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Persistent Organic Pollutants Review Committee
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Item 5 (b) (i) of the provisional agenda*

Technical work: consideration of chemicals proposed for listing in Annexes A, B and/or C to the Convention: Dechlorane Plus (CAS No. 13560-89-9) and its syn-isomer (CAS No. 135821-03-3) and anti-isomer (CAS No. 135821-74-8)

Proposal 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) in Annexes A, B and/or C to the Stockholm Convention on Persistent Organic Pollutants

Note by the Secretariat

I. Introduction

1. Norway has submitted a proposal 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) in Annexes A, B and/or C to the Convention pursuant to paragraph 1 of Article 8 of the Convention (see annex to the present note). The proposal is being circulated as submitted and has not been formally edited. The Secretariat's verification of whether the proposal contains the information specified in Annex D to the Convention is set out in document UNEP/POPS/POPRC.15/INF/6.

II. Proposed action

2. The Committee may wish:

(a) To consider the information provided in the present note;

(b) To decide whether it is satisfied that the proposal fulfils the requirements of Article 8 of and Annex D to the Convention;

(c) To develop and agree on, if it decides that the proposal fulfils the requirements referred to in subparagraph 2 (b) above, a workplan for preparing a draft risk profile pursuant to paragraph 6 of Article 8.

* UNEP/POPS/POPRC.15/1.

Annex

Proposal 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) in Annexes A, B and/or C to the Stockholm Convention on Persistent Organic Pollutants

1. Introduction

1. The commercial mixture Dechlorane Plus (DP) is a polychlorinated flame retardant that has been in use since the 1960s. Since then, research has confirmed its global ubiquitous distribution (Wang et al., 2016) and several studies have shown increasing or stable time trend of DP (Olukunle et al., 2018; Liu et al., 2016; Vorkamp et al., 2018; Li et al., 2016).

2. 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 non-plasticizing flame retardants in polymeric systems such as nylon and polypropylene plastic (reviewed in Wang et al., 2016; OxyChem, 2019). This nomination report specifically addresses the information requirements and screening criteria of Annex D to the Stockholm Convention on Persistent Organic Pollutants and summarizes relevant evidence relating to the screening criteria for persistence, bioaccumulation, adverse effects and long-range transport.

Chemical identity

3. The "Dechlorane Plus"™ (CAS No. 13560-89-9) is a commercially available formulation that contains two stereoisomers, syn-DP (CAS No. 135821-03-3) and anti-DP (CAS No. 135821-74-8). 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 65% anti-DP and 35% syn-DP (OxyChem, 2013).

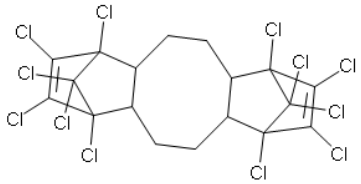
4. The chemical identity of DP and the modelled and experimental physicochemical properties for its two isomers are listed in tables 1 and 2, below.

Table 1. Chemical identity of Dechlorane Plus.

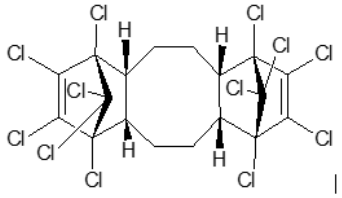
CAS number:	13560-89-9, 135821-03-3, 135821-74-8
IUPAC name:	1,6,7,8,9,14,15,16,17,17,18,18-dodecachloropentacyclo-[12.2.1.16,9.02,13.05,10] octadeca-7,15-dien
EC number:	236-948-9
EC name:	1,6,7,8,9,14,15,16,17,17,18,18-Dodecachloropentacyclo[12.2.1.16,9.02,13.05,10] octadeca-7,15-diene
Molecular formula:	$C_{18}H_{12}Cl_{12}$
Molecular weight:	653.73
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;
Trade names	Dechlorane Plus 25 (Dech Plus); Dechlorane Plus 35 (Dech Plus-2); DP-515; Dechlorane 605; Dechlorane A; DP; DDC-CO¸ Escapeflam DK-15 (China)

Table 2. Overview of selected physicochemical properties.

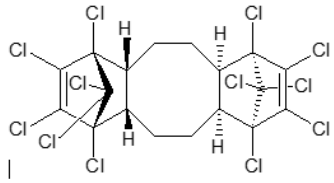
Property	Value	Reference
Physical state at 20°C and 101.3 kPa	Solid white powder	ECHA, 2017
Melting/freezing point	340–382°C 350°C	ECHA, 2017 OxyChem datasheet (2007)
Vapour pressure	0.006 mm Hg (at 200°C)	OxyChem datasheet (2007)
Water solubility	<1.67 ng/L (20–25°C) 0.044 µg/L–249 µg/L (insoluble)	ECHA, 2017 OxyChem datasheet
n-Octanol/water partition coefficient, K_{OW} (log value)	9.3	OxyChem datasheet (2007)
Octanol-air partition coefficient K_{OA} (log value)	12.26	OxyChem datasheet (2007)
Sediement/water partition coefficient K_P (log value)	6.65	OxyChem datasheet (2007)



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



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



Anti- (or exo) Dechlorane Plus (Cas No. 135821-74-8)

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

2. Global consumption and use

5. DP is classified as a low production volume chemical in the European Union (EU), while it is a high production volume chemical in the United States (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 recently been reported to be 300–1000 tonnes per year by Anpon since 2003 (Wang et al., 2010a). The global annual production approximates 5000 tonnes (Ren et al., 2008). Based on a survey conducted for the year 2011, between 1 to 10 tonnes of DP, including DP in some products and/or manufactured items, were imported into Canada (Canada, 2016). DP is registered

(March 2019) under REACH by two Dutch companies at an estimated tonnage of 100 – 1000 per annum. Sweden registered a use of 5 tonnes in 2006, after a decrease from 11 tonnes in 2005 (Kaj et al., 2010).

6. 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) but has also been marketed 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). According to information provided in a Canadian assessment, DP is used in Canada as a flame retardant in automobile manufacturing (Canada, 2016).

7. 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 up to 40 % in silicon rubber (OxyChem, 2007). According to manufacturer literature (OxyChem, 2007), DP is manufactured for use solely by industrial customers.

3. National and international administrative actions

8. In 2018, DP was identified as Substances of Very High Concern (SVHC) and added to the REACH Candidate List due to their persistent and bioaccumulative properties (ECHA, 2017a). Inclusion on this list means the substances can be subject to further review and only used for specific authorized purposes under strictly controlled conditions. Moreover, upon request industry is obliged to inform consumers on the occurrence of the listed substances in consumer articles above a concentration of 0.1%. In September 2018, DP was included in European Chemicals Agency (ECHA)'s Draft Recommendation of Priority Substances to be included in Annex XIV of the REACH Regulation (List of Substances Subject to Authorization). The Public consultation closed in December 2018 and the final decision on including DP in Annex XIV will be made by the Commission after voting in the REACH Committee. There is no harmonised classification for Dechlorane Plus (CAS No. 13560-89-9) in the EU, but 78 notifiers have classified the substance as Acute Toxicity Category 4, H332 Harmful if inhaled (ECHA, 2018). In Norway DP was recently (January 2019) added to the national list of priority substances with a national goal to phase out the use by 2020 (Norwegian Environment Agency 2019).

9. DP is listed on Canada's Domestic Substances List (DSL), and has been manufactured or imported in quantities of 0.1 ton or more (Environment and Climate Change Canada (ECCC), 2019a). An updated screening assessment report is expected to be published in Spring 2019 in Canada. If the assessment concludes that DP meets criteria for toxicity, the ECCC will propose to amend the regulations to prohibit the manufacture, import, use, sale and offer for sale of DP and all products containing the substance (ECCC, 2019b).

10. In the United States, 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 US EPA with production, import and use volumes, as well as other relevant information (as referenced in Canada, 2016.

4. Information on DP and how it fulfils the Annex D screening criteria

4.1 Persistence

11. 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 and therefore hydrolysis is not expected to be a relevant degradation process (Canada, 2016; ECHA, 2017b). Photodegradation in water was determined in a study utilizing a light source that provided high photon fluxes in the solar spectral region above 290 nm. In the study, DP was exposed to light for 168 hours, after which the photolysis half-life was calculated from the analytical results. The photolytic half-life in water was estimated to be >24 years (Chou et al., 1979; OxyChem, 2004; Sverko et al., 2011).

12. Degradation of DP in aging soils have 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). Model simulations

were then developed to investigate the transformation. The model simulations indicated that transformation rates were inconstant and distinguishable 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).

13. DP was also found to be persistent in suspended sediment with a half-life approximately 17 years (Sverko et al., 2008). Lake sediment was used in a 6-week study to determine the extent of partitioning of DP from water into soil. The calculated sorption partition coefficient (K_P) was 4.5×10^6 (+/- 1.9×10^6) and DP preferentially adsorbed to the sediment. In three aerobic and one anaerobic biodegradation studies, DP was tested for degradation in water incubated with sewage sludge organisms for up to 6 weeks. Either DP was not biodegradable or minimally biodegradable (with CO_2 as a possible metabolite), or its extremely low water solubility prevented the bacteria in the domestic sewage from contacting and degrading the test substance (OxyChem, 2004).

Conclusion on persistence according to the criteria in Annex D

14. DP fulfils the criteria for persistence with half-lives in water greater than two months (>24 years) and half-life in sediment and soil greater than six months (180 days). Furthermore, DP was recently identified as a very persistent (vP) substance in the EU (ECHA, 2017a).

4.2 Bioaccumulation

15. DP was first detected in archived fish (walleye) from Lake Erie in 2006, which suggested that DP was at least bioavailable (Hoh et al., 2006). Thereafter, different levels of DP were obtained in various biota samples including the human body (Reviewed in Wang et al., 2016), confirming bioavailability of DP in both the terrestrial and aquatic biotas.

16. The log octanol-water partition coefficient (log K_{OW}) for DP is reported to be 9.3 (OxyChem, 2004). The high log K_{OW} and the very low water solubility (Table 2) indicate that DP is very hydrophobic and therefor partition into organic matter such as binding to particles. This is further supported by the relatively high sediment-water partition coefficient K_P of 6.65 (log value). These properties make aqueous laboratory studies very difficult to carry out reliably due to the potential difficulties in maintaining stable exposure levels at concentrations under the water solubility.

17. Chemicals are bioaccumulative when the bioaccumulation factor (BAF) is greater than 5000 (Kelly et al., 2007) corresponding to a log BAF of 3.70. Aqueous exposure is expected to be of limited importance in terms of bioaccumulation potential, and whilst significant uptake has been shown to occur in fish (Guo et al., 2017; Malak et al., 2018; Kurt-Karakus et al., 2019), there are currently few reliable bioconcentration factor (BCF) data from fish (ECHA, 2017b). However, in a recent study, BAF in aquatic organisms in an urban catchment in Singapore were reported. DP isomers exhibited relatively high BAF values, exceeding the BAF threshold (i.e. BAF > 5000 L/kg on a wet wt. basis) with the anti-DP BAF-value ranged from 1.7×10^5–3.0×10^9 L/kg wet weight, which indicates great bioaccumulation potential of anti-DP (Wang and Kelly, 2018). Furthermore, the bioaccumulation potential of both syn- and anti-DP were investigated in amphipods in the Norwegian Arctic (Carlsson et al., 2018) and the DP isomers (syn- and anti-) showed high log BAFs (amphipod-water) at 8.9 and 9.1, respectively indicating possible bioaccumulation.

18. In a dietary bioaccumulation laboratory test using juvenile Rainbow Trout (Oncorhynchus mykiss), BMF>1 (5.2 for the syn- and 1.9 for the anti-isomer) was reported (Tomy et al., 2007). In the same study, a 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. A depuration half-life above around 8-10 days is suggestive of a lipid-normalised and growth-corrected BCF above 5 000 L/kg according to the analysis by the Environment Agency in UK (EA, 2012).

19. Several field studies report biomagnification factors (BMF) and trophic magnification factors (TMF) in various food webs. A number of 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). Stereoselective bioaccumulation has been observed in some studies but for all studies available, BMFs or TMFs are above 1 regardless of isomer (anti- or syn-) (Tang et al., 2018; Na et al., 2017; Wu et al., 2010).

20. TMF>1 for both DP isomers in a Chinese fresh water food web has been reported (Wu et al., 2010). Furthermore, Tomy et al. (2007) investigated food web samples from to Canadian Lakes, the trophic level-adjusted BMFs ranged between <0.1–11 (BMF 11 for anti-isomer) in the Lake Winnipeg food web and between 0.1–12 (BMF 12 for syn-isomer) in the Lake Ontario food web. In a recent study from Lake Ontario, tropic level adjusted TMFs > 1 for DP (syn- and anti) was reported in a pelagic food web containing invertebrates, forage fish and lake trout (Kurt-Karakus et al., 2019).

Trophic magnification was reported in a fresh water food web from a highly contaminated site in South China, and TMFs of syn-, anti- and total DP were in the range of 6.5 to 11.3 (Wu et al., 2010). Trophic magnification in an aquatic food chain that receives discharges from the DP production facility in China was investigated. Sample numbers were low, but a TMF of 1.9 for anti-DP was reported (Wang et al., 2015). In a study of biota samples from an estuary in South China, TMF for DP was determined to be 2.31 and estimated BMFs for various predator-prey combinations were in the range 1.27 to 11.8 (Sun et al 2015). In a more recent study, BMFs for DP from a similar food web sampled in the South China Sea were reported to be in the range 2.3–7.1 (Sun et al., 2017). BMF ranging 1.02–1.23 in serum was observed for DP after reaching steady-state in a carp exposure study (Tang et al., 2018). In tissues (carcass, liver, gonad, gastrointestinal track, and gill) were steady-state were not reached, BMFs were below 1. Trophic magnification was investigated in marine food webs (algae, limpet, starfish, gammarid, krill, cod, penguin, seal, and stercorarius) in Antarctica (Na et al., 2017). Trophic level adjusted magnification was observed in the range 2.87–3.34 with anti-DP having the highest magnification potential. In this study biomagnification of PCBs were also investigated and for comparison, DP magnification was similar to that of highly chlorinated PCBs (Na et al., 2017).

21. Bioaccumulation and maternal transfer have been investigated in amphibians collected from a highly contaminated site in South China (Wu et al., 2018). Amphibians occupy an important trophic position in the food chain 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 the present study (Wu et al., 2018).

22. Available monitoring studies show that DP are widely dispersed in the environment (reviewed in Canada, 2016). Evidence from around the world shows that uptake can clearly occur in a wide range of wildlife species throughout freshwater and marine aquatic food webs (reviewed in Wang et al., 2016). Field monitoring data suggest that DP is bioavailable and can achieve a relatively high body burden in some cases that may be expected to be associated with toxic effects (ECHA, 2017b).

23. DP is bioavailable to humans and has also been detected in several human tissues (reviewed in Wang et al., 2016). 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).

Conclusion on bioaccumulation according to the criteria in Annex D

24. BAF over 5000 and log K_{OW} > 5 show that DP fulfils the criteria on bioaccumulation under Annex D of the Convention. BMFs and TMFs > 1 have been reported for a number of organisms and food webs and this further support the conclusion that DP is bioaccumulative. Furthermore, DP was recently identified as a very bioaccumulative (vB) substance in the EU (ECHA, 2017a).

4.3 Potential for long-range environmental transport

25. Due to its low vapour pressure and a high log octanol-air partitioning coefficient (K_{OA}), DP is expected to be associated with particles in the atmosphere (reviewed in de Wit et al., 2011 and Sverko et al., 2011). This expectation is confirmed by monitoring studies which have reported mean fractions of particle bound DP in air ranging from 79 to as high as 97 % and 99 % (Hoh et al., 2006; Ren et al., 2008; Möller et al., 2010). Sorption to particles may slow down reaction rates and therefore the actual half-lives in the air may be greater, facilitating the long-range transport of DP on particles (reviewed in Sverko et al., 2011). Other chemicals with similarly low vapor pressures, such as decabromodiphenyl ether (decaBDE), show significant long-range atmospheric transport potential (Breivik et al., 2006). Modelling data reported in Sverko et al., 2011 indicate that DP have transport and persistence properties similar to many of the listed persistent organic pollutants (POPs). DP is found predominantly in the particulate phase also in water (Möller et al., 2010).

26. Monitoring data suggest that DP is a global pollutant that can undergo long-range atmospheric transport to remote areas where it is detected high frequency (reviewed in AMAP, 2017; see also Möller et al., 2010; 2011a,b; 2012; Na et al., 2015; Yang et al., 2013b; Wang et al., 2010b). Detection in Arctic media include air, snow, soil, sediment water and biota (reviewed in AMAP, 2017). In a Norwegian screening assessment of various species (birds, mink and polar bears) sampled at Svalbard, DP was detected in all species at 80–100 % frequency in 2017 (Norwegian Environment Agency, 2018). DP have also been reported in various environmental matrixes in Antarctica (soil, lichen, mosses), with detection frequencies of 100 % (Gao et al., 2018; Kim et al., 2018). In a remote mountain area in Tibet, DP was detected in 89 % of all lichen samples collected (Yang et al., 2016).

27. Findings by Möller et al. (2010) suggests that DP is susceptible to long-range transport via particles in both air and water. In this study, marine boundary layer air and surface seawater was

sampled during a cruise from the East Greenland Sea and in the Northern and Southern Atlantics toward Antarctica. As discussed by Na et al. (2015), the fraction of anti-DP (f_{anti}) in samples collected by Möller et al., 2010 along the Atlantic transect (England-Spain-west-Africa-Antarctica) ranged from 0.63 to 0.3 and was 0.35 for most samples collected from the Southern Atlantic Ocean (see Na et al., 2015). 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). According to Möller et al., 2010, these findings reflect a stereoselective depletion of anti-DP likely caused by UV degradation during long-range transport. Möller et al., 2010 also observed decreasing air concentrations of DP with increasing latitude. This trend was not reflected in the seawater samples. The analysis of air mass back trajectories showed the influence of 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 (Möller et al., 2010). Both water and air concentrations were higher at stations near the coast than in the open sea. Decreasing DP concentrations with increasing latitude were also observed in the Chukchi/Bering Sea, however, concentrations were near detection limits (Möller et al., 2011a). The authors also observed correlations with PBDEs, possibly indicating the same source regions as for DP.

28. As reviewed by AMAP (2017), DP isomer ratios in Arctic air have in some studies been reported to be in the same range as found in the commercial mixture (Xiao et al., 2012a; Salamova et al., 2014; Vorkamp et al., 2015). More recently, Carlsson et al., 2018 observed a lower fraction of anti-DP in air and water from the Longearbyen settlement at Svalbard than in the technical mixture. More specifically, f_{anti} values of 0.5 were reported for both matrixes as compared to an f_{anti} value of 0.75 for the commercial mixture. The findings are consistent with long-range transport of DP to the Arctic, however according to the authors, contribution from local sources cannot be excluded. In another study from Svalbard, 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 is significantly different 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 the authors propose that long-range atmospheric transport is the main source to DP in these matrixes at Svalbard. Bird and reindeer dung had f_{anti} values of 0.67 and 0.66, a finding that the authors attributes 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.

29. In the study by Gao et al., (2018), DP was widely detected in soil and lichen in various locations on the Fildes Peninsula in Antarctica. Calculated fractions of anti-DP (f_{anti}) ranged from 0.16 to 0.58 and from 0.10 to 0.44, in soil and lichen respectively, with median values of 0.37 and 0.24, and were consistent with DP being brought to the peninsula by long-range atmospheric transport. 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 Georg 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 to DP in the area (Kim et al., 2018).

30. 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., 2016). Human activity in the area is limited and contamination levels of several POPs is low (Yang et al., 2013b; Wang et al., 2010b). Long-range air transport has been suggested as the predominant source to semi-volatile organic compounds in the area. In a study investigating halogenated flame retardants in air and soils from Mt. Gongga on the eastern Tibetan plateau, DP levels from below detection limit to 11.5 pg/m^3 and 8.3 pg/g in air and soils, 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 was 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 a study on lichen from the southeast Tibetan Plateau, DP levels in the range from 20-1121 pg/g were detected (Yang et al., 2013b). The average concentration was 318 pg/g and DP concentrations seemingly decreased with increasing altitude. According to the authors the distribution pattern for DP observed in this study and

knowledge of the monthly average surface wind vector field in the study area found in an earlier study (Yang et al., 2013b) 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., 2016a). The authors 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 in the area. 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 United States (Qiu and Hites, 2008) but significantly higher than in moss from Ny-Ålesund at Svalbard (Na et al., 2015).

Conclusion on long-range environmental transport according to the criteria in Annex D

31. Long-range environmental transport is a source to DP detected in the Arctic, in the Antarctic and in remote alpine ecosystems on the Tibetan plateau. However, DP levels in these areas are also influenced by human and animal activities in the respective areas, as well as melting ice. DP has a low vapour pressure and a high log K_{OA}. Similar to other extremely hydrophobic non-volatile compounds such as e.g. decaBDE, DP can be transported over long geographical distances in association with airborne particles under dry conditions and high wind speeds (Brevik et al., 2006, Cao et al., 2014, Hoh et al., 2006)

4.4 Adverse effects

32. Although it has been produced for several decades, DP was first identified as an environmental pollutant of concern in 2006 (Hoh et al., 2006). Toxic effects have been less studied but shows that DP has the potential to elicit toxic effects in different organisms. As further described below, reported effects include oxidative stress and oxidative damages, neurotoxicity and potential for endocrine disruption. DP has also been reported to cross the blood-brain barrier and to be maternally transferred to off-spring in several species. Available acute toxicity studies in experimental animals suggested low concern for acute toxicity via oral, inhalation or dermal routes of exposure (ECHA, 2017b; Canada 2016). However, for bioaccumulative and persistent compounds chronical exposure would be more appropriate to describe potential hazard to the environment and humans.

33. DP has low water solubility, and will in the aquatic environment partition to particles, sediment and biota. Toxicity data on sediment dwelling organisms are lacking, but recent publications on effects in green macro algae and bivalves indicate that DP induce oxidative stress and damages to these primary producers in the ecosystem (Gong et al., 2018; Gagne et al., 2017; Baron et al., 2016). In one instance, potentially negative effects on photosynthetic endpoints was reported at low doses (Gong et al., 2018). Oxidative stress has also been observed after exposure to DP in fish (Chen et al., 2017; Hang et al., 2013; Kang et al., 2016), marine bivalves (Baron et al., 2016; Gagne et al., 2017), earthworm (Zhang et al., 2014; Yang et al., 2016), bird (Li et al., 2013a) and mice (Wu et al., 2012), and are further described below.

34. Marine bivalves (*Mytilus galloprovincialis*) orally exposed to DP-concentrations of 0, 5.6, 56, and 100 µg/L for 6 days by spiked agarose, showed DNA strand-breaks in hemocytes for all doses. No clear dose response was observed, although all doses were significantly different from negative control; 13, 23 and 18%, respectively ($p<0.001$). Increased micronuclei formation was observed only for highest dose (100 ug/L) tested (Baron et al., 2016). In a more long-term study, fasting blue mussels (*Mytilus edulis*) were exposed to 0, 0.001, 0.01, 0.1 and 1.0 µg DP/L for 29 days. 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 ug/L), while cyclooxygenase activity (COX) was significantly decreased by 44% at the 1 µg/L dose ($p=0.002$) (Gagne et al., 2017). The effect on COX could have impact on the control of spawning process and thus reproduction in blue mussels (Matsutani and Nomura, 1987).

35. Short time 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). The 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 at al doses (15, 30 and 60 µg/L at 96 hours post fertilization). Axonal growth-related gene-expression was altered in highest dose (60 µg/L), oxidative stress markers as reactive oxygen species (ROS) and malondialdehyde (MDA) as well as apoptotic mRNA markers were increased by the two highest doses (30 µg/L and 60 µg/L) (Chen et al., 2017).

36. In adult zebrafish receiving 3 µg/g by gavage twice in a 6 days study, increased catalase activity was observed in liver indicating oxidative stress response (Kang et al., 2016). In juvenile Chinese sturgeon (*Acipenser sinensis*) treated with DP at doses of 1, 10, and 100 mg/kg wet weight

(ww) for 14 days, 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 (Lian et al., 2014).

37. Oxidative stress responses and neurotoxicity has been observed in earthworm (Zhang et al., 2014; Yang et al., 2016b). No acute toxicity was observed in earthworms (*Eisenia fetida*) 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 as such as superoxide dismutase (SOD), MDA, glutathione, glutathione-peroxidase, catalase and 8-OHdG in tissue as well as tail DNA in comet assay of isolated coelomocytes. Furthermore, acetylcholinesterase (AChE) and cellulase activity of earthworms was reduced even by the low dose indicating potential neurotoxic effect in earthworms (Yang et al., 2016). The overall 28-day no-effect concentration (NOEC) for these responses would be <0.1 mg/kg (ECHA, 2017b).

38. According to available assessments, laboratory studies with mammals suggests that DP is not carcinogenic, mutagenic or toxic to reproduction (as reviewed in ECHA, 2017b; Canada, 2016). Other effects in mammals has however been reported.

39. Wu et al. (2012) report liver impairments in mice at high-dose exposure. Following 10 days' oral exposure, oxidative stress and damage was induced in male mouse livers at all doses (500, 2000 or 5000 mg/kg-bw per day), with altered hepatic carbohydrate, lipid, nucleotide and energy metabolism as well as signal transduction processes.

40. In a 90-days oral study with rats exposed to DP at 0, 1, 10 or 100 mg/kg/day, 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 and thyroid stimulating hormone (TSH) were increased. In the liver, gene-expression of several key enzymes were altered (Li et al., 2013b). However, no significant changes in absolute body or liver weight or liver histopathology was observed.

41. Some indication of endocrine disruption potential of DP has been observed. In a short-time oral exposure study of adult zebrafish, transcriptional responses of both thyroid and sex hormone related genes in brain were observed, suggesting possible thyroid and sex hormone disrupting potentials of DP (Kang et al 2016). The fish were given doses of 0, 0.3, 1, or 3 μg/g of zebrafish ww by gavage on day 0 and 2. On day six of the experiment, induction of Cyp 19b (brain type of aromatase) was observed at all doses at body residual levels that were environmentally relevant. In addition, brain estrogen receptor alpha mRNA-levels were elevated at the lowest dose (0.3 μg/g). Plasma thyroxine (T4) concentrations increased along with up-regulation of corticotropin releasing hormone and TSH-b genes in brain (Kang et al., 2016), which both are involved in fine-tuning of thyroid hormone pathway in non-mammalian vertebrates (De Groef et al., 2006).

42. An association between serum thyroid hormone 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-years residents' group was 2–3 fold higher than the 3-years residents group (geometric mean and range was 13.5, 1.28–900 and 3.68, 1.69-11.6 ng/g lipid weight, 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 thyroid stimulating hormone was significantly lower in maternal serum in the group that had lived 20 years in the area (n=44), than for those whose mothers had been residents to the area for 3 years (n=22), (p=0.046). When concentrations of polybrominated diphenylethers was 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).

43. Adding to the above-mentioned indications of effects on sex- and thyroid hormones and liver metabolism, potential obesogenic effects have been indicated for DP. In a mechanistic *in vitro* study, DP was shown to activate 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 end points were independent of PPAR-γ activation, suggesting that other potential modes of actions of DP may be involved (Peshdary et al., 2019).

44. 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). Detection of DP in bird eggs show that maternal uptake and transfer to eggs occur also in

birds (Guerra et al., 2011; Zeng et al., 2016). 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). DP and other flame retardants (FR) were found in developing embryos of female sharks, demonstrating maternal transfer *in utero* (Marler et al., 2018). 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 FR input to offspring during the first few years of life.

Conclusion on adverse effects according to the criteria in Annex D

45. DP is maternally transferred to offspring in several species and is also shown to cross the blood-brain barrier in fish and amphibians. Oxidative stress responses have been observed in all tested organisms after DP exposure, in addition, potential endocrine and neurodevelopmental effects are indicated

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

46. Based on existing data DP can be considered to meet the screening criteria in Annex D for persistence, bioaccumulation, long-range transport and adverse effects under the Stockholm Convention.

47. DP has been detected in remote regions such as the Arctic and Antarctic. Tropic magnification and bioaccumulation have been reported in organisms from these regions suggesting that wild life in these vulnerable areas are exposed to DP that potentially can lead to adverse effects if no action is taken.

48. Although DP has been produced and used for almost half a century, its first environmental detection was only reported in 2006 in the Great Lakes Basin in North America (Hoh et al., 2006). However, since then research has confirmed its global ubiquitous distribution (Wang et al., 2016) and observed increasing trends (Liu et al., 2016; Li et al., 2016; Vorkamp et al., 2018) as well as unchanged long-term trends (Olukunle et al., 2018) have been reported.

49. Available toxicity date indicate concern for potential adverse effect to the environment and humans. Adding to the concern is the lack of toxicity information on DP biproducts such as 1,3 or 1,5-Dechlorane Plus monoadduct (DMPA) that in some cases are detected in greater levels than DP in environmental samples (Sverko et al., 2011; Tomy et al., 2013), and the structural similarities with already listed substances such as aldrin, mirex, chlordecone.

50. DP is transferred to the developing child 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.

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

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