Polybrominated dibenzo-*p*-dioxins and dibenzofurans and mixed polybrominated/chlorinated dibenzo-*p*-dioxins and dibenzofurans

2nd draft revised risk profile

February 2026

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Executive summary

1. At its twentieth meeting, the POPs Review Committee evaluated the proposal by Switzerland to list polyhalogenated dibenzo-*p*-dioxins and dibenzofurans in Annex C to the Stockholm Convention. The Committee concluded that these compounds meet the screening criteria specified in Annex D to the Convention (decision POPRC-20/4). At its twenty-first meeting in 2025, the Committee decided to defer its decision on the draft risk profile to its twenty-second meeting (decision POPRC-21/1).

2. The scope of the evaluation covers polybrominated dibenzo-*p*-dioxins and dibenzofurans (PBDD/Fs) and mixed polybrominated/chlorinated dibenzo-*p*-dioxins and dibenzofurans (PBCDD/Fs), excluding polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDD/Fs). PBDD/Fs and PBCDD/Fs are a group of chemicals characterized by the presence of bromine atoms or both bromine and chlorine atoms attached to their dibenzo-*p*-dioxin and dibenzofuran structures. They share similar properties and structures with their chlorinated counterparts but include bromine atoms, which influence their environmental fate and toxicity.

3. PBDD/Fs and PBCDD/Fs are not intentionally produced, except for use as analytical standards, but are formed as unintentional by-products during various industrial and combustion processes. These processes include waste incineration, metallurgical operations, and thermal processing of brominated flame retardants (BFRs). Important sources are commercial polybrominated diphenyl ether (PBDE) mixtures where they are present as by-product, photolytic and thermal degradation of BFRs and unintentional production in industrial thermal processes. PBDD/Fs can also be formed when plastics containing direct precursors such as PBDEs are subject to recycling processes or are exposed to sunlight. PBDD/Fs and PBCDD/Fs are neither traded, nor stockpiled except for small quantities for analytics and research purposes. Their uses are limited to research and their presence in materials where BFRs are incorporated. Research indicates that some PBDD/Fs and PBCDD/Fs might originate from natural sources like algae or cyanobacteria, where bromophenols undergo biogenic formation. The corresponding congeners are usually less brominated, ranging from mono- to pentabrominated and occur in significantly lower levels, often only trace amounts, compared to the higher substituted congeners from anthropogenic sources.

4. Releases of PBDD/Fs and PBCDD/Fs can occur when materials are heated or burnt, including in waste incineration but also in accidental or other types of uncontrolled fires (such as open burning of waste), or exposed to sunlight, especially when BFRs or other brominated (and chlorinated) aromatics are present or in relation with the production and use of BFRs. PBCDD/Fs can also be found in landfill leachate. The release to the environment occurs primarily through industrial activities such as incineration, metallurgical processes, and e-waste recycling. Estimates based on production and use of PBDEs in 2001 suggest that global annual emissions of PBDFs could have reached 2,300 kg. Based on a worst-case assessment, the potential PBDD/F content that could be emitted from tetrabromobisphenol A (TBBPA) was estimated at around 7,700 kg in 1999. The estimated annual input of PBDD/Fs into the atmosphere from secondary nonferrous smelting industries in China was 3.5 g TEQ (toxic equivalency) in 2018.

5. Tetra- to octahalogenated PBDD/Fs and PBCDD/Fs meet the criteria for persistence according to Annex D criteria. Di- and trihalogenated PBDD/Fs and PBCDD/Fs are also likely to meet the persistence criteria. Using a read-across approach from empirical data for suitable analogue PCDD/Fs, and supported by the QSARs and subsequent extrapolation, the half-life values in water and soil are estimated to be greater than the relevant thresholds of two and/or six months, respectively, for di- to octahalogenated PBDD/Fs and PBCDD/Fs. Based on modelling, congeners of PBDD/Fs and PCBDD/Fs are predicted to exhibit similarly low levels of biodegradation as congeners of PCDD/Fs. Moreover, the photodegradation of TeBDDs in a thin layer of soil exposed to daylight was found to be slow.

6. Di- to octahalogenated PBDD/Fs and PBCDD/Fs fulfil the criteria for bioaccumulation according to Annex D criteria. PBDD/Fs and PBCDD/Fs have estimated log $K_{OW} \geq 5$ and are therefore considered likely to bioaccumulate. They exhibit similar uptake and elimination patterns as PCDD/Fs in fish and mammals. PBCDD/Fs have physicochemical properties in the range between PCDD/Fs and PBDD/Fs. Since both PCDD/Fs (as recognized POPs) and PBDD/Fs are bioaccumulative, it is very likely that this is also the case for PBCDD/Fs. PBDDs appear to be more resistant to mammalian metabolism than PCDDs. The estimated half-lives for TeBDF and TeBDD in humans range from 1 to 11 years. These long elimination half-lives in humans, suggest that these substances are highly bioaccumulative. The occurrence in human tissues and in breast milk adds to the concern.

7. Di- to octahalogenated PBDD/Fs and PBCDD/Fs fulfil the criteria for long-range environmental transport. Their estimated gas-phase half-lives are above 2 days. Due to their physicochemical properties, they are expected to occur predominantly particle-bound in the atmosphere. The atmospheric half-lives will therefore be higher than those calculated for the gas phase. PBDD/Fs in air from background areas confirms the atmospheric long-range transport. PBDD/Fs have been detected in biota (including *Globicephala melas,* long-finned pilot whale) and ornithogenic soils

in remote regions. PBDD/Fs are also subject to long-range environmental transport in marine (micro)plastics. Moreover, PBDD/Fs could be formed from PBDEs or other BFRs once they reach remote areas, e.g. due to photolytic degradation. The occurrence of PBDD/Fs and PBCDD/Fs in migratory species, provides evidence that PBDD/Fs also undergo long-range environmental transport via such species.

8. PBDD/Fs and PBCDD/Fs have been detected in various environmental compartments, including sediments, air and biota. PBDD/Fs have also been detected in high-trophic level species such as seals and whales. They were also found in soil, wastewater, sludge, fly and bottom ash, various animal and plant species, food and feed, indoor dust and in humans.

9. Exposure of the general population to PBDD/Fs takes place primarily by consumption of food products. Further, oral uptake from children's toys made from recycled plastics containing BFRs, contributes to daily intake. Firefighters and workers in the e-waste sector show higher PBDD/F levels in their blood compared to PCDD/Fs. PBDD/Fs are also present in human milk, with varying contributions to the total TEQ levels across regions. Monitoring of human adipose tissue indicates low PBDD/F TEQ concentrations. Dietary exposure estimates for the UK range from 0.19 to 1.64 pg TEQ per kg body weight per day. The contribution to the total TEQ from PBDD/Fs, PBCDD/Fs and PBBs was consistently around 30% for all age groups and for average and high consumers. PBDD/Fs and probably also PCBDD/Fs may make a significant contribution to dietary exposure to dioxin-like compounds. Concentrations of 2,3,7,8-substituted PBDDs and PBDFs in human food, tissues and milk can contribute significantly to the total dioxin-like toxicity. Reported PBDD/F dietary intakes suggest that some population groups, in particular young children, may exceed the tolerable weekly intake for dioxin-like contaminants (2 pg TEQ/kg bw/week). The release of PBDD/Fs and PCDD/Fs, unintentionally formed from the incineration of Br- and Cl-containing solid wastes, poses significant risks to the environment and human health including vulnerable groups. Traditional consumption of whale blubber in the Faroe Islands and other communities in remote regions can already surpass safe limits of consumption as a result of the PBDD/F content in whale blubber from remote regions and could cause adverse effects as a result of long-range environmental transport and exposure to PBDD/Fs in a remote region.

10. PBDD/Fs and PBCDD/Fs cause adverse effects similar to those of PCDD/Fs with similar relative effect potencies (REPs) for causing adverse effects in mammals and fish. In mammals, the mode of action, type of toxicity and potency of 2,3,7,8-substituted PBDDs and PBDFs are similar to their chlorinated analogues including lethality, wasting, teratogenesis, reproductive effects, immunotoxicity such as thymic atrophy, endocrine disruption, and increased hepatic porphyrins.

11. It is concluded that tetra- to octahalogenated PBDD/Fs and PBCDD/Fs 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.

1. Introduction

12. In May 2024, Switzerland submitted a proposal to list polyhalogenated dibenzo-*p*-dioxins and dibenzofurans in Annex C to the Stockholm Convention on Persistent Organic Pollutants. The proposal (UNEP/POPS/POPRC.20/5) was submitted in accordance with Article 8 of the Convention and reviewed by the POPs Review Committee (POPRC) at its twentieth meeting in September 2024. At its twenty-first meeting in 2025, the Committee adopted decision POPRC-21/1, by which it decided, among other things, to defer its decision on the draft risk profile to its twenty-second meeting and to establish an intersessional working group to review and update the draft risk profile, in particular with regard to information on long-range environmental transport and adverse effects on human health of PBDD/Fs and PBCDD/Fs, in accordance with Annex E to the Convention. Further, the Committee invited Parties and observers to submit information specified in Annex E, in particular information on long-range environmental transport and adverse effects on human health.

13. The scope of the evaluation covers polybrominated dibenzo-*p*-dioxins and dibenzofurans (PBDD/Fs) and mixed polybrominated/chlorinated dibenzo-*p*-dioxins and dibenzofurans (PBCDD/Fs), excluding polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDD/Fs), which are already listed in Annex C to the Convention.

14. Based on the evaluation, the risk profile concludes that tetra- to octahalogenated PBDD/Fs and PBCDD/Fs meet the criteria for persistence, bioaccumulation, potential for long-range environmental transport and adverse effects (see paragraph 147).

1.1 Chemical identity

15. There are 75 polybrominated dibenzo-*p*-dioxin (PBDD) and 135 polybrominated dibenzofuran (PBDF) congeners (Ballschmiter & Bacher, 1996). There are 1550 mixed polybrominated/chlorinated dibenzo-*p*-dioxin (PBCDD) and 3050 mixed polybrominated/chlorinated dibenzofuran (PBCDF) congeners (WHO, 1998; L. Yang et al., 2021a).

16. The general chemical structures of PBDD/Fs and PBCDD/Fs are shown in Figure 1:

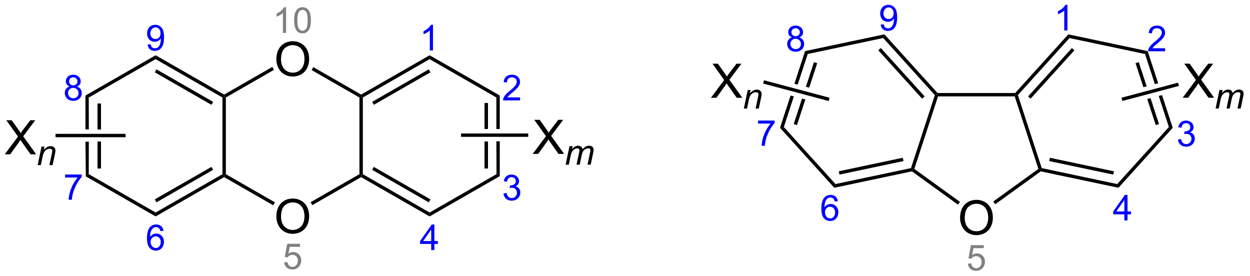

Figure 1. General chemical structures of polybrominated and mixed polybrominated/chlorinated dibenzo-*p*-dioxins (left) and polybrominated and mixed polybrominated/chlorinated dibenzofurans (right) (X = Cl, Br; $1 \leq m + n \leq 8$; where Br ≥ 1; possible positions are indicated by blue numbers).

17. PBDD/Fs and PBCDD/Fs are structurally analogues of PCDD/Fs, sharing similar physicochemical properties. However, distinct differences arise from the larger size and higher molecular weight of the bromine atoms compared to the chlorine atoms. Consequently, PBDD/Fs (and PBCDD/Fs) exhibit higher molecular weights, lower vapor pressures, higher organic carbon adsorption constants ($\log K_{OC} = 4.1–6.8$), and lower water solubilities compared to analogous PCDD/F congeners. Additionally, the larger bromine atom enhances the affinity of PBDD/Fs (and PBCDD/Fs) for attaching to particles (Yang et al., 2021a), such as aerosols or suspended solids.

18. As there is no intentional production of PBDD/Fs and PBCDD/Fs other than for use as analytical standards (Kannan et al., 2012), there are no trade names or commercial names associated with these substances.

Due to the large number of individual congeners, the PBDD/F congeners, for which in the review by van den Berg et al., (2013) relative effect potency (REP) ranges in mammals were compared with the corresponding PCDD/F congeners, were selected for a closer assessment in this risk profile (see Table 1). Six additional PBDD/F congeners were added to Table 1 to cover all levels of bromination of PBDD/Fs. World Health Organisation (WHO) (1998) provides a more extensive list of CAS numbers for PBDD/F and PBCDD/F congeners.

19. To address data limitations, a read-across approach has been implemented in this document based on guidance on grouping of chemicals from the OECD (2025). According to this guidance, substances whose physicochemical, toxicological and ecotoxicological properties are similar or follow a regular pattern as a result of structural similarity may be considered as a group and appropriate for read-across. In this case, information on PCDD/Fs, which are already listed in Annex C to the Convention, were used as analogues. It is appropriate to take such an approach as PBDD/Fs and PBCDD/Fs are PCDD/F analogues with chlorine atoms partially or completely substituted with bromine atoms, which produce a similar level of dioxin-like toxicity. They have similar physicochemical properties (see paragraph 17), low chemical transformation and biological degradation rates and are lipophilic and persistent

(Yang et al., 2021a), which are all relevant factors in bioaccumulation. As discussed in paragraphs 117 and 120, like their chlorinated analogues, PBDD/Fs and PBCDD/Fs have the ability to bind to the aryl hydrocarbon receptor (AhR), which leads to similar (eco)toxic effects. The use of PCDD/Fs as reference substances for a read-across approach has also been recommended previously by Mennear & Lee (1994) who consider PBDD/Fs as human carcinogens based on structural similarity since the carcinogenic potential of the PCDD/Fs in animals is established (IARC Working Group on the Evaluation of Carcinogenic Risks to Humans, 1997; Mennear & Lee, 1994) as described in paragraph 133.

Table 1. Commonly used names, acronyms and CAS Registry Numbers of selected PBDD/F congeners (additional congeners, not reviewed by van den Berg et al. (2013), are marked with an asterisk). The IUPAC names contain the terms "oxanthrene" and "dibenzo[*b*,*d*]furan" instead of "dibenzo-*p*-dioxin" and "dibenzofuran", respectively.

Name	Acronym	CAS RN
2-bromodibenzo-*p*-dioxin*	2-MoBDD	105906-36-3
2,7-dibromodibenzo-*p*-dioxin*	2,7-DiBDD	39073-07-9
2,3,7-tribromodibenzo-*p*-dioxin	2,3,7-TrBDD	51974-40-4
2,3,7,8-tetrabromodibenzo-*p*-dioxin	2,3,7,8-TeBDD	50585-41-6
1,2,3,7,8-pentabromodibenzo-*p*-dioxin	1,2,3,7,8-PeBDD	109333-34-8
1,2,3,4,7,8-hexabromodibenzo-*p*-dioxin	1,2,3,4,7,8-HxBDD	110999-44-5
1,2,3,6,7,8-hexabromodibenzo-*p*-dioxin	1,2,3,6,7,8-HxBDD	110999-45-6
1,2,3,7,8,9-hexabromodibenzo-*p*-dioxin	1,2,3,7,8,9-HxBDD	110999-46-7
1,2,3,4,6,7,8-heptabromodibenzo-*p*-dioxin*	1,2,3,4,6,7,8-HpBDD	110999-47-8
octabromodibenzo-*p*-dioxin	OBDD	2170-45-8
2-bromodibenzofuran*	2-MoBDF	86-76-0
2,7-dibromodibenzofuran*	2,7-DiBDF	65489-80-7
2,3,8-tribromodibenzofuran*	2,3,8-TrBDF	84761-82-0
2,3,7,8-tetrabromodibenzofuran	2,3,7,8-TeBDF	67733-57-7
1,2,3,7,8-pentabromodibenzofuran	1,2,3,7,8-PeBDF	107555-93-1
2,3,4,7,8-pentabromodibenzofuran	2,3,4,7,8-PeBDF	131166-92-2
1,2,3,4,7,8-hexabromodibenzofuran	1,2,3,4,7,8-HxBDF	129880-08-6
1,2,3,4,6,7,8-heptabromodibenzofuran	1,2,3,4,6,7,8-HpBDF	107555-95-3
octabromodibenzofuran	OBDF	103582-29-2

20. PBCDD and PBCDF congeners with the same substitution pattern and with alternating chloro and bromo substituents were selected for closer evaluation (for structural formulae see UNEP/POPS/POPRC.22/INF/#, Figure 1–4).

1.2 Conclusion of the POPs Review Committee regarding Annex D information

21. At its twentieth meeting, the POPs Review Committee evaluated the proposal by Switzerland to list polyhalogenated dibenzo-*p*-dioxins and dibenzofurans in Annex C to the Stockholm Convention. The Committee concluded that PBDD/Fs and PBCDD/Fs meet the screening criteria specified in Annex D to the Convention (decision POPRC-20/4).

1.3 Data sources

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

(a) The proposal to list polyhalogenated dibenzo-*p*-dioxins and dibenzofurans by Switzerland (UNEP/POPS/POPRC.20/5);

(b) Information submitted by Parties and observers according to Annex E to the Convention and in response to the invitation for comments on the draft risk profile. Annex E information was provided by: Australia, Canada, El Salvador, Egypt, Ethiopia, the European Union (EU), Germany, Italy, Japan, Mauritius, Monaco, Morocco, Netherlands, Norway, Oman, Peru, Saudi Arabia, Spain, Sweden, Tajikistan, United Kingdom, a best available techniques (BAT)/best environmental practice (BEP) expert, the Bromine Science and Environmental Forum (BSEF), the Cancer Prevention and Education Society, the International Pollution Elimination Network (IPEN)/Alaska Community Action on Toxics (ACAT) and Arnika, MedTechEurope;

(c) Peer-reviewed scientific journals, as well as information from reports and other grey literature.

1.4 Status of the chemical under national or regional regulations

23. In Germany, substances, mixtures and articles may not be placed on the market if they contain five PBDD and three PBDF specific congeners[1] above defined thresholds in the range from 1 to 100 µg/kg (Bundesrepublik Deutschland, 2017; EFSA Panel on Contaminants in the Food Chain (CONTAM) et al., 2024; Kannan et al., 2012). These congeners are among those listed in Table **1**. Further, in Germany, for specific installations processing pre-shredded material containing brominated flame retardants, annual measurements of PBDD/Fs in the exhaust gas are required (TA Luft, 2021).

24. In the Netherlands, all PBDD/Fs are indicated as extremely hazardous substances: persistent, bioaccumulative and very toxic. Releases of such substances are bound to obligations related to minimizing their emissions and discharges into the environment[2] (Ministerie van Binnenlandse Zaken en Koninkrijksrelaties, Nederlanden, 2018).

25. Decision SC-5/5 encouraged Parties and other relevant stakeholders to implement where appropriate, taking into account national circumstances, the recommendations set out in the Annex to decision POPRC-6/2 on the elimination from the waste stream of brominated diphenyl ethers that are listed in Annex A to the Convention. The recommendations included generating and collecting information on releases of PBDD/Fs in emissions to air and in the solid residues from thermal processes used in treating materials contaminated with brominated diphenyl ethers and to identify appropriate disposal options to ensure that brominated diphenyl ethers and/or PBDD/Fs are not released (UNEP/POPS/COP.5/15, 2015). The POPRC discussed the toxicity and unintentional formation of PBDD/Fs and PBCDD/Fs and their presence in polybrominated diphenyl ethers (PBDEs) at POPRC 6 (UNEP/POPS/POPRC.6/2, 2010).

26. In the EU, emissions to air of PBDD/Fs are to be monitored once every six months in waste incineration plants. The monitoring only applies to the incineration of waste containing BFRs or to plants using boiler bromine addition with continuous injection of bromine (European Commission, 2019).

2. Summary of the information relevant to the risk profile

2.1 Sources

2.1.1 Production, trade, stockpiles

27. As the chemical is not intentionally produced, no information was available on intentional production, trade or stockpiles. In their Annex E submissions, several Parties and one observer (Canada, El Salvador, Egypt, Ethiopia, Germany, Japan, Mauritius, Monaco, Morocco, Netherlands, Norway, Saudi Arabia, Tajikistan, United Kingdom, MedTechEurope) state that there is no evidence indicating that PBDD/Fs or PBCDD/Fs have been or are being intentionally produced in their countries.

28. There are several anthropogenic sources of PBDD/F emissions to the environment. For example, PBDD/Fs and PBCDD/Fs can be formed unintentionally when materials are heated or burnt, including in accidental or other types of uncontrolled fires, especially when brominated flame retardants (BFRs) or other brominated (and chlorinated) organics are present (EFSA Panel on Contaminants in the Food Chain (CONTAM) et al., 2024; Kannan et al., 2012; Secretariat of the Stockholm Convention, 2008; US EPA, 1997). Similarly to their chlorinated counterparts, PBDD/Fs (and, when chlorine is present, PBCDD/Fs) can be formed in thermal processes from (a) brominated precursors (such as bromophenols, polybrominated diphenyl ethers etc.), (b) in a *de novo* syntheses from the pyrolysis of chemically unrelated bromine containing compounds, such as bromocarbons and, (3) the burning of non-brominated organic matter such as polystyrene, cellulose, lignin, coal and particulate carbon in the presence of bromine donors

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[1] 2,3,7,8-Tetrabromodibenzo-*p*-dioxin; 1,2,3,7,8-pentabromodibenzo-*p*-dioxin; 2,3,7,8-tetrabromodibenzofuran; 2,3,4,7,8-pentabromodibenzofuran; 1,2,3,4,7,8-hexabromodibenzo-*p*-dioxin; 1,2,3,7,8,9-hexabromodibenzo-*p*-dioxin; 1,2,3,6,7,8-hexabromodibenzo-*p*-dioxin and 1,2,3,7,8-pentabromodibenzofuran.

[2] The Dutch national system for priority substances distinguishes various classes of substances, resulting in various limits for emissions to the environment together with an obligation to minimize these emissions. A substance falls into the “extremely hazardous substances”, or in Dutch “Extreem Risicovolle Stoffen (ERS)” class based on extreme persistence, toxicity, and bioaccumulation behavior. The emission limit value for ERS substances is expressed in TEQ (toxic equivalency). For an annual emission load of more than 20 milligrams (mg) TEQ, an emission limit value of 0.05 nanograms (ng) TEQ/Nm^3 applies.

(NATO/CCMS (1988)). Further, they can be formed in (industrial) chemical processes, where the propensity to generate PBDD/Fs (and, when chlorine is present, PBCDD/Fs) during synthesis of chemical compounds decreases in the following order: bromophenols > bromobenzenes > aliphatic brominated compounds > inorganic brominated compound (Annex E, BSEF, 2025; Fiedler, 2003). Since PBDD/Fs and PBCDD/Fs are only formed unintentionally, they are neither traded, nor stockpiled with the exception of small quantities for analytics and research purposes.

29. According to Yang et al. (2021a), there are three dominant sources for PBDD/Fs that contribute to their presence in the environment: (1) commercial PBDE mixtures (presence as a by-product), (2) photolytic and thermal degradation of BFRs and (3) unintentional production in industrial thermal processes.

30. Lu et al. (2017) found that mechanochemical non-combustion technologies, used to treat contaminated soil, have the potential to form high levels of PBDD/Fs and PBCDD/Fs depending on the temperature. However, they found that an efficient cooling system could prevent the formation of PBCDD/Fs (Lu et al., 2017).

31. PBDD/Fs can be formed when plastics containing direct precursors such as PBDEs are subject to recycling processes such as extrusion and molding (Hamm et al., 2001; Ortuño et al., 2015; Weber & Kuch, 2003; Zennegg et al., 2014).

32. Japan identified cement manufacturing facilities (due to waste incineration) and flame-retardant plastic manufacturing and processing facilities as main sources of PBDD/F emissions to the atmosphere as well as general and industrial waste incinerators, steelmaking electric furnaces, aluminium secondary refining facilities and sewage treatment plants (due to sludge incineration) as secondary sources. In regard to sources to water, Japan identified sewage treatment facilities and flame-retardant manufacturing and handling facilities as main sources of PBDD/F emissions, as well as flame-retardant fiber processing facilities, flame-retardant plastic manufacturing and processing facilities, cement manufacturing facilities and flame-retardant resin manufacturing facilities as secondary sources (Annex E, MoE Japan, 2024). Ethiopia, Morocco and Tajikistan identified cement production, secondary metal smelting, mining and smelting processes, brick kilns and waste incineration, open burning, accidental fires, electricity production and heating and oil/lubricant combustion, vehicle emissions as possible sources of PBDD/Fs and PBCDD/Fs (Annex E, Ethiopia, 2025; Annex E, Morocco, 2025; Annex E, Tajikistan, 2025).

33. PBDD/Fs can be generated during the production of BFRs. Levels of PBDFs in commercial octa- and decabromodiphenyl ether (decaBDE) mixtures were generally in the µg/g range (Hanari et al., 2006; Ren et al., 2011; Suzuki et al., 2021). Tetrabrominated PBDDs can be formed as by-products during the synthesis of 1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE) (Ren et al., 2017). Two tetrabrominated congeners, namely 1,3,6,8- and 1,3,7,9-TeBDD, have been identified in commercial BTBPE with concentrations of 2.5 and 1.5 µg/g, respectively (Ren et al., 2017). Moreover, PBDD/Fs are also formed during thermal treatment of acrylonitrile butadiene styrene (ABS) containing BTBPE (Zhan et al., 2019). PBDD/Fs can also be present in other BFRs. Low levels of PBDD/Fs have been detected as impurities in TBBPA (European Commission, 2006). Since PBDD/Fs are present as impurities in commercial mixtures of BFRs, they subsequently occur in household products and house dust (Neuwahl et al., 2019; Tue et al., 2019).

34. Studies show that certain BFRs can act as precursors for PBDD/Fs, although the yield may vary depending on the substance and temperature (Zhang et al., 2016). Several studies have investigated photolytic and thermal degradation of BFRs and BFR-containing products (e.g. incorporated in plastics) with a special focus on PBDEs as precursors for PBDD/Fs (Ebert & Bahadir, 2003; Long et al., 2025). The formation of PBDD/Fs during pyrolysis was investigated at different temperatures and carrier gas compositions. During pyrolysis of a polybutylene terephthalate/decabromobiphenyl blend (estimated content of decabromobiphenyl 5.7%), PBDFs are formed at a µg/g level. Even when oxygen is present in the carrier gas, PBDDs are formed at a much lower level than PBDFs. In the presence of 10% oxygen, the maximum yield of tetra- to octaBDFs is 70 µg/g at 600 °C (Luijk & Govers, 1992). During pyrolysis, highly brominated PBDEs favor the cleavage of ether bonds to form polybrominated benzenes, whereas less brominated PBDEs are more likely to transform into PBDD/Fs (Liang et al., 2020). PBDFs are formed when a salmon sample spiked with decaBDE is heated at 200 °C. Each minute of heating 1 g salmon fillet spiked with 200 ng BDE-209 generated about 0.5‰ PBDFs relative to the initial amount of decaBDE (Vetter et al., 2015). From a mechanistic point of view, the formation of PBDFs from PBDEs requires the intramolecular elimination of Br_2, HBr or H_2 for cyclization. For example, the elimination of two bromine substituents in the 2,2′ position of BDE-209 could result in the formation of octabrominated PBDFs (Kajiwara et al., 2008). Wang et al. (2024) investigated mechanisms and kinetics of the formation of PBDD/Fs from PBDEs by pyrolysis and combustion using density functional theory calculation and determined that involvement of polymers and metals significantly reduces the energy barriers for the formation of PBDD/Fs. The presence of polymers lowers the optimal formation temperature of PBDFs from 600 °C to 350–400 °C.

35. Experimental studies have shown that during photodegradation of decaBDE incorporated into high-impact polystyrene (HIPS), concentrations of PBDF increased by approximately 40 times after one week of exposure to natural sunlight, while concentrations of BDE-209 (the dominant congener in commercial decaBDE mixtures) decreased. In this study, tri- to octabrominated PBDF congeners were detected, with hepta- and octabrominated congeners comprising around 40% of the total PBDF levels in the irradiated samples (Kajiwara et al., 2008).

Photolytic transformation of technical decabromodiphenyl ether (decaBDE) was also demonstrated in textiles containing flame retardants exposed to natural sunlight. Exposure of a textile treated with technical decaBDE resulted in the formation of PBDFs as products of photodecomposition of polybrominated diphenyl ethers present in the technical decaBDE. After 329 days of exposure, the total PBDF concentration reached a maximum of 27,000 ng/g, which was approximately 10 times the initial concentration. During the experiment, di- to hexa-BDF congener concentrations increased continuously. PBDFs were formed as a result of sunlight exposure during normal use of products treated with technical decaBDE (Kajiwara et al., 2013). Kajiwara et al. (2013) suggest that for assessing the risks associated with normal use, disposal and recycling of consumer products, close attention should be paid to the fact that PBDFs are produced by sunlight exposure of plastics flame-retarded with PBDEs.

36. PBDD/Fs and/or precursors (e.g. bromophenols) can be formed as a result of pyrolysis and combustion of BFRs and materials containing BFRs such as TBBPA, tetrabromobisphenol A-bis(2,3-dibromopropyl ether) (TBBPA-BDBPE), BTBPE, polybrominated biphenyls (PBBs) or hexabromocyclododecane (HBCD) (Altarawneh et al., 2019; Die et al., 2022; Dumler et al., 1989; Ebert & Bahadir, 2003; Hsiao & Lin, 2025; J. Huang et al., 2023; J. Liu et al., 2025a; W.-J. Liu et al., 2016; Luijk & Govers, 1992; O'Keefe, 1978; Ortuño et al., 2014a; J. Wang et al., 2024b; X. Wang et al., 2018a; Wichmann et al., 2002). Details are provided in UNEP/POPS/POPRC.22/INF/#.

37. The presence of both bromine and chlorine in waste streams that are treated by combustion may result in the formation of PBCDD/Fs. The ratio of bromine and chlorine in the combustion process largely corresponds to the ratio of bromine and chlorine in the PBCDD/Fs formed. During insufficiently controlled combustion processes, significant amounts of PBDD/Fs may be formed, including those with the most toxic dioxin-like properties. The origin of these PBDD/Fs may partly be explained by their presence as contaminants in the commercial PBDE flame-retardant mixtures in household waste, but *de novo* synthesis should also be considered as a significant source (Neuwahl et al., 2019). The release of PBDD/Fs and PCDD/Fs, unintentionally formed from the incineration of Br- and Cl-containing solid wastes, poses significant risks to the environment and human health (Li et al., 2025b).

38. Another source of bromine in incineration processes may be the use of techniques for the enhanced oxidation of mercury (thereby enabling efficient mercury removal in the downstream flue gas control system of selected hazardous waste incinerators) by means of high-temperature bromide injection (Neuwahl et al., 2019). PBCDD/Fs have been detected in fly ash of municipal waste incinerators from the US, Japan and Canada (Huang et al., 1992). Tu et al. (2011) detected PBDD/Fs in incineration residues and off gases of two municipal solid waste incinerators (MSWIs) and a coal fired power plant. For example, they reported PBCDD/F concentrations ranging from 0.021 to 0.167 ng/Nm^3 (normal cubic meter) in the stack flue gases of these facilities. As combustion processes generally take place in the presence of an excess of chlorine, PBCDD/Fs are usually formed predominantly. The total amounts of PBDD/Fs and PBCDD/Fs formed during thermal processes depend largely on the chemical composition of the waste and the specific conditions of the thermal treatment (Altarawneh, 2022; Ebert & Bahadir, 2003; Hamm et al., 2001; Weber & Greim, 1997; Weber & Kuch, 2003).

39. Li et al. (2025a) investigated the formation of PBDD/Fs after the aromatic bromination of dibenzo-*p*-dioxins and dibenzofurans in MSWI flue gas with a focus on the catalytic role of various metal salts. They can be abundant (dibenzo-*p*-dioxins: 0.5–203 ng/Nm^3, dibenzofurans: 360–1,120,000 ng/Nm^3) and potentially be another precursor of PBDD/Fs. Mechanistic analysis revealed that positions 1/4/6/9 and 2/3/7/8 on aromatic structures are highly susceptible to bromination substitution reactions, with the reactivity influenced by the metal type. For some metallurgical industries, PBDD/Fs and PBCDD/Fs were found to be emitted in similar or higher amounts than PCDD/Fs (Du et al., 2010). PBDD/Fs that were found to be formed during combustion and other anthropogenic processes present a congener distribution in which furans are more prevalent, as well as congeners with higher levels of substitution (tetra- to octa-) (Arnoldsson, 2012a).

40. Research conducted in the Baltic Sea indicates that, similar to some PCDD/Fs, certain PBDD/Fs and PBCDD/Fs can originate from natural sources (Haglund, 2010a; Haglund et al., 2007; Malmvärn et al., 2008; Unger et al., 2009). The congeners from natural sources are usually less brominated than those formed anthropogenically, ranging from mono- to pentabrominated. Those found in highest concentrations in Baltic Sea monitoring were tribrominated. Primary producers such as cyanobacteria, algae or aquatic sponge could be potential sources of naturally occurring PBDDs, suggesting a biogenic formation (Arnoldsson, 2012a; Unger et al., 2009). One study proposed a biosynthetic route for PBDDs found in Baltic Sea biota involving bromoperoxidase-mediated formation of bromophenols (such as 2,4,6-tribromophenol) followed by condensation of two bromophenols to form various PBDD congeners, particularly TrBDDs (Haglund et al., 2007). As natural bromophenols are expected to be primarily *ortho*- and *para*-substituted, natural formation of the PBDD congener analogous to the most toxic 2,3,7,8-TeCDD appears unlikely. Indeed, 2,3,7,8-TeBDD was only found in perch from one location, at 0.011 pg/g ww in a composite sample. Most biota samples in this study had total PBDD concentrations in the range of n.d. – 0.075 ng/g ww, though one composite mussel sample had a total PBDD concentration of 4.1 ng/g ww, of which ~85% was 1,3,7- and 1,3,8-TrBDD. A follow-up study proposed a range of natural formation mechanisms to better account for the observed PBDD congener patterns found in Baltic biota, including enzymatic and light-induced coupling of widespread

environmental bromophenols or hydroxylated diphenyl ethers (OH-PBDEs), and debromination of PBDDs to create lesser brominated products (Haglund, 2010a).

41. In another study by Unger et al. (2009), a range of PBDD/Fs and PCBDD/Fs were found in sponges from the Baltic Sea. Consistent with other Baltic Sea biomonitoring studies, 1,3,8- and 1,3,7-TrBDDs were the most abundant congeners at 43 ng/g$_{\text{extractable organic matter (EOM)}}$ and 84 ng/g$_{\text{EOM}}$ respectively. Other di- and tetrabrominated PBDDs were found at below 5 ng/g$_{\text{EOM}}$ each (3.2 ng/g$_{\text{EOM}}$ 2,7/2,8-DiBDD, 0.87 ng/g$_{\text{EOM}}$ 1,8-DiBDD, 4.5 ng/g$_{\text{EOM}}$ 2,3,7-TrBDD, 3.1 ng/g$_{\text{EOM}}$ 1,2,4,7/1,2,4,8-TeBDD, 3.2 ng/g$_{\text{EOM}}$ 1,2,3,7-TeBDD and 1.7 µg/kg$_{\text{EOM}}$ 1,2,3,8-TeBDD). Some PBCDDs with unknown substitution patterns (USP) were also found (ClBr$_x$DD, x = 2–3) (0.76 ng/g$_{\text{EOM}}$ ClBr$_2$DD$_{\text{USP 1}}$, 2.3 ng/g$_{\text{EOM}}$ ClBr$_2$DD$_{\text{USP 2}}$, 1.3 ng/g$_{\text{EOM}}$ ClBr$_2$DD$_{\text{USP 3}}$ and 1.9 ng/g$_{\text{EOM}}$ ClBr$_2$DD$_{\text{USP 6}}$). No PBDF congeners were present in concentrations higher than 0.5 ng/g$_{\text{EOM}}$. The PBDD/Fs formed in anthropogenically related (combustion) processes are mostly PBDFs and highly brominated (tetra- to hepta-) congeners, whereas studies in biota linked to halogenated natural particles, e.g. in Baltic biota, have shown lower PBDF to PBDD ratio content and that PBDDs are mainly less brominated (mono- to tetra-) congeners (Arnoldsson, 2012a).

42. Goto et al. (2017) investigated the occurrence of mono- to octabrominated PBDD/Fs in Tokyo Bay, Japan, using ^{210}Pb-dated sediment cores covering the period 1895–2000. In the sediment cores, PBDF and PBDE concentrations increased drastically after the 1960s and reached the highest levels in the late 1990s. PBDD levels showed only marginal fluctuation over the past century, with the predominance of 2,7-/2,8-DiBDDs and 1,3,7-/1,3,8-TrBDDs in all the sediment layers. In the last decades, PBDF levels are higher than PBDD levels. It is noteworthy that the PBDD congeners were also found in the sediment layers corresponding to the pre-industrial era, supporting their natural formation in the coastal environment. The contribution of the anthropogenic PBDFs to the total toxic equivalent (TEQ) increases consistently since their first occurrence. In the most recent sediment core samples of the investigation period, from 1998, 1999 and 2000, the PBDF contribution to the total TEQ ranges from 0.29 to 1.03%. The authors concluded that lower brominated PBDDs are predominantly of natural origin, whereas for PBDFs, anthropogenic sources dominate.

2.1.2 Use

43. No information was received indicating that there has been or there are any intentional uses of PBDD/Fs and/or PBCDD/Fs with the exemption of small quantities for analytics and research purposes (Annex E submissions in 2024 and 2025).

44. PBDEs, which are precursors of PBDD/Fs or, when chlorine is present, of PBCDD/Fs, are already restricted to a large extent as tetra-, penta-, hexa-, hepta- and decaBDE are listed with specific exemptions in Annex A (Elimination) to the Convention. Previously, commercial PBDE mixtures were used as additive flame retardants and had a variety of applications including in plastics/polymers/composites, textiles, adhesives, sealants, coatings and inks (UNEP/POPS/POPRC.4/15/Add.1, UNEP/POPS/POPRC.2/17/Add.1, UNEP/POPS/POPRC.10/10/Add.2). However, although production of most products containing PBDEs has ceased over the past few decades, certain products containing PBDEs are still in use and PBDEs are present in consumer products made from plastics derived from electronic waste (e-waste) (Behnisch et al., 2023; Grechko et al., 2022; Petrlik et al., 2019). In consumer products with PBDE concentrations between 169 and 593 µg/g, levels of PBDD/Fs ranged from 690 to 3,821 pg WHO-Toxic Equivalent (TEQ)/g (Petrlik et al., 2019).

45. Many substances that have been used to replace PBDEs are also brominated (Cowell et al., 2017) and may contribute to PBDD/F releases either as impurities or as a result of their formation during pyrolysis and combustion (Altarawneh et al., 2019; Hanari et al., 2006; Zhou & Liu, 2018). Other aromatic BFRs that can decompose to precursors of PBDD/Fs or, when chlorine is present, to precursors of PBCDD/Fs are applied to a variety of polymers used in textiles, electrical and electronic equipment and automotive articles (ECHA, 2024). For example, TBBPA can be used in enhancing the fire resistance of electrical and electronic devices and systems that incorporate printed circuit boards, including household appliances like televisions, vacuum cleaners, and washing machines. Further TBBPA can be used in office and communication devices such as photocopy machines, computers, printers, and fax machines, alongside equipment used in automotive, aviation and entertainment sectors (Ortuño et al., 2014a). Moreover, TBBPA has been detected, among other BFRs, at concentrations of hundreds of µg/g in consumer products made from recycled plastics derived from e-waste (Behnisch et al., 2023; Grechko et al., 2022). Behnisch et al. (2023) reported TBBPA concentrations ranging from < LOD to 372 µg/g, PBCD/Fs from 110 to 17,000 pg biological equivalence[3] (BEQ)/g, PCDD/F from 1.9 to 26.1 pg TEQ/g, and PBDD/F from 56 to 13,900 pg TEQ/g. In comparison, Grechko et al. (2022) found TBBPA levels ranging from 0.5 to 185 µg/g), and PBDD/F (sum of 17 congeners) ranging from 56 to 14,100 pg TEQ/g. PBDD/Fs can also be formed in the presence of other substances that contain bromine. This has for

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[3] In bioassays, such as the DR CALUX method, it is possible to determine the total dioxin-like activity or thyroid-hormone transport inhibition directly based on the toxic potency of a complex mixture of contaminants in a sample and express it as biological equivalence (BEQ) concentrations, expressed as, e.g., pg 2,3,7,8-TCDD BEQ/g (Behnisch et al. (2023).

example been shown through studies on vehicle exhaust where it was linked to dibromoethane, an additive used in leaded petrol (Haglund et al., 1988; UK HPA, 2025).

2.1.3. Releases to the environment

46. As the use of BFRs has become widespread over the past 50 years (Cowell et al., 2017), it is no longer uncommon to find them in municipal incineration waste and with metals recycled by heat treatment in metallurgy. The emission of PBDD/Fs is influenced by many parameters, including the type of combustion and the type of material (INERIS, 2020). In a Swedish study, Remberger et al. (2014) concluded that PBDFs will be emitted into air and water. PBDFs were generally found in samples from wastewater treatment plants (WWTPs), storm water, indoor air and dust, indicating that they may be emitted and transported to the environment via these sources and pathways. Therein, the authors conclude that BFRs and PBDFs emitted to indoor air and to dust may be transported to outdoor air via ventilation. The presence of these pollutants in influent, effluent and sludge from municipal WWTPs also indicates that diffusive emissions from household products occur and that these chemicals may be emitted to the environment from WWTPs. The presence of the BFRs and PBDD/Fs in storm water and fire extinguishing water shows that these may also be pathways to the environment (Remberger et al., 2014).

47. The occurrence of PBDD/Fs in sediments was positively correlated with local development levels and concentrations were higher from highly urbanized/industrialized areas in China (Liu et al., 2023). PBDE-related products/industries were the greatest contributors to contamination with PBDD/Fs in this area, followed by bromo-phenol/benzene-related products/industries. These results indicate that the main contamination sources of PBDFs were technical PBDE formulations, especially decaBDE (Goto et al., 2017). Significant correlation between PBDFs and PBDEs was also observed in sediment samples of British lakes, whereas sources of PBCDD/Fs were not related to PBDEs and differ from those of PBDFs (Peters et al., 2021).

48. Based on the production and use of commercial PBDE mixtures in 2001, the potential global annual emissions of PBDFs were calculated to be 2,300 kg based on an annual global demand for commercial PBDE mixtures of 66,110 tonnes (Hanari et al., 2006). A similar calculation can be made for potential global annual emissions of PBDD/Fs from TBBPA. Evidence on impurities of PBDD/Fs in TBBPA is available in the EU risk assessment report: TBBPA (European Commission, 2006). According to this source, several studies have investigated the levels of PBDD/Fs present in TBBPA. Some of them have indicated that the levels are low (often below detection limits). A few studies have occasionally indicated the presence of some congeners at more elevated levels. Highest levels were found in TBBPA which contained PeBDF to OBDF in a range between 1 and 31.5 ng/g with a sum concentration of 63.6 ng/g (Thoma et al., 1986a and Dummler et al., 1989b as cited in European Commission, 2006). Global TBBPA consumption in 1999 was 121,300 tonnes. For a worst-case assessment, the sum concentration of 63.6 ng/g PBDD/F and a global consumption of 121,300 tonnes can be used. The potential PBDD/F content which could be emitted from TBBPA results in an annual amount of 7,700 kg at that time (in 1999). Global production volumes of TBBPA have increased since then and were around 260,000 tonnes in 2024 (Market Growth Reports, 2026). This is related to correspondingly increasing quantities of PBDD/Fs which can potentially be released. Abbasi et al. (2019) presented an inventory of PBDEs in the anthroposphere and the environment. The global in-use and waste stocks of the sum of 5 BDEs and BDE-209 were estimated to be ~425 kt and ~113 kt, respectively, in 2018. A total of 6 and 10.5 kt of the 5 BDEs and BDE-209, respectively, has been emitted to the atmosphere by 2018. More than 70% of PBDE emissions during production and use occurred in the industrialized regions, while more than 70% of the emissions during waste disposal occurred in the less industrialized regions. A total of 70 kt of these BDEs has been recycled within products since 1970. As recycling rates are expected to have increased under the circular economy, an additional 45 kt of PBDEs (mainly BDE-209) may reappear in new products (Abbasi et al., 2019). Hence, the earlier listing of PBDEs including decaBDE under the Convention contributes to a decrease of PBDE-related PBDD/F-releases, since they are important precursors to PBDD/Fs. However, they are still present in products, waste and the environment (UNEP/POPS/POPRC.20/INF/11). PBDD/Fs are not routinely monitored in all regions.

49. Plastics from cathode ray tube (CRT) casings containing PBDEs or other BFRs sampled in Nigeria were found to contain a mean concentration of 41,000 ng/g PBDD/Fs. It was estimated that the CRT casings stockpiled in Nigeria contain between 2 and 8 tonnes of PBDD/Fs. PBDE-treated plastic contained ca. 1,000 ppm (µg/g) of PBDD/Fs when related to the PBDE content. Taking into account the total global production of decaBDE (ca. 1.25 million tonnes) and c-octaBDE (ca. 110,000 tonnes), it is possible that the total quantity of PBDD/Fs in these products might be around 1,000 tonnes (Sindiku et al., 2015). PBDFs of different bromination degrees (60–80% octabrominated PBDF as predominant congener) from TV casing samples were observed during photodegradation of TV casings with BDE-209 incorporated in concentrations up to 10% by weight (Kajiwara et al., 2008). Plastics containing PBDEs or PBDD/Fs can thus contribute to releases to the environment.

50. Another study evaluated the occurrence of PBDEs and PBDD/Fs in polyurethane foam (PUF) from seats of end-of-life vehicles (ELVs). The results showed that specific PBDD congeners were generally below the detection limit or found in low concentrations in foam samples from ELVs. However, in two samples with the highest PBDE concentrations, PBDD and PBDF concentrations were higher. Maximum TEQ PBDD/F concentrations of 2.4 ng

TEQ/g and 0.62 ng TEQ/g were observed, representing total concentrations of PBDD/Fs of about 410 ng/g and 140 ng/g, respectively (Redin et al., 2017). Products containing PBDD/Fs can contribute to their releases.

51. Current global plastic recycling practices may contribute to the formation of PBDD/Fs from BFRs, leading to cross-contamination of new products and unintentional releases into the environment. The formation of PBDD/Fs has been observed in two South African e-waste recycling processes and under controlled laboratory conditions (Zennegg et al., 2014). In this study, formation rates of PBDFs between $2\cdot10^{-5}$ and $2\cdot10^{-4}$ per $\sum$PBDE/min were calculated during thermal exposure in the recycling processes and under laboratory conditions. The PBDF pattern was dominated by penta-, hexa-, hepta- and octabrominated congeners in all samples (Behnisch et al., 2023; Fernandes et al., 2023; Li et al., 2020). Emissions are especially high during uncontrolled e-waste recycling; however, large emissions have also been measured in controlled recycling facilities that seem to be connected to the dust released from the interior of the e-waste during dismantling processes (Lundstedt, 2016). In 2022, PBDD/Fs were measured in raw and clean gases at a shredder plant in Germany, using refrigeration scrap and aluminium scrap as feedstock. PBDDs were not detected. PBDFs (tetra- to octa-congeners) were only detected in the raw gas at concentrations of 0.61 and 2.10 ng/Nm^3 (communication from the German Environment Agency, 2025).

52. Unintentional production from thermal processes, especially incineration of e-waste containing PBDEs, is a relevant source of PBDD/F emissions. Among the multiple types of e-waste, CRT monitors, printed circuit boards, and plastics containing flame retardants used in electronic housing are more prone to produce PBDD/Fs during recycling processes, such as dismantling, mechanical recycling, thermal processes or thermal stress, chemical separation and extraction, and disposal of residues by combustion or in landfill (Duan et al., 2011; Weber & Kuch, 2003). PBDD/Fs and PBCDD/Fs could be present in imported plastic or e-waste and in stockpiles such as e-waste, cables, plastics, rubber and BFR-treated foams (Annex E, Ethiopia, 2025; Annex E, Tajikistan, 2025). Large quantities of brominated flame retardants and other brominated additives have been imported in articles (e.g. in EEE, vehicles or building materials) into countries around the world (e.g. the total estimated PBDE import from 2000 to 2022 to Mauritius amounted to 44,687 kg; Annex E, Mauritius, 2025) and are globally present in waste streams. These brominated articles and waste streams constitute sources where PBDD/F and PBCDD/F formation and release can occur under thermal treatment, open burning or due to photodegradation of PBDEs.

53. In some countries, people use plastic waste as an alternative fuel to wood, which produces a range of pollutants including PBDD/Fs and PBCDD/Fs (Gündoğdu, 2024). The growing amount of plastic waste has led to a rising prevalence of the open burning of plastic waste, especially in locations lacking formal waste management systems (Bharadwaj et al., 2025; Petrlik et al., 2019), leading as a consequence to increasing releases of PBDD/Fs and PBCDD/Fs from open burning of plastic waste. Uncontrolled thermal disposal of e-waste and dismantling/recycling of e-waste containing brominated and/or chlorinated flame retardants can also lead to the formation of PBDD/Fs and PBCDD/Fs (ECHA, 2023b) and has been reported to emit PBDD/Fs at levels up to 50–500 times higher than those of PCDD/Fs (Yang et al., 2021a). Similar results were observed in a study by Ortuño et al. (2014b), where PBDD/Fs were formed as a result of the pyrolysis and combustion of waste printed circuit boards. The total content of eleven 2,3,7,8-substituted congeners (Te- through HpBDD/Fs) was 7,240 and 3,250 ng WHO$_{2005}$-TEQ[4]/kg sample before and after metal removal, respectively. An incomplete and uncontrolled incineration of plastics containing BFRs can form high amounts of PBDD/Fs, which can be released to the environment (Ebert & Bahadir, 2003).

54. In a one-year air sampling study around a large-scale MSWI, the concentrations of PBDD/Fs and PCDD/Fs showed spatially decreasing trends with the distance away from the MSWI (Li et al., 2023). Liu et al. (2025b) investigated the generation characteristics of PBDD/Fs and PCDD/Fs under varying incineration conditions within the normal range of municipal solid waste. They selected a long-term operating MSWI in south China, investigated the generation of PBDD/Fs, and compared them to those of PCDD/Fs. Generally, PBDD/Fs exhibited mass-based emission factors (mass-EFs) one order of magnitude higher than those of PCDD/Fs and were predominantly released via bottom ash. Both PBDD/Fs and PCDD/Fs showed significant changes in emissions with the variations in waste load, O$_2$ content, and waste composition (feedstock type represented as the industrial/household waste weight ratio), particularly PBDD/Fs. PBDD/Fs derived from furnaces demonstrated higher sensitivity to waste load, compared to and composition, whereas PCDD/Fs were primarily influenced by O$_2$ levels, followed by waste composition; PCDD/Fs consistently exhibited increasing mass-EFs under unconventional conditions and PCDD/Fs in filtered fly ash were highly sensitive to all the three variables, while filtered PBDD/Fs remained insensitive to these conditions. Data comparison between raw flue gas and filtered fly ash indicated secondary generation of PBDD/Fs and PCDD/Fs during gas purification, as well as the insufficient dioxin-trapping efficiency of bag filters, particularly for PBDD/Fs (Liu et al., 2025b).

55. During the investigation on the feasibility of thermal valorisation of automobile shredder residue, different oxygen ratios resulted in a significant difference in PBDD/F releases. The highest concentrations of PBDD/Fs found were at low oxygen ratios (λ=0.35) at 600 °C and highly brominated (hepta- and octabrominated) dibenzofurans were obtained in 10–50 times higher concentration than the lower brominated congeners. In comparison, at 850 °C

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[4] See van den Berg et al. (2013).

PBDD/Fs were almost only found at over-stochiometric amounts of oxygen (λ=1.5), but the concentrations were still lower by a factor of 100 compared to the highest concentrations found at 600 °C (Rey et al., 2016).

56. Use of plastics containing BFRs as an alternative fuel to wood in some countries (e.g. burning of e-waste) have reportedly led to soils contaminated with PBDD/Fs and PBCDD/Fs (Ma et al., 2009; Weber et al., 2015). The highest PBDD/F levels (500 pg (TEQ)/g fat) in free-range chicken eggs, which are sensitive indicators of POP contamination in soils/dust and represent an important human exposure pathway, were found in an e-waste and automotive waste scrap yard in Ghana (Petrlik et al., 2025). High concentrations in such eggs were also found in the surroundings of e-waste recycling sites in Kalasin province, Thailand (Dvorská et al., 2023). PBCDD/Fs were also found to occur in e-waste sites at elevated levels. PBCDD/Fs have been detected in soil samples of high ash content collected near a plastic recycling facility after a fire accident in Canada (Myers et al., 2012). Moreover, PBCDFs were substantial TEQ contributors in open burning areas (total TEQs were 0.42–2.6 times those of PBDFs based on roughly estimated concentrations) (Tue et al., 2019).

57. PBDD/Fs were monitored in a cement production plant, which used petroleum coke as primary fuel, but also alternative fuels such as solid recovered fuel, automotive shredder residue, sewage sludge, waste tires and meat and bone meal. The monitoring was performed over the period of a year. PBDD/F emission values found ranged from 0.01 to 0.44 pg I-TEQ/m³. Due to no availability of bromine in the cement plant, the observed values stem from the materials used as fuels. The congener distribution was also investigated, and it was apparent, that there was a clear predominance of OBDD and OBDF, followed by HpBDD. Further, in all samples the PBDF/PBDD ratio found was >1, meaning that furan concentrations consistently exceed dioxin concentrations (Conesa et al., 2016). Yang et al. (2019) identified the cement kiln backend as a primary formation site for PBDD/Fs, where highly brominated congeners (predominantly HpBDFs) are produced through the thermal transformation of PBDE precursors via HBr and Br_2 elimination pathways. There were significant correlations between the decaBDE and HpBDF concentrations, which is an indicator of transformation from PBDEs to PBDD/Fs (Yang et al., 2019).

58. Metallurgical facilities including sinter plants, electric arc furnaces (EAFs), secondary aluminium smelters and secondary copper smelters were identified as a significant source of PBDD/Fs into their surrounding atmospheric environments in Chinese Taiwan (Wang et al., 2008). Copper smelters can be treating high inputs of BFR-containing plastics and secondary copper smelting processes are a source of PBDD/F emissions (Lin et al., 2022; Wang et al., 2015; Wu et al., 2020). The composition of the raw materials used in secondary aluminium plants in China was found to be a key factor influencing the amounts of PBDD/Fs emitted (Wang et al., 2016). The estimated annual input of PBDD/Fs into atmosphere from secondary nonferrous smelting industries in China was 3.5 g TEQ in 2018 (Yang et al., 2021b). PBDD/Fs can be formed during thermal metallurgical processes (Yang et al., 2021b). Wang et al. (2010a) calculated emission rates of PBDD/Fs from metallurgical processes, including EAFs, fly ash treatment plants and sinter plants. They found average emission rates from 25 to 750 µg/h (corresponding to 0.46 to 6.67 µg TEQ/h) for the investigated metallurgical facilities in Chinese Taiwan with the highest values for sinter plants. PBDD/Fs and PBCDD/Fs are released from the incineration of WEEE (Dresch et al., 2014) and co-disposal of HBCD in a municipal solid waste incinerator showed increased concentrations of PBDEs and PBDD/Fs after the air pollution control devices (Xie et al., 2024).

59. Stack gases from three EAFs in steel producing facilities in China have been analyzed to determine, whether they significantly contributed to the emissions of PBDD/Fs. Average emission rates during the preheating stages of the furnaces revealed average emissions of 271 to 9,470 pg/Nm³ (corresponding to 10.8 to 791 pg/Nm³ TEQ). These concentrations were reduced to 262 pg/Nm³ (4.5 pg/Nm³ TEQ) during the smelting stage. The wide fluctuations were caused by different factors, such as feeding materials and the working temperature of the bag filters. Scrap steel raw materials are complex and may contain BFRs, which can be or decompose to precursors of PBDD/Fs. The emissions primarily concentrated in the preheating stage and contributed up to 57.7% of total TEQ, driven largely by feedstock heterogeneity and operational temperatures (Shen et al., 2021). These monitoring activities do not represent the overall situation in China. In a UK iron ore sinter plant, mean PBDD/F concentrations in stack emissions were 0.14 ng WHO-TEQ/m^3 (range: 0.03–0.39). PBDD/F emission concentrations were approximately ten times lower than their PCDD/F homologues. Laboratory tests were carried out to study possible formation mechanism of PBDD/Fs in this study. It was observed that PBDD/F formation increased with increasing amounts of bromide in the raw sinter mix indicating a possible *de novo* formation. In contrast, the addition of PBDEs did not lead to an increased formation of PBDD/Fs in these experiments, which suggests that PBDEs may not act as PBDD/F precursors during sintering (Drage et al., 2014). Wang et al. (2018b) found that while desulfurization in Chinese iron ore sintering plants removes 53.6%–97.1% of PBDD/Fs from stack gases, these pollutants are primarily transferred to solid residues; consequently, the environmental release is shifting from gaseous emissions to increasing levels in desulfurization products and fly ash (M. Wang et al., 2018b). Li et al. (2015) found that converter steelmaking processes are a source of PBDD/Fs and PBDEs in China, though their emission levels and toxic equivalents (0.32–4.33 pg TEQ/Nm³) are notably lower than those of electric arc furnaces and sintering processes, with PBDEs identified as the predominant brominated pollutants in the stack gas. Average emission factors of the PBDD/Fs in five converter steelmaking processes were used to estimate the emission masses of PBDD/Fs in China. Based on the annual crude steel output of China in 2012 (5.67×10^8 t), preliminary estimates of the annual atmospheric emissions of PBDD/Fs from secondary ventilation systems in converter steelmaking in China were 0.2 kg (3.3 g TEQ) (S. Li et al., 2015).

60. The review by Mwangi et al. (2015) concluded that, based on the few studies available, it is evident that diesel vehicle emissions are a relevant source of PBDD/Fs, PBBs and PBDEs. The mass concentrations of PBDD/Fs (417 pg/Nm^3) are relatively higher than those of chlorinated pollutants such as PCDD/Fs (156 pg/Nm^3) and PCBs (267 pg/Nm^3). These values highlight the importance and the contribution of the less understood and investigated brominated pollutants compared with their chlorinated homologues. Bromine is contained in natural and anthropogenic solid fuels such as peat, coal, wastes and solid recovered fuel. Also incineration of solid fuels (including waste) can contribute to releases of PBDD/Fs and PBCDD/Fs. Natural sources and concentration of Br are typically so low that in order for the content to be significant (practical implications to combustion system) some type of anthropogenic influence is required (Vainikka & Hupa, 2012a, 2012b).

61. For the sources identified in Japan (see paragraph 32), emission of PBDD/Fs into the atmosphere and water were monitored in Japan from 2003 to 2020. Considering tetra- to octahalogenated congeners, the annual releases to the atmosphere amounted to 1.96 g TEQ/year. This value decreased after regulation of decaBDE (in 2018) to 1.48 g TEQ/year. In water, the annual release of 3.08 g TEQ/year decreased to 1.46 g TEQ/year (Annex E, MoE Japan, 2024). Compared to PCDD/Fs emissions, these releases in terms of TEQ are lower. For example, in 2014, total PCDD/F emissions in Japan into air and water amounted to 121 to 123 g-TEQ (NIP Japan, 2016). Using the annual releases before the decaBDE regulation as a basis for comparison, it can be concluded that the contribution of the PBDD/Fs to the total TEQ was about 4%. Multiple PBCDD/Fs congeners were found in leachate from an industrial landfill in Japan indicating that landfills might be an additional source for their release into the environment (Hashimoto et al., 2021).

62. Appropriate risk management measures such as BAT/BEP developed to limit the formation of PCDD/Fs can also limit the formation of PBDD/Fs and PBCDD/Fs but may not be sufficient in all cases. BSEF provided specific information indicating that BFRs can be handled during their end-of-life phase without formation of significant dioxin/furan levels (Annex E, BSEF, 2024; Brusselaers et al., 2006; Dresch et al., 2014; Vehlow et al., 2002). However, based on their findings (see paragraph 54), Liu et al. (2025b) state that MSWI-derived PBDD/Fs in bottom ash and other MSWI-derived wastes warrant significant attention. Results published by Kajiwara et al. (2021) show that, compared to baseline tests, increased levels of PBDD/Fs were found in bottom ash and sealing water (used to cool the bottom ash), while they were lower in stack gas and fly ash during incineration of plastic television housing waste containing decaBDE at commercial-scale industrial waste incineration plants. These results suggest that any PBDD/Fs that were present in the source materials or generated in the furnace and that were not combusted tended to be readily discharged into bottom ash and sealing water. However, destruction efficiencies were over 99.9999% for decaBDE and discharge of PBDD/Fs was negligible. PBDD/Fs are present at levels of 0.83–1.4 ng/g dry matter in bottom ash and of 6.9–9.6 ng/L in sealing water. Information on technologies for the destruction or irreversible transformation of specific POP-BFR wastes, is provided in subsection IV.G.2 of the Basel Convention general technical guidelines on the environmentally sound management of wastes consisting of, containing or contaminated with POPs (UNEP/CHW.16/6/Add.1/Rev.1, 2023).

63. A study carried out in the UK to determine the concentrations of key pollutants in solid residues and air emissions of energy from waste facilities showed very low concentrations of PBDD/Fs as partial products of combustion with no consistent trend, when POP-containing WEEE plastic was incinerated (WRC, 2024). The measurements found that concentrations of PBDD/Fs in stack emissions ranged between 0.009 and 0.0711 ng/m^3. In air pollution control residues, the concentrations of PBDD/Fs were generally lower than their chlorinated analogues and ranged between 158 and 531 ng/kg. The concentrations increased after addition of the POP-containing plastic fraction produced from treatment of WEEE. Concentrations of PCDD/Fs and PBDD/Fs were generally very low across all site incinerator bottom ash samples. PBDD/F concentrations were at levels that ranged between 266 and 1450 ng/kg. In baseline boiler ash, the concentrations of PBDD/Fs (69–150 ng/kg) were lower than their chlorinated analogues. Other studies on municipal solid waste have shown that the bottom ash might contain higher concentrations of PBDD/Fs and that there is a secondary formation of PBDD/Fs during cleaning of flue gas (Hsieh et al., 2023; M.-Y. Liu et al., 2025b; L.-C. Wang et al., 2010c).

2.2. Environmental fate

2.2.1 Persistence

64. The photodegradation half-life of TeBDDs in a 5 mm layer of soil exposed to daylight was found to be between 3 and 6 months. The soil used came from a remote rural site and was air-dried at room temperature for 14 days and ground in a pestle and mortar. Portions of the soil were given in 5 mm deep aluminium trays. The samples were spiked with 40 µL each of PBDD/PBCDD mixture in hexane. The blank (unspiked) sample, and one spiked sample were extracted immediately. Half of the remaining spiked samples were kept in the dark indoors at room temperature, the other half was placed outdoors in a position where it was exposed to full sun. Samples were removed from the outdoor site and from the dark indoors at intervals over a 3-month period. Extracts were subjected to multi-layer and Florisil column chromatography and GC-MS analysis (Chatkittikunwong & Creaser, 1994). Since the photic zone in soils is only 0.2–0.4 mm (direct photolysis) and indirect photolysis is restricted to 2 mm (Hebert & Miller, 1990), photodegradation is not relevant in deeper soil layers.

65. For PBDD/Fs and PBCDD/Fs, direct photolysis is a relevant consideration as they absorb UV light at visible wavelengths. The direct photolysis of PBDDs/PBDFs was studied in various environments, including organic solvents, quartz surfaces, soil, and soot particles. Photodegradation was fastest in organic solvents under UV or sunlight, leading to reductive debromination and formation of lower brominated congeners. For example, half-lives for PBDDs/PBDFs in dodecane solutions exposed to laboratory daylight were estimated to be in the order of 100 to 1000 h for tri- to hexahalogenated congeners (Chatkittikunwong & Creaser, 1994). The calculated half-lives, assuming a first-order kinetic scheme, were in the range of minutes when direct sunlight or UV light and quartz vials were used. In contrast, photolysis was significantly slower in soil, suggested to be in the range 600 to 4000 hours for tri- to hexahalogenated congeners of PBDDs and PBCDDs (Chatkittikunwong & Creaser, 1994), and on airborne particles. The degradation rate depends on bromine substitution, with higher brominated congeners breaking down more rapidly. PBDD/Fs degraded faster than their chlorinated analogues, and photolytic degradation was more effective in controlled solutions than in real environmental conditions (WHO, 1998) where PBDD/Fs occur predominantly particle-bound, e.g. to airborne or waterborne particles.

66. During photolysis, PBCDD/Fs preferentially lose their bromine atoms and therefore are transformed into PCDD/Fs. Consistently short photolytic half-lives (0.5–4 min) were observed for the mono-B-tetra-CDD/Fs and di-B-tetra-CDD/Fs tested in iso-octane, whereas the resulting TeCDD/Fs were much more stable. Such a transformation of PBCDD/Fs to PCDD/Fs can also occur during incineration processes (WHO, 1998). Due to the large variation in the light available for different environmental compartments, these data cannot be used on their own within persistence assessments.

67. PBDD/Fs and PBCDD/Fs have been found in sediment cores from English lakes (Peters et al., 2021). Similarly, PBDD/Fs, including DiBDDs and TrBDDs, have been detected in sediment cores from Tokyo Bay (Goto et al., 2017). Their presence in sediment slices that were deposited several decades ago provide indirect evidence of their persistence under these conditions.

68. Since insufficient data is available for brominated analogues, information on PCDD/Fs is used for read-across. The chlorinated analogue, 2,3,7,8-TeCDD, is very resistant to biodegradation. In lake waters and sediments, a half-life of 550–590 days was observed for 2,3,7,8-TeCDD. In lake water alone, about 70% of the 2,3,7,8-TeCDD remained after 589 days. Using an outdoor pond as a model aquatic ecosystem, and dosing it with ^{14}C-labelled 2,3,7,8-TeCDD, the half-life was estimated to be approximately 1 year. Although biodegradation may have occurred, it is almost impossible to estimate the biodegradation half-life of 2,3,7,8-TeCDD in aquatic systems from this experiment according to the EHC 88 evaluation (WHO, 1989).

69. Biodegradation is more relevant than photolysis in many media, including sediment, deeper layers of soil and surface water. In the absence of an experimental dataset, biodegradation half-lives of PBDD/Fs and PBCDD/Fs were roughly estimated using the BIOWIN3 scores derived from the BIOWIN v4.10 module implemented in EPI Suite (US EPA, 2017). This was done for the PBDD/F congeners listed in Table 1 as well as for the analogous PBCDD/F and PCDD/F congeners. These scores were then converted into half-lives in water using the equation developed by Scheringer et al. (2012). Note that the use of the equation derived by Rorije et al. (2011) would result in half-lives that are higher by a factor of 1.69 to 2.19. These approaches were designed for screening purposes, and the authors note that the underlying ratings from BIOWIN3 are only semi-quantitative. The half-lives in water were converted into half-lives in soil by multiplying them by the factor of 1.85 (Boethling et al., 1995; Rorije et al., 2011). The authors indicate that the factor was designed for use in chemical screening and in setting priorities for further review. The estimated half-lives in water are greater than the threshold of two months for all PBDD/F and PBCDD/F congeners except the monobrominated ones as well as DiBCDFs. For soil, the estimated half-lives of all but the mono- and dihalogenated PBDD/F and PBCDD/F congeners exceed the threshold of six months (for numerical values see UNEP/POPS/POPRC.22/INF/#, Table 1 and Table 2).

70. The persistence of PBDD/Fs and PBCDD/Fs was further evaluated on the basis of empirical data available for PCDD/Fs. The relative duration of the half-lives of analogue chemicals, which differ only in the type of halogen atoms, is considered to be better captured by BIOWIN3 than their absolute values. As shown in Figure 2, the estimated half-lives of PBDD/F and PBCDD/F congeners are consistently higher than those of analogous PCDD/F congeners calculated by the same method. Therefore, the actual half-lives of PBDD/F and PBCDD/F congeners may be as high as those determined for their analogue PCDD/F congeners. Indicative half-lives of PCDD/F congeners relevant to the Baltic environment at an annual average temperature of 7 °C were suggested based on different photolysis and biodegradation data available in the literature (Sinkkonen & Paasivirta, 2000). Even though these half-lives consider biodegradation and photolysis, they agree quite well with the BIOWIN3 estimates for water, although the agreement decreases for higher chlorinated congeners (see Figure 2 and UNEP/POPS/POPRC.22/INF/#, Table 2). For soil, on the other hand, the PCDD/F half-lives derived by Sinkkonen & Paasivirta (2000) for the Baltic environment are considerably higher than those estimated based on the BIOWIN3 scores. Another data set of PCDD/Fs half-lives in soil is available from a more recent study (Terzaghi et al., 2020). Data were derived from a greenhouse experiment performed with an aged, contaminated soil and different plant species under semi-field conditions in Northern Italy over a period of 18 months. These extrapolated half-lives are closer to those estimated based on the BIOWIN3 scores but are mostly higher and do not show a clear trend.

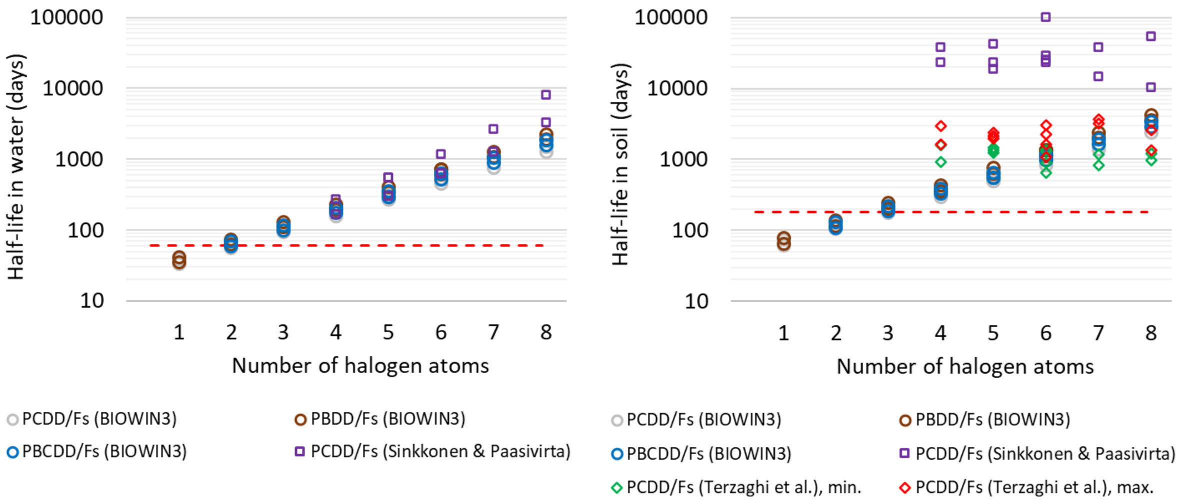

Figure 2. Estimated biodegradation half-lives of PBDD/F and PBCDD/F congeners in water (left) and soil (right), compared to those of their chlorinated analogues (PCDD/Fs) and grouped according to the number of halogen atoms. Empirically derived half-lives of PCDD/F congeners (Sinkkonen & Paasivirta, 2000; Terzaghi et al., 2020) are also displayed. Dashed lines represent the thresholds for persistence according to Annex D (60 and 180 days).

71. According to additional biodegradation predictions provided by the UK (Annex E, UK, 2024), PBDD/Fs and PCBDD/Fs are, similar to PCDD/Fs, not readily biodegradable in water, calculated using BIOWIN2 and BIOWIN6 for PBDD/Fs and calculated using BIOWIN2 for PCBDD/Fs (for values see UNEP/POPS/POPRC.22/INF/#, Table 3 and Table 4).

72. In conclusion, the available data suggests that tetra- to octahalogenated PBDD/Fs and PBCDD/Fs are persistent. They are at least as persistent as their chlorinated analogues. They meet the criteria for persistence with estimated biodegradation half-life values ranging from 175 to 2267 days in water and from 324 to 4195 days in soil. Di- and trihalogenated PBDD/Fs and PBCDD/Fs are also likely to meet the persistence criteria, with estimated biodegradation half-life values ranging from 59 to 130 days in water and from 109 to 241 days in soil, but with less confidence. On the other hand, monobrominated dibenzo-*p*-dioxins and dibenzofurans are unlikely to meet these criteria. The presence of PBDD/Fs in specific environmental compartments, such as in historic sediment cores, as compiled in section 2.3.1, provides further evidence for this conclusion.

2.2.2 Bioaccumulation

73. The log K_{OW} values of the PBDD/F congeners, estimated using the KOWWIN v1.69 module implemented in EPI Suite (US EPA, 2017), range from 4.9 to 11.5, and those of the analogous PBCDD/F congeners from 5.6 to 10.5 (see UNEP/POPS/POPRC.22/INF/#, Table 5). It is noted that the molecular weights of HpBDD, OBDD and OBDF are above the upper end of the values in the KOWWIN training set (720 g/mol), which might imply a somewhat lower accuracy for these compounds. Nonetheless, it is still safe to state that their log K_{OW} values are above 5. The KOWWIN estimates are in line with other predicted values of 5.23 to 7.90 for mono- to tetrabrominated PBDD/Fs using an older version of the same model (Arnoldsson, 2012a).

74. Although experimental BCF/BAF values are not available, empirical data to assess bioaccumulation exists for aquatic species. The halogen substitution pattern influences the retention of these substances. PBDDs with unsubstituted vicinal positions, i.e. Di- and TrBDDs, are less strongly retained than other congeners in fish, but eliminated and metabolized (Arnoldsson et al., 2012b). The maternal transfer from fish to eggs was found to be negatively correlated with the degree of bromination. Thereby, concentrations in fish and eggs were examined during a 6- or 12-week uptake period, followed by a 6-week elimination period via a food uptake study (Arnoldsson et al., 2012b). Nominal concentrations of 1 ng congener per gram of feed were tested. Similar to OECD TG 305, fish were fed with 2% of their body weight each day. Half-lives were in the order PCBs > PCDD/Fs > PBDD/Fs. For example, the half-life of 2,3,7,8-TeCDD was 36 days, the half-life for 2,3,7,8-TeBDD was 12 days and the half-life for 2,3,7,8-TeBDF was 5.0 days.

75. On the other hand, PBDDs appear to be more resistant to mammalian metabolism than PCDDs (Birnbaum et al., 2003; Bjurlid et al., 2018a). PBDD/Fs have been found in organisms that are high up in the food web, including 22 pooled samples from 69 ringed seals (*Pusa hispida botnica*) from the Baltic Sea from 1974 to 2015 (Bjurlid et al., 2018b) and juvenile male pilot whales (*Globicephala melas*) caught around the Faroe Islands during the period 1997–2013 (Bjurlid et al., 2018a). 1,2,3,4,6,7,8-HpBDF was the most abundant congener in both studies. The concentration range for ∑PBDD/Fs was 0.080–71 pg/g lw (lipid weight) (median 3.6 pg/g lw) in 20 of 26 whale blubber samples, and 0.5–52.3 pg/g lw (median 8.6 pg/g lw) in the seal blubber (22 pooled samples from 69 seals). In the whale

blubber, the PBDD/F-TEQ (using the same TEF as for PCDD/Fs) ranged from 0.0039 to 4.7 pg TEQ/g lw, with average and median values of 0.34 pg TEQ/g lw and 0.043 pg TEQ/g lw, respectively (Bjurlid et al., 2018a, 2018b). These data demonstrate the concentrations are log normally distributed. In both species, no correlation was found between concentrations of PBDD/Fs and PBDEs, which indicates differences in exposure or tentatively differences in metabolism of PBDEs and PBDD/Fs (Bjurlid et al., 2018a). In Japan and Sweden, PBDD/Fs have also been found in human adipose tissue (Choi et al., 2003; Ericson et al., 2008; Hagberg et al., 2011; Lundstedt, 2016). These data show widespread global occurrence and bioavailability in different species and trophic levels.

76. In terrestrial mammals, the strongest evidence on bioaccumulation was found for some tri- and tetra-brominated congeners (notably 2,3,7,8-TeBDD) showing uptake and tissue distribution while data for heavier congeners (5–8 Br) was not available. Elimination half-lives for 2,3,7,8-TeBDD in rat liver, feces and adipose tissue were estimated to be 17, 18, and 58 days, respectively. They are similar to those reported for 2,3,7,8-TeCDD in liver and feces, but higher than those in adipose tissue. Despite differences in early retention, half-lives of 2,3,7,8-TeBDF and 2,3,7,8-TeCDF in liver are comparable (WHO, 1998). While 2,3,7,8-TeCDF is metabolized rather rapidly in rats and mice with a half-life of less than 2 days, 2,3,7,8-TeBDF appears to be much more resistant to metabolism, resulting in a longer half-life (Birnbaum et al., 2003). In a study from Tue et al. (2024) the terminal half-life in plasma of TeBDF in mice is 14 days. The elimination half-life of 2,3,7,8-TeBDD in rats is between 2 and 3 weeks, similar to that of 2,3,7,8-TeCDD. If the relationship between rat and human half-lives for 2,3,7,8-TeCDD holds for 2,3,7,8-TeBDD, this would suggest a half-life in the range of 5 to 10 years in humans for TeBDD (Birnbaum et al., 2003). According to ECHA (2023a), a terminal elimination half-life greater than 4 days in rats (whole-body) and/or greater than 50 days in humans is indicative of high bioaccumulation (vB).

77. Elimination half-lives ranging from 1.1 to 1.9 years and from 2.9 to 10.8 years have been determined for 2,3,7,8-TeBDF and 2,3,7,8-TeBDD, respectively, in workers assigned to the extrusion of plastic resins containing PBDEs (Zober et al., 1992). A chemist who had synthesized 2,3,7,8-TeBDD in 1956 was found to have markedly elevated blood levels 35 years after exposure (WHO, 1998). The estimated half-lives for humans are 3–11 years (mean: 5.9 years) for 2,3,7,8-TeBDD and of 1–2 years (mean: 1.5 years) for 2,3,7,8-TeBDF (WHO, 1998).

78. After rats were exposed via gavage, the major tissue depots of radioactivity were liver, adipose tissue, and skin and the major route of excretion was the feces where between 30% and 80% of the dose was found 72 hours after administration (Diliberto et al., 1993; Mennear & Lee, 1994).

79. The long elimination half-lives of PBDD/Fs in mammals, suggest that these substances are highly bioaccumulative. The occurrence in human tissues and in breast milk (see paragraph 112) adds to the concern. PBCDD/Fs have physicochemical properties in the range between PCDD/Fs and PBDD/Fs, including a log K_{OW} at or greater than 5. Since both PCDD/Fs (as a recognized POP) and PBDD/Fs are bioaccumulative, it is very likely that this is also the case for PBCDD/Fs. Moreover, in combination with the high toxicity of PBDD/Fs and PBCDD/Fs (see below), even moderate bioaccumulation can lead to body concentrations that elicit adverse effects. Furthermore, PBCDD/Fs may photochemically be transformed to PCDD/Fs. Likewise, higher brominated PBDD/Fs may form lower brominated PBDD/F congeners (WHO, 1998).

80. In conclusion, the long elimination half-lives, the estimated log K_{OW} values above 5 and the monitoring data demonstrate that di- to octahalogenated PBDD/Fs and PBCDD/Fs fulfil the criteria for bioaccumulation.

2.2.3 Potential for long-range environmental transport

81. PBDD/Fs and PBCDD/Fs undergo atmospheric oxidation by reaction with OH radicals. For PBDD/F congeners, the rate was found to be dependent on the bromination pattern (Zeng & Wang, 2018). The AOPWIN v1.92 module implemented in EPI Suite (US EPA, 2017) was applied to calculate the gas phase half-lives of PBDD/F and PBCDD/F congeners, based on second-order rate constants for the reaction with OH radicals. A 24-hour average concentration of photochemically generated OH radicals in air of 7.5×10^5 OH radicals/cm^3 (as implemented in AOPWIN) was used. The complete training sets for AOPWIN's estimation methodology are not available. Therefore, the reliability of the predictions is unknown. The half-lives obtained are 1.8 to 504 days for PBDD/F congeners and 2.8 to 435 days for PBCDD/F congeners. These estimates are slightly higher than for the analogous PCDD/F congeners (see UNEP/POPS/POPRC.22/INF/#, Table 6), which are recognized POPs. PBDD/Fs can undergo photolysis (Yang et al., 2021a). However, due to their physicochemical properties, the PBDD/F and PBCDD/F congeners are expected to occur predominantly particle-bound in the atmosphere (WHO, 1998). The fractions of PBDD/Fs and PBCDD/Fs sorbed to particles are higher than for PCDD/Fs, in line with their lower vapor pressures (Yang et al., 2021a). As the sorbed fraction undergoes little or no atmospheric oxidation, the atmospheric half-lives will be higher than those calculated for the gas phase. For PBDD/Fs and PBCDD/Fs, direct photolysis is a relevant consideration as they absorb UV light at visible wavelengths. Section 2.2.1 indicates that direct photolysis in sunlight is rapid in quartz vials, but significantly slower when the substances are adsorbed to soil or airborne particles. The limited available data suggests the half-life in air is likely to exceed two days where PBDD/Fs and PBCDD/Fs are bound to particles.

82. A recent modelling study concludes that each of the investigated PBDD/F congeners (tetra- to heptabrominated) exceeds the modeled accumulation levels of other POPs in remote regions when emitted to air or water, but not to soil. For the modelling, selected brominated dioxins and furans (PBBD/Fs) were screened for their potential to be dispersed, transferred to, and accumulated in remote regions. The screening was carried out applying a set of long-range transport potential (LRTP) metrics, collectively referred to as the emissions fractions approach (EFA), as implemented in a modified version of the OECD P_{OV}[5] and LRTP Screening Tool. The Tool is parameterized to reflect the environmental characteristics of the global environment, and it includes one air, one water, and one soil compartment. Input parameters were the log K_{OW}, the log K_{AW} and the degradation half-lives in air, water, and soil. The required input parameters were retrieved from EPI Suite using SMILES (Breivik et al., 2023). As input values such as degradation half-life in air have uncertainties associated with them, the modelling results should be treated with caution.

83. Another modelling study by Kawai et al. (2014) compared the long-range transport potential of groups of halogenated POPs and POP-like chemicals using a new metric, the global average of imported fractions in receptor regions (GIF). The authors used a 3D atmospheric transport model (FATE) to simulate the transport and fate of various chemicals, including PCBs, PCDDs, PCDFs, and PBDEs, and compared the performance of the new GIF metric with existing metrics. The GIF of PBDDs and PBDFs was 0.09 to 0.21 and 0.11 to 0.21, respectively, which overlap with the ranges obtained for PCDDs (0.17–0.22) and PCDFs (0.13–0.27). It should be noted that the number of congeners investigated for PBDDs (20) and PBDFs (31) was higher than for PCDDs (5) and PCDFs (13), respectively. The GIF is defined as a global average of imported fractions in receptor regions weighted by the contents. GIFs for investigated POPs range from 0.08 (for BDE-209) to 0.5 (for PCB-36). A higher GIF means a higher LRTP.

84. According to Lundstedt (2016), the strong dominance of PBDFs, over PBDDs, in studies from different regions around the world, such as China, Japan and Sweden supports the theory that most PBDD/Fs in the atmosphere, even in remote areas, originate from BFR-containing materials. Air concentrations of PBDD/Fs were similar in the city of Gothenburg on the Swedish west coast (number of samples: 4 in total; years of monitoring: 2011–2012; concentrations measured: 1.5–1.8 fg/m^3 for ∑PBDD and 2.1–510 fg/m^3 for ∑PBDFs) and in Råö (number of samples: 16; years of monitoring: 2011–2012; concentrations measured: 1.1–310 fg/m^3 for ∑PBDD and 34–1200 fg/m^3 for ∑PBDFs), the background monitoring station on the Swedish west coast. The concentrations of PBDD/Fs in air samples from Pallas (number of samples: 2; years of monitoring: 2011–2012; concentrations measured: < LOQ for ∑PBDDs[6] and 39–91 fg/m^3 for ∑PBDFs), a remote station in northern Finland were lower, but indicate long-range environmental transport (Lundstedt, 2016; Remberger et al., 2014). According to this data, PBDF levels in remote areas (Pallas) are approximately 4 to 10 times lower than close to sources (Gothenburg and Råö). PBDDs were generally found at lower levels and were not found above detection limits in Pallas. PBDD/Fs were found in all air and deposition samples from the background areas and the importance of atmospheric long-range transport to Sweden was confirmed (Remberger et al., 2014). The authors noted that there appeared to be an annual variation in both PBDD/F and PCDD/F concentrations, with higher levels during winter. They also noted that significantly elevated levels were detected in May–June 2011, which coincided with extensive forest fires in southern Russia and Ukraine (Remberger et al., 2014). Another study indicating long-range transport of PBDD/Fs in air masses was conducted near Chinese Taiwan, where Chao et al. (2014) measured 0.0013 pg WHO-TEQ/m^3 in the air over the ocean. Only the highly brominated PBDF congeners were detectable. 1,2,3,4,7,8-HxBDF, 1,2,3,4,6,7,8-HpBDF and OBDF were found in the atmosphere over both the ocean and land. The mean atmospheric PBDD/F TEQ levels for the ocean and land were comparable. The authors conclude that halogenated POPs (note: including the PBDD/Fs) could easily reach remote ocean sites via atmospheric transport (Chao et al., 2014). However, to clarify the contribution of other potential nearby sources of PBDD/Fs and possible transport pathways, further analysis would be required. In conclusion, the evidence particularly demonstrates the presence of PBDFs in the atmosphere, also in remote areas. It could therefore be concluded that there is evidence for long-range atmospheric transport of PBDFs.

85. PBDD/Fs are present in commercial PBDE mixtures (Hanari et al., 2006; L. Yang et al., 2021a) and other commercial BFRs, e.g. in TBBPA (Thoma et al., 1986a and Dumler et al., 1989b as cited in European Commission, 2006). As a consequence, PBDD/Fs are present as a by-product in plastics containing brominated flame retardants and can contribute to PBDD/F releases to the environment. Significant quantities of waste plastics, including plastics containing brominated flame retardants such as PBDEs or TBBPA, are released into the environment. Multiple studies have confirmed the presence of PBDEs in marine plastic litter in varying concentrations, including in remote locations, including buoyant microplastics (Tanaka et al., 2019; Turner, 2022). Regarding PBDEs, estimates associated with the input of PBDEs to the ocean when “bound” to marine plastics and microplastics range from about 360 to 950 tonnes per year based on the annual production of plastics and PBDEs over the past decade, and from

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[5] “P_{OV}“ means “overall persistence”

[6] Some individual PBDD congeners could be quantified: 2,3,7,8-TeBDD (0.059–0.10 fg/m^3), 1,2,3,7,8-PeBDD (0.049–0.13 fg/m^3), 1,2,3,4,7,8- and 1,2,3,6,7,8-HxBDD (0.19–0.51 fg/m^3), 1,2,3,7,8,9-HxBDD (0.048–0.39 fg/m^3), 1,2,3,4,6,7,8-HpBDD (0.65–0.82 fg/m^3).

about 20 to 50 tonnes per year based on the abundance and distribution of PBDEs in marine plastic litter (Turner, 2022). Marine (micro)plastics are subject to long range-environmental transport and plastic fragments have been consistently detected since the 1970s on remote beaches, in sea water, sea ice and glacier ice in remote regions as well as in wildlife living in less populated to remote areas (Andrade et al., 2021; Turner, 2022). In conclusion, PBDD/Fs are subject to long-range environmental transport since they are present and transported in marine plastics flame retarded with BFRs such as PBDEs or TBBPA.

86. In addition, PBDEs incorporated into plastic have shown to transform to PBDFs due to photolytic degradation (Kajiwara et al., 2008, 2013; see paragraphs 35 and 49). Photolytic transformation of BFR-containing materials occur; however, quantitative data demonstrating significant formation yields of PBDD/Fs under realistic outdoor conditions representative of remote environments are limited. Moreover, mechanistic assessments of BFR phototransformation suggest that some proposed direct-photolysis routes are not necessarily dominant (e.g., pathways with high activation barriers), and that alternative reactions such as debromination and indirect processes via reactive oxygen species may be more favourable (S. Wang et al., 2019). In conclusion, photolytic formation of PBDD/Fs in BFR-containing plastics after their transport to remote regions may occur, however, the relative contribution of photolysis-driven formation to long-range environmental transport remains uncertain and is likely to be limited.

87. As mentioned in paragraph 75, PBDD/Fs have been detected at median concentrations of 3.6 pg/g lw in the blubber of long-finned pilot whales (*Globicephala melas*) caught around the Faroe Islands (Bjurlid et al., 2018a). These pilot whales occur regularly in the Faroese archipelago. Their migration is related to prey distribution and ocean temperature. If their favourite squid prey is not available, pilot whales leave to feed elsewhere. Though they are observed in the coastal areas of the Faroe Islands, they are not restricted to this area and they migrate throughout a much larger area in the North Atlantic (Abend & Smith, 1999; Bloch et al., 2003; Buckland et al., 1993). In conclusion, the occurrence of PBDD/Fs in these whales and their migration activities provides specific evidence that PBDD/Fs undergo long-range environmental transport via migratory species.

88. PBDD/Fs were identified in lichens (5.7–16.2 pg/g), soils (0.25–2.62 pg/g), and ornithogenic soils (0.77–1.61 pg/g) in the Antarctic coastal environment (Mwangi et al., 2016). The authors extrapolated that the relatively elevated PBDD/F contents in samples from sampling sites near research bases could be from the burning of waste containing plastics used for generating electricity and domestic heating in the Antarctic research stations. Therefore, due to the proximity of these sites to contamination sources, no conclusions can be drawn on the long-range transport potential of PBDD/Fs from this study. However, significant PBDD/F contents were also found in ornithogenic soil samples collected from sites where penguin colonies are located and which are not located near the research bases. The significant contents in these soils indicate long-range environmental transport. They are likely due to the penguins enriching the soil with pollutants through their fecal matter. After deposition, POPs enter the food chain, and they have been detected in various biota, including krill (Bengtson Nash et al., 2008). Penguins are flesh-eating birds that mainly prey on krill (Corsolini et al., 2007; Mwangi et al., 2016) facilitating the bioaccumulation of POPs and their subsequent transfer to terrestrial soils via fecal matter.

89. In summary, it can be stated that PBDD/Fs have been detected in biota and soil in remote regions (see paragraphs 75 and 88). Detection was at low concentrations, however, it should be noted that the Stockholm Convention does not require a concentration threshold for the criterion of long-range environmental transport. Furthermore, PBDD/Fs and PBCDD/Fs exhibit significantly higher degree of toxicity in comparison to the majority of acknowledged POPs, rendering even low concentrations potentially hazardous. For PBCDD/Fs, specific monitoring data are not available due to analytical challenges (see paragraph 91). Estimated gas-phase half-lives of PBDD/F and PBCDD/F congeners range from 1.8 to 504 days. However, the substances are predominantly particle-bound in the atmosphere and model results suggest that the half-life in air is likely to exceed the two days threshold for long-range environmental transport in air (see paragraph 81). Modelling provides further evidence for the potential for long-range environmental transport of PBDD/Fs and PBCDD/Fs (see paragraphs 82 and 83). PBDD/Fs in air from background areas confirms the importance of atmospheric long-range transport (see paragraph 84). PBDD/Fs are also subject to long-range environmental transport in marine (micro)plastics (see paragraph 85) and PBDD/Fs undergo long-range environmental transport via migratory species (see paragraph 87).

90. In conclusion, di- to octahalogenated PBDD/Fs and PBCDD/Fs fulfil the criteria for long-range environmental transport.

2.3 Exposure

2.3.1 Environmental monitoring data

91. Monitoring data for PBCDD/Fs in the environment and the human food chain are limited. This is mainly due to the complex analytical methods required and the lack of analytical standards (Annex E, BSEF, 2025; Kannan et al., 2012; Myers et al., 2012; van den Berg et al., 2013; Weber et al., 2015). It should be emphasized here that their chlorinated analogues, PCDD/Fs, are well studied and monitored in the environment and human food chain; they are ubiquitously distributed at global level. Recently, a new workflow to screen PBCDD/Fs by two-dimensional gas

chromatography/time-of-flight mass spectrometry and post-data processing has been suggested (Hashimoto et al., 2021).

92. Limited historical data on the occurrence of PBDD/Fs and PBCDD/Fs have been recorded, most of which confirm emissions from incineration processes containing chlorine and bromine in the raw materials (M. Wang et al., 2023; M.-S. Wang et al., 2010b). Goto et al. (2017) investigated the occurrence of mono- to octabrominated PBDD/Fs in Tokyo Bay, Japan, using ^{210}Pb-dated sediment cores covering the period 1895–2000. PBDDs were more abundant than PBDFs before the 1960s. PBDF levels were found to increase up to the most recent sediment layer. Peters et al. (2021) analyzed temporal trends of PBDEs, PBDD/Fs and PBCDD/Fs using ^{210}Pb-dated cores from English lakes between 1950 and 2015 and found that PBDEs increased towards the present and were significantly correlated ($R = 0.88–0.98$; $p < 0.05$) with concentrations of PBDFs ranging from 0 to 19,000 pg/g organic carbon, whereas PBCDD/Fs concentrations decreased. The authors suggest that the sources of PBCDD/Fs are not related to PBDEs, and that they differ from those of PBDFs. They also argue that examining PBDF homologue patterns implies the importance of emissions from combustion activities such as waste incineration and iron ore sintering (Peters et al., 2021).

93. Several researchers have investigated the distribution of PBDD/F contaminants in sediment samples from different global locations. Xiao et al. (2016) found that intensive e-waste recycling in southern China has led to severe PBDD/F contamination in workshop dust, soils and sediments, with the estimated daily intake of PBDD/Fs for local residents and children through soil/dust exposure exceeding that of their chlorinated counterparts (PCDD/Fs). Xiao et al. (2016) reported high mean concentrations in in surface sediment from Chinese e-waste areas Longtang (527 pg/g), that correspond to the mean concentration found by Liu et al. (2022) in another Chinese e-waste area Guiyu (355 pg/g). A level of 1,860 pg WHO-TEQ/g of PBDD/Fs was measured in sediment from a pond inside a dumpsite in Kalasin Province, Thailand, where e-waste is burned (Dvorská et al., 2023). A very high PBDD/F concentration in sediment up to 10,000 pg/g was found at a coastal area of Sweden adjacent to some metal smelters that partially used e-waste as feed stocks (Lundstedt, 2016). Edgbaston Pool, a lake in a highly populated area in the UK, had anomalously high levels up to 7,100 pg/g (Peters et al., 2021). Levels from other industrialized regions show a great diversity suggesting different pollution profiles in different urban and industrial landscapes. Examples are South Korea Sea Coast (n.d. to 460 pg/g) (Terauchi et al., 2009), Hong Kong Coast (China) (11.0–217 pg/g) (Terauchi et al., 2009), Japan's Osaka Bay (2.4–590 pg/g) (Takigami et al., 2005), Tokyo Bay (2.2–66 pg/g) (Goto et al., 2017), Baiyangdian (China) (n.d. to 280 pg/g) (Zhao et al., 2019), and the Pearl River Delta (China) (1.57–1300 pg/g) (Liu et al., 2023).

94. In seawater at Kaohsiung Harbor (KH, Chinese Taiwan) total dioxin-like toxicity from PCDD/Fs, PCBs and PBDD/Fs ranged from 0.00936 to 0.167 pg WHO-TEQ/L, with PBDD/Fs accounting for 19% of total toxicity. The authors state that PBDEs in seawater can transform into PBDD/Fs upon UV light exposure, highlighting another potential pathway for the persistence and spread of these harmful contaminants in the environment (Wardiani et al., 2024). The PBDD/F levels totalled 0.0169 and 0.00195 pg WHO-TEQ/L at sampling sites KH-1 and KH-2, respectively, but ranged from 0.00224 to 0.0206 pg WHO-TEQ/L for sampling sites KH-3, KH-4, and KH-5, with KH-5 predictably accounting for the maximum.

95. Haglund et al. (2010b) reported PBDD levels in perch for the timeframe between 1990 and 2005 to be in the range from 0.4 to 73 pg/g fresh weight (fw). PBDD/Fs have been found in a broad range of marine biota, including pilot whales, seals, fish, shellfish and mussels (Bjurlid et al., 2018a, 2018b; Fattore et al., 2018; Fernandes et al., 2009; Haglund et al., 2007). The data suggest that typically PBDFs occur more frequently and generally at higher levels than PBDDs, which indicates in many cases anthropogenic sources. For details see UNEP/POPS/POPRC.22/INF/#, Table 7. Some species present in the Baltic Sea show a prevalence of less brominated congeners (mono- to penta-), which combined with an analysis of the spatial distribution, suggests local natural biogenic production, for example in red algae (*Ceramium tenuicorne*), blue mussels or aquatic sponge (*Ephydatia fluviatilis*) (Arnoldsson et al., 2012b; Haglund et al., 2007; Unger et al., 2009). Malmvärn et al. (2018) detected Tr- to PeBDFs (concentrations: 0.03 ng/g fw for TrBDDs) in red algae from the Baltic Sea. In Baltic wild salmon (*Salmo salar*), 3-B-2,7,8-tri-CDF was detected in the concentration range from 0.039 to 0.075 pg/g fw (Zacs et al., 2013). 22 other PBCDD/Fs were not detected. The fish were sampled from two Latvian rivers. PBDD/Fs were present above the relatively high LOD values (range 0.003–4.5 pg/g fw) in 5 out of 65 samples of wild and farmed fish from the Mediterranean Sea. The concentration values of detected congeners ranged from 0.01 to 0.89 pg/g fw and the highest values were found in a wild species *(Sarda sarda*) (Fattore et al., 2018).

96. Between 1974 and 2013, PBDD/F concentrations in Baltic ringed seals ranged from 0.5 to 52.3 pg/g lw (Bjurlid et al., 2018b). While visual inspection of the data showed declining levels for other pollutant groups (e.g., PCDD/Fs, PCBs), PBDD/F concentrations exhibited a slight increase starting in 2004. However, the overall low concentrations of PBDD/Fs precluded the identification of long-term temporal trends. Falandysz et al. (2020) studied the PBDD/F concentrations in cod liver oil sourced from the Baltic Sea between 1972–2001. All five PBDDs analysed were below detection limit. Of six PBDFs analysed, four were below detection limit. No significant trend could be established.

97. Wang et al. (2008) investigated PBDD/Fs in different atmospheric environments in Chinese Taiwan. The mean PBDD/F concentrations in the atmosphere of the rural, urban, industrial, and science park areas were 11, 24, 46, and 95 fg/Nm^3, respectively. According to the same authors, PBDF concentrations in stack flue gases of full-scale MSWIs amounted to 2.28 pg/Nm^3 (Wang & Chang-Chien, 2007). PBDFs were all much more dominant than PBDDs in the atmosphere, and their mass fractions increased with PBDD/F concentrations. In another study in Kyoto, Japan on atmosphere and bulk deposition also PBDFs were predominant and PBDDs were detected only at trace levels (Hayakawa et al., 2004).

98. Since 2002, Japan has surveyed PBDD/Fs in exhaust gas, drainage and air inside buildings of selected facilities focusing on the life cycle of products that contain BFRs. The facilities to be surveyed are selected each year. The results are published (in Japanese) (NIP Japan, 2016). Monitoring data are available from Japan starting from 2002 for PBDD/Fs in the surrounding areas of selected facilities focusing on the life cycle of products that contain BFRs. Concentration ranges in the atmosphere from 0.10 to 990 pg/m^3 (average 22 pg/m^3) with higher concentrations around a flame-retardant resin manufacturing facility (2002) and flame-retardant manufacturing facilities (2003 and 2006) were found. In dust fall, a range from 140 to 180,000 pg/m^2/day (average 16,000 pg/m^2/day) was found with higher concentrations at a home appliance recycling facility (2002) and a flame-retardant resin manufacturing facility (2002). In public water bodies (waters for public use such as rivers, lakes, ports and harbors, coastal seas, etc., including public waterways connected thereto, irrigation waterways and other public-use waterways) a range from n.d. to 7,400 pg/L (average 330 pg/L) was found with higher concentrations around a flame-retardant textile processing facility and around a sewage treatment facility. In public water sediments, a range from n.d. to 13,000 pg/g dw (average 630 pg/g) was found with higher concentrations around a flame-retardant resin manufacturing facility (tribromophenols end-treated TBBPA type epoxy resin), a flame-retardant resin manufacturing facility (TBBPA epoxy resin), a home appliance recycling facility and a flame-retardant textile processing facility (Annex E, MoE Japan, 2024). PBDD/F levels in dust from the Ban Nong Bua dumpsite reached 16,900 pg WHO-TEQ/g dw, while levels in ash from the same site were 17,500 pg WHO-TEQ/g dw. In the shredding plant at an e-waste site in Kalasin Province, Thailand, PBDD/F levels were slightly above 426 pg WHO-TEQ/g dw (Dvorská et al., 2023). At a different location near the e-waste processing plant in Khao Hin Son, Thailand, PBDD/F levels ranged from 4 to 10 pg WHO-TEQ/g dw (Petrlik, Boontongmai, et al., 2022).

99. PBDD/Fs have also been found in soil (Amuah et al., 2024; Liu et al., 2025c; Tue et al., 2016; Zhou et al., 2021). Amuah et al. (2024) summarized data from previous studies in Agbogbloshie, Ghana. Total PBDD recorded levels in soil in pg/g dw ranged from n.d.–5.0 (non-e-waste site) to 1.3–490 (non-burning e-waste site) and 120–4,000 (open burning e-waste site). Tue et al. (2016) further recorded total PBDF levels in soil in pg/g dw ranging from n.d.–100 (non-e-waste site) to 2,600–83,000 (non-burning e-waste site) and 83,000–3,800,000 (open burning e-waste site). The total levels (ng/g) of PBDD/Fs (83–3800) > PCDD/Fs (18–520) > dl-PCBs (3.4–83) were detected in the open burning areas of the Agbogbloshie e-waste site. The constituents in non-burning areas (PBDD/Fs: 2.6–83 ng/g; PCDD/Fs: 1.0–7.0 ng/g; dl-PCBs: 1.4–7.5 ng/g) were 1–2 orders lower than those in open burning areas, but about two orders higher than those in non-e-waste locations in Accra (n.d. to 100 ng/g; 8.0–300 ng/g and 2.7–290 ng/g) (Amuah et al., 2024). PBCDD/Fs have been found in soil samples after a plastic recycling-plant fire in Canada within the ng/g range with mono-B-di-CDFs as most prevalent congeners (8.6 to 180 ng/g) (Myers et al., 2012).

100. PBDD/Fs have also been found in soil, wastewater, sludge, fly ash, various animal and plant species, food and feed, indoor dust and humans (Altarawneh et al., 2019; Bell et al., 2023; Lundstedt, 2016; Ma et al., 2009). Bell et al. (2023) analyzed disposed waste incineration residues in Chinese Taiwan for PBDD/F residues. PBDD/Fs were detected in ash samples from two of five sampled localities, with concentrations up to 61 pg WHO-TEQ/g dw. Ma et al. (2009) reported total PBDD/F (10 congeners) concentrations in leaves (n=6) (from a nearby e-waste recycling facility) of 113–818 pg/g dw, in an electronic shredder (n=5) of 392–18,500 pg/g dw, in soil samples (n=10) of 716–800,000 pg/g dw, in workshop-floor dust (n=5) of 89,600–143,000 pg/g dw, and in soil from a chemical-industrial complex (n=12) up to 427 pg/g dw. PBDD/Fs were also found in blood, viscera and meat samples of cattle raised in one Indonesian landfill (n=2) with concentrations up to 38.6 pg/g fat in viscera with 1,2,3,4,6,7,8-HpBDD as the predominant congener (Ismawati et al., 2024). PBDD/F levels ranging from below the LOQ to 1.7 pg WHO-TEQ/g fw were measured in apple snails from the vicinity of a dumpsite where e-waste is burned in Thailand (Dvorská et al., 2023). Apple snails are part of the diet for local community.

2.3.2 Human exposure

101. The TEQ scheme has enabled the setting of tolerable intake thresholds for PCDD/Fs and dioxin-like PCBs. In 2001, the Joint FAO/WHO Expert Committee on Food Additives (JECFA) established a provisional tolerable monthly intake (PTMI) for dioxins and dioxin-like compounds of 70 pg/kg bw per month (JECFA, 2002). More recently, the European Food Safety Authority (EFSA) has set a tolerable weekly intake for dioxin-like contaminants of 2 pg TEQ/kg bw/week as set by the EFSA Panel on Contaminants in the Food Chain (CONTAM) et al., 2018). Various authors (van den Berg, 2013; Mortimer, 2013) propose to apply the same scheme for PBDD/Fs.

102. Assuming a 70 kg person and considering the tolerable weekly intake of 2pg/TEQ bw/week and the average PBDD/F TEQ concentration value found in pilot whale blubber (0.34 pg TEQ/g lw; see paragraph 75), the amount of

blubber that a person could consume before surpassing the tolerable weekly intake limit would correspond to 59 g/day. In a worst-case scenario, applying the maximum PBDD/F TEQ concentration found in the blubber (4.7 pg/TEQ/g lw), a person consuming more than 4.3 g blubber day would exceed the tolerable weekly intake of 2pg/TEQ bw/week. Faroese adults showed an average consumption of 7 g whale blubber per day (Weihe et al., 1996). According to dietary recommendations issued in 1997, pregnant women should not consume pilot whale meat and blubber and many Faroese people, particularly women and children, have stopped eating pilot whale. However, according to a study with 200 randomly chosen adults, still 17% of Faroese people eat pilot whale more than once a month (Ceta Journal, 2013) and could therefore exceed the tolerable weekly intake of 2 pg/TEQ bw/week. Also other communities in remote regions consume undisclosed species of whales (Chukmasov et al., 2019; Eduard, 2021; Kozlov & Zdor, 2003; Siddiqui et al., 2024) in large quantities, especially indigenous populations in Arctic regions. In addition, POPs have been found in other species of migratory whales consumed by indigenous populations, and it is very likely that PBDD/Fs are also present in these species (Chukmasov et al., 2019). In conclusion, this demonstrates that traditional consumption of whale blubber, such as in the Faroe Islands, and consumption of whale in other communities in remote regions can already surpass safe limits of consumption as a result of the PBDD/F content in whale from remote regions and can cause adverse effects as a result of long-range environmental transport and exposure to PBDD/Fs in a remote region (i.e. without considering the TEQ contribution from PBCDD/Fs and other dioxins or dioxin-like compounds).

103. Fish are a major source for the intake of PBDEs. It was experimentally determined that PBDFs are formed when a salmon sample spiked with decaBDE is heated at 200 °C. Although concentrations of decaBDE in wild catch were lower than the spiking level (200 ng/g fish), it was concluded that due to the formation of PBDFs in this process, the consumption of heated, decaBDE-contaminated fish may add to the uptake of dioxin-like compounds with our diet (Vetter et al., 2015).

104. Levels of PBDD/Fs in food samples from Italy showed that terrestrially raised food had low or nondetectable levels of contamination compared to fish and shellfish with concentrations of <0.001 to 4.58 pg/g in fish. The dietary exposure (upper bound) of different Italian population groups, resulting from these occurrence levels was estimated using TEQ approach that is commonly used for dioxin-like contaminants and ranged from 0.17 to 0.42 pg TEQ/kg bw/day. The authors conclude that the PBDD/F exposure levels alone would likely exceed the EFSA tolerable weekly intake of 2 pg/kg bodyweight/week for some of the assessed population sub-groups, while cumulative intake including PCDD/Fs and PCBs would exceed this threshold for all assessed population subgroups (Diletti et al., 2020).

105. In the dietary study conducted by Mortimer et al. (2013), the total diet was estimated by measuring a range of contaminants in composites representing 19 food groups (such as bread, cereals, carcass meat, offal, poultry, fish, etc.) in the UK. Dietary exposures for average and high-level (97.5th percentile, P97.5) consumers were estimated by using the Intake Programme (software developed for the Food Standards Agency). The estimated dietary exposures to total PBDD/F/PBB TEQ/kg bw/day were estimated for all age groups (toddlers, children aged 4–6 years, children aged 7–10 years, children aged 11–14 years, youths, adults and the elderly), applying the TEFs for chlorinated compounds to their brominated and mixed halogenated analogues. The highest dietary exposure estimates were found for toddlers (mean, upper bound (UB): 0.95, P97.5, UB: 1.64 pg WHO_{2005}-TEQ/kg bw/day) and children aged 4–6 years (mean, UB: 0.84, P97.5, UB: 1.29 pg WHO_{2005}-TEQ/kg bw/day). The contribution to the total TEQ from the PBDD/Fs/PBCDD/Fs/PBBs was consistently around 30% for all age groups and for average and high consumers. The authors of the study concluded that brominated and probably also mixed halogenated dioxins, furans and biphenyls may make a significant contribution to dietary exposure to dioxin-like compounds.

106. PBDD/Fs are present in most commonly consumed foods contributing to the dietary intake of PBDD/Fs (Fernandes & Falandysz, 2021; Rose et al., 2010). Analysis of various food samples (eggs, milk, fish, shellfish, pork, beef and poultry) showed mean TEQs for the PBDD/Fs ranging from 0.15 to 0.48 pg/g fat upper bound in the animal products and 0.03–0.05 pg/g whole weight in shellfish and fish. The most frequently detected congeners were 2,3,7,8-TeBDF, 2,3,7,8-TeBDD as well as 1,2,3,4,6,7,8-HpBDD/F. 2,3,4,7,8-PeBDF, 1,2,3,4,7,8-HxBDF and OBDF were also detected in some samples (Ceci et al., 2022). In another study, PBDD/Fs were found at higher concentrations than their chlorinated analogues in lower-lipid food groups like bread, cereal, potatoes, and fresh fruit, with the most abundant congener being 1,2,3,4,6,7,8-HpBDF (Bramwell et al., 2017). In the UK and Ireland, as observed in other studies, PBDFs were more prevalent than PBDDs, with the most frequently found congeners 2,3,7,8-TeBDF, 2,3,4,7,8-PeBDF, 2,3,8-TrBDF and 1,2,3,4,6,7,8-HpBDF (Rose & Fernandes, 2010). In the research by Teebthaisong et al. (2021), fifteen pooled samples of free-range chicken eggs (from 14 hot spots around the world) and two reference samples from supermarkets were analyzed for PBDD/Fs. The highest level of PBDD/Fs were measured in eggs from Agbobloshie, Ghana (300 pg TEQ/g fat), an e-waste and end-of-life vehicles scrapyard followed by eggs from the vicinity of waste incinerators in Wuhan (27 pg TEQ/g fat). Fernandes et al. (2023) investigated the uptake of PBDD/Fs, PCDD/Fs and other chemicals from recycled poultry bedding material into chicken muscle tissue, liver and eggs in the UK. The average TEQ contribution of the brominated contaminants was 33% and 44% in eggs and tissues respectively (> 96% thereof was from PBDD/Fs). In the Netherlands, eggs and poultry meat (“broilers”) was found to be heavily contaminated with 2,3,7,8-TeBDF and contaminated poultry feed was identified as a probable source (Dirks et al., 2024). The dioxin-like activity in two egg samples reached 21 and 18 pg BEQ/g fat, exceeding the instrumentally measured sum of PCDD/Fs, dl-PCBs, and PBDD/Fs (19.6 and 11 pg WHO-TEQ/g fat), suggesting the

presence of additional dioxin-like compounds such as PBCDD/Fs. This is supported by the high BFR concentrations observed in both samples (Petrlik et al., 2025). In a study investigating the occurrence of halogenated aromatic compounds in five common fish species in South China, PBDD/Fs showed dominance in highly brominated PBDFs, and their pollution was closely associated with PBDEs. PBDD/Fs contributed significantly higher to total TEQ and intake risks (45.6% on average) than the other halogenated aromatic compounds (Zhang et al., 2024).

107. PBDD/Fs as well as PBDEs, TBBPA and other BFRs were found in children's toys and other consumer products made from recycled plastics sampled from 26 countries around the world. Solvent extraction was used to analyse the toys and showed levels of PBDD/Fs ranging from 110 to 17,000 pg BEQ/g (DR CALUX) and from 56 to 13,900 pg TEQ/g (GC-HRMS). In the short term, only a fraction of the PBDD/Fs will be bioavailable as BFRs migrate slowly out of the plastic (Sun et al., 2019; Sun & Zeng, 2022). Ingestion of contaminated plastics by mouthing of such products by a 12-month-old child (9 kg body weight, WHO standard) can contribute to the total daily intake for PCDD/Fs (Behnisch et al., 2023; Budin et al., 2020).

108. In the serum of firefighters, levels of PBDD/Fs were found to be one order of magnitude higher than those of PCDD/Fs (Shaw et al., 2013). Firefighters are exposed to flame retardants, PBDD/Fs and PCDD/Fs. PBDD/Fs were found in relatively higher concentrations in air and on surfaces of personal protection equipment than PCDD/Fs and thus may be of greater concern with regard to occupational exposures (Fent et al., 2020). An International Agency for Research on Cancer (IARC) monograph addresses exposure of firefighters to PBDD/Fs (IARC, 2023).

109. PBDF and PBDD congeners were present in blood specimens of chemical workers during extrusion blending of resins containing decaBDE. Concentrations in blood lipid ranged from non-detectable to 112 pg/g and from non-detectable to 478 pg/g for 2,3,7,8-TeBDF and 2,3,7,8-TeBDD, respectively and correlated well with assignments in the extruder work area and provided biological half-life estimates of between 1.1 and 1.9 years for 2,3,7,8-TeBDF and between 2.9 and 10.8 years for 2,3,7,8-TeBDD. The overall clinical assessment of the study was that the immune system of exposed employees was not adversely impacted at these burdens of PBDFs and PBDDs (Zober et al., 1992). However, the sample size was small, with only 42 participants, and only effects on the immune system were investigated in the study.

110. According to Peecher et al. (2022), blood serum of Vietnamese women from the e-waste sector was analyzed for nine PBDD/F congeners. Significant differences between non-recycler and recycler (recruited from nearby commune) concentrations in lipids were seen. PBDD concentrations of 1,2,3,7,8-PeBDD in non-recyclers showed a median of 0 (and a geometric mean less than LOD) and in recyclers a median of 0 (but with geometric mean at 5.7). 2,3,7,8-TeBDF concentrations were 0 in non-recyclers (with a geometric mean at less than LOD) and 0 in recyclers (but with geometric mean at 7.5). Regarding 1,2,3,7,8-PeCDF, non-recyclers had lower concentrations at 1.3 pg/g compared to recyclers at 2.6 pg/g lipid. In non-recyclers, for PBDDs a geometric mean <LOD was measured, the total PBDD TEQ accounted for 19.5 pg/g and 48.8% of the total dioxin and the total PBDF TEQ for 11.2 pg/g and 28.1%. Thus, PBDD/Fs would be responsible for 76.9% and PCDD/Fs for 23.1% of a combined TEQ consisting of brominated and chlorinated 2,3,7,8-substituted dibenzo-*p*-dioxins/dibenzofurans. In recyclers, total PBDD TEQ was 14.3 pg/g and 43.8% and PBDF TEQ was 8.5 pg/g and 26.0% of the total dioxin load, making PBDDs/PBDFs responsible for 69.8% of their total TEQ (Peecher et al., 2022).

111. The results of the analyses of PBDD/Fs in indoor, outdoor air and indoor dust of gymnasiums in Chinese Taiwan showed that indoor-dust, indoor $PM_{2.5}$, and outdoor $PM_{2.5}$ levels of PBDD/Fs were 37.8 ± 13.7 pg WHO_{2005}-TEQ g^{-1} and 0.00650 ± 0.00340 and 0.00469 ± 0.00101 pg WHO_{2005}-TEQ m^{-3}, respectively (Su et al., 2022). A study by Tue et al. (2013) analyzed house dust from New York State for dioxin-like activities and dioxin-related compounds (DRCs), including PBDD/Fs. PBDD/F concentrations varied widely (0.33–150 ng/g, median 2.1 ng/g), with contributions to BEQ ranging from <1% to 21%. The sum of the concentrations of thirteen 2,3,7,8-substituted PBDD/F congeners that were analyzed in ambient air samples from the northwest suburbs of Beijing was in the range from 1171 to 2424 fg/m^3 (42.2 to 109.3 fg TEQ m^3) (Zhang et al., 2015).

112. The occurrence in human tissues and in breast milk demonstrates bioavailability and exposure to sensitive life stages. According to several studies monitoring the occurrence of PBDD/Fs in human milk, their contribution to total TEQ levels varies. In Sweden, PBDD/Fs in samples from 10 volunteers from Örebro contributed an average of 40% to the TEQ (Bjurlid, 2018c). In an older Swedish study, the levels of PCDD/Fs decreased over time in breast milk samples from 1997 to 2011. In the same study, PBDD/Fs were found above the detection limit in only one sample from 2010, contributing with 1.6% to the total TEQ (Haglund et al., 2014). In Germany, the median (95th percentile) concentrations of PBDD/F congeners in samples from 100 volunteers in Bavaria were at 0.62 (2.69) pg WHO_{2005}-TEQ/g lw contributing approximately 11% (Fromme et al., 2022). Shanghai samples contained three PBDF congeners (no PBDDs) – 2,3,7,8-TeBDF, 1,2,3,4,6,7,8-HpBDF, and OBDF – with mean concentrations of 3.2 pg/g (detected in 72.5% of the samples), 9.5 pg/g (100%) and 28 pg/g (67.5%), respectively (Lin et al., 2021). In a global WHO monitoring study of 38 pooled samples from 28 countries, PBDD/F concentrations contributed about 10% to overall TEQ on average (Schächtele et al., 2023).

113. Additional human monitoring studies are focusing on PBDD/Fs in human adipose tissue. Because of the complexity of plasma samples, the limit of detection for PBDD/Fs in plasma is relatively high (20 pg/g lw), which has

contributed to the scarcity of available data on PBDD/Fs in human plasma (Yang et al., 2021a). Hagberg et al. (2011) reported that ten human adipose tissue samples were analyzed for PBDD/Fs. In all samples, 2,3,7,8-TeBDF was present in levels ranging from 0.23 to 0.54 pg/g lw and PeBDFs were present in nine out of ten samples. The levels of 2,3,7,8-TeBDF, 1,2,3,7,8- and 2,3,4,7,8-PeBDF were similar to those reported in the previous screening study (Ericson Jogsten et al., 2010). Similarly, Thatcher et al. (2015) examined PCDDs and PBDDs in the visceral and subcutaneous adipose tissue and liver biopsies, reporting that PBDD/F TEQ concentrations were relatively low, contributing less than 5% of adipose tissue TEQ and under 10% of liver TEQ, with 2,3,7,8-TeBDF and 2,3,4,7,8-PeBDF being the most frequently detected compounds. Another Swedish study found 2,3,7,8-TeBDF in eight out of nine human adipose tissue at concentrations between 0.27 and 2.24 pg/g lw (Ericson et al., 2008).

114. In the EU, there is an ongoing interlaboratory study on the determination of brominated contaminants in powdered infant milk that also includes PBDD/Fs and PBCDD/Fs (Annex E, BAT/BEP Expert, 2024).

2.4 Hazard assessment for endpoints of concern

2.4.1 Adverse effects of dioxins, dioxin-like substances and their mechanisms of action

115. Adverse effects of dioxins and dioxin-like substances have been well-established in experimental animal studies, wildlife and humans (White & Birnbaum, 2009). In experimental animal studies, a wide range of adverse effects have been demonstrated for PCDD/Fs, that include lethality, wasting, teratogenesis, reproductive effects, immunotoxicity, such as thymic atrophy, endocrine disruption (decreases in T4 and vitamin A), hepatotoxicity and carcinogenicity. In wildlife, reproductive failure, developmental malformations, immune suppression, and population-level effects in fish, birds, and mammals from contaminated ecosystems, with patterns consistent with those observed in laboratory animals have been observed. In humans, the European Food Safety Authority concluded that there is a likely causal relationship between early-life exposure to dioxins and dioxin-like PCBs and impaired semen quality (EFSA Panel on Contaminants in the Food Chain (CONTAM) et al., 2018). Furthermore, there were suggestive associations with endocrine, neurodevelopmental, immunological, metabolic and cancer outcomes, but that could not clearly be concluded to be causal.

116. In experimental animal studies, essentially all adverse effects observed following exposure to PCDD/Fs have also been observed for PBDD/Fs (Birnbaum et al., 2003; WHO, 1998). The WHO states that “Generally, the studies performed with 2,3,7,8-substituted PBDDs/PBDFs showed typical TeCDD-like effects in experimental animals”.

117. Dioxin-like toxicity occurs when dioxins and all dioxin-like substances, via the same mechanism of action, bind to the aryl hydrocarbon receptor (AhR) (Van den Berg et al., 1998). This binding activates the AhR, causing it to translocate to the nucleus, heterodimerize with aryl hydrocarbon receptor nuclear translocator (ARNT), and then regulate the expression of genes involved in adaptive and toxic responses. This AhR-mediated dysregulation of gene expression is the fundamental mechanism by which dioxins cause adverse effects like chloracne, cancer promotion, and developmental toxicity (Piwarski & Salisbury, 2023). PBDD/Fs are also ligands to the AhR (van den Berg et al., 2013) and can thus themselves provoke similar effects as PCDD/Fs and in combination with PCDD/s contribute to the overall exposure and effects of dioxin-like substances (Birnbaum et al., 2003).

118. As PCDD/Fs are already listed in Annex C to the Convention, read-across from the structurally related chlorinated analogue, TeCDD, to inform on adverse effects of PBDD/Fs is used. The replacement of chlorine with bromine, as in the case of PBDD/Fs, is expected to produce a similar level of dioxin-like toxicity. Therefore, the use of the relative effect potency (REP) and toxic equivalency factor (TEF) schemes[7] for PBDD/Fs should be considered applicable (van den Berg et al., 2013).

2.4.2 Dioxin-like hazard characteristics of PBDD/Fs

119. The most toxic congeners are those substituted at positions 2, 3, 7 and 8. There are seven 2,3,7,8-substituted PBDDs and ten 2,3,7,8-substituted PBDFs, as well as 337 possible 2,3,7,8-substituted PBCDDs and 647 possible 2,3,7,8-substituted PBCDFs (WHO, 1998).

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[7] According to the WHO-UNEP TEF scheme, the toxicity of a mixture of dioxins and/or dioxin-like compounds can be expressed in a single number – the toxic equivalency (TEQ). It is a single figure resulting from the product of the concentration and individual toxic equivalency factors (TEFs) of each congener. TEFs express the toxicity of PCDD/Fs and dioxin-like PCBs in terms of the most toxic PCDD/F congener, 2,3,7,8-TeCDD.Since available data for fish toxicity were considered to be too limited for inclusion in the WHO-UNEP TEF scheme, van den Berg et al. (2013) decided not to derive specific WHO-UNEP TEFs for fish. However, for ecotoxicological risk assessment, they recommend the use of specific REPs from fish embryo assays. Based on the limited mammalian REP database for PBDD/Fs, Van den Berg et al. (2013) concluded that sufficient differentiation from the present TEF values of the chlorinated analogues was not possible. However, the REPs for PBDD/Fs in mammals closely follow those of the chlorinated analogues, at least within one order of magnitude. Therefore, van den Berg et al. (2013) recommended the use of similar interim TEF values for brominated and chlorinated congeners for human risk assessment, pending more detailed information in the future.

120. Like PCDD/Fs, PBDD/Fs and PBCDD/Fs have the ability to activate the AhR signal (Behnisch et al., 2003).Binding to the AhR by PBDDs and PBCDDs varies with the number and position of the halogen atoms. While 2,3,7,8-TeBDD has a slightly lower binding affinity than 2,3,7,8-TeCDD, 2,8-di-B-3,7-di-CDD has twice the binding affinity of 2,3,7,8-TeCDD (Birnbaum et al., 2003).

2.4.3 Experimental studies on dioxin-like toxicity of PBDD/Fs

121. Adult zebrafish being exposed to 2,3,7,8-TeBDD for 9-weeks via feed (nominal concentrations: 3, 30 and 300 ng/g food dw) showed a significant dose and time dependent reduced spawning success and significantly altered ovarian morphology (Norman Haldén et al., 2011). The high dose group (300 ng/g food dw) stopped spawning after one week of exposure and the medium group (30 ng/g food dw) after five weeks of exposure. The reduction of vitellogenic oocytes was significantly larger in the medium group than in the low dose group. Fish from the high dose showed high mortality (23%) and were therefore sampled three weeks before the other groups. Also, body weight was lower than the pre-exposure weight in the high and medium group.

122. For tetra- to hexahalogenated PBCDDs, REPs ranging from 0.19 to 0.96 have been calculated (Eljarrat & Barceló, 2004). A higher toxicity of a PBCDF congener, 3,7,8-tri-B-2-CDF, compared to 2,3,7,8-TeCDD was observed *in vivo* in early-life stages of Japanese medaka (*Oryzias latipes*) using aqueous exposure with a REP_{water} of 3.3 and a REP_{egg} of 4.6 (Nakayama et al., 2022). Furthermore, PBCDD/Fs may photochemically be transformed to PCDD/Fs (WHO, 1998) which are listed as POPs in Annex C to the Convention.

123. To understand the impact of 1,3,7-TrBDD on bird health, the effects of 1,3,7-tribromodibenzo-*p*-dioxin (1,3,7-TrBDD) on chicken embryos was investigated comparing these effects with those induced by 2,3,7,8-TeCDD. The study identified numerous differentially expressed genes (DEGs) in the 1,3,7-TrBDD-treated groups, including cytochrome P450 1A4 and 1A5, suggesting activation of the AhR. Furthermore, the transcriptome results suggest that 1,3,7-TrBDD has a unique impact on insulin signaling, peroxisome pathways, and some metabolism related pathways in chicken embryos that are activated via a different mode from TCDD, although the effects on the cancer- and metabolism-related pathways were similar for both compounds. According to the authors, both 1,3,7-TrBDD and 2,3,7,8-TeCDD can induce similar transcriptomic changes related to cancer and metabolism, however, 1,3,7-TrBDD exhibits less severe phenotypic effects compared to the more fatal effects of TCDD (Park et al., 2022).

124. 2,3,7,8-TeBDD is a similarly powerful inducer of ethoxyresorufin *O*-deethylase (EROD), a widely used biomarker for exposure for substances that can bind to the AhR, in rat liver as 2,3,7,8-TeCDD after a single subcutaneous injection (Nagao et al., 1990). The induction of EROD was confirmed in rats and zebrafish being exposed to 2,3,7,8-TeBDD (Mason et al., 1987; Mennear & Lee, 1994; Norman Haldén et al., 2011; Weber & Greim, 1997). AhR regulated genes were also induced in rats and zebrafish being exposed to 2,3,7,8-TeBDD with similar affinities like PCDDs (Mennear and Lee, 1994; Norman Haldén et al., 2011). Hepatic EROD activity and hepatic expression of AhR-regulated genes indicate if brominated dioxins can activate gene transcription through the AhR pathway (Norman Haldén et al., 2011). Mason et al. (1987) demonstrated that a series of PBDDs and PBCDDs produced body weight loss and thymic atrophy in immature male Wistar rats. However, no doses were specified for 1,3,7,8-TeBDD; 2,3,7,8-TeBDD; 1,2,3,7,8-PeBDD; 1,2,4,7,8-PeBDD; 2,3-di-B-7,8-di-CDD; and 2-B-3,7,8-tri-CDD (Mason et al., 1987).

125. A comparative acute oral toxicity study of 2,3,7,8-TeCDF and TeBDF (Moore et al. 1979, cited in (Mennear & Lee, 1994; WHO, 1998) on male Hartley guinea pigs exposed to single doses of 0, 0.47, 1.58, 4.74, or 15.81 µg/kg of TeBDF or 0, 1, 5, 10, or 15 µg/kg of TeCDF showed similar effects such as decreased body weight gain after the lower doses (1.0 µg/kg for TeCDF and 4.74 µg/kg for TeBDF) and prompt body weight loss after the higher doses (10 µg/kg for TCDF and 15.8 µg/kg for TeBDF). When compared on an equimolar basis, the LD_{50} levels in the study are similar between the chlorinated and brominated homologues.

126. The evidence shows that PBDD/Fs and PBCDD/Fs can induce developmental neurotoxicity (2,3,7,8-TeBDF) and neurobehavioral changes like decreased exploratory behaviour in a novel environment in adulthood and ultrasonic vocalization (USV) during infancy (2,3,7,8-TeBDF) in mouse offspring perinatally exposed (Kimura et al., 2020, 2023). Liver response, such as an increase in liver size and disrupted gene expression has been reported in mice after TeBDF or TeCDD exposure (Kimura et al., 2022). 2,3,7,8-TeBDD caused typical 2,3,7,8-TeCDD-like effects, including wasting syndrome, thymus atrophy, and liver toxicity. Additionally, liver damage described as peliosis hepatis, which has not been reported after exposure of rats to 2,3,7,8-TeCDD, was observed. The pattern of lesions (lethality, histopathology, liver and thymus weights) found in guinea pigs after a single exposure and in rats after short-term exposure to 2,3,7,8-TeBDF was similar to that of 2,3,7,8- TeCDF (WHO, 1998).

127. Van den Berg et al. (2013) compiled relative potency factor (REP) ranges for brominated compounds from a range of *in vitro* and *in vivo* studies and compared them with their respective chlorinated compounds. The available evidence shows that many REPs for PBDD/Fs in mammals are similar to, or at least within one order of magnitude of those for their chlorinated analogues. Similarly, based on embryo injection, the REPs for causing larval mortality in different strains of rainbow trout are mostly within one order of magnitude for PBDD/Fs and their corresponding PCDD/F congeners (van den Berg et al., 2013). Budin et al. (2021) determined to what extent TEF values were

comparable to human-based REPs and if the default assignment of TEFs to PBDD/Fs, based on their chlorinated counterparts, is a valid approach. According to their results, and pending the generation of a larger database of REPs from additional studies, the TEF values assigned to PBDD/Fs seem to be appropriate for human hazard assessment of PBBD/Fs mixtures (Budin et al., 2021).

128. Mason et al. (1987) compared the relative abilities of TeCDD and TeBDD to bind to rat liver cytosolic receptors (hydroxylapatite receptor binding assay) and to induce AHH (aryl hydrocarbon hydroxylase) and EROD *in vitro*. The affinities of the compounds for receptor binding sites appeared to be equivalent between TeCDD and TeBDD, but the chlorine derivative appeared to be somewhat more potent as an inducer of AHH and EROD. Ao et al. (2009) investigated the immunotoxic effects on mice of the 2,3,7,8-TeBDD and 1,2,3,7,8-PeBDD, in comparison with those of 2,3,7,8-TeCDD and 1,2,3,7,8-PeCDD to gain insight into the potency of brominated dioxins for immunotoxicity. The mice were immunized with ovalbumin (OVA) and dosed with 1, 3 or 10 µg/kg. TeCDD or TeBDD showed similar effects on thymus weight and cell number, spleen weight and cell number, and OVA induced-IL-5 production by splenocytes. PeCDD also showed effects similar to those of TeCDD. On the other hand, PeBDD showed somewhat dose-independent effects. A study by Frawley et al. (2014) showed suppression of the immunoglobulin M (IgM) antibody response and Ttr gene expression, and upregulated phase I XME genes of female B6C3F1/N mice by 2,3,7,8-TeCDD, 2,3,7,8-TeBDF, 2,3,7,8-TeCDF, 1,2,3,7,8-PeBDF, 2,3,4,7,8-PeBDF, 2,3,4,7,8-PeCDF and 2,3-di-B-7,8-di-CDD. The study found that although 2,3,7,8-TeCDD was the most potent of the compounds tested, the brominated analogues were generally more potent than their chlorinated analogues.

129. In the study by Hardy et al. (1990), TeBDF was administered orally to Sprague Dawley rats (5 days/week for 4 weeks) at 0, 1, 10, 50, 150, or 500 µg/kg/day. All animals of the high dose (500 µg/kg/day) died; 70% of the animals receiving 150 µg/kg/day died, and those dosed with 50 µg/kg/day or more exhibited decreased body weight gain. Mean thymus weight (relative to body weight) was decreased in males by 150 µg/kg/day and in females by 10 and 50 µg/kg/day. Liver, kidney, spleen, adrenal gland, heart, and thymus from animals in the control, and 1, 10, and 50 µg/kg/day groups were examined. Treatment related alterations were noted in the liver and thymus at 50 µg/kg bw/day and to a lesser extent at 10 µg/kg bw/day with hepatic lesions characterized as panlobular hypertrophy of the hepatocytes with associated panlobular hepatocyte vacuolation and focal necrosis. Thymic atrophy consisting of overall depletion of the lymphoid elements was present in all 50 µg/kg male and female rats from which thymus was available and in most animals from the 10 µg/kg dose group. No treatment-related alterations were observed in the 1 µg/kg group. Ivens et al. (1993) conducted a 13-week repeated toxicity study including accumulation and elimination of TeBDD in male and female Wistar rats with oral doses of 0.01, 0.1, 1, 3, and 10 µg/kg. The authors stated that animals dosed with either 0.01 or 0.1 µg/kg exhibited no treatment-related signs of toxicity; however, doses of 0.1 µg/kg or higher produced decreases in serum thyroxin in both sexes. An increase in triiodothyronine (T3) was seen in animals with doses of 1 or 3 µg/kg while tetraiodothyronine (T4) was decreased. The major effects noted in the 3 and 10 µg/kg groups included decreased body weight gain, poor general health, icterus, and after 12 weeks, increased plasma alkaline phosphate, alanine aminotransferase, and aspartate aminotransferase in the 3 µg/kg group animals that survived to the scheduled termination of the study. Activities of microsomal enzymes (ethoxyresorufin *O*-deethylase, ethoxycoumarin *O*-deethylase, aryl hydrocarbon hydroxylase, UDP-glucuronyltransferase) investigated in liver, lung and kidney were dose-dependently elevated after 13 weeks of treatment. At a dose of 3.0 µg/kg, activities were below those of the dose 1.0 µg/kg. According to the authors, this is probably due to liver toxicity.

130. Effects on fertility were found in Wistar rats after daily oral administration of TeBDD at 3 or 10 µg/kg bw/day for 13 weeks. The effects included dose dependent decreases in spermatogenic activity in the testes and defective or necrotic spermatocytes in the epididymis (Ivens-Kohl et al. 1989 as cited in WHO (1998)).

131. In addition to the developmental effects of 2,3,7,8-TeBDF; 1,2,3,7,8-PeBDF; and 2,3,4,7,8-PeBDF were studied on pregnant C_{57}BL/6N mice on gestation day 10, with doses ranging from 0 to 192 µg/kg for 2,3,7,8-TeBDD and from 0 to 4,000 µg/kg for 2,3,7,8-TeBDF, 1,2,3,7,8-PeBDF, and 2,3,4,7,8-PeBDF (Birnbaum et al., 1991; Mennear & Lee, 1994). All doses of each compound significantly increased liver weights in the dams but produced no other evidence of maternal toxicity. TeBDD and TeBDF caused dose-related increases in foetal weights, and 500 µg/kg or more of TeBDF significantly increased embryo/foetal mortality. The most sensitive measure of developmental toxicity was the induction of hydronephrosis. The incidence was increased by each compound, and ED$_{50}$ doses were estimated to be 9 µg/kg for TeBDD, 12 µg/kg for TeBDF, 340 µg/kg for 1,2,3,7,8-PeBDF, and 4,350 µg/kg for 2,3,4,7,8-PeBDF. The authors estimated the ED$_{50}$ dose for TeCDD to produce hydronephrosis was 4 µg/kg. Thus TeBDD, as the most potent inducer of hydronephrosis among the brominated congeners tested, displays a similar potency as the concomitantly tested TeCDD. All of the PBDDs and PBDFs caused significant increases in the incidence of cleft palate, which the authors suggested may indicate these compounds produce their effects through the same mechanism. For TeCDD cleft palate (CP) occurred at doses ≥1 µg/kg/day and hydronephrosis (HN) at doses ≥0.5 µg/kg bw/day. For the other congeners the incidence of HN was also significantly increased above background levels at the following doses (µg/kg bw/day): 2,3,7,8-TeBDD, 3; 2,3,7,8-TeBDF, 25; 1,2,3,7,8-PeBDF, 500; 2,3,4,7,8-PeBDF, 400. The LOELs (µg/kg bw/day) for CP were: 2,3,7,8-TeBDD, 48; 2,3,7,8-TeBDF, 200; 1,2,3,7,8-PeBDF, 4,000; 2,3,4,7,8-PeBDF, 2400. The results of the study by Ding et al. (2007) suggest that PBDD/Fs are able to modulate steroidogenic gene expression, which may lead to endocrine disruption. According to WHO (1998), 2,3,7,8-

TeBDF administered orally to Sprague-Dawley rats for 4 weeks caused dose-dependent growth retardation and histopathological changes in liver and thymus. The NOAEL for these effects was established at 1 µg/kg bw/day.

132. Endocrine effects were observed in studies with rats, where thyroid hormones were affected after single and subchronic exposures to 2,3,7,8-TeBDDD. Four weeks after single oral doses of 10, 33, or 100 µg/kg bw, T3 was increased and T4 was reduced dose-dependently in the serum of female and male Wistar rats (*n* = 5) (Ivens et al., 1992 as cited in WHO (1998)). Wistar rats (10/male and female rats per group) treated with daily doses of 0.01–3.0 µg/kg bw/day for 3 months showed reduced T4 and increased T3 levels at ≥ 0.1 µg/kg bw/day (Löser & Ivens, 1989; Ivens et al., 1993 as cited in WHO (1998)).

133. The carcinogenicity of PCDD/Fs in animals is well established. 2,3,7,8-TeCDD is known to be a human carcinogen within the U.S. National Toxicology Program (NTP). The authors of Mennear & Lee (1994) consider PBDD/Fs as human carcinogens as well based on structural similarity and read-across since the carcinogenic potential of the PCDD/Fs in animals is established (Mennear & Lee, 1994; IARC Working Group on the Evaluation of Carcinogenic Risks to Humans, 1997). Experimental studies in mice have shown carcinogenic potential of PBDD/Fs, after a permanent cell line was established from peritoneal macrophages of mice treated with TeBDD. Animals being exposed to these cells after subcutaneous injection developed tumours at the injection sites after three weeks (Massa et al., 1992; WHO, 1998).

134. According to the authors, there is sufficient evidence to conclude that concentrations of 2,3,7,8-substituted PBDDs and PBDFs in human food, tissues and milk can contribute significantly to the total dioxin-like toxicity. In mammals, the mode of action, type of toxicity and potency of 2,3,7,8-substituted PBDDs and PBDFs are similar to their chlorinated analogues (van den Berg et al., 2013).

135. Reported PBDD/F dietary intakes suggest that some population groups, in particular young children, may exceed the tolerable weekly intake for dioxin-like contaminants (2 pg TEQ/kg bw/week as set by the EFSA Panel on Contaminants in the Food Chain (CONTAM) et al., 2018), even for mean consumption estimated with lower bound data (Fernandes & Falandysz, 2021). PBCDD/Fs may significantly contribute to the overall dioxin-like exposure (Fernandes et al., 2014; van den Berg et al., 2013). This is in line with another study, which concluded that PBDD/Fs, and probably also PBCDD/Fs, may significantly contribute to dietary exposure to dioxin-like compounds (Mortimer et al. 2013).

136. [There is sufficient evidence to conclude that,] due to their toxicity, which is similar to that of PCDD/Fs for most congeners (Birnbaum et al., 2003; Bjurlid, 2018c; Mennear & Lee, 1994; van den Berg et al., 2013), it can be concluded that tri- to octahalogenated PBDDs and PBCDDs as well as tetra- to heptahalogenated PBDFs and PBCDFs fulfil the criteria for adverse effects on [human health and/or] the environment.

3. Synthesis of information

137. There are several anthropogenic sources of PBDD/Fs and emissions to the environment including (1) commercial PBDE mixtures (presence of PBDD/Fs as by-product), (2) photolytic and thermal degradation of BFRs, and (3) unintentional production in industrial thermal processes. PBDD/Fs and PBCDD/Fs can also be formed when mechanochemical non-combustion technologies are used to treat contaminated soil. PBDD/Fs can be formed when plastics containing direct precursors such as PBDEs are subject to recycling processes or as a result of photolytic and thermal degradation of BFRs and BFR-containing products. PBDD/Fs and PBCDD/Fs can form unintentionally during heating or burning, particularly when BFRs or other brominated/chlorinated materials are involved. The yield of PBDD/Fs and PBCDD/Fs varies. PBDD/Fs and PBCDD/Fs can also form during the production of BFRs, such as BTBPE, and through degradation of PBDEs and TBBPA (another BFR) under various conditions. The presence of chlorine and bromine influences the ratio of PBDD/Fs to PBCDD/Fs formed. Burning of fuel appears to be a further source, although the significance is unclear.

138. Research indicates that some PBDD/Fs and PBCDD/Fs originate from natural sources like algae or cyanobacteria, from condensation of natural bromophenols or other reactions. The corresponding congeners are usually less brominated, ranging from mono- to pentabrominated, with tribrominated congeners appearing to be the most abundant.

139. No evidence was received indicating any intentional uses of PBDD/Fs or PBCDD/Fs with the exception of small quantities for analytics and research purposes. The uses of PBDEs, which are precursors of PBDD/Fs or, when chlorine is present, of PBCDD/Fs, are already restricted to a large extent as tetra-, penta-, hexa-, hepta- and decaBDE are listed in Annex A (Elimination) to the Convention. Other BFRs that can decompose to precursors of PBDD/Fs or, when chlorine is present, of PBCDD/Fs are used in a variety of applications including printed circuit boards, electrical enclosures, automotive applications, building materials, and textiles for public buildings and specialized sectors like military and medical, aerospace adhesives, coatings, and structural parts of aircraft and rail vehicles.

140. Releases of PBDD/Fs and PBCDD/Fs can occur when materials are heated or burnt, including in accidental or other types of uncontrolled fires, especially when BFRs or other brominated (and chlorinated) aromatics are present or

in relation with the production and use of BFRs. PBCDD/Fs can also be found in landfill leachate. Appropriate risk management measures such as BAT/BEP developed to limit the formation of PCDD/Fs can also limit the formation of PBDD/Fs and PBCDD/Fs. This is supported by low levels of PBDDs and PBCDDs detected where controls are in place, including municipal incinerators. Thermal disposal of e-waste can release PBDD/Fs at levels significantly higher than PCDD/Fs. Estimates based on production and use of PBDEs in 2001 alone suggest that global annual emissions of PBDFs could have reached 2,300 kg. Global PCDD emission is estimated to be 17,000 kg annually (Mathew et al., 2025). PBDD/Fs have been found in plastics from e-waste, like CRT casings, with stockpiled e-waste in places like Nigeria containing large amounts of these pollutants. Uncontrolled incineration of plastics with BFRs, including in fires in private residencies, can also produce harmful levels of PBDD/Fs. Recycling e-waste and materials like automobile shredder residue may lead to cross-contamination and environmental release of PBDD/Fs. Metallurgical facilities, such as steel plants and other non-ferrous smelting plants, have also been identified as significant sources of PBDD/F emissions. Other sources include landfills and industrial sites, especially in urbanized areas.

141. PBDD/Fs and PBCDD/Fs are persistent (tetra- to octahalogenated congeners). Using a read-across approach from empirical data for suitable analogues PCDD/Fs, and supported by the QSARs and subsequent extrapolation, the half-lives in water and soil are estimated to be greater than the relevant thresholds of two and/or six months, respectively, for tetra- to octahalogenated PBDD/Fs and PBCDD/Fs. Based on BIOWIN2, -3 and -6 QSARs, congeners of PBDD/Fs and PBDD/Fs are predicted to exhibit similarly low levels of biodegradation as congeners of PCDD/Fs.

142. PBDD/Fs and PBCDD/Fs (di- to octahalogenated congeners) have log $K_{\mathrm{OW}} \geq 5$ and, according to Annex D criteria, are considered likely to bioaccumulate. Elimination half-lives of PBDD/Fs in mammals are significantly longer than those of PCDDs. The estimated half-lives for humans range from 1 to 11 years. The long terminal elimination of half-lives of PBDD/Fs in mammals suggests that these substances are highly bioaccumulative. PBCDD/Fs have physicochemical properties (such as molecular size, halogenation patterns, and metabolic stability) in the range between PCDD/Fs and PBDD/Fs. Since both PCDD/Fs (as recognized POPs) and PBDD/Fs are bioaccumulative, it is very likely that this is also the case for PBCDD/Fs. PBDD/Fs have been detected in high-trophic level species such as seals and whales, and human tissues in Japan and Sweden. The occurrence in human tissues and in breast milk demonstrates bioavailability and adds to the concern of exposure to sensitive life stages. Overall, these compounds show longer elimination half-lives compared to PCDDs, indicating their higher resistance to metabolism and a higher potential for bioaccumulation.

143. PBDD/Fs and PBCDD/Fs have the potential for long-range environmental transport via atmosphere, oceans and migratory species. Their estimated gas-phase half-lives range from 1.8 to 504 days. Due to their physicochemical properties, they are expected to occur predominantly particle-bound in the atmosphere. The atmospheric half-lives will therefore be higher than those calculated for the gas phase and exceed the two days threshold for long-range environmental transport in air. PBDD/Fs detected in air from background areas confirm the atmospheric long-range transport as a contributing pathway. A recent modelling study concludes that each of the investigated PBDD/F congeners (tetra- to heptabrominated) exceeds the accumulation levels of other POPs in the remote region when emitted to air or water. PBDD/Fs have been detected in biota (including *Globicephala melas,* long-finned pilot whale) and ornithogenic soils in remote regions. PBDD/Fs are also subject to long-range environmental transport in marine (micro)plastics. Moreover, PBDD/Fs could be formed from PBDEs or other BFRs once they reach remote areas, e.g. due to photolytic degradation. The occurrence of PBDD/Fs in pilot whales (*Globicephala melas*) caught around the Faroe Islands, provides evidence that PBDD/Fs also undergo long-range environmental transport via migratory species.

144. PBDD/Fs and PBCDD/Fs have been detected in various environmental compartments, including sediments, air and biota. Elevated concentrations are often linked to industrial activities such as related to the life cycle (manufacturing, use, disposal, recycling) of products that contain BFRs including e-waste processing and incineration. Studies show regional variation, with high levels in several countries in places like e-waste and automotive scrap yards, e-waste recycling areas, areas adjacent to some metal smelters that partially used e-waste as feed stocks and open burning areas. PBDD/Fs have also been found in soil, wastewater, sludge, fly ash, various animals including predators such as seals or whales and plant species, food and feed, indoor dust and in humans.

145. Total PBDD/F exposure of the general population includes proximal sources of PBDD/Fs and sources of PBDD/Fs that are due to LRET. Proximate sources of PBDD/Fs include exposure to air after accidental incineration of home materials containing BFRs, occupational exposures like firefighters or e-waste sector workers, and through the consumption of locally sourced food products like eggs, milk, some fish and meat. Higher levels of PBDD/Fs were detected in food sourced from industrial areas, such as e-waste and incinerator sites. Ingestion of children's toys and other consumer products made from recycled plastics containing BFRs, can contribute to the total daily intake for PBDD/Fs, especially through mouthing habits. Various TEQ levels of exposures to PBDD/Fs can occur through

human milk depending on region. LRET includes exposures to PBDD/Fs in air that traveled from distant industrial sites and through the consumption of migratory fish. Inclusion of PBDD/Fs in the WHO-UNEP TEF concept can improve human risk assessment for dioxins and furans by including compounds that work through a common mode of action. Using the TEF concept, PBDD/Fs, and likely PCBDD/Fs, may make a significant contribution to dietary exposure to dioxin-like compounds. Dietary exposure estimates for the UK range from 0.19 to 1.64 pg TEQ per kg body weight per day. The contribution to the total TEQ from the PBDD/Fs, PBCDD/Fs and PBBs was consistently around 30% for all age groups and for average and high consumers. Finally, biomonitoring studies found firefighters and workers in the e-waste sector show higher PBDD/F levels in their blood compared to PCDD/Fs. Monitoring of human adipose tissue indicates low PBDD/F TEQ concentrations.

146. PBDD/Fs and PBCDD/Fs cause adverse effects similar to that of PCDD/Fs with similar REPs for causing adverse effects in mammals and fish. In mammals, the mode of action, type of toxicity and potency of 2,3,7,8-substituted PBDDs and PBDFs are similar to their chlorinated analogues including lethality, wasting, teratogenesis, reproductive effects, immunotoxicity such as thymic atrophy, endocrine disruption (decreases in T4 and vitamin A), and increased hepatic porphyrins (Birnbaum et al., 2003; WHO, 1998). There is sufficient evidence to conclude that concentrations of 2,3,7,8-substituted PBDDs and PBDFs in human food, tissues and milk can contribute significantly to the total dioxin-like toxicity. Reported PBDD/F dietary intakes suggest that some vulnerable population groups, in particular young children, may exceed the tolerable weekly intake for dioxin-like contaminants (2 pg TEQ/kg bw/week). In its preamble, the Stockholm Convention acknowledges that the Arctic ecosystems and indigenous communities are particularly at risk because of the biomagnification of persistent organic pollutants and that contamination of their traditional foods is a public health issue. Scientific evidence demonstrates that traditional consumption of whale blubber such as in the Faroe Islands and other communities in remote regions can already surpass safe limits of consumption as a result of the PBDD/F content in whale blubber from remote regions and could cause adverse effects as a result of long-range environmental transport and exposure to PBDD/Fs in a remote region.

4. Concluding statement

147. In conclusion, PBDD/Fs and PBCDD/Fs are estimated to be released in significant quantities from anthropogenic sources. Tetra- to octahalogenated PBDD/Fs and PBCDD/Fs are considered to fulfil the criteria for persistence, bioaccumulation, long-range environmental transport and adverse effects on [human health and/or] the environment under the Convention. They are persistent in the environment for a long time, increasing the probability of exposure to the environment and humans. They bioaccumulate in biota, including in humans with very long half-lives. Modelling studies have shown that they have the potential to be transported over long distances. PBDD/Fs have been detected in some environmental compartments including in air, ornithogenic soils and marine mammals in remote regions. PBDD/Fs and PBCDD/Fs can significantly contribute to the overall dioxin-like exposure, and they may cause adverse effects also in remote regions.

148. Therefore, it is concluded that tetra- to octahalogenated PBDD/Fs and PBCDD/Fs are likely, [as a result of their long-range environmental transport], to lead to significant adverse [human health and/or] environmental effects such that global action is warranted [, while acknowledging a lack of full scientific certainty in certain aspects of the risk profile].

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[8] While the titles of referenced sources cannot be altered, any mention of "Taiwan" or "Hong Kong" within this document is to be understood as referring to "Chinese Taiwan" and "Hong Kong (China)," respectively, in accordance with United Nations General Assembly resolution 2758 (XXVI) of 25 October 1971.

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