Dechlorane Plus and its *syn*- and *anti*-isomers

Draft risk profile

Prepared by the intersessional working group of the
Persistent Organic Pollutants Review Committee

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Executive summary

1. The Persistent Organic Pollutants Review Committee at its fifteenth meeting 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 to the Secretariat the information specified in Annex E, before 2 December 2019.

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. Additionally, the raw material hexachlorocyclopentadiene, used in the production of DP, may contain low levels of a number of impurities. DP and its isomers are only known intentionally produced.

4. DP is produced by the Occidental Chemical Company (OxyChem) in Niagara Falls, New York, and at the Anpon manufacturing plant in Huai'an in China. Production in the United States started in 1986 and in China in 2003. DP is considered a high production volume chemical in the United States with an estimated annual production of 450–4500 tonnes. In China, 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 disposal and recycling activities. Releases from use include industrial uses as well as releases from consumer products. DP has been detected globally in many locations, ranging from production and recycling sites to urban, rural and remote areas, including measurements in air, water, soil, wastewater, sludge, biosolids, landfill leachate, indoor and outdoor dust, wildlife, as well as in humans.

6. On a global scale, the highest DP concentrations were detected in the United States and China close to known production sites or electronic waste (e-waste) treatment facilities. DP is also found in remote regions. Several monitoring studies document the long-range transport of DP to remote regions via the atmosphere, ocean currents and possibly also via migratory birds. Modelling suggests DP has transport and persistence properties similar to listed POPs. In remote areas, DP has been detected in different environmental matrices and biota in the Arctic, Antarctic and mountain regions in Tibet. The atmospheric half-life of DP, based on gas phase reactions and not considering particle-bound DP, is calculated to be about 14 hours, i.e. below the criterion of two days set in Annex D (d) (iii) of the Convention. However, modelled half-lives in air are largely based on gaseous 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 (K_{oa}). 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 DPis 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 has very low water solubility and a high octanol-water partition coefficient (log K_{ow}) and is thus expected to bind to organic matter in soil and sediments, and therefore it is probably not easily bioavailable for microorganisms. Furthermore, DP has no functional groups that are susceptible for hydrolysis and is therefore not expected to hydrolyse. DP is detected in deep soil and sediment cores, indicating its persistence. Modelling predicted that aerobic biodegradation of DP would be very slow and similar to analogue chemicals (chlordane, heptachlor, dieldrin/endrin, aldrin and endosulfane) that are already listed under the Stockholm Convention.

8. Although the water solubility is low, bioconcentration factors above 5000 have recently been reported for DP in fish studies. A depuration half-life (ranging from 6 to 180 days) corresponding to a bioconcentration factor above 5 000 L/kg has been reported for DP, as well as biomagnification and trophic magnification factors above 1 in a variety of food chains and locations, including in Antarctica. DP has been identified as a very persistent and very bioaccumulative substance (vPvB) in the European Union.

9. 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 with its *syn*- and *anti*-isomers. DP has been detected in human blood and breast milk in many regions. Moreover, the foetus may be exposed to DP via the umbilical cord blood, and breast milk may be an important source of exposure for infants. Highest DP levels have been observed in occupationally exposed workers or residents living near production facilities and e-waste recycling sites in China.

10. Available studies in experimental animals suggest low concern for acute toxicity via oral, inhalative or dermal routes of exposure in humans or to organisms, but long-term studies exiding 90 days are missing. However, DP show structural similarities with already listed POPs such as aldrin, chlordane, heptachlor, mirex, chlordecon

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which have neurotoxic and some also carisinogenic properties. Studies with DP report effects including oxidative damage, neurotoxicity and potential for endocrine disruption. Oxidative stress has been observed in marine macroalgae, fish, marine bivalves, earthworm, birds, and mice after exposure to DP. In a marine macroalgae exposed to DP, photosynthesis was reduced at low concentrations. Neurotoxicity was observed in fish and potential neurotoxicity indicated in earth worms. Potential endocrine modulative effects have been described in fish and some studies indicate possible endocrine modulative effects 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 also 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.

11. DP is widely detected in the global environment, including in remote regions. It is also persistent, bioaccumulative, toxic to animals and potentially to humans. It is transported to locations far from its production site and use. Therefore, it is concluded that DP, with its *syn*- and *anti*-isomers, are likely, as a result of their long-range environmental transport, to lead to adverse human health and/or environmental effects such that global action is warranted.

1. Introduction

12. In May 2019, Norway submitted a proposal to include 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. Dechlorane Plus is commonly abbreviated as DP or DDC-CO. In this risk profile DP is used.

13. The commercial mixture "Dechlorane Plus"™ is a polychlorinated flame retardant that has been in use since the 1960s. Since then, research has confirmed its globally ubiquitous distribution (Wang et al., 2016) and several studies have shown increasing or stable time trends of DP levels (see section 2.3.1 Environmental levels and trends).

14. DP 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 (reviewed in Wang et al., 2016; OxyChem, 2019).

1.1 Chemical Identity

15. The "Dechlorane Plus"™ technical mixture (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). The isomers are present in the technical product 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).

16. The chemical identity of DP and the modelled and experimental physicochemical properties for DP with its two isomers are listed in Tables 1 and 2, below.

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

CAS number:	13560-89-9, 135821-03-3, 135821-74-8
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-dien
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-dodechydro-1,4:7,10-dimethanodibenzo[a,e]cyclooctene; Dodecachlorododecahydrodimethanodibenzocyclooctene

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Trade names	Dechlorane Plus 25 (Dech Plus); Dechlorane Plus 35 (Dech Plus-2); DP-515; Dechlorane 605; Dechlorane A; DP; Escapeflam DK-15 (China); PyroVex SG (grade 515, 25 and 35)

Table 2: Overview of selected physicochemical properties Dechloran Plus with its *syn*-DP and *anti*-DP isomers

Property	Value	Reference
Physical state at 20 °C and at 101.3 kPa	Solid white powder	ECHA, 2017b
Melting/freezing point	340-382 °C 350 °C	ECHA, 2017b OxyChem datasheet (2007)
Vapour pressure	0.006 mm Hg ≙ 0.8 Pa (at 200 °C)	OxyChem datasheet (2007)
Water solubility*	<1.67 ng/L (20 – 25 °C) 0.044 – 249 µg/L (insoluble)	ECHA, 2017b OxyChem datasheet (2007)
n-Octanol/water partition coefficient, K_{ow} (log value)	9.3	OxyChem datasheet (2007)
n-Octanol-air partition coefficient K_{oa} (log value)	12.26	OxyChem datasheet (2007)
Sediment/water partition coefficient K_p (log value)	6.65	OxyChem datasheet (2007)
Air-water partition coefficient K_{aw} (log value)	The following log K_{AW} values at 25 °C are estimated based on the Henry's Law constant: $\geq 1.39\times10^5$ Pa·m^3/mol 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, 2017b

*There is some uncertainty in the precise value for water solubility (Chou et al. (1979). 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
*** Calculates the Henry's Law constant (air/water partition coefficient) using both the group contribution and the bond contribution methods.

17. Dechlorane Plus is produced by the Diels-Alder reaction of two moles of hexachlorocyclopentadiene with one mole of 1,5-cyclooctadiene. The commercial technical hexachlorocyclopentadiene may contain low levels of a number of impurities such as tetrachloroethylene, *hexachlorobutadiene,* pentachlorocyclopentenone, octachlorocyclopentadiene, *hexachlorobenzene, pentachlorobenzene and mirex*, where those in italic are listed POPs (US.EPA, 1978). 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 may also occur as impurities in the commercial substance. For example, Li et al. (2013b) found a mono-dechlorinated substance (DP-1Cl or DP-Cl_{11}) in the commercial substance produced by Jiangsu Anpon Co. Ltd., China.

18. Quantitative structure-activity relationship (QSAR) models (Sverko et al., 2011; Feo et al., 2012) and voluntary testing of DP by its manufacturer (EHSI, 2004) have indicated that DP has physical-chemical characteristics typical of other POPs, including high lipophilicity (log K_{OW} ~9), resistance to biodegradation, and bioaccumulation (log BAF ~5 in fish) (EHSI, 2004; Sverko et al., 2011; Feo et al., 2012).

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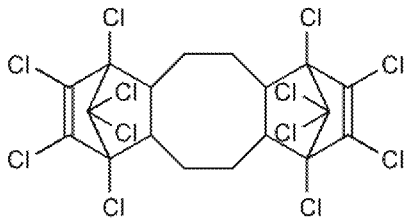
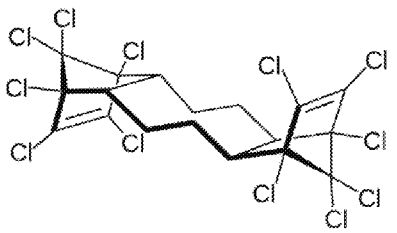
Dechlorane Plus (CAS no. 13560-89-9)



Anti- (or *exo*) Dechlorane Plus (CAS no. 135821-74-8)



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

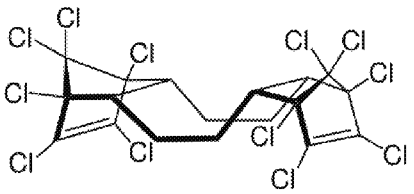


Figure 1: Structural formula of Dechlorane Plus and its two isomers

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 Dechlorane Plus (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 Dechlorane Plus and its *syn*-isomer and *anti*-isomer met the screening criteria specified in Annex D (Decision POPRC-15/4). It was 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) Nomination report on Dechlorane Plus and its *syn*-isomer and *anti*-isomer submitted by Norway (UNEP/POPS/POPRC.15/3);

(b) Information submitted by the following Parties and Observers according to Annex E of the Convention: Belarus, Canada, Egypt, Germany, Hungary, Monaco, Netherlands, New-Zealand, Romania, Thailand, Qatar, Republic of Korea, State of Palestine, International POPs Elimination Network (IPEN and Alaska Community Action on Toxics (ACAT);

(c) Documents supporting the identification of Dechlorane Plus as a Substance of Very High Concern (SVHC) under the European Chemicals Agency (ECHA) Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) regulation (ECHA, 2017b,c);

(d) Screening assessment report and risk management approach from Canada (Canada, 2019a; 2019b);

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

1.4 National and international administrative actions

21. In 2018, based on an Annex XV dossier and Risk Management Options Analysis prepared by the UK, DP (including its *syn-* and *anti*-isomers) were identified as Substances of Very High Concern (SVHC) and added to the REACH Candidate List due to their very persistent and very bioaccumulative properties (ECHA, 2017a). Inclusion on this list means the substances can be subject to further review. Moreover, suppliers of articles containing a SVHC in a concentration above 0.1% (weight by weight) have to provide professional users and distributors with enough information to allow the safe use of the article. As a minimum, the name of the substance in question has to be communicated. Upon request from a consumer, the supplier has to provide the same information, within 45 days. Based on its intrinsic properties in combination with high volume and widespread use, ECHA recommended in October 2019 to include DP in Annex XIV of the REACH Regulation (List of Substances Subject to Authorization). To align risk management activities within the European Union (EU) and the potential listing under the Stockholm

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Convention, an Annex XV REACH restriction dossier for DP will now be prepared by Norway. A restriction proposal is expected to be submitted in April 2021. This will affect the process regarding possible inclusion of DP in REACH Annex XIV. There is no harmonised classification for Dechlorane Plus (CAS 13560-89-9) in the EU, but 78 notifiers have classified the substance as Acute Toxicity Category 4, H332 Harmful if inhaled (ECHA, 2018).

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

23. 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 and offer for sale of DP and all products containing the substance (Canada, 2019b).

24. In the United States (US), DP is listed under the Toxic Substances Control Act (TSCA) inventory and is subject to the Chemical Data Reporting Rule, which requires manufacturers and importers to provide the United States Environmental Protection Agency (US EPA) with production, import and use volumes, as well as other relevant information ([HYPERLINK "https://www.epa.gov/tsca-inventory"])).

25. DP is listed on New Zealand's Inventory of Chemicals but does not have its own approval under the HSNO (Hazardous Substances and New Organisms) Act. This means it can only be used as a component of products that are covered by group standards (Annex E information, New Zealand). [HYPERLINK]

26. In Thailand, at the national level, DP is not yet classified under the Hazardous Substance Act B.E. 2535 (1992). However, DP is regulated under the notification of the Ministry of Industry, Thailand on Account no 5.6, requiring manufacturers and importers to report the fact of importing chemicals/products with quantity more than 1000 kg/year. At present, DP has not yet been controlled and there is no data on import volume and usage (Annex E information, Thailand).

27. DP has been listed on the International Chemical Secretariat's (ChemSec) Substitute It Now (SIN) List since 2014 (SIN LIST). The SIN List consisting of chemicals that have been identified by ChemSec as being SVHC, based on the criteria defined within REACH, the EU chemicals legislation.

2. Summary of the information relevant to the risk profile

2.1 Sources

2.1.1 Production, trade, stockpiles

28. DP is registered (last accessed 4 January 2020) under REACH by a Dutch company at an estimated tonnage of 100 – 1000 tonnes per annum. Sweden registered use of 5 tonnes in 2006, after a decrease from 11 tonnes in 2005 (Kaj et al., 2010). DP is classified as a high production volume chemical (>500 tons) in the US (Sverko et al., 2011). In the US, DP is manufactured by Occidental Chemical Company (OxyChem) in Niagara Falls, New York, and the annual productions were estimated to be 450–4500 tonnes since 1986 (Qiu et al., 2007). Furthermore, manufacturing in China has been reported to be 300–1000 tonnes per year by Anpon in Huai'an since 2003 (Wang et al., 2010a). The global annual production is approximately 5000 tonnes (Ren et al., 2008). 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 into Canada by a few companies (Canada, 2019a; 2019b).

2.1.2 Uses

29. The use of DP comes exclusively from anthropogenic sources, as there are no natural sources of DP. DP has been manufactured for close to 60 years and is currently marketed as an alternative/replacement for decabromodiphenyl ether (decaBDE) as well as a substitute for Mirex (Hoh et al., 2006). Furthermore, uses in a range of flame-retardant applications of electronic wiring and cables, automobiles, plastic roofing materials, and hard plastic connectors in televisions and computer monitors, wire coatings, and furniture has also been reported (Zhang et al., 2015). In a recent study from China, DP was detected in various building material (Hou et al., 2018). Samples of wallpaper (non-wowen, 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 panel) contained DP in the range n.d. – 5.1 ng/g (Hou et al., 2018). According to information provided in a Canadian assessment, DP is used in Canada as a flame retardant in automobile manufacturing (Canada, 2019b). Annex E information from Netherlands confirms use in automobiles in the powertrain, cooling, chassis and bodywork parts.

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30. As a flame retardant, DP is used in many polymeric systems. 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 copolymer (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). According to manufacturer literature (OxyChem, 2007), DP is manufactured for use solely by industrial customers.

31. Under REACH, DP is registered for the following product categories; polymer preparations and compounds, semiconductors, adhesives, sealants. However, use in articles is also indicated, e.g. electric cabels, electronic articles, plastic products and glued articles in aircraft parts. For furter information go to ECHA web site (https://echa.europa.eu/da/registration-dossier/-/registered-dossier/11906/3/1/4).

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 monitor and laptop casings (Taverna et al., 2017).

33. A recent paper reported detection of DP in plastic castings from two TVs and one computer, but not in circuit boards from similar equipment collected from a recycling facility in China (Li et al., 2018, 2019c). In addition, DP was detected in recycled ABS from Guangdong Province (China) (Cao et al., 2020).

34. PyroVex® SG is an additive flame-retardant product that, according to the supplier Velsicol Chemicals LLC, contains > 99% DP (Velsicol Chemicals LLC, accessed 06.02.2020). PyroVex® SG is typically mixed into plastics and other materials and can be used alone or be formulated with complementary chemistries to yield the market's performance requirements. It is marketed as an ideal product for use in the manufacture of flame retarded wire and cable, as an excellent flame retardant for electrical connectors used in computers and a myriad of 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. Identified applications where DP was used as an additive flame retardant included; in the aviation and automotive industries, in polymeric compounds for cable insulation and minor use within the development of fireworks (https://echa.europa.eu/-/public-consultation-to-identify-nine-new-substances-of-very-high-concern).

36. Based on information gathered from a 'Statistical Survey of Chemicals' under the 'Chemicals Control Act', no DP was manufactured in the Republic of Korea since 2010. However, 10 tonnes of DP were imported in 2014 and 2016 (Annex E information, Republic of Korea).

2.1.3. Releases and emissions to the environment

37. The occurrence of the commercial mixture DP and its isomers in the environment is a result of anthropogenic production, use and disposal. DP and its isomers are not known to be unintentionally produced.

38. A number of activities such as production, use, recycling, waste handling, as well as leachate and run-off from landfills, wastewater treatment plants (WWTPs) and more, can lead to releases of DP to the environment (Wang et al., 2016; Wang and Kelly, 2017). DP has been detected in sludge (Barón et al., 2014a; Norwegian Environment Agency, 2018b; 2019a) and biosolids (Davis et al., 2012) from WWTPs. In addition, sludge from WWTPs contaminated with DP is potentially used as soil enrichment (Wu et al., 2017). In China, about 10 – 15 % of sewage sludge generated per year is used as fertilizer (Dai, 2011 reviewed in Ji et al., 2018) and can therefore present a potential risk if it contains hazardous substances (Ji et al., 2018). Furthermore, DP (*syn*-DP and *anti*-DP) has been detected in the particulate fraction of storm water in Oslo, Norway (Norwegian Environment Agency, 2018b; 2019a). In these studies, *anti*-DP was present in higher concentration (approx. 3x) than *syn*-DP.

39. An investigation of chlorinated flame retardants was conducted in samples from Canadian WWTPs. Detection frequency of total DP (*syn*- and *anti*) 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). The occurrence of DP in a conventional WWTP in Shanghai has also been examined. 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}), and the average of f_{syn} ranged from 0.16 to 0.33, which fell in the range of two commercial DP mixtures. The annual release of DP via sewage sludge from all WWTPs in Shanghai were estimated to 164.8 g (Xiang et al., 2014).

40. DP is used as a flame retardant in electrical and electronic (EE) 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) was 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

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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 household products, which are 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 (Newton et al., 2015; Wong et al., 2017, Cao et al., 2014). In China, detections in indoor dust included an office, a hotel, a kindergarten classroom, and a student dormitory, as well as one outdoor (road dust) microenvironment. The highest concentration was found in hotel dust (124 000 ng/g) followed by dormitory dust (14 200 ng/g), and kindergarten classroom dust (231 ng/g) (Cao et al., 2014). For additional information on releases and emission from manufacture, WWTP, e-waste facilities, landfills and such (see Table 1 in UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION]).

2.2. Environmental fate

2.2.1 Persistence

42. DP is chemically stable in various environmental compartments with minimal or no anaerobic degradation (reviewed in Wang et al., 2016). DP does not contain any functional groups that are susceptible to hydrolysis and therefore hydrolysis is not expected to be a relevant degradation process (Canada, 2019a; ECHA, 2017b). Due to DP's very low water solubility and high log Kow, it is not likely to be bioavailable for microorganisms and is expected to bind to organic carbon in soil and sediments (Wang et al., 2016). Furthermore, studies with fish indicate a very low potential for biotransformation and provide supporting evidence that DP is metabolically stable, and therefore persistent in the environment (Tomy et al., 2008, ECHA, 2017b). In addition, field studies provide some limited evidence that DP is persistent since it was found in a sediment core layer corresponding to around 1980 (Qiu et al., 2007).

43. The photodegradation of DP has been investigated by irradiation, using a xenon lamp. Rapid photodegradation was found under 200–750 nm light, while the degradation became much slower when the range of light wavelength changed to 280–750 nm. The quantum yields of 200–280 nm (UV-C) were about 2–3 orders of magnitude higher than 280–320 nm, and no yields were detected in 320–750 nm range (Wang et al., 2013b). These findings suggest that DP is considerably photo-stable in the environment and will not degrade with normal illumination on the terrestrial surface. However, DP degradation under UV-C irradiation was so fast that photodegradation would be a potential removal process. The sequential appearance where one to four chlorine (Cl) are removed from DP and exchanged with hydrogen (H) suggested that the reaction was leading to photodechlorination of DP (Wang et al., 2013b). Dechlorination was also observed in a different study where *anti*-DP, *syn*-DP, and commercial DP solutions were exposed to UV light. Dechlorination products were mainly [−1Cl+1H], [−2Cl+2H] (Li et al., 2013b).

44. Using modelling (BIOWIN v4.10 (U.S. EPA, 2012), it was predicted that aerobic biodegradation of DP would be very slow (ECHA, 2017b). The results from modelling are the same as shown for a number of analogue chemicals such as chlordane, heptachlor, dieldrin/endrin, aldrin and endosulfane that are already listed as POPs under the Stockholm Convention. Modelling studies were also used to estimate half-lives for anti-DP and the commercial mixture of DP in water, soil and sediment (Zhang et al., 2016; Annex E information, The Netherlands). Physical–chemical properties were predicted using three chemical property estimation tools: EPISuite, SPARC and Absolv in this study and the half-lives in water, soil and sediment were estimated as 180, 3650 and 1621 days, respectively, for both isomers (Zhang et al., 2016).

45. Degradation of DP in aging soils has recently been investigated. A slight decrease in total DP concentrations suggested that only limited degradation occurred during the course of the study, with only 4.2-8.2% of initial DP having degraded after 260 days (Cheng et al., 2019). Following 260 days of ageing of soil containing 3 different cocentration of DP (0.1, 1, 10 μg/g DP soil), 89.6 % to 98.8 % were consistently obtained suggesting that only a small proportion of DP might degrade in soil over time under the influence of ageing. Model simulations were then developed to investigate the transformation. The model simulations indicated that transformation rates were inconstant and statistically distinguishable from one another over time, and half-lives of DP were estimated to range from 1325 to 2948 days, further indicating its environmental persistence in aging soils (Cheng et al., 2019). It has also been reported that *syn*-DP varied little with depth down to 100 cm in soil suggesting that isomer-specific degradation or preferential adsorption does not play a significant role in soil (Wang et al., 2010a).

46. Modelling of degradation potential and microbial metabolic pathways suggests that biodegradation is likely to be very slow, and there is low propability that DP will degrade any faster than analogue substances listed under the Stockholm Convention. Data showing lack of degradation in soil and sediments over time as well as low ability to biotransform in fish support the conclusion that DP is very persistent.

2.2.2 Bioaccumulation and pharmacokinetics

47. The log K_{ow} for DP is reported to be 9.3 (OxyChem, 2007). The high log K_{ow} and the very low water solubility (Table 2) indicates that DP is very hydrophobic and therefore partitions into organic matter such as binding to particles. This is further supported by the relatively high log sediment-water partition coefficient (log K_p) of 6.65. These properties make aqueous laboratory studies very difficult to carry out due to the potential difficulties in maintaining stable exposure levels at concentrations below the water solubility. 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).

48. A recent laboratory-scale microcosm fish study reported bioconcentration factor (BCF) values exceeding the bioconcentration criteria set by the Stockholm Convention (BCF >5000) (Wang et al., 2019). In this study, where carp were exposed continuosly to DP through water, reported BCF values were 5700 and 9300 L/kg for *syn*- and *anti*-DP, respectively (Wang et al., 2019). DP showed slightly shorter depuration times (*syn*- and *anti*-DP were 6.3 and 7.2 days, respectively) when muscle was analysed, than reported in Tomy et al. (2008).

49. 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 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). Results from a food web study from Antarctica were similar to those of Tomy et al., (2008) but showed tissue-specific variations (Tang et al., 2018). 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 in tissues. In another study, the elimination half-life for *syn*-DP was about 179 days (d) in liver, 44 d in muscle and 24 d in serum, and for *anti*-DP, 54 d in muscle and 25 d in serum. Depuration time for *anti*-DP for the liver was not reported (Li et al., 2013b). Furthermore, model calculation of the oral up-take of DP showed great agreement between modelling (Larisch and Goss, 2018) and experimental data (Tomy et al., 2008) indicating that up-take of super-hydrophobic chemicals like DP from food does take place without any anomalies. Uptake of DP is slow but will eventually result in substantial bioconcentration when substances are not metabolized (Larisch and Goss, 2018).

50. 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 and marine waters, as BMFs (Tomy et al., 2007; Wu et al., 2018; Sun et al., 2015, Sun et al., 2017) and TMFs (Sun et al., 2015; Kurt-Karakus et al., 2019; Na et al., 2017) are reported to be greater than 1, including in one study in Antarctica (Na et al., 2017). 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 an aquatic food web study from China, TMFs were reported in the range 1.9 – 3.1. In this study, biomagnification potential (TMF ranging 0.53 – 5.4) were also reported for DPMA (impurity in commercial DP), 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. Only one terrestrial study is available showing BMFs above 1 (Yu et al., 2013). For further details on the bioaccumulation studies (see Table 2 in UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION]).

51. The isomer composition of DP in environmental samples can be different from the technical products because of their biota isomer-selective uptake or elimination, biodegradation, and stereospecific photodegradation (Wang et al., 2015). Studies have shown that *syn*-DP has a stronger ability of stereoselective enrichment than *anti*-DP in aquatic organisms, such as in walleye (Hoh et al., 2006), oysters (Jia et al., 2011), rainbow trout (Tomy et al., 2008) and seal (Klosterhaus et al., 2012), but opposite results were observed in Chinese sturgeon (Peng et al., 2012), and carp (Wang et al., 2020). 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), and the type of wildlife and tissue (Peng et al., 2014; Zhang et al., 2011; Zheng et al., 2014a) are the main factors for the stereoselective bioaccumulation of DP in organisms.

52. Chicken that were bred in the vicinity of an e-waste facility were naturally exposed to DP through sand and food and showed preferred accumulation of the *anti*-DP isomer (Li et al., 2014). Furthermore, the study revealed tissue specific accumulation of *anti*-DP with highest concentration of anti-DP in the brain, fat and liver. The study also showed that *anti*-Cl_{11}-DP was absorbed through the diet rather than metabolized from DP *in vivo* (Li et al., 2014). This information is strengthened by the fact that *syn*-Cl_{11}-DP and *anti*-Cl_{11}-DP were detected in commercial DP-25, hence, it cannot be excluded that these chemicals originated from commercial products (Li et al., 2013b). In a study from 2011, chicken eggs were injected with DP (Crump et al., 2011). In hepatic tissue of the *in ovo* exposed chicken embryos the relative proportion of the two isomers significantly shifted, the proportion of *syn*-DP increased from 0.34

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to 0.65, as *anti*-DP decreased (0.66 to 0.35), resulting in *syn*-DP isomer beeing predominant in this study. While in a recent study, hens and fertilized eggs were exposed to DP isomers and no differences were observed in absorption of the isomers (Li et al., 2019a). However, the results found that *anti*-DP had a slightly longer half-life that *syn*-DP, and that stereo-selective excretion of *syn*-DP, rather than metabolism of *anti*-DP played a more prominent role in isomer-specific bioaccumulation of DP in chickens. A similar stereoselectivity was shown for seagrass (marine algae) exposed to DP isomers, where *syn*-DP was stereo-selectively excreted (Zhao et al., 2014).

53. There is some indication of biotransformation of DP in isomer-specific studies (Hoh et al., 2006; Tomy et al., 2007; Tomy et al., 2008; Zhu et al., 2014). DP has been shown not to metabolize easily in biota (Tomy et al., 2008; Xian et al., 2011) and the results from Tomy et al. suggest enzyme-induced metabolism of DP in biota might be low, if it does occur. 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), but some studies suggest they are formed through biotic or abiotic processes prior to uptake or even through analytical impurities (Sverko et al., 2008, 2010; Tomy et al., 2008; Zheng et al., 2010, 2014b). 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).

54. DP was first detected in archived fish (walleye) from Lake Erie in 2006, 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). Evidence from around the world shows 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). 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) and cord serum (Ben et al., 2014) 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).

55. In summary, the long depuration half-life, which is indicative of a BCF above 5 000 L/kg, has been reported for DP. BCF above 5000 L/kg have been reported for DP in fish studies as well as BMFs and TMFs > 1 for several organisms and food webs, and this further supports the conclusion that DP is bioaccumulative. 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). Additionally, DP was recently identified as a very bioaccumulative (vB) substance in the EU (ECHA, 2017a).

2.2.3 Potential for long-range transport

56. Due its high hydrophobicity and low vapour pressure (see Section 1.1), DP is mainly adsorbed 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 % (e.g. Hoh et al., 2006; Ren et al., 2008; Möller et al., 2010). Measurements in seawater confirm this 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 (Möller et al., 2010). In Atlantic seawater the particle bound fraction was on average 58% for *syn*-DP and 75 % for *anti*-DP.

57. The modelled half-life (QSAR estimate) of DP in air is 13.68 hours (Sverko et al., 2011), i.e. below the criterion of two days set in Annex D (d) (iii) of the Convention. However, modelled half-lives in air are largely based on gaseous phase reactions and do not take into account possibly longer half-lives following sorption to particles, which is presumed to be the primary mode of transport for DP (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 the long-range transport of DP on particles (Sverko et al., 2011; Canada, 2019a).

58. Modelling performed with the OECD P_{OV} and LRTP Screening Tool, a software tool for screening chemicals for persistence and long-range transport potential (Wegmann et al., 2009), suggest DP has transport and persistence properties within the range reported for listed POPs (Sverko et al., 2011). The characteristic travel distance DP in air using this model was calculated to 2508 km by Canada (2019a). This is below the value of 5097 km determined by model results for reference chemicals. However, the transfer efficiency for DP was calculated to be 9.7%, which is above the value of the reference substance PCB-28. The high transfer efficiency calculated indicates that DP may be deposited to some degree in remote regions. Furthermore, an overall persistence of 213 days is predicted for DP emission to air, with the model estimating 98.82% of the substance in air being partitioned to aerosols. As discussed above, this is in line with reported monitoring data (Hoh et al., 2006; Ren et al., 2008; Möller et al., 2010; AMAP 2017, Canada 2019a) and suggests that particle-bound transport may be important for long-range transport of DP.

59. As indicated by ECHA (2017d) the results generated using the OECD P_{OV} and LRTP Screening Tool are uncertain, largely because most of the input parameters are estimated, and may be of limited value for substances such as as DP that have low vapour pressure and adsorb strongly to particulates in the air (see also Sverko et al., 2011).

Substances with low vapour pressure that adsorb strongly to particulates in the air can undergo long-range transport to remote regions when atmospheric conditions permit (e.g. during dry periods), and their long-range transport is likely to be governed by the fate of the particulates to which they bind (ECHA 2017d).

60. 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, lichen, mosses, southern elephant seal and Antarctic fur seal (Möller et al., 2010; Gao et al., 2018; Kim et al., 2018; Aznar-Alemany et al., 2019); and in air, soil and lichen in a remote mountain region in Tibet (Yang et al., 2016a; Liu et al., 2018).

61. 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 in terrestrial and freshwater species, e.g. 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).

62. 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 3. UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION]). Reported detection frequencies vary from non-detected 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).

63. Möller et al. (2010) were the first to show the potential for DP to undergo long-range environmental transport, as indicated by the detection of DP in remote ocean areas of the Arctic and Antarctica. In this study, marine boundary layer air and surface seawater samples were collected during a sea expedition from the East-Greenland Sea, and in the Northern and Southern Atlantics toward Antarctica. DP concentrations ranged from 0.05 to 4.2 pg/m^3 in the atmosphere and from not detected to 1.3 pg/ L in seawater. The reported concentrations in air and seawater samples suggest that transport may occur both via air and seawater, but also shows a preferential partitioning to particles in both air and seawater with mean reported particle bound fractions of 79 and 81 % for *syn*- and *anti*-DP in air, and (as discussed already above), from 58-97 % for *syn*- and from 75-80% *anti*-DP in ocean water. In the East-Greenland Sea, the authors observed decreasing concentrations of DP in air with increasing latitude. The authors suggest that the findings reflect stereoselective depletion of *anti*-DP likely caused by UV degradation during long-range transport. 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.

64. On a polar expedition cruise from the East China Sea to the Arctic (Möller et al., 2011), DP was detected in both the atmosphere and in seawater along with other dechloranes and the dechlorination product C_{11}DP. Detections of DP were mainly in the particulate phase. Concentrations ranged from 0.01 to1.4 pg/m^3 in air and from 0.006 to 0.4 seawater pg/ L in seawater. The Asian continent was identified as a source of DP and other dechloranes in the marine environment studied and the air-seawater exchange indicated deposition of DP from air to water. DP concentrations in air in the Chukchi/Bering Sea 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.

65. 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 Achipelago 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 3 and 4, UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION]).

66. As discussed by Vorkamp et al. (2019a), the highest DP levels in Arctic air to date were detected on Greenland in 2012. Reported ΣDP 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; Vorkamp et al., 2019a). Other detections in Arctic air include Alert in the Canadian Arctic and Little Fox Lake in the Canadian Sub-Arctic (Xiao et al., 2012, Yu et al. 2015), and Longearbyen on Svalbard (Salamova et al., 2014). Mean ΣDP concentrations reported for these sites were of ~0.25, ~0.096, ~0.75 and 1.2 pg/m^3, respectively. DP was also detected in air from Pallas in Northern-Finland at a mean ΣDP concentration of 0.039 pg/m^3 (Haglund et al., 2016).

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67. Studies from other remote areas documenting long-range environmental transport of DP via atmospheric transport are also available. 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 and can be attributed to long-range environmental transport. 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.

68. 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; Vorkamp et al., 2018; Løseth et al., 2019; Briels et al., 2019; Mo et al., 2019; Desjardins et al., 2019). Birds have previously been identified 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) and may represent an additional transport pathway for DP to remote regions.

69. In a study on DP in birds in remote regions, Vorkamp et al. (2018) detected DP in peregrine falcon eggs collected in Kujalleq in South-Greenland in the low Arctic. Greenland peregrine falcons are migratory birds that spend the northern hemisphere winter in the Caribbean and South America. Thus, the DP in their eggs likely reflect exposure experienced at wintering and breeding grounds as well as during migration. This finding suggests migratory birds as possible biovectors for long-range transport of DP. It is however not possible to quantify the input of DP to remote regions via this transport pathway from the limited available literature. However, earlier studies suggest that input via migrating birds in some instances may be considerable and can contribute to augmentet levels locally. In Ellasjøen at Svalbard, seabird guano was found to account for approximately 14% of the POPs contaminant inventory of the lake catchment area, approximately 80% of the POPs contaminant inventory of the lake itself and was found to be approximately thirty times more efficient as a contaminant transport pathway for POPs compared to atmospheric long-range transport (Evenset et al., 2007). In a study by Desjardins et al. (2019), DP and other halogenated flame retardants were detected in guano of urban-adapted ring-billed gulls from a colony on Deslauriers Island near Montreal, Canada, and were similarly to the findings from Ellasjøen at Svalbard postulated to contribute to the augment exposure locally (Desjardins et al., 2019). In this study, the total amount of halogenated flame retardants deposited by the entire ring-billed gull colony (64,980 gulls, both sexes combined) through guano was estimated to 1 g during a 28-day period.

70. 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*-DP 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 the commercial mixture (Xiao et al., 2012; Salamova et al., 2014; Vorkamp et al., 2015), but has in other studies including air (Möller et al., 2010; Möller et al., 2011; Carlsson et al., 2018) and other environmental matrices (Carlsson et al., 2018; Na et al., 2015), been reported to differ from the commcerical mixtures. As discussed by Na et al. (2015), the f_{anti} values in samples collected by Möller et al. (2010) along the Atlantic transect (England-Spain-West-Africa-Antarctica) ranged from 0.3 to 0.63 and was 0.35 for most samples collected from the Southern Atlantic Ocean (Na et al., 2015) i.e.. the f_{anti} values were significantly lower than that of commercial DP produced by OxyChem for which f_{anti} values of 0.65-0.8 have been reported (Guerra et al., 2011), and lower than the f_{anti} value of 0.59 reported from DP produced by the Anpon facility in China (Wang et al., 2010a) and are consistent with long-range environmental transport. Carlsson et al. (2018) observed a lower fraction of *anti*-DP in air and water from the Longyearbyen settlement at Svalbard than in the technical mixture. More specifically, f_{anti} values of 0.5 were reported for both matrices as compared to a f_{anti} value of 0.75 for the commercial mixture. These findings were by the authors considered indicative of long-range transport as a source to the DP detected in the study, however, but the authors state that contribution from local sources cannot be excluded. Na et al. (2015) reported that the DP profile observed in surface seawater, sediment, soil, moss and reindeer dung collected near the Ny-Ålesund settlement at Svalbard is significantly different from that of the commercial product. Based on calculated f_{anti} values of 0.36, 0.21, 0.18, 0.27 and 0.43 for seawater, sediment, soil, moss and air, respectively, the authors propose that long-range atmospheric transport is the main source of DP in these matrices on Svalbard. Bird and reindeer dung had f_{anti} values of 0.67 and 0.66, a finding that the authors attribute to the migratory pattern of the birds and reindeers and changes in isomer ratios due to bioaccumulation/ biotransformation in the organisms. The DP levels reported from Svalbard in this study were significantly lower than those reported from areas in Europe and Asia with higher human activity and more local sources.

71. Studies reporting DP isomer fractions are also available from the Antarctic. Gao et al. (2018), report f_{anti} values in the range 0.16 to 0.58 and 0.10 to 0.44, respectively, for soil and lichen from Antarctica, with median values of 0.37 and 0.24. The authors suggest that the findings are consistent with DP being brought to the Fildes Peninsula in Antarctica by long-range atmospheric transport. However, the concentration of DP in soil and lichen differed between the sampling sites, indicating that the measured DP levels were also affected by anthropogenic and animal activities in

the area (Gao et al., 2018). 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. The sampling sites were in maritime Antarctica, which has a milder climate than continental Antarctica, and were distributed over a distance of 200 km. Based on geographical differences in DP levels and f_{anti} values, the authors proposed that long-range transport, human activities, melting glacier water and biological activities, e.g. from penguins, were possible sources of DP in the area (Kim et al., 2018).

2.3 Exposure

2.3.1 Environmental levels and trends

72. DP is ubiquitous in the environment (Sverko et al., 2011; Wang et al., 2016). Itis detected 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. On a global scale, the highest DP concentrations were generally detected in the US and China with concentrations up to several ng m^{-3}, close to the known production sites or e-waste treatment facilities (Sverko et al., 2011). DP levels in remote regions and in the oceans are generally much lower than levels reported in source regions near production sites and urban centres, although high levels in some instances have also been observed at remote sites (reviewed in Sverko et al., 2011; Canada, 2019a). As further detailed below, due to limited data of temporal and spatial trends for DP, the results are equivocal.

73. Several DP monitoring studies have been carried out in the vicinity of DP manufacturers and e-waste facilities in the US and in China (Sverko et al., 2011; Ji et al., 2018). Evidently, high levels of DP have been reported in various matrices from these areas, indicating that environmental levels are impacted by DP production (Wang et al., 2016). The soil concentration of DP near the production facility in China was 1200 ng/g and the amount decreased by an order of magnitude within 7.5 km (Wang et al., 2010c). A very high DP concentration (3327 ng/g) was found at an e-waste recycling site in Qingyuan, China, while DP levels fell dramatically with increasing distance away from the recycling site, suggesting that e-waste recycling activities are an important source of DP emissions and potential exposure (Yu et al., 2010). With the continuous phasing out of the production of decaBDE and the demand for alternatives, it is likely that DP and decabromodiphenyl ethane (DBDPE) will become the major halogenated flame retardants present in the Chinese environment in the near future (Zhu et al., 2014).

74. As indicated above, DP is detected in remote areas globally. The recent review of Arctic data conducted under the AMAP assessment of "Chemicals of Emerging Arctic Concern", 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 (see Table 3 in UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION]). For example, in a study investigating halogenated flame retardants 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 8.3 pg/g in air and soil, respectively, were reported (Liu et al., 2018). According to the authors, the concentrations of DP in this study were higher than in the Great Lakes region (0.14–4.0 pg/m^3) (Sverko et al., 2011), but were 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 (7740–26,700 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 20 to 1121 pg/g, with an average concentration of 318 pg/g, and decreased with increasing altitude. The concentrations reported in this study were comparable, but slightly higher than in tree barks 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). The Tibetan Plateau is the world's highest elevation plateau. It is considered as one of the most remote and isolated areas on earth, and ideal for the study of long-range transport of POPs (Yang et al., 2016a). Human activity in the area is limited and long-range air transport has been suggested as the predominant source of semi-volatile organic compounds in the area (Wang et al., 2010b; Yang et al., 2013; Yang et al., 2016a; Liu et al., 2018).

75. 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 dry weight (dw)) (Wang et al., 2012), in lichen in the southeast Tibetan Plateau (167 pg/g) (Yang et a., 2016a), and lower than in tree bark from areas in the northeastern US (0.03–115 ng/g) and Korea (1.4 ng/g), which are sites influenced by DP manufacturing facilities (Qiu and Hites 2008).

76. In a Norwegian screening assessment of emerging environmental pollutants in the Norwegian Arctic, DP was detected in all species monitored. The detection frequency 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-detectable to 6.9 ng/g wet weight (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). DP were 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 comparison, 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.

77. In a report by Vorkamp et al. (2019) investigating DP in air and biota from Greenland, *anti*-DP was detected in 92 % of all air samples from 2016 and in 46 % of the air samples collected in 2014. 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 lipid weight (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 was 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.

78. DP was also previously detected in air and biota samples from Greenland along with other flame retardants (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*-DP 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 of *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.

79. 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.

80. 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 ΣDP 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).

81. Aznar-Alemany et al. (2019) investigated DP and other dechloranes in dead seals from the South Shetland Islands on the Antarctic Peninsula. Only Dechlorane 602 and *anti*-DP were found. 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 (LOQ) in muscle, brain and fur. Dechlorane 602 appeared in greater concentrations than *anti*-DP, especially in adipose tissue. Reported average dechlorane 602 concentrations were 1.48 ±0.35 ng/g lw (range n.d.–1.72 ng/g lw) in fat of southern elephant seal and 1.92 ± 1.66 ng/g lw (range n.d.–3.19 ng/g lw) in fat of Antarctic fur seal. In other tissues of both species, dechlorane 602 levels was always below 0.55 ng/g lw.

82. 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). In addition, 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 (See Table 8 in UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION]). 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 ΣDP 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 long-range transport. 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).

83. 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) and (2011) observed decreasing concentrations of DP in air, but not in water, with increasing latitude. 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.

84. Temporal trend data for DP are equivocal. DP concentrations in gas and particle phase air samples in Harbin, Heilongjiang Province, China, from 2008 to 2013 were reported (Li et al., 2016). The results showed significantly increasing trends for both *syn*-DP and *anti*-DP. There was a significant difference in the doubling times between *syn*-DP ($t2 = 2.3 \pm 0.37$, $r = 0.39$, $p = 0.001$) and *anti*-DP ($t2 = 1.8 \pm 0.25$, $r = 0.43$, $p = 0.001$) but the doubling times of both DP isomers were shorter than reported for other flame retardants in the study, suggesting that DP levels increased more rapidly in the atmosphere. Strong local sources were identified based on the air parcel backward trajectories and the potential source contribution function (Li et al., 2016). In a study from North America, DP concentrations in air (vapor plus particle phase) were increasing during the period 2005-2013, with doubling times of 4–6 years at 3 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 (Liu et al., 2016). In a study from Canadian sites (Burnt Island, Egbert and Point Petre) in the Great Lake Basin, the levels of *syn*-DP and *anti*-DP in the atmosphere for the period 2005 – 2013 were decreasing, with a half-life of 3.4 years at Burnt Island and 14 years at Point Petre, which is closer to patterns seen in urban centers (Shunthirasingham et al., 2018). The declines have leveled off for these compounds after 2010, however this is not in line with other studies from the same region (Olukunle et al., 2018). While the levels of *syn*-DP were decreasing, with a half-life of 6.6 years at the Point Petre, they were increasing, with a doubling time of 7.6 years at the Burnt Island. The results from Burnt Island are consistent, with regrad to the observed increased levels of DP, with studies by Salamova and Hites (2011) who reported an increasing trend for the *syn*-DP with a doubling time of 9.5 years at US Great Lakes sites. However, although the investigated sites are in the same region, they are not identical.

85. When compared to the global data, levels at Chicago, Cleveland, and 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 and in a typical urban city in China (Li et al., 2016) may indicate that DP is being increasingly used, possibly as a replacement for BDE-209 (Sverko et al., 2011). In a study, using a multiple linear regression model of DP concentrations to isolate the variabilities due to sampling date and population near the sampling site, DP data of precipitation, vapor and particle phase from the Great Lakes where investigated in the period 2005 – 2015 (Olukunle et al., 2018). DP concentrations in all phases were changing as a function of sampling date, indicating that the input of this compound into the environment is continuing, presumably because its use and production are not regulated (Olukunle et al., 2018). On the other hand, 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 and sediment in the Great Lakes area are not clear, but as indicated by Vorkamp et al. (2018) it is possible that the matrices reflect different emission sources.

86. There is also some information on timetrends 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 1986–2014 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). In contrast, 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; however, the authors highlight that underlying timetrends may have been oscured by large intraspecies variation in concentration levels (Báron et al., 2015). No differences were found in white stork. In the same study the authors also report a higher BDE-209/ DP ratio in black kites in 1999 than in 2011. Although the difference was not significant, it suggests a decrease in BDE-209 levels relative to DP possibly reflecting a shift in the use of these flame retardants. In a study on 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. The finding was assumed to indicate a shift in the use of flame retarndants from PBDEs to other alternatives such as DP in China. A study reporting levels and trends of different flame retardants in the German and polar environments from 2005 to 2015/16 found declining trends for DP in some of the investigated biological matrices (i.e. tree leaves, tree shoots, herring gull eggs) and increasing trends in bream filet and blue mussel soft body (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 noted by the authors herring gulls feed in the marine environment, e.g. on blue mussels, but may also have a terrestrial diet. Although it is not clear to which extent the herring gulls investigated in this study feed in the terrestrial environment, transmitter-tracked female North Sea herring gulls have been shown to spend of 70% of their time in such costal terrestrial and urban habitats (e.g. harbors, food disposal facilities, recycling facilities and industrial parks), prior to the breeding season (Schwemmer et al., 2019). Thus, one possibility is that the time trends reported by Dreyer et al. (2019) reflect differences in DP emission and deposition between terrestrial and aquatic environments.

87. In a study with global coverage, Rauert et al. (2018) report air monitoring data for DP and other flame retardants from 48 global sites covering a range of location types (background sites (n=31), polar (n=4), agricultural (n=2), rural (n=5), urban (n=6) across all five UN regions. Samples were collected over a year in three-month periods using polyurethane foam passive air samplers. The study is part of the Global Atmospheric Passive Sampling (GAPS) programme and generally showed low detection rates for all flame retardants at background sites and lower mean concentrations and frequency for all novel flame retardants, compared to PBDEs (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 (19 in quarter one and four, 7.5 pg/m^3 in quarter two and 116 pg/m^3 in quarter three in 2014). In Paris, concentrations of all novel flame retardants detected were elevated in 2014 compared to earlier measurements from 2005. β-BDE-DBCH, BEH-TEBP and anti-DP dominated the profile. The syn-DP isomer was below method detection limits. In addition, reduced PBDE levels were observed. The increasing concentrations of the replacement flame retardants and the reduced levels of the PBDEs may be indicating the shift away from the use of PBDEs to alternative flame retardants in this urban area.

2.3.2 Human exposure

88. Humans may be exposed to DP from indoor dust, food, air, water, soil, sediment and breast milk. DP has been detected in 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 9 in UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION]). 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 kindergarten classrooms (mean 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*-isomers of DP were detected in 100% of car dust samples reaching 5 and 6 ng/g, respectively (Hassan and Shoeib, 2015). 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 ΣDP 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 was 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).

89. In foodstuff from Belgium, DP was found in 5% of the 1289 samples (firm cheese, quail eggs and pork, mean ΣDP levels were 339, 637 and 331 pg/g ww, respectively) (Poma et al., 2018). In 175 samples of 35 different foodstuff from the retail market in Korea, the mean ΣDP was in the range from ND-170 pg/g ww, and *syn*- and *anti*-DP was detected in 83.4 and 79.4% of the foodstuff, respectively (Kim et al., 2014). Daily intake of DP was estimated to 11.2×10^3 pg/day, which was 3 orders of magnitude higher than for other dechloranes. Furthermore, grain was the most contributing food group to daily intake for Korean population.The highest level of DP was found in bovine liver ΣDP 170 pg/g ww followed by spanish mackerel, oysters and dried anchovy with 135, 81 and 78 pg/g ww, respectively (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 ΣDP (Malak et al., 2019). In a market basket study of 123 food samples from Japan, DP was detected in Groups Ш (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 (11 pg/m^3) in Osaka city was comparable to that reported for urban areas in China despite no DP manufactoring facilities in Japan. The estimated daily intake of ΣDP (750 pg/day) via inhalation and diet was approximately one percent of that for ΣPBDE (62 ng/ day) (Kakimoto et al., 2014).

90. In a study comparing human exposure at a production facility and an e-waste facility in Southern China, high levels of DP were found in foodstuff 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 1270, 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). Estimated dietary intake dose accounted for more than 99% and 93% of the daily intake to people included in the study from the production site and e-waste site, respectively, which assumed that consumed food was produced locally (Wang et al.,

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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 site was 0.03 $\times 10^{-3}$ 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 flame retardants, including DP. High dietary intake of halogenated flame retardants (HFRs) via home-produced eggs in Baihe village was estimated revealed potential health concerns for local residents due to high consumption of PBDE (Huang et al., 2018). No temporal trend was observed for DP levels however, the highest levels of DP were observed in eggs from 2016, mean and SD was 5413 ± 1978 ng/g lw (Huang et al., 2018).

91. 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 ΣDP 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 ΣDP around 7 ng/g lw compared to mean ΣDP 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, while PBDEs steadily decreased with time (Kim et al., 2016).

92. 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).

93. 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 flame retardants 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 median 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 living in that city for <3 years (R(3) group) (Ben et al., 2013). For overview see Table 10 in UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION]).

94. 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).

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2.4 Hazard assessment for endpoint of concern

95. Available studies show that DP has the potential to elicit 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 defense and repair mechanisms. However, reactive oxygen species are important signaling molecules that could trigger unwanted effects by activating signaling cascades regulating cell death and cell survival and by causing damage to DNA, protein, telomeres, cells, tissues etc. Oxidative stress is linked to ageing and disease and could be the extra stressor that lead to "overload" in the wildlife with multiple stressors. As further described below, reported effects include oxidative damages to most tested organisms, neurotoxicity in zebrafish and carp (Chen et al., 2017; Li et al., 2019b) and potential for endocrine modulating effects in zebrafish (Kang et al., 2016) and humans (Ben et al., 2014; Guo et al. 2019) and immunemodulating effects in carp (Li et al., 2019b).

2.4.1 Toxicity to aquatic organisms

96. DP has very low water solubility (< 44 ng/l, see Table 2) and will partition to particles, sediment and biota in the aquatic environment. Observed levels in natural water may be higher than in pure rinsed water, e.g. in Laizhou Bay water, median DP was 11.7 ng/L for sum filtered and particulate, range of ΣDP in filtered and particulate water samples were nd to 10 ng/L and 0.890 to 346 ng/L, respectively (Zhen et al., 2018).

97. Toxicity data on sediment dwelling organisms are lacking, and the Canadian screening assessment used toxicity data on the DP analogue chlordane to fill the data-gap as a worst-case-scenario for sediment dwelling organisms and estimated a predicted no effect level (PNEC) of 0.0129 mg/kg dw for DP (Canada, 2019b).

98. Two studies on aquatic algaes indicate potentially negative effects on photosynthetic endpoints at low doses of DP (Gong et al., 2013; 2018). The freswater algae *P. subcapitata* was exposed to nominal concentrations of DP of 13.51, 135.1 and 1 351 ng/L for up to 72 h (Gong et al., 2013). Esterase activity had a significant but transient increased at 2h by the two highest doses and 135.1 ng/L at 24 h. ROS level was significantly increased at all doses at 48h (Gong et al., 2013). At the highest dose, average chlorophyll a content was significantly increased at 48 and 72h indicating a hormesis respons to DP exposure (Gong et al., 2013). In juvenile marine green macroalgae *U. pertusa* the enrichment of DP affected physiological responses of photosynthesis and caused oxidative stress (Gong et al., 2018). Following a 21-day uptake (DP doses of 10^{-8} to 10^{-6} mol/L equal to ~6.53 to 653 µg/L) and 21-day depuration, *anti*-DP was prone to bioaccumulate. ΣDP accumulated (after exposure to 10^{-8} mol/L) were 127 and 206 ng/g ww on day 7 and 21, respectively. 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 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 enrichment of DP (Gong et al., 2018). Marine macroalgae constitute the main source of biomass production and thus support the structure and function of the coastal marine ecosystem. Adverse effects on macroalgae may also have a potential impact on other levels of organisms.

99. 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 immediate after dosing were 0, 0.4 ± 0.3, 0.3 ± 0.2 and 0.7 ± 0.5 ug/L, and measured levels in mussel after 23 h exposure were 4.7±3.1, 8.8±2.1 and 21±9.1 ug/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. Gills were identified as the most responsive tissue. Lipid peroxidation in gills were found to increase by 82% and 67% 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% 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 give oxidative stress and add to the multiple stressors that mussels have to use energy on to compensate.

100. Four studies with embryo/larval zebrafish with waterborn 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 study. Kang et al. (2016) exposed to nominal concentrations of 0.4, 0.8 or 1.6 mg/L DP from 4 to 144 hours post fertilization (hpf) and measured water concentration of ΣDP was 140, 248, 267 µg/L DP at 0 h, and 28.8, 70.5 and 71.7 µg/L at 48 h, respectively. 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 up to 7 days (168 hpf).

101. 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 hyperactive response to dark-stimuli-activation at 120 hpf after exposure to 6.4 μM DP (4.18 mg/L). Chen et al., (2017) observed significant increase in spontane movements at 24 hpf for all 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 periode. 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 all doses 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 as reactive oxygen species (ROS) and MDA as well as apoptotic messenger ribonucleic acid (mRNA) markers were increased at the two highest doses (30 μg/L and 60 μg/L) (Chen et al., 2017).

102. Co-exposure of zebrafish embryos to 6 to 96 hours post fertilization (hpf) to DP (60 μg/L) and/or 3-methyl phenanthren (3-MP) (5 or 20 μg/L) 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). Relative ROS level in whole fish was significantly elevated compared to control from all exposure at 96 hpf, but no significant alteration was observed for co-exposure. As in the previous study, (Chen et al., 2017), significant increase in spontaneous movement occured at 24hpf with 60 ug/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 $Ca2^{+}$ homeostasis was investigated at 96hpf, and a significant synergistic increase was observed brain intracellular Ca^{2+} levels, and a synergistic reduction in Ca2+ 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%, respectiviely. 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.

103. Studies also show that DP can cross the blood-brain barrier in fish (Zhang et al., 2011) as well as in frogs (Li et al., 2014), 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 coutours, 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. 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). CYP1B1 gene expression was significantly upregulated at all timepoints and concentrations in both the liver and brain. In addition, were CYP2B and CYP3A1 and the apoptosis related factors bax and bcl-2 altered in liver. These results indicate that DP exposure perturbsmetabolism in the liver and brain, inhibits antioxidant enzyme activity, increases lipidperoxidation, promotes inflammation and induces cell apoptosis in juvenile carp (Li et al., 2019b).

104. In juvenile Chinese sturgeon treated with DP at doses of 1, 10, and 100 mg/kg ww for 14 days via a single intra peritoneal 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 induced apoptosis in the liver (Liang et al., 2014).

105. 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 increasei of SOD activity was observed for all doses 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 six, 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, had elevated levels at the lowest dose (0.3 μg/g). Plasma thyroxine (T4) concentrations increased along with up-regulation of corticotropin releasing hormone and TSH-β genes in brain (Kang et al., 2016), which are both involved in the fine-tuning of the thyroid hormone pathways in non-mammalian vertebrates (De Groef et al., 2006).

106. 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 flame retardants were found in developing embryos of female sharks, demonstrating maternal transfer *in utero* (Marler et al., 2018).

2.4.2 Toxicity in terrestrial organisms

107. Oxidative stress responses and neurotoxicity have been observed in earthworms (Zhang et al., 2014; Yang et al., 2016b). No acute toxicity was observed in earthworms exposed to DP at 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 responses is <0.1 mg/kg (ECHA, 2017b).

108. No overt 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).

109. 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.

110. 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. 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.

2.4.3 Human toxicity

111. Available assessments and laboratory studies with mammalssuggests that, DP is not carcinogenic, mutagenic or toxic to reproduction (as reviewed in ECHA, 2017b, c; Canada 2019a). Other potential adverse effects in mammals have however been reported as oxidative stress, liver imparement and endocrine effects as described below.

112. 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.

113. 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 (as summarized in ECHA, 2017b,c; Canada 2019, OxyChem 2004b). 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 of the substance. 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). Further toxicity testing has been required by ECHA following

a compliance check on the REACH registration dossier. The deadline for the registrant to provide these data is 21/12/2020 (ECHA website).

114. Wu et al. (2012) report liver impairments in mice at high-dose exposure. Following a 10-day oral exposure, oxidative stress and damage was induced 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. 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. Microarray analysis indicates that DP alters hepatic carbohydrate, lipid, nucleotide and energy metabolism as well as signal transduction processes.

115. 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, aspartate aminotransferase and creatine kinase were reduced significantly at highest dose, and serum glucose significantly increased. A non-significant increase in thyroid stimulating hormone (TSH) were also observed. 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.

116. 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 description of the studies are also limited. In a 13 weeks oral feeding study in rats, absolute and relative liver weight was significant increased in males only, at 500 mg/kg, but no associated histopatology observed (Oscarson, 1975, as described in Oxychem (2004b)). In a repeate-dose inhalation toxicity study, rats were exposed to 0, 640 or 1524 mg DP as dust/m^3, 6 h per day, 5 d per week for 28 days (Bishop 1975). Significant increase in absolute and relative liver weight was observed at both doses. Corresponding hepatocytomegaly (swelling of livercells 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. Effect of lungs 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 repeted-dermal study, male and female rabbits were dermal exposed to DP i 3% aqueous methylcellulose at 0, 500 or 2000 mg/kg bw d, for 5 days per week, in 4 weeks on 20% of the total body surface (Trzyna 1975). None to minemal erythrema occured at the aplication site. In females, 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 for highest dose. However, no histopatology effects were observed in any of the organs ((Trzyna 1975, as described in OxyChem 2004b).

117. In a combined repeated-dose/reproductive/developmental toxicity screening test, conducted according to OECD guidelines, 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). In the repeated dose toxicity (RDT) phase, 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 were observed across all dose groups including controls, which were linked to gavage administration errors. The authors identified a no-observable-effect level (NOEL) of 5000 mg/kg-bw/day. A significant increase in absolute and relative ovary weight was observed at 1500 mg/kg groupe, and a significant increase in absolute and relative thyroid/parathyroid weight was observed in females in the 750 mg/kg groupe treated for 28 days in the toxicity phase. A non-significant increase in 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 increase from 1500 mg/kg bw, mean pup weight increased 3, 15 and 13% compared to control for the 750, 1500 and 5000 mg/kg groups. This effect may be a result of strengthening of uterus function as a result of DPs' ability to activate the peroxisome proliferator-activated receptor PRAR-γ as described in paragraph below (Hewitt et al., 2006; Díaz et al., 2012; Peshdary et al., 2019).

118. A few studies have investigated possible association between DP and the thyroid hormone (TH) homeostasis and lipoproteins in humans. An association between serum TH and DP levels was reported in human mother-infant pairs near an e-waste recycling area in China (Ben et al., 2014). ΣDP concentration in maternal sera from the 20-year residents' group was 2 to 3-fold higher than the 3-year residents' group (geometric mean and range was 13.5, 1.28-900 and 3.68, 1.69-11.6 ng/g lw, respectively). *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). 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). 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). In another study comparing serum levels of adults living in an e-waste 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 iodothyroine deiodinase 1 were observed in highly exposed individuals (Guo et al., 2019). The *syn*- and *anti*-DP mean and range in serum was 57 (12-1000) and 58 (11-1450) ng/g lw in residents from the e-waste area, and 3.2 (0.36-12) and 5.9 (0.67-38) ng/g lw for the control group (Guo et al., 2019). Another study of pregnant women in Wenling, China observed no association between serum thyroid hormones and DP levels. The mean and range of ΣDP in serum were 13.9 (0.41-252) ng/g lw (Yin et al., 2020). However, a positive relationship between DP serum level and apolipoprotein A was observed (p= 0.095) for total DP and 0.045 for *syn*-DP (Yin et al., 2020). 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) in a study from US (Wang et al., 2020).

119. Adding to the above-mentioned indications of effects on sex- and thyroid hormones and liver metabolism, potential obesogenic effects have been observed for DP. In a mechanistic *in vitro* study, DP was shown to activate the peroxisome proliferator-activated receptor PRAR-γ and induce adipogenesis (observed as lipid accumulation and upregulation of adipogenesis mRNA and protein markers) of both murine and human preadipocytes. However, some DP-mediated adipogenic endpoints were independent of PPARγ activation, suggesting that other potential modes of actions of DP may be involved (Peshdary et al., 2019).

2.4.4. Other concerns

120. 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 81 for details). DPMA may also be under-reported because of destructive sample preparation methods that may degrade it (Rjabova et al., 2016). Based on predictive models, DPMA is potential both bioaccumulative and very persistant on the basis of quantitative structure activity relationship screening. No information is available on its toxicity to human and the environment, but due to its structural similarities 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 in see Figure 1 and Table 11 in UNEP/POPS/POPRC.16/INF/… [NOTE: INF IN PREPARATION])

3. Synthesis of information

121. DP has been detected in remote regions far from sites of production and use, including in the Arctic, the Antarctic and high-mountain regions in Tibet. Available monitoring data provides evidence for ongoing long-range environmental transport 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 comparable to listed POPs and, in some studies, reported to be in the same range as in source retions. Furthermore, DP is currently a high production volume chemical that is marketed as a replacement for already regulated flame retardants, suggesting that use could increase in the future. Due to the continued and possibly increasing input of DP to remote regions, concentrations may increase and could lead to adverse effects in the future if no action is taken.

122. Although DP has been produced and used for almost half a century, its first environmental detection was reported in 2006 in the Great Lakes Basin in North America (Hoh et al., 2006). Since then has its global ubiquitous distribution has confirmed (Wang et al., 2016). Due to limited data temporal and spatial trends for DP are equivocal; while increasing trends are observed in some studies (Liu et al., 2016; Li et al., 2016; Vorkamp et al., 2018) other studies report unchanged levels (Olukunle et al., 2018).

123. Available toxicity data indicates concern for potential adverse effects to the environment and humans due to oxidative stress that impacts several biological processes.

124. Adding to the concern for adverse effects is the lack of toxicity information on DP by-products such as 1,3- or 1,5-Dechlorane Plus monoadduct (DPMA) that in some cases is detected in greater levels than DP in environmental samples (Sverko et al., 2011; Tomy et al., 2013; Guerra et al., 2011), and the structural similarity of DP to already listed substances such as aldrin, mirex, and chlordecone. Furthermore, mixture toxicity effects that could increase DP toxicity and bioaccumulation andvisa versa for other pollutants have been indicated (Chen et al., 2019).

125. 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 e-waste recycling sites and production plants. However, DP is also detected in dust, sludge and wastewater from WWTPs at sites far from production sites and e-waste recycling sites, indicating emission and exposure from consumer products.

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126. Due to the POP properties and risks related to its widespread production and use, international action is warranted to control further release of DP.

Table 5. POP characteristics of DP and its *syn*- and *anti*-isomer

Criterion	Meets the criterion (Yes/No)	Remarks
Persistence	Yes	• Modelling of degradation potential and microbial metabolic pathways which suggests that biodegradation is slow, and low probability that DP will degrade any faster than structural analogues that are listed under the Stockholm Convention (ECHA, 2017b; Zhang et al., 2016) • Limited degradation in ageing soil with only 4.2-8.2% of initial DP having degraded after 260 days (Cheng et al., 2019) • Lack of degradation in soil over time (Wang et al., 2010a) and low ability to biotransform in fish (Tomy et al., 2008) support the conclusion that DP is persistent
Bio-accumulation	Yes	• Log K_{ow} of 9.3 (OxyChem, 2007) indicating high potential for bioaccumulation • Depuration half-life, which is corresponding to a BAF > 5000 (Tomy et al., 2008; Tang et al., 24.018; Li et al., 2013b) • BMFs/TMFs > 1 reported for several species and food webs (Tomy et al., 2007; Wu et al., 2018; Sun et al., 2015, Sun et al., 2017; Kurt-Karakus et al., 2019; Na et al., 2017)
Potential for Long-Range Environmental Transport	Yes	• DP is detected in the environment and biota in remote areas; in the Arctic, Antarctic and a remote mountain regtion in Tibet (Canada 2019a; AMAP 2017; Möller et al. 2010, 2012; Gao et al. 2018; Kim et al. 2018; Na et al. 2017; Yang et al. 2016; Liu et al. 2018). • Long-range environmental transport is facilitated by DP binding to particles in the atmosphere and in seawater (e.g. Möller et al., 2010; Möller et al., 2011; Möller et al., 2012; Sverko et al., 2011; Canada 2019a), and may possibly also occur via migrating birds (Vorkamp et al. 2018). • Concentrations in air decrease with increasing latitude (Möller et al. 2010, 2011).
Adverse effects	Yes	• Neurotoxicity as observed in zebrafish (Chen et al., 2017; 2019) and carp (Li et al., 2019b). • Potential for endocrine modulating effects as observed in zebrafish (Kang et al., 2016) and humans (Ben et al., 2014; Guo et al. 2019; Peshdary et al., 2019). • Immunemodulating effects as observed in carp (Li et al., 2019b). • Oxidative stress as observed in fish (Chen et al., 2017; Hang et al., 2013; Kang et al., 2016; Li et al., 2019b), 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).

4. Concluding statement

127. 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 has the potential to undergo long-range environmental transport, making emissions and releases of this substance a transboundary pollution problem including in remote areas. Globally, DP is detected in humans, wildlife and environmental samples in all global regions, including measurements in the Arctic and Antarctic.

128. In humans, DP has been detected in umbilical 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. However, DP is also detected in dust, sludge and wastewater from WWTPs at sites far from production sites and e-waste recycling sites, indicating emission and exposure from consumer products.

129. Available scientific literature suggests that there is a potential risk for adverse effects for developing aquatic species and organisms due to oxidative stress that that affects important biological processes. The concern for adverse effects furthermore relates to observed effects on the liver, endocrine modulating effects, in particular to thyroid hormone system and PPAR to both human health and the environment, as well as neurotoxic and neurodevelopmental

effects in aquatic organsims. Furthermore, observed effects on photosynthetic activity in marine macroalgae suggest possible impacts on aquatic ecosystems mediated by negative effects on primary production.

130. Mixture toxicity effects such as increased toxicity and bioaccumulation have been indicated for DP and suggest an additional cause for concern. Furthermore, the combined effect of climate change and other environmental stressors adds to the risk posed by environmental pollutants by affecting the vulnerability and adaptability of organisms as shown for Arctic organisms.

131. Based on the persistence, bioaccumulation, adverse effects observed in organisms and the widespread occurrence in environmental compartments including at remote regions, it is concluded that DP and its *syn*- and *anti*-isomers are likely, as a result of their long-range environmental transport, to lead to significant adverse human health and environmental effects such that global action is warranted.

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