e-waste recycling region (n=54) and a control region (n=58).Lower levels of TSH, thyroid binding globulin and mRNA expression of thyroid receptor (TRα) and higher level of mRNA gene expression iodothyroine deiodinase 1 (ID1) were observed in highly exposed individuals (Guo et al., 2019). For serum DP levels, a nonmonotonic associated was observed for serum TSH and a negative association observed for serum thyroxine binding globulin (TBG) and gene expression of ID1 and TRα was positive and negative associated with serum DP levels, respectively (Guo et al., 2019). Association between serum DP levels and gene expression of ID1 was also observed in children (n= 112) from the same area (Guo et al., 2020). No association was observed between serum TH hormones and DP levels, but a nonmonotonic associated was observed between level of *anti*-DP and TBG. Positive correlation was observed between serum DP levels and gene expression of ID1 (Guo et al., 2020). Another study of pregnant women (n=64) in Wenling, China observed no association between serum thyroid hormones and DP levels. However, a positive relationship between DP serum level and apolipoprotein A was observed (p= 0.095) for total DP and for *syn*-DP (p=0.045) (Yin et al., 2020). In a study from the US, a positive association (b= 0.089, p= 0.023) between *anti*-DP levels in the subjects' wrist band and blood levels of TSH was found for women (n= 62) (Wang et al., 2020).

142. In another epidemiology study comparing groups from e-waste dismantling (n= 54) and control area (n=58), Guo et al (2018b), showed significant negative association between DP serum levels and follicle stimultating hormon (FSH) in both female and male. In male, a positive association between anti-DP and testosterone and a non-monotonic association between testosterone and syn-DP serum levels were observed (Guo et al., 2018b). In a generalized additive model, a significant nonmonotonic association was observed for estrogen and DP in female. However, the study did not report menstrual cycle as one of the listed confoundes so the female data may have some limitations.

2.4.5 Other concerns

143. The DP by-products 1,3- and 1,5-Dechlorane Plus monoadduct (DPMA) have in some cases been detected in greater levels than DP in environmental samples (Sverko et al., 2011; Tomy et al., 2013; Guerra et al., 2011), and also in Antarctic biota (Wolschke et al., 2015) (see paragraph 89 for details). DPMA may also be under-reported because of destructive sample preparation methods (Rjabova et al., 2016). Based on predictive QSAR modelling studies, DPMA is potentially both bioaccumulative and persistent. No information is available on its toxicity to humans and the environment, but based on its structural similarity to aldrin and heptachlor it might be epoxidated in the environment to form substances that could be neurotoxic and/or cause hepatotoxicity (ECHA 2017b,c). For additional information see Figure 1 and Table 16 in UNEP/POPS/POPRC.17/INF/xx.

3. Synthesis of information

144. DP is a commercially available additive chlorinated FR that is marketed and used as a replacement for already regulated FRs in plastic polymers. Known applications include the aviation and automotive industries, electrical wire and cable coatings, plastic roofing materials, connectors in TV and computer monitors.

145. DP is detected in the environmental samples, humans and wildlife at different locations around the globe. It is a global contaminant and has been found in remote regions far from sites of production and use, including in the Arctic, the Antarctic and high-mountain regions in Tibet. DP has also been detected in dust, sludge and wastewater from WWTPs at sites far from production and e-waste recycling sites, indicating emission and exposure from consumer products.

146. Available monitoring data provides evidence for ongoing LRET. Bioaccumulation and trophic magnification are reported in several studies, including in biota in remote regions. DP is environmentally persistent, and although existing environmental levels in remote regions are generally low, there is reason for concern. DP levels in remote regions are in some studies reported to be in the same range as in source regions. DP is persistent in the environment and bioaccumulates in biota. The levels are likely to increase over time as a result of continued use and could possibly lead to adverse effects in biota in the future if no action is taken.

147. Although DP has been produced and used for almost half a century, the first environmental detection of DP was reported in 2006 in the Great Lakes Basin in North America (Hoh et al., 2006). Further research has identified DP as a global pollutant (Wang et al., 2016). Due to limited data, temporal and spatial trends for DP are equivocal; while increasing trends are observed in some studies, other studies report unchanged levels (see section 2.3.1).

148. Available short-term toxicity data indicates concern for potential adverse effects to the environment and humans. One relevant mode of action could be the induction of oxidative stress that influences several biological processes. Effects on the developing nervous system and brain have been indicated in fish. DNA damage, increased mortality rate, delayed hatching time, reduced hatching rate, decreased body length, and increased morphological

deformities was observed in carp embryos. DP has been shown to activate peroxisome proliferator-activated receptor gamma and inhibit insulin signalling *in vitro* in mammalian adipose cells. DP has been shown to impair liver function in mice and at low doses alter insulin secretion and induce glucose intolerance. In addition, changes in biomarkers involved in insulin resistance and adipocyte dysfunction were also observed. A few studies indicate potential effects to the sex- and thyroid hormone pathway in humans. Furthermore, a co-exposure study on zebrafish indicated that other pollutants may increase bioaccumulation, resulting in greater toxicity. Further data, including studies of longer duration and possibly different routes of exposure would provide better insight into the toxicity of DP.

149. Adding to the concern for adverse effects are the DP by-products, such as 1,3- or 1,5-DPMA which have structural similarities to substances such as aldrin and heptachlor that are already listed by the Stockholm Convention (Table 14, UNEP/POPS/POPRC.17/INF/xx). There are no toxicity data for these monoadducts, but in some cases these are detected in greater levels than DP in environmental samples (Sverko et al., 2011; Tomy et al., 2013; Guerra et al., 2011).

150. DP is transferred to developing offspring during pregnancy via blood, and after delivery via breast feeding. Maternal transfer to eggs has been described for fish, bird and amphibians leading to exposure during sensitive life stages. High concentrations of DP have been detected in environmental samples and humans living near DP production plants and e-waste recycling sites.

151. Due to its POP properties and its widespread production and use, global action is warranted to prevent further releases of DP.

4. Concluding statement

152. DP is emitted into the environment from human activities, e.g. from manufacturing, use and disposal and management of waste. DP is persistent, bioaccumulative and undergoes LRET, making emissions and releases of this substance a transboundary pollution problem. Globally, DP is detected in humans, wildlife and environmental samples in all global regions, including in the Arctic and Antarctic.

153. In humans, DP has been detected in hair, umbilical cord blood, serum and breast milk. High concentrations of DP have been detected in environmental samples and humans living near e-waste recycling sites and production plants. DP is also detected in dust, sludge and wastewater from WWTPs indicating emission and exposure from consumer products throughout their lifecycle.

154. Available scientific literature suggests that there is a potential risk for adverse effects for developing aquatic species and organisms due to oxidative stress that may affect important biological processes. In aquatic organisms, effects on the developing nervous system and brain have been indicated as well. Furthermore, observed effects on photosynthetic activity in marine macroalgae suggest possible impacts on aquatic ecosystems mediated by negative effects on primary production. The concern for potential adverse effects to humans relates to adipose tissue, liver and endocrine modulating effects, in particular to the insulin and peroxisome proliferator-activated receptor gamma pathways.

155. Mixture toxicity effects such as increased toxicity and bioaccumulation have been indicated for DP in one study and suggest an additional cause for concern. Climate warming may exacerbate the mobilization and release of DP in polar environments as DP deposited in sea ice, glaciers, and permafrost are released into freshwater and marine environments. The combined effect of climate change and other environmental stressors adds to the risk by affecting exposure levels, the vulnerability and adaptability of organisms, particularly in the polar regions.

156. Based on evidence for persistence, bioaccumulation and adverse effects of DP observed in some organisms and its widespread occurrence in the global environment including at remote regions, [it is concluded that DP is likely, as a result of its long-range environmental transport, to lead to significant adverse human health and/or environmental effects such that global action is warranted].

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