STUDY TO SUPPORT THE REVIEW OF WASTE RELATED ISSUES IN ANNEXES IV AND V OF REGULATION (EC) 850/2004

Final report

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Date: January 2019

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Study to support the review of waste related issues in Annexes IV and V of Regulation (EC) 850/2004

Final report

Written by:

RAMBØLL

January 2019
ABSTRACT

To reduce POP emissions into the environment Annexes IV and V of Regulation (EC) No 850/2004 on persistent organic pollutants (POPs) establish the obligation to generally destroy or irreversibly transform the POP content of waste above a certain threshold limit (‘LPCL’). In exceptional cases waste containing POPs above the limits may be otherwise managed with specified operations for specified waste types if destruction or irreversible transformation do not represent the environmentally preferable option and the concentration of the POPs is below a maximum threshold value (‘MPCL’). In this study information from competent authorities, scientific institutions, literature and other experts was collected, analysed and summarised. Further, mass flows for newly listed and candidate POPs were established and, finally, recommendations for the environmentally sound management of waste containing POPs were elaborated. Suggestions for limit values (LPCL and MPCL) were derived using the methodology developed by BiPRO GmbH in 2005, which assesses the economic, environmental, health, waste management, legal and recycling implications. Additionally, the identification of action needed to fully implement the listing of the new POPs and the potential listing of the candidate POPs in the annexes to Regulation (EC) No 850/2004 and other relevant EU legislation was identified and an overview demonstrating the need for action within relevant legislation was illustrated.

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<td>(P)NEC</td>
<td>(Predicted) no effect concentration</td>
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<td>ABS</td>
<td>Acrylonitrile-Butadiene-Styrene</td>
</tr>
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<td>ACEA</td>
<td>European Automobile Manufacturers Association</td>
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<tr>
<td>AFFF</td>
<td>Aqueous Film Forming Foam</td>
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<td>APC</td>
<td>Air pollution control</td>
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<td>APFO</td>
<td>Ammonium Perfluorooctanoate</td>
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<td>ASR</td>
<td>Automotive shredder residues</td>
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<tr>
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<td>Best Available Techniques</td>
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<td>BEP</td>
<td>Best Environmental Practices</td>
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<td>Brominated flame retardants</td>
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<td>Br</td>
<td>Bromine</td>
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<td>Bromine Recovery Unit</td>
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<td>BTBPE</td>
<td>1,2-Bis(2,4,6-tribromophenoxy)ethane</td>
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<td>BTFs</td>
<td>Bio transfer factors</td>
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<td>C&amp;D</td>
<td>Construction &amp; Demolition</td>
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<td>Chemical Abstracts Service</td>
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<td>CEWEP</td>
<td>Confederation of European Waste-to-Energy Plants</td>
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<td>CLRTAP</td>
<td>Convention on Long Range Transboundary Air Pollution</td>
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<td>COP</td>
<td>Conference of the Parties</td>
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<td>DBDPE</td>
<td>Decabromodiphenylethane</td>
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<td>DCBP</td>
<td>Dichlorobenzophenone</td>
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<td>DecaBDE</td>
<td>Decabromodiphenyl ether</td>
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<td>DEFRA</td>
<td>Department for Environment, Food and Rural Affairs</td>
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<td>DIP</td>
<td>Direct injection probe</td>
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<tr>
<td>dl-PCB</td>
<td>Dioxin-like polychlorinated biphenyl</td>
</tr>
<tr>
<td>dm</td>
<td>Dry matter</td>
</tr>
<tr>
<td>dw</td>
<td>Dry weight</td>
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<td>ECB</td>
<td>European Central Bank</td>
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<td>Electron capture detection</td>
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<td>ECNI</td>
<td>Electron capture negative ion</td>
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<td>EEA</td>
<td>European Environmental Agency</td>
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<td>EEE</td>
<td>Electrical and electronic equipment</td>
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<td>European Electronics Recyclers Association</td>
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<td>EINECS</td>
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<td>ELV</td>
<td>End of Life Vehicles</td>
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<td>EPA</td>
<td>Environmental Protection Agency</td>
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<td>E-PRTR</td>
<td>European Pollutant Release and Transfer Register</td>
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<td>EPS</td>
<td>Expanded polystyrene</td>
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<td>EQS</td>
<td>Environmental quality standards</td>
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<td>ESIS</td>
<td>European chemical Substances Information System</td>
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<td>ESM</td>
<td>Environmental Sound Management</td>
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<td>ETICS</td>
<td>External Thermal Insulation Composite Systems</td>
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<td>EU</td>
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<td>EuRIC</td>
<td>European Recycling Industries' Confederation</td>
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<td>FEDEREC</td>
<td>French trade association for recycling</td>
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<tr>
<td>FGT</td>
<td>Flue-gas treatment</td>
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<td>FR</td>
<td>Flame retardant</td>
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<td>GC-MS</td>
<td>Gas chromatograph-mass spectrometry</td>
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<td>Abbreviation</td>
<td>Description</td>
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<tr>
<td>HBCDD</td>
<td>Hexabromocyclododecane</td>
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<td>HEC</td>
<td>Halogen electrolytic conductivity detector</td>
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<td>HIPS</td>
<td>High-impact polystyrene</td>
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<td>HPLC</td>
<td>High-performance liquid chromatography</td>
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<tr>
<td>HPV</td>
<td>High Production Volume Chemical</td>
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<td>HR</td>
<td>High resolution</td>
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<td>IA-MS</td>
<td>Ion attachment-mass spectrometry</td>
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<td>IEC</td>
<td>International Electrotechnical Commission</td>
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<td>IED</td>
<td>Industrial Emissions Directive</td>
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<td>IPEN</td>
<td>International POPs Elimination Network</td>
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<td>ISO</td>
<td>International Standards Organization</td>
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<tr>
<td>LC</td>
<td>Liquid chromatography</td>
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<td>LCCP</td>
<td>Long chain chlorinated paraffin</td>
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<td>LHAs</td>
<td>Large household appliances</td>
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<td>LOD</td>
<td>Limit of detection</td>
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<td>LOQ</td>
<td>Limit of quantification</td>
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<td>LOW</td>
<td>Liste of waste</td>
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<td>LPCL</td>
<td>Low POP concentration limits</td>
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<td>LPVC</td>
<td>Low Production Volume Chemical</td>
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<td>LRMS</td>
<td>Low resolution mass spectrometry</td>
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<td>MAC-EQS</td>
<td>Maximum allowable concentration environmental quality standards</td>
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<td>MDL</td>
<td>Method detection limit</td>
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<td>Ministry of the Environment</td>
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<td>MPCL</td>
<td>Maximum POP concentration limits</td>
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<td>MS</td>
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<td>Term</td>
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<td>MS-ESI</td>
<td>Mass spectrometry with electrospray ionization</td>
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<td>MSWI</td>
<td>Municipal Solid Waste Incineration</td>
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<td>Nordic COM</td>
<td>Nordic Council of Ministers</td>
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<tr>
<td>OECD</td>
<td>Organisation for Economic Co-operation and Development</td>
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<td>PBDD</td>
<td>Polybrominated dibenzo-dioxins</td>
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<td>PBDEs</td>
<td>Polybrominated diphenyl ethers</td>
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<td>PBDF</td>
<td>Polybrominated dibenzofurans</td>
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<tr>
<td>PBT</td>
<td>Persistent, Bioaccumulative &amp; Toxic</td>
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<td>PID</td>
<td>Photoionization detector</td>
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<td>POP</td>
<td>Persistent Organic Pollutant</td>
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<td>PP</td>
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<td>Post Shredder Technologies/Treatment</td>
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<td>Risk management evaluation</td>
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</tr>
<tr>
<td>SHF</td>
<td>Shredder heavy fraction</td>
</tr>
<tr>
<td>SLF</td>
<td>Shredder light fraction</td>
</tr>
<tr>
<td>SVHC</td>
<td>substances of very high concern</td>
</tr>
<tr>
<td>TDI</td>
<td>Tolerable daily intake</td>
</tr>
<tr>
<td>TEF</td>
<td>Toxic equivalency factor</td>
</tr>
<tr>
<td>TEQ</td>
<td>Total dioxin toxic equivalence</td>
</tr>
<tr>
<td>TOF-MS</td>
<td>time of flight Mass Spectrometry</td>
</tr>
<tr>
<td>TÜV</td>
<td>Technical Inspection Association (Technischer Überwachungsverein)</td>
</tr>
<tr>
<td>TWI</td>
<td>Tolerable weekly intake</td>
</tr>
<tr>
<td>UBA</td>
<td>German Environmental Agency (Umweltbundesamt)</td>
</tr>
<tr>
<td>UHPLC</td>
<td>High-performance liquid chromatography</td>
</tr>
<tr>
<td>UNECE</td>
<td>United Nations Economic Commission for Europe</td>
</tr>
<tr>
<td>UNECE</td>
<td>United Nations Economic Commission for Europe</td>
</tr>
<tr>
<td>UPE</td>
<td>Unsaturated polyester</td>
</tr>
<tr>
<td>UTC</td>
<td>Unintentional Trace Contamination</td>
</tr>
<tr>
<td>VECAP</td>
<td>Voluntary Emissions Control Action Programme</td>
</tr>
<tr>
<td>vPvB</td>
<td>very Persistent &amp; very Bioaccumulative</td>
</tr>
<tr>
<td>WEEE</td>
<td>Waste Electrical &amp; Electronic Equipment</td>
</tr>
<tr>
<td>Acronym</td>
<td>Description</td>
</tr>
<tr>
<td>---------</td>
<td>-----------------------------------</td>
</tr>
<tr>
<td>WFD</td>
<td>Waste Framework Directive</td>
</tr>
<tr>
<td>WWTP</td>
<td>Wastewater treatment plant</td>
</tr>
<tr>
<td>XPS</td>
<td>Extruded polystyrene foam</td>
</tr>
<tr>
<td>XRF</td>
<td>X-ray fluorescence</td>
</tr>
</tbody>
</table>
EXECUTIVE SUMMARY

The information and views set out in this study are those of the author(s) and do not necessarily reflect the official opinion of the Commission. The Commission does not guarantee the accuracy of the data included in this study. Neither the Commission nor any person acting on the Commission’s behalf may be held responsible for the use which may be made of the information contained therein.

Persistent Organic Pollutants (hereinafter "POPs") are a group of organic compounds that possess toxic properties, persist in the environment, bioaccumulate through the food web and pose a risk to human health and the environment. These chemicals are transported across international boundaries far from their sources through air, water and migratory species. The European Union is party to two international legally binding instruments related to POPs, the "Protocol on Persistent Organic Pollutants of the Convention on Long-Range Transboundary Air Pollution" of UNECE and the global "Stockholm Convention on persistent organic pollutants". Both aim to reduce and eliminate the production, use and release of POPs in the territories of participating parties. Waste management is considered an essential measure to control releases of POPs into the environment throughout their life-cycle, and comprehensive regulation of contamination of waste by POPs has the potential to reduce the presence of POPs in the environment.

In 2017, three substances were added to the Annexes to the Stockholm Convention by the Conference of the Parties and proposals for three candidate substances had been submitted. As any decision to amend the Stockholm Convention must be transposed and implemented by parties within one year after parties were notified of that decision (unless a party is a so-called opt-in party), a study contract has been awarded to Ramboll to support the Commission in the analysis of waste-related issues of specific POPs. Relevant for this study are the following nine substances and substances groups:

- Newly listed POPs: Decabromodiphenylether (decaBDE) and short-chain chlorinated paraffins (SCCPs) in Annex A and Hexachlorobutadiene (HCBD) in Annex C to the Stockholm Convention;
- Candidate POPs: Dicofol, Pentadecafluorooctanoic acid (PFOA, perfluorooctanoic acid) and its salts and PFOA-related compounds, Perfluorohexanoic acid (PFHxS) and its salts and PFHxS-related compounds;
- Already listed POPs: Hexabromocyclododecane (HBCDD), Polychlorinated Biphenyls (PCB), Polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/PCDF).

The main objective of the present study is to provide to the Commission necessary scientific information in order to amend the POP Regulation in view of the listing of new substances. This includes (1) Assessing the actions necessary for the implementation of the listing of the new (and potentially of the candidate) POPs in the annexes to the POP Regulation and other relevant EU legislation; (2) Providing guidance on sound management of waste containing the new and candidate POPs; (3) Compilation and evaluation of existing data on the new POPs and the candidate POPs, as well as the proposal of concentration limits and the assessment of their impact; (4) Reviewing of the existing limit values for certain already listed substances on the basis of new scientific data and studies and, if necessary, the proposal of new limit values and assessment of their impact; and (5) Dissemination of information through preparation and performance of a stakeholder workshop and the setting up of a project website.

Legal Analysis

Actions needed to fully implement the listing of the new POPs and the candidate POPs in the Annexes of the POP Regulation and other relevant EU legislation were identified.
The overall conclusion of the project team is that the REACH Regulation (EC) No 1907/2006 is directly affected by the results of the last Conference of the Parties of the Stockholm\(^1\) and Basel\(^2\) Conventions. Further, possibly affected EU legislation includes the RoHS Directive 2011/65/EU, the WEEE Directive 2012/19/EU, the End of Life Vehicles Directive 2000/53/EC, the List of Waste Commission Decision 2014/955/EU and the Sewage Sludge Directive 86/278/EEC. More information on possibly affected EU regulations can be found in Table 1.

With regards to the REACH Regulation, required actions are to remove the restriction regarding decaBDE\(^3\) from Annex XVII of REACH and to list the POP in the relevant Annex in the POP Regulation. The implementation of the decision adopted under the Stockholm Convention requires the listing of the chemical in the POP Regulation, which will replace the listing in Annex XVII of REACH.

**Compilation and Assessment of Information**

To establish an information basis for the assessment, information from competent authorities, scientific institutions, literature and other information sources was compiled on:

- life cycle and lifetime of products and articles in which the substances were used;
- concentration levels of the substances in products and articles at the end of their life cycle;
- types of articles recycled, extent of recycling and types of articles produced from recycled materials;
- occurrence and levels of newly listed POPs and candidate POPs in different waste streams (and categories), in recycled articles and in articles currently in use as well as occurrence and levels in different articles that could be recycled in the near future.

Current and past waste management operations and existing recycling pathways were investigated. For obtaining information from Member States and other relevant stakeholders a non-formal consultation was carried out via a questionnaire. The summaries of the answers to the questionnaire were published at a dedicated project website.\(^4\) Using the information obtained, mass flows for new and candidate POPs were established where reasonably possible.

In October 2018, an expert workshop was organised in Brussels where the preliminary results of the study were presented and discussed. New and relevant information provided by stakeholders during or after the workshop was further assessed and considered in the final report.

**Derivation of limit values**

Limit values in accordance with Annex IV of the POP Regulation as well as for specific waste treatment operations, ensure pollutant removal to the largest extent possible on the one hand and environmentally sound recycling processes on the other hand. A distinction is made between “low POP concentration limits” (LPCLs) in accordance with Art. 7(4)a of the POP Regulation and “maximum POP concentration limits” (MPCLs) of substances listed in Annex IV of the POP Regulation.

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\(^1\) The following Decisions of the COP of the Stockholm Convention have been analysed: UNEP/POPS/COP.8/SC-8/4; UNEP/POPS/COP.8/SC-8/7; UNEP/POPS/COP.8/SC-8/10; UNEP/POPS/COP.8/SC-8/11; UNEP/POPS/COP.8/SC-8/12

\(^2\) The following Decisions of the COP of the Basel Convention have been analysed: UNEP/CHW/COP.13/BC-13/4; UNEP/CHW.13/6/Add.1/Rev.1;

\(^3\) PFOA is also listed under Annex XVII of REACH. As such, removal of its listing may also become necessary upon its listing as a POP under the Stockholm Convention and subsequently the POP Regulation.

\(^4\) See http://pops-and-waste.bipro.de/
 Relevant waste streams and product categories, as well as typical waste amounts arising from relevant applications during a reference year, were used to derive scenarios for the substances linked to possible LPCLs. Based on the data collected and evaluated in the detailed mass flows, it was possible to draw conclusions on expected effects of different limit values on specific and overall mass flow of the POP substances in question. Thus, it was possible to derive conclusions on economic impacts such as cost changes in waste treatment costs and possible restraints in disposal/destruction capacity which constitute limiting criteria in the assessment of feasible limit values.

Generally, the stricter the LPCL, the more wastes may be classified as POP waste and consequently will require destruction or irreversible transformation such as incineration, physicochemical treatment or disposal at hazardous waste landfills in hard rock formation or underground.

Concentration limits for LPCLs and MPCLs were derived using a methodology developed by BiPRO in 2005. The methodology uses a number of lower and upper limitation criteria to derive a range for a possible concentration limit. This approach enables on the one hand that the limit value can be implemented realistically in the light of the available data, and on the other hand, that human health and the environment are protected from POPs to a large extent. The results of the assessment of the limitation criteria and the recommendation for LPCLs and MPCLs are shown in the following table (please note explanations in the subsequent text):

<table>
<thead>
<tr>
<th></th>
<th>decaBDE</th>
<th>SCCPs</th>
<th>HCBD</th>
<th>Dicofol</th>
<th>PFOA</th>
<th>PFHxS</th>
<th>HBCDD</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Result lower limitation criteria in mg/kg (the LPCL should be at or above the result)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(A) Analytical potential</td>
<td>200</td>
<td>0.1</td>
<td>0.001</td>
<td>0.03</td>
<td>1</td>
<td>1</td>
<td>100</td>
</tr>
<tr>
<td>(B) Background contamination</td>
<td>≥20</td>
<td>1.0</td>
<td>0.1</td>
<td>3</td>
<td>0.01</td>
<td>0.01</td>
<td>2</td>
</tr>
<tr>
<td>(DR) Disposal and recovery capacities</td>
<td>0.3</td>
<td>1</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>10</td>
</tr>
<tr>
<td>(E) Economic feasibility</td>
<td>200</td>
<td>420</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>10</td>
</tr>
<tr>
<td><strong>Result upper limitation criteria in mg/kg (the LPCL should be at or below the result)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(LV) Limit values</td>
<td>1,000</td>
<td>10,000</td>
<td>100</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
<tr>
<td>(R) Risks</td>
<td>n.a.</td>
<td>18,000</td>
<td>200</td>
<td>200</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
<tr>
<td><strong>Currently established concentration limits (in mg/kg)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Current LPCL:</td>
<td>-</td>
<td>10,000</td>
<td>100</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
<tr>
<td>Current MPCL:</td>
<td>-</td>
<td>10,000</td>
<td>1,000</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
<tr>
<td><strong>Recommended concentration limits (in mg/kg)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LPCL:</td>
<td>200 to 1,000¹</td>
<td>420 to 10,000</td>
<td>100</td>
<td>50</td>
<td>50/2,000²</td>
<td>50/2,000³</td>
<td>100 to 1,000</td>
</tr>
<tr>
<td>MPCL:</td>
<td>10,000⁴</td>
<td>10,000</td>
<td>1,000</td>
<td>5,000</td>
<td>50/2,000²</td>
<td>50/2,000³</td>
<td>1,000</td>
</tr>
</tbody>
</table>

n.a.: Not applicable; ¹ for decaBDE or the sum of POP-BDEs; ² 50 mg/kg for PFOA and 2,000 mg/kg for PFOA-related substances; ³ 50 mg/kg for PFHxS and 2,000 mg/kg for PFHxS-related substances; ⁴-integrated in the MPCL for the sum of concentrations of tetraBDE, pentaBDE, hexaBDE, heptaBDE and decaBDE.

**Further explanations and recommendations for the environmentally sound management (ESM).**

**DecaBDE**

The current POP Regulation does not contain a limit value for decaBDE. Based on the assessments in the present study a range of 200 mg/kg to 1,000 mg/kg for decaBDE or the sum of POP-BDEs is
proposed. The relevant standards (IEC 62321-3-1:2013 and IEC 62321-6:2015) should be validated for concentration levels in line with the LPCL which will be selected. However, implementation can already be based on existing analytical methods.

Several considerations are relevant when considering a value within the proposed range including the precautionary principle, economic, technical and practical feasibility and efficiency.

Based on the consideration that decaBDE contributes via debromination to the toxicity of the sum of all POP-BDEs, we suggest that decaBDE should be integrated in the MPCL at a level of 10,000 mg/kg for the sum of the concentrations of tetraBDE, pentaBDE, hexaBDE, heptaBDE, and decaBDE.

Regarding ESM for WEEE plastics, there is a risk that contaminated recyclates enter the product, cycle. The risk can be minimised as far as possible through a higher yield of separation of WEEE plastics containing decaBDE. Considering that a complete stop of the recycling of WEEE plastic is not a desirable option, a higher yield of separation of decaBDE from the waste stream should be achieved by an increased share of identification and separation of decaBDE-containing WEEE prior to shredding processes based on knowledge (information on relevant WEEE and components) and efforts (e.g. invest more time in the screening process) and/or based on improved identification and separation technology (e.g. invest in more and better separation or pre-treatment technology). A share of around 6 % of plastics from WEEE (low bromine fraction) is landfilled. Landfilling of plastics from WEEE treatment should be reduced since remaining decaBDE is not destroyed. Used EEE is frequently exported from the EU. Export of plastics from WEEE treatment should be reduced since adequate ESM is not necessarily ensured in all importing countries.

Regarding ESM of End of Life Vehicles, a relevant share of decaBDE within the shredder light fraction is not destroyed and could, therefore, enter recyclates. Risk can be minimised through a higher yield of separation of decaBDE rich fractions by means of an increased share of identification and separation of decaBDE-containing WEEE prior to shredding processes based on knowledge (information on relevant WEEE and components) and efforts (e.g. invest more time in the screening process) and/or based on improved identification and separation technology (e.g. invest in more and better separation or pre-treatment technology). High-density fractions should be treated appropriately to destroy or irreversibly transform its decaBDE content. Landfilling should be reduced. Similar to the considerations for used WEEE, export of used cars should be reduced.

Particular attention should be given to the ESM of decaBDE-containing plastics from construction and demolition waste. The occurrence of decaBDE in this waste stream becomes relevant and will increase over the next 20 years. Options to identify and separate plastics containing decaBDE and other POP-BDE have to be explored to make progress in the ESM of construction and demolition waste.

**SCCP**

The current LPCL for SCCPs is 10,000 mg/kg. Based on the assessments in the present study a range of 420 mg/kg to 10,000 mg/kg for SCCP is proposed. Theoretically lowering the LPCL to values higher 1,500 mg/kg would have no impact on the historically relevant uses and related waste streams, however, higher SCCPs concentration (several thousand mg/kg) have been reported during enforcement activities in EU MS. Therefore, a possible limit value below 10,000 mg/kg could still have an impact on the produced and/or recycled products contaminated with or containing SCCPs entering the EU market. The potential impact cannot be quantified based on the currently available information but is likely linked to comparably low changes in costs for waste management compared to the current situation and is not expected to result in bottlenecks regarding disposal and recovery capacities in the EU.

Lowering the LPCL below 1,500 mg/kg (e.g. 1,000 mg/kg or 500 mg/kg) would potentially restrict the placing on the market and use of articles containing SCCPs in concentrations lower than 0.15 % by weight and probably also the alternative MCCPs as this is the amount of SCCPs that may be present as an impurity in an article produced with MCCPs (according to the EU POP Regulation).
limit value below 1,500 mg/kg would theoretically still have no impact on the management of waste streams resulting from historic applications (e.g. rubber from conveyor belts and sealants and adhesives) but would have an impact on certain produced and/or recycled products contaminated with or containing SCCPs entering the EU market. Further, selecting a limit value below 1,500 mg/kg but above 420 mg/kg, which is the estimated theoretical average SCCPs concentration in rubber waste in the EU, would imply a comparably low change in costs for waste management. A limit value above 420 mg/kg would also not result in any disposal and recovery capacity bottlenecks in the EU.

However, a POP concentration limit below 420 mg/kg (e.g. 100 mg/kg) would result in an additional cost increase which is considered not economically feasible. All other implications described in previous section also apply here (e.g. related to the placing on the market and use of articles containing SCCPs in concentrations lower than 0.15 %). The waste disposal and recovery capacities are still estimated to be sufficient even at potential limit values below 420 mg/kg but are estimated not sufficient at values below 1 mg/kg.

For SCCPs, considering that no indication was found that the current MPCL is inadequate, it is proposed to maintain the current value of 10,000 mg/kg, as established within the Regulation.

Recommendations for environmentally sound management to minimise the risk include the separation of SCCP-containing rubber conveyor belts from underground mining and sealants and adhesives in C&D waste. Separation as far as possible should be aimed for considering the limitation of available separation techniques specific to SCCP. For rubber conveyor belts, metal fraction of the conveyor belt should be separated from the rubber fraction as far as possible. Possible arising dust generation should be avoided as far as possible and inhalation of contaminated dust can be avoided by the application of breathing masks. Other waste streams (e.g. resulting from consumer products) might be difficult to control as they mostly stem from imported products. It is expected that most of the relevant waste is managed together with municipal solid waste (which is largely incinerated and landfilled in the EU). However, it cannot be generally excluded that certain waste streams are also recycled (possibly leading to accumulation of SCCPs in new recycled products). A measure to control the concentration of import products and related waste is only feasible if reliable analytical methods/standards are used to distinguish between MCCPs and SCCPs.

**HCB**

HCB is listed in the EU POP Regulation with an LPCL of 100 mg/kg. Based on the assessments in the present report, a concentration range of 0.1 to 100 mg/kg for the LPCL is possible. There is no indication that there is the need to adjust the established concentration limit. It is proposed to maintain the LPCL for HCB at 100 mg/kg.

HCB is listed in the EU POP Regulation with an MPCL of 1,000 mg/kg. There is no indication that there is a need to adjust the established concentration limit and that there would be beneficial impact on environment or health from lowering this concentration limit. It is proposed to maintain the MPCL for HCB at 1,000 mg/kg.

Waste streams are not considered relevant in the EU. Generally, recommendations on ESM e.g. from general and specific Basel Convention guidelines on POPs apply.

**Dicofol**

For dicofol, no limit values are currently established under the EU POP Regulation. Based on the evaluation of the lower and upper limitation criteria, a potential LPCL for dicofol could be set in a range between 3 and 200 mg/kg. It is proposed to establish the LPCL for dicofol at 50 mg/kg.
A possible LPCL of 50 mg/kg, which is well within the proposed range, would align with the LPCLs currently established for other POP pesticides.

An MPCL, as in the case of the LPCL, of 5,000 mg/kg, in alignment with other POP pesticides is proposed to be established for dicofol.

Waste streams are not considered relevant in the EU. Generally, recommendations on ESM e.g. from general and specific Basel Convention guidelines on POPs apply.

**PFOA**

The current POP Regulation does not yet contain a limit value for PFOA, its salts and related compounds. Due to a limitation of available quantitative information no upper limitation criteria could be established. In the absence of upper limiting criteria, the same LPCL as established for PFOS is recommended, i.e. 50 mg/kg for PFOA and 2,000 mg/kg for PFOA-related substances. A higher LPCL for PFOA-related substances is recommended because PFOA-related compounds only degrade to a certain and unknown extent to PFOA and because it is a concentration limit for the sum of PFOA-related compounds.

Also, for the MPCL the same concentration limit as established for PFOS is recommended, i.e. an MPCL of 50 mg/kg for PFOA and 2,000 mg/kg for PFOA-related substances a MPCL.

Based on the currently available information and data, it was not possible to investigate waste streams in detail as it has been done for other substances in this report. Most of the relevant waste streams are/will be typically incinerated. Controlled incineration with high temperatures (e.g., at 1,000°C) is effective to destroy PFOA and to prevent the formation of PFOA from the thermolysis of highly fluorinated polymers. For PFOS, pre-treatment is mentioned as an important aspect of environmental sound disposal and methods should be selected based on the nature and type of the PFOS waste. This can also apply to PFOA waste.

The need for additional information is strongly supported by the project team, especially information and data on production and use of PFOA in different application areas is still very limited or in certain cases entirely missing, but essential for a well justified derivation of POP waste limit values. It can be expected that EU member states will continue with or will initiate projects related to PFOA in articles, wastes and recyclates which will help to improve the current situation regarding the data availability for the EU.

**PFHxS**

The current POP Regulation does not yet contain a limit value for PFHxS, its salts and related compounds. Due to the limited data availability, it was not possible to derive any of the upper limiting criteria for PFHxS, its salts and PFHxS-related compounds. In the absence of upper limiting criteria, the same LPCL as established for PFOS is recommended, i.e. 50 mg/kg for PFHxS and 2,000 mg/kg for PFHxS-related substances. A higher LPCL for PFHxS-related substances is recommended because the compounds only degrade to a certain and unknown extent to PFHxS and because it is a concentration limit for the sum of PFHxS-related compounds.

Information on production and use of PFHxS, its salts and related compounds in the EU is very scarce and mostly qualitative. Based on the available information it is considered that most identified waste streams will in the future typically be incinerated. For other relevant waste streams, it can be assumed that these will be collected and managed by specialised companies (e.g. for metal plating).

Information, especially on production and use is still very limited or entirely missing but is essential for a well justified derivation of POP waste limit values. It can be expected that EU member states will continue with or will initiate projects related to PFHxS in articles, wastes and recyclates which will help to improve the current situation regarding the data availability.
**HBCDD**

HBCDD is listed in the EU POP Regulation with an LPCL of 1,000 mg/kg. Based on the results of the present study a range of 100 to 1,000 mg/kg HBCDD is proposed as LPCL. To decide where within this possible range the LPCL should be established, several feasibility and efficiency considerations should be taken into account. The relevant standard (IEC 62321-3-1:2013) should be validated for concentration levels in line with the LPCL which will be selected. However, implementation can already be based on existing analytical methods.

The MPCL in Annex V of the EU POP regulation is established at 1,000 mg/kg. There is no indication that the current MPCL is not appropriate. It is proposed to maintain the established concentration limit 1,000 mg/kg for HBCDD.

EPS and XPS in construction and EPS plastic are considered as the most relevant waste streams. Regarding their ESM, the risk of EPS/XPS being recycled can be minimised by separating EPS/XPS insulation materials to the largest extent possible in construction and destruction and by treating them in an appropriate way.

An innovative process for the recycling of EPS containing HBCDD is CreaSolv®. In the process HBCDD is destroyed to a large extent while PS and bromine can be recycled and used for further applications.

About one third of EPS packaging waste is recycled. Nowadays HBCDD is not used in EPS packaging anymore. However, in practice, EPS containing HBCDD insulation materials are unintentionally mixed and recycled with packaging waste. Thus, waste which is not contaminated or contains only minor amounts of HBCDD (EPS packaging waste), is mixed with HBCDD-containing waste (EPS from C&D) and in consequence, contaminated recycled PS pellets are produced.

This practice entails a further transfer of the HBCDD content into various plastic products. To minimise the risk, better separation of EPS waste from C&D and packaging EPS waste should be strived for. To this end specific measures should be implemented to (1) raise the awareness about the necessity to separate EPS from construction at waste producers, recyclers and waste collection facilities and to (2) increase the use of screening methods to identify HBCDD-containing EPS at collection facilities and recycling facilities.

Almost 30% of EPS/XPS panels containing HBCDD is landfilled. Related risks can be minimised through minimising their landfill.

**PCBs and PCDD/F**

The LPCL for non-dioxin like-PCBs is established at 50 mg/kg. The LPCL for PCDD/Fs is established at 15 µg TEQ/kg.

The review of the LPCL for PCBs showed, that there is a need to consider this group of substances in bilateral way, meaning that non-dioxin-like (ndl-) and dioxin-like (dl-) PCBs should be considered independently from each other. The structure of the dl-PCBs is similar to dioxin and related to the same toxic mode of action. Thus, the following recommendations can be made.

For ndl-PCBs, there is no indication, that the LPCL (50 mg/kg) should be adjusted. DL-PCBs however, need to be addressed separately from ndl-PCBs.

For this there are two options, either, dl-PCBs can be integrated in the TEQ based LPCL of PCDD/F or an individual TEQ based LPCL for dl-PCBs can be established which might lay in the range of 10-times higher than the LPCL of PCDD/F, as the highest TEF of all dl-PCBs is 0.1 for PCB 126. However, recent information indicates that the toxicity of this dl-PCB congener may be overestimated.
For **PCDD/PCDFs** there are indications, based on scientific input, that the current LPCL of 15 µg TEQ/kg is too high to prevent potential adverse effect on environment.

Fly ashes from municipal solid waste incineration are considered an important mass flow in this respect. Available data indicate that 11% and 3% of such fly ashes would exceed an LPCL of 5 µg TEQ/kg and 10 µg TEQ/kg respectively. This indicates that if the LPCL for PCDD/Fs would be lowered correspondingly, the majority of the fly ash can still be used according to current treatment. Stakeholders, however, claim when the LPCL of PCDD/Fs is lowered, the safe recycling (e.g. as filler in asphalt) could be hampered and that a proper data collection of PCDD/F content in fly ashes should be conducted before suggesting a new LCPL value, with a proper assessment of the impact on environment and human health when current practices are followed.

For **PCDD/Fs**, based on the results of this study a lower value in the range for example of 5-10 µg/kg TEQ should be considered. In the case, that dl-PCBs are included in the LPCL of PCDD/Fs it might be reasonable to consider a value at the upper limit of this range.
Les informations et points de vue exposés dans cette étude sont ceux des auteurs et ne reflètent pas nécessairement l’opinion officielle de la Commission. La Commission ne garantit pas l’exactitude des données incluses dans cette étude. Ni la Commission ni aucune personne agissant au nom de la Commission ne peut être tenue pour responsable de l’utilisation qui pourrait être faite des informations dans l’étude.

Les polluants organiques persistants (ci-après "POP") sont un groupe de composés organiques qui possèdent des propriétés toxiques, persistent dans l'environnement, s'accumulent dans les organismes vivants par l’intermédiaire du réseau trophique et constituent un risque pour la santé humaine et pour l'environnement. Ces substances chimiques sont transportées loin de leurs sources, au-delà des frontières nationales, par l’air, l’eau et les espèces migratrices. L’Union européenne a ratifié deux traités internationaux relatifs aux POP, le "«Protocole sur les polluants organiques persistants de la Convention sur la pollution atmosphérique transfrontière de longue portée» et la «Convention de Stockholm sur les Polluants Organiques Persistants». Les deux visent à réduire et à éliminer la production, l’utilisation et le rejet de POP sur les territoires des parties signataires. La gestion des déchets est considérée comme une mesure essentielle pour contrôler le rejet de POP dans l’environnement pendant leur cycle de vie, et une réglementation exhaustive de la contamination des déchets par les POP pourrait réduire leur présence dans l’environnement.

En 2017, trois substances ont été ajoutées aux annexes de la Convention de Stockholm par la Conférence des Parties et des propositions ont été soumises pour trois substances candidates. Puisque tout décision d’amendement de la Convention de Stockholm doit être transposé et mis en œuvre par les parties dans un délai d’un an compter de la notification de cette décision aux parties (à moins qu’une partie ne soit une partie dite «opt-in party »), Ramboll a été chargé d’étudier pour la Commission certaines questions liées à des déchets de POP spécifiques. Les neufs substances et groupes de substances suivants ont été étudiés :

- POP nouvellement listés : Décabromodiphényléther (décaBDE) et les chloroalcanes à chaine courte (SCCP) à l’annexe A et Hexachlorobutadiène (HCBD) à l’annexe C de la Convention de Stockholm;
- POP candidats : Dicofol, acide pentadécafluorooctanoïque (PFOA, acide perfluorooctanoïque) et ses sels et composés apparentés, acide perfluorohexanoïque (PFHxS) et ses sels et composés apparentés PFOA;
- POP déjà inscrits sur la liste : hexabromocyclododécane (HBCDD), biphényles polychlorés (PCB), dibenzo-p-dioxines polychlorées et dibenzofurannes (PCDD/PCDF).

L'objectif principal de la présente étude est de fournir à la Commission les informations scientifiques nécessaires à l’amendement du Règlement POP en vue de l’inscription de nouvelles substances. Ceci implique (1) l’évaluation des mesures nécessaires à la mise en œuvre de l’inscription des nouveaux POP (et les POP candidats potentiels) dans les annexes du règlement POP et d’autres législations de l’UE pertinentes; (2) la définition des orientations concernant la bonne gestion des déchets contenant les nouveaux POP et les POP candidats; (3) la compilation et l’évaluation les données existantes sur les nouveaux POP et les POP candidats, ainsi que la proposition des limites en ce qui concerne leur contenu et l’évaluation de leur impact; (4) la révision des valeurs limites existantes pour certaines substances déjà inscrites, basées sur de nouvelles données et études scientifiques et, si nécessaire, la proposition de nouvelles valeurs limites et l’évaluation de leur impact; et (5) la diffusion d'information par la préparation et la réalisation d’un atelier réunissant les parties intéressées et la création d’un site web du projet.

Analyse juridique
Les actions nécessaires pour mettre pleinement en œuvre la liste des nouveaux POP et des POP candidats inscrits aux annexes du Règlement POP et d'autres législations de l'UE pertinentes ont été identifiées.

La conclusion générale de l'équipe du projet est que le règlement REACH (CE) n° 1907/2006 est directement affecté par les résultats de la dernière conférence des parties aux Conventions de Stockholm\(^5\) et de Bâle\(^6\). D'autres législations de l'UE pourraient être affectées notamment la directive RoHS 2011/65/UE, la directive DEEE 2012/19/UE, la directive 2000/53/CE relative aux véhicules hors d'usage, la décision 2014/955/UE de la Commission sur la liste des déchets et la directive 86/278/CEE relative aux boues d'épuration. Vous trouverez de plus amples informations sur les législations européennes susceptibles d'être affectées dans Table 1.

En ce qui concerne le Règlement REACH, les mesures nécessaires consistent à supprimer les restrictions concernant le décaBDE\(^7\) de l'annexe XVII de REACH et à inscrire ce POP dans les annexes correspondantes du règlement POP. La mise en œuvre de la décision adoptée en vertu de la Convention de Stockholm nécessite l’inscription du produit chimique dans le règlement POP, qui remplacera l’inscription à l’annexe XVII de REACH.

**Compilation et évaluation de l'information**

Afin d'établir une base d'information pour l'étude, des informations provenant des autorités compétentes, des institutions scientifiques, de la littérature et d'autres sources d'information ont été regroupées à propos :

- Du cycle de vie et de la durée de vie des produits et articles dans lesquels les substances ont été utilisées;
- Des niveaux de concentration des substances dans les produits et les articles à la fin de leur cycle de vie;
- Des types d'articles recyclés, de la dimension du recyclage et des types d'articles produits à partir de matériaux recyclés;
- De la présence et des niveaux de concentration de POP nouvellement inscrits et des POP candidats dans différents flux de déchets (et catégories), dans les articles recyclés et dans les articles actuellement utilisés, ainsi que la présence et les niveaux dans différents articles susceptibles d'être recyclés dans un avenir proche.

Les gestions des déchets actuelles et passées ainsi que les voies de recyclage existantes ont été étudiées. Afin d’obtenir des informations des États membres et des autres parties intéressées, une consultation informelle a été réalisée au moyen d’un questionnaire. Les résumés des réponses de ce questionnaire ont été publiées sur un site internet consacré au projet\(^8\). Sur la base des informations obtenues, les débits massiques pour les nouveaux POP et les POP candidats ont été établis, dans la mesure du possible.

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\(^6\) Les décisions suivantes de la COP de la Convention de Bâle ont été analysées: UNEP/CHW/COP.13/BC-13/4; UNEP/CHW.13/6/Add.1/Rev.1;

\(^7\) PFOA figure également à l’annexe XVII de REACH. En tant que tel, il pourrait également être nécessaire de retirer son inscription de la liste après son inscription en tant que POP au titre de la Convention de Stockholm, puis du règlement relatif aux POP.

\(^8\) Voir http://pops-and-waste.bipro.de/
En octobre 2018, un atelier d’experts a été organisé à Bruxelles où les résultats préliminaires de l’étude ont été présentés et discutés. Les nouvelles informations et celles pertinentes fournies par les parties prenantes pendant ou après l’atelier ont ensuite été évaluées et prises en compte dans le rapport final.

Dérivation des limites de concentration

Les valeurs limites conformes à l’Annexe IV du Règlement POP ainsi que pour des opérations spécifiques de traitement des déchets assurent, d’une part, l’élimination des polluants autant que possible et, d’autre part, des procédés de recyclage respectueux de l’environnement. Une distinction est faite entre les « low POP concentration limits» (LPCL) conformément à l’art. 7(4)a) du Règlement sur les POP et «maximum POP concentration limits» (MCPL) des substances énumérées à l’annexe IV du Règlement sur les POP.

Les flux de déchets et les catégories de produits, ainsi que les quantités de déchets représentatifs provenant d’applications appropriées au cours d’une année de référence, ont été utilisés pour établir des scénarios pour les substances liées à d’éventuelles LPCL. Sur la base des données collectées et évaluées dans les débits massiques détaillés, il a été possible de déduire les effets attendus des différentes valeurs limites sur le débit massique spécifique et en général des substances POP en question. Ainsi, il a été possible de tirer des conclusions sur les incidences économiques telles que l’évolution des coûts de traitement des déchets et les éventuelles restrictions de la capacité d’élimination/destruction qui constituent des critères limitatifs pour l’évaluation des valeurs limites réalisables.

En règle générale, plus la LPCL est stricte, plus des déchets doivent être classés comme déchets de POP et, par conséquent, nécessiteront une destruction ou une transformation irréversible, tels que l’incinération, le traitement physico-chimique ou le stockage dans des sites de décharges pour déchets dangereux, dans des formations rocheuses souterraines, profondes et sûres.

Des limites de concentration pour les LPCL et les MPCL ont été dérivées à l’aide d’une méthodologie mise au point par BiPRO en 2005. Cette méthodologie utilise un certain nombre de critères de limitation inférieurs et supérieurs pour calculer une rangée de valeurs pour une limite de concentration possible. Cette approche permet, d’une part, que la valeur limite puisse être appliquée de manière réaliste à la lumière des données disponibles et, d’autre part, que la santé humaine et l’environnement soient protégés des POP dans une large mesure. Les résultats de l’évaluation des critères de limitation et de recommandation pour les LPCL et les MPCL sont présentés dans le tableau ci-dessous (veuillez noter les explications données dans le texte suivant):

<table>
<thead>
<tr>
<th></th>
<th>decaBDE</th>
<th>SCCPs</th>
<th>HCBD</th>
<th>Dicofol</th>
<th>PFOA</th>
<th>PFHxS</th>
<th>HBCDD</th>
</tr>
</thead>
<tbody>
<tr>
<td>Résultat des critères de limitation inférieurs en mg/kg (la LPCL doit être égale ou supérieure au résultat)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(A) Analytical potential</td>
<td>200</td>
<td>0.1</td>
<td>0.001</td>
<td>0.03</td>
<td>1</td>
<td>1</td>
<td>100</td>
</tr>
<tr>
<td>(B) Background contamination</td>
<td>≥20</td>
<td>1.0</td>
<td>0.1</td>
<td>3</td>
<td>0.01</td>
<td>0.01</td>
<td>2</td>
</tr>
<tr>
<td>(DR) Disposal and recovery capacities</td>
<td>0.3</td>
<td>1</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>10</td>
</tr>
<tr>
<td>(E) Economic feasibility</td>
<td>200</td>
<td>420</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>10</td>
</tr>
<tr>
<td>Résultat des critères de limitation supérieurs en mg / kg (la LPCL devrait être égale ou inférieure au résultat)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(LV) Limit values</td>
<td>1,000</td>
<td>10,000</td>
<td>100</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
<tr>
<td>(R) Risks</td>
<td>n.a.</td>
<td>18,000</td>
<td>200</td>
<td>200</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
</tbody>
</table>
Explications et recommandations complémentaires concernant la gestion écologiquement rationnelle (GER)

DécaBDE

Le Règlement POP actuel ne contient pas de valeur limite pour le décaBDE. D’après les évaluations de la présente étude, nous proposons une fourchette de 200 mg/kg à 1 000 mg/kg pour le décaBDE ou la somme des POP-BDE. Les normes correspondantes (CEI 62321-3-1:2013 et CEI 62321-6: 2015) devraient être validées pour les niveaux de concentration conformes à la LPCL qui sera sélectionnée. Cependant, la mise en œuvre peut déjà être basée sur les méthodes analytiques existantes.

Plusieurs considérations sont pertinentes lorsque l’on considère une valeur dans la fourchette proposée, y compris le principe de précaution, la faisabilité économique, technique et pratique ainsi que l’efficacité.

Compte tenu du fait que le décaBDE contribue par débromation à la toxicité de la somme de tous les POP-BDE, nous suggérons que le décaBDE soit intégré dans la MPCL à un niveau de 10 000 mg/kg pour la somme des concentrations de tétraBDE, pentaBDE, hexaBDE, heptaBDE et décaBDE.

En ce qui concerne la GER des déchets plastiques d’équipements électriques et électroniques (DEEE), il existe un risque que des produits recyclés contaminés entrent dans le cycle du produit. Le risque peut être minimisé grâce à un taux plus élevé de séparation des plastiques des DEEE contenant du décaBDE, dans la mesure du possible. Étant donné qu’un arrêt complet du recyclage du plastique des DEEE n’est pas une option souhaitable, un meilleur rendement de séparation du décaBDE du flux de déchets devrait être obtenu en augmentant la part d’identification et de séparation du décaBDE contenant des DEEE avant les processus de broyage sur la base des connaissances (informations sur les DEEE et composants pertinents) et des efforts (par exemple, investir plus de temps dans le processus de tri) et/ou des technologies améliorées d’identification et de séparation (par exemple, investir davantage et mieux dans une technique de tri ou de traitement préliminaire). Une part d’environ 6 % des plastiques issus des DEEE (fraction à faible teneur en brome) est mise en décharge. La mise en décharge des plastiques issus du traitement des DEEE devrait être réduite car le décaBDE restant n’est pas détruit. Les EEE usagés sont fréquemment exportés hors de l’UE. L’exportation de plastiques issus du traitement des DEEE devrait être réduite, étant donné que la GER adéquate n’est pas nécessairement assurée dans tous les pays importateurs.

En ce qui concerne la GER des véhicules en fin de vie, une part importante du décaBDE dans la fraction légère de déchiquetage n’est pas détruite et pourrait donc entrer dans les produits
recyclés. Le risque peut être minimisé par un rendement plus élevé de séparation des fractions riches en décaBDE grâce à une part accrue de la technologie post-déchiquetage appliquée à la fraction légère de déchiquetage. Les fractions à haute densité devraient être traitées de manière appropriée pour détruire ou transformer irréversiblement leur teneur en décaBDE. La mise en décharge devrait être réduite. Comme pour les DEEE usagés, l'exportation de voitures d'occasion devrait être réduite.

Il convient d'accorder une attention particulière à la GER du décaBDE contenant des plastiques provenant de déchets de construction et de démolition. La présence de décaBDE dans ce flux de déchets augmentera continuellement au cours des 20 prochaines années. Il faut donc explorer les options pour identifier et séparer le décaBDE et les autres plastiques contenant des POP-BDE afin de réaliser des progrès dans la gestion écologiquement rationnelle des déchets de construction et de démolition.

Les chloroalcanes à chaine courte (SCCP)

La LPCL actuelle pour les SCCP est de 10 000 mg/kg. Sur la base des évaluations effectuées dans le cadre de la présente étude, une fourchette de 420 mg/kg à 10 000 mg/kg est proposée pour les SCCP. Théoriquement, abaisser la LPCL à des valeurs supérieures à 1 500 mg/kg n'aurait aucune incidence sur les utilisations historiquement pertinentes et les flux de déchets connexes, mais des concentrations de SCCP plus élevées (plusieurs milliers de mg/kg) ont été signalées lors d'activités de contrôle dans les États membres de l'UE. Par conséquent, une valeur limite inférieure à 10 000 mg/kg pourrait encore avoir un impact sur les produits fabriqués et/ou recyclés contaminés par ou contenant des SCCP entrant sur le marché communautaire. L'impact potentiel ne peut être quantifié sur la base des informations actuellement disponibles, mais serait probablement lié à des variations relativement faibles des coûts de gestion des déchets par rapport à la situation actuelle et ne devrait pas entraîner de goulots d'étranglement au niveau des capacités d'élimination et de valorisation dans l'UE.

Abaisser la LPCL en dessous de 1 500 mg/kg (par exemple 1 000 mg/kg ou 500 mg/kg) pourrait limiter la mise sur le marché et l'utilisation d'articles contenant moins de 0,15% (en poids) de SCCP et probablement aussi les alternatives au MCCP, car c'est la quantité de SCCP qui peut être présente sous forme d'impuretés dans un article produit avec des MCCP (selon le règlement POP de l'UE). Une valeur limite inférieure à 1 500 mg/kg n'aurait théoriquement toujours aucune incidence sur la gestion des flux de déchets résultant d'applications historiques (par exemple, le caoutchouc provenant des bandes transportueuses et des produits d'étanchéité et adhésifs), mais aurait un impact sur certains produits fabriqués et/ou recyclés contaminés par ou contenant des SCCP entrant sur le marché communautaire. En outre, le choix d'une valeur limite inférieure à 1 500 mg/kg mais supérieure à 420 mg/kg, qui est la concentration moyenne théorique estimée des SCCP dans les déchets de caoutchouc dans l'UE, entraînerait un changement relativement faible des coûts de gestion des déchets. Une valeur limite supérieure à 420 mg/kg n'entraînerait pas non plus de goulots d'étranglement au niveau des capacités d'élimination et de valorisation dans l'UE.

Toutefois, une concentration limite de POP inférieure à 420 mg/kg (par exemple, 100 mg/kg) entraînerait une augmentation supplémentaire des coûts ce qui n'est pas considéré comme une solution économiquement réalisable. Toutes les autres implications décrites dans la section précédente s'appliquent également ici (par exemple, liées à la mise sur le marché et à l'utilisation d'articles contenant des SCCP à des concentrations inférieures à 0,15 %). Les capacités d'élimination et de valorisation des déchets sont toujours jugées suffisantes même lorsque les valeurs limites potentielles sont inférieures à 420 mg/kg, mais ne le sont pas lorsque les valeurs sont inférieures à 1 mg/kg.
En ce qui concerne les SCCP, étant donné qu'aucune indication n'a été trouvée indiquant que la limite maximale de résidus actuelle est insuffisante, il est proposé de maintenir la valeur actuelle de 10 000 mg/kg, telle qu'établie dans le Règlement.

Les recommandations pour une gestion écologiquement rationnelle afin de minimiser le risque comprennent la séparation des bandes transportrices en caoutchouc contenant des SCCP des mines souterraines et des produits d'étanchéité et adhésifs dans les déchets de C&D. Dans la mesure du possible, la séparation devrait tenir compte de la limitation des techniques de séparation disponibles propres aux SCCP. Pour les bandes transportées en caoutchouc, la fraction métallique de la bande transportée devrait être séparée, dans la mesure du possible, de la fraction en caoutchouc. Il convient d'éviter autant que possible tout dégagement de poussière et l'inhalation de poussières contaminées devrait être évitée par l'application d'un masque respiratoire. D'autres flux de déchets (résultant par exemple de produits de consommation) pourraient être difficiles à contrôler car ils proviennent essentiellement de produits importés. On s'attend à ce que la pluspart des déchets concernés soient gérés conjointement avec les déchets solides municipaux (qui sont en grande partie incinérés et mis en décharge dans l'UE). Toutefois, il n'est généralement pas exclu que certains flux de déchets soient également recyclés (ce qui pourrait entraîner l'accumulation de SCCP dans de nouveaux produits recyclés). Une mesure de contrôle de la concentration des produits importés et des déchets connexes n'est réalisable que si des méthodes/normes d'analyse fiables sont utilisées pour distinguer les MCCPs des SCCPs.

**HCBD**

L'HCBD figure dans le règlement POP de l'UE avec une LPCL de 100 mg/kg. D'après les évaluations présentées dans le présent rapport, une fourchette de concentration de 0,1 à 100 mg/kg pour la LPCL est possible. Rien n'indique qu'il soit nécessaire d'ajuster la limite de concentration établie. Il est proposé de maintenir la LPCL pour l'HCBD à 100 mg/kg.

L'HCBD figure dans le règlement POP de l'UE avec une MPCL de 1 000 mg/kg. Rien n'indique qu'il soit nécessaire d'ajuster la limite de concentration établie et que l'abaissement de cette limite de concentration aurait des effets bénéfiques sur l'environnement ou la santé. Il est proposé de maintenir la MPCL pour l'HCBD à 1 000 mg/kg.

Les flux de déchets ne sont pas considérés comme pertinents dans l'UE. En règle générale, les recommandations relatives à la GER, par exemple des directives générales et spécifiques de la Convention de Bâle sur les POP, s'appliquent.

**Dicofol**

Pour le dicofol, aucune valeur limite n'est actuellement établie en vertu du règlement POP de l'UE. D'après l'évaluation des critères de limites inférieure et supérieure, une LPCL potentielle pour le dicofol pourrait être établie dans une fourchette allant de 3 à 200 mg/kg. Il est proposé d'établir la LPCL pour le dicofol à 50 mg/kg.

Une LPCL possible de 50 mg/kg, qui se situe bien à l'intérieur de la fourchette proposée, s'alignerait sur les LPCL actuellement établies pour d'autres pesticides POP.

Il est proposé d'établir une LCPL, comme dans le cas de la LPCL, de 5 000 mg/kg, en alignement avec les autres pesticides POP pour le dicofol.

Les flux de déchets ne sont pas considérés comme pertinents dans l'UE. D'une manière générale, les recommandations relatives à la gestion écologiquement rationnelle des déchets, par exemple les recommandations générales et spécifiques de la Convention de Bâle sur les POP, s'appliquent.
**PFOA**

Aucune valeur limite pour le PFOA, ses sels et ses composés apparentés, n’est établi dans le Règlement POP actuel. En raison de la disponibilité limitée des données quantitatives, aucun critère de limitation supérieure n’a pu être établi. En l’absence de seuil supérieur établi, il est recommandé d’utiliser la même LPCL que celle établie pour le PFOS, c’est-à-dire 50 mg/kg pour le PFOA et 2 000 mg/kg pour les substances apparentées au PFOA. Une LPCL plus élevée pour les substances apparentées au PFOA est recommandée parce qu’elles ne se dégradent que dans une certaine mesure et de façon inconnue en PFOA et parce que c’est une limite de concentration pour la somme des substances apparentées au PFOA.

Pour le MCPL il est recommandé de fixer la même limite de concentration que celle établie pour le PFOS, c'est-à-dire 50 mg/kg pour le PFOA et 2 000 mg/kg pour les substances apparentées au PFOA.

Sur la base des informations et des données actuellement disponibles, il n'a pas été possible d'étudier en détail les flux de déchets comme cela a été fait pour d'autres substances dans ce rapport. La plupart des flux de déchets concernés sont ou seront généralement incinérés. L'incinération contrôlée à haute température (p. ex., à 1 000 °C) est efficace pour détruire le PFOA et empêcher la formation de PFOA de la thermolyse de polymères hautement fluorés. En ce qui concerne le PFOS, le prétraitement est mentionné comme un aspect important de l'élimination écologique et les méthodes devraient être choisies en fonction de la nature et du type des déchets de PFOS. Cela peut également s'appliquer aux déchets de PFOA.

Le besoin d'informations supplémentaires est fortement soutenu par l'équipe du projet, en particulier les informations et les données sur la production et l'utilisation de PFOA dans différents domaines d'application sont encore très limitées ou, dans certains cas, totalement manquantes, mais essentielles pour un choix dûment justifié des valeurs limites pour les déchets POP. On peut s'attendre à ce que les États membres de l'UE poursuivent ou lancent des projets liés au PFOA dans les articles, les déchets et les produits recyclés qui contribueront à améliorer la situation actuelle concernant la disponibilité des données pour l'UE.

**PFHxS**

Aucune valeur limite pour le PFHxS, ses sels et ses composés apparentés, n’est établi dans le Règlement POP actuel. En raison de la disponibilité limitée des données, il n’a pas été possible d’établir des critères limitatifs supérieurs pour le PFHxS, ses sels et ses composés apparentés. En l’absence de critères limitatifs supérieurs, il est recommandé d’utiliser la même LPCL que celle établie pour le PFOS, c'est-à-dire 50 mg/kg pour le PFHxS et 2 000 mg/kg pour les substances apparentées au PFHxS. Une LPCL plus élevée pour les substances apparentées au PFHxS est recommandée car les composés ne se dégradent que dans une certaine mesure et de façon inconnue en PFHxS et parce que c’est une limite de concentration pour la somme des substances apparentées au PFHxS.

Les informations sur la production et l'utilisation du PFHxS, de ses sels et des composés apparentés dans l'UE sont très rares et essentiellement qualitatives. Sur la base des informations disponibles, il a été considéré que la plupart des flux de déchets identifiés seront à l’avenir incinérés. Pour les autres flux de déchets pertinents, on peut supposer qu'ils seront collectés et gérés par des entreprises spécialisées (par exemple, pour la métallisation).

Les informations, en particulier sur la production et l'utilisation, sont encore très limitées, voire inexistantes, mais elles sont essentielles pour définir des valeurs limites dûment justifiées pour les déchets de POP. On peut s'attendre à ce que les États membres de l'UE poursuivent ou lancent des projets liés à la présence de PFHxS dans les articles, les déchets et les produits recyclés, qui contribueront à améliorer la situation actuelle en matière de disponibilité des données.
**HBCDD**

Le HBCDD figure dans le Règlement POP de l’UE avec une LPCL de 1 000 mg/kg. Compte tenu des résultats de la présente étude, une fourchette de 100 à 1 000 mg/kg de HBCDD est proposée comme LPCL. Afin de décider à quel endroit de cette fourchette la LPCL devrait être établie, plusieurs considérations de faisabilité et d’efficacité doivent être prises en compte. La norme correspondante (CEI 62321-3-1: 2013) devrait être validée pour les niveaux de concentration conformément au LPCL qui sera sélectionné. Cependant, la mise en œuvre peut déjà être basée sur les méthodes analytiques existantes.

La MPCL figurant à l’annexe V du Règlement POP de l’UE est fixée à 1 000 mg/kg. Rien n’indique que la LPCL actuelle ne soit pas appropriée. Il est proposé de maintenir la limite de concentration établie à 1,000 mg/kg pour la MPCL de l’HBCDD.

Le EPS et le XPS dans la construction ainsi que le plastique EPS sont considérés comme les flux de déchets les plus importants. En ce qui concerne leurs GER, le risque de recyclage des EPS/XPS peut être minimisé en séparant au maximum les matériaux isolants EPS/XPS dans la construction et la destruction et en les traitant de manière appropriée.

CreaSolv® est un procédé innovant pour le recyclage du EPS contenant du HBCDD. L’HBCDD est largement détruit tandis que le PS et le brome peuvent être recyclés et utilisés pour d’autres applications.

Environ un tiers des déchets d’emballages en EPS sont recyclés. Aujourd’hui le HBCDD n’est plus utilisé dans les emballages en EPS. Toutefois, dans la pratique, HBCDD contenant des matériaux isolants EPS sont mélangés par inadvertance et recyclés avec les déchets d’emballage. Ainsi, les déchets qui au départ ne sont pas contaminés ou ne contiennent que de faibles quantités d’HBCDD (déchets d'emballages en EPS) sont mélangés à des déchets contenant du HBCDD (EPS from C&D) et, par conséquent, des granulés/pellets de PS recyclés contaminés sont produits.

Cette pratique implique un transfert supplémentaire d’HBCDD dans divers produits en plastique. Pour minimiser le risque, il conviendrait de s’efforcer de mieux séparer les déchets de EPS des déchets de C&D et des déchets d'emballages en EPS. A cette fin, des mesures spécifiques devraient être mises en œuvre pour (1) sensibiliser à la nécessité de séparer l’EPS de la construction chez les producteurs de déchets, les recycleurs et les installations de collecte des déchets et pour (2) accroître l'utilisation des méthodes de criblage pour identifier les EPS contenant du HBCDD dans les installations de collecte et de recyclage.

Près de 30 % des panneaux EPS/XPS contenant du HBCDD sont mis en décharge. Les risques connexes peuvent être réduits en minimisant leur mise en décharge.

**PCBs and PCDD/F**

La LPCL pour les PCB qui ne ressemblent pas aux dioxines (ndl-PCBs) est établie à 50 mg / kg. La LPCL pour PCDD/F est établie à 15 µg TEQ / kg.

L’examen des LPCL pour les PCB a montré qu’il est nécessaire d’envisager ce groupe de substances de manière bilatérale, ce qui signifie que les ndl-PCBs qui ressemblent aux dioxines (dl-PCBs) doivent être considérés indépendamment. La structure des dl-PCB est similaire à celle de la dioxine et correspond au même mode d’action toxique. Ainsi, les recommandations suivantes peuvent être faites.

Pour les ndl-PCB, rien n’indique que la LPCL (50 mg / kg) devrait être ajustée. Les dl-PCB doivent cependant être traités séparément des ndl-PCB.

Pour cela, il y a deux options: les dl-PCB peuvent être intégrés dans la LPCL de PCDD/F basées sur le TEQ, ou une LPCL individuel pour les dl-PCB basés sur le TEQ peut être établie par exemple 10 fois plus élevé que le Les LPCL de PCDD/F, parce que le TEF le plus élevé de tous les dl-PCB étant
de 0,1 pour le PCB 126. Cependant, des informations récentes indiquent que la toxicité de ce congénère de dl-PCB pourrait être surestimée.

D'après les données scientifiques, il apparaît que, pour les PCDD/PCDF, la LPCL actuelle de 15 µg TEQ/kg est trop élevée pour prévenir tout effet négatif possible sur l'environnement.

Les cendres volantes provenant de l'incinération des déchets solides municipaux sont considérées à cet égard comme un flux de masse important. Les données disponibles indiquent que 11% et 3% de ces cendres volantes dépasseraient une LPCL de 5 µg TEQ/kg et de 10 µg TEQ/kg, respectivement. Cela indique que si la LPCL pour les PCDD/F était abaissée de manière correspondante, la majorité des cendres volantes peut toujours être utilisées conformément au traitement en cours. Les parties prenantes, cependant, affirment que, lorsque la valeur LPCL des PCDD/F est abaissée, le recyclage en toute sécurité (par exemple comme agent de remplissage dans l'asphalte) peut être entravé et qu'une collecte de données appropriée sur le contenu en PCDD/F dans les cendres volantes doit être effectuée avant de proposer une nouvelle LCPL, avec une évaluation appropriée de l'impact sur l'environnement et la santé humaine lorsque les pratiques actuelles sont suivies.

Pour les PCDD/F, sur la base des résultats de cette étude, une valeur inférieure comprise dans l'intervalle, par exemple de 5 à 10 µg/kg, devrait être envisagée. Dans le cas où les dl-PCB sont inclus dans la LPCL des PCDD / F, il pourrait être raisonnable d'envisager une valeur à la limite supérieure de cette fourchette.
1. BACKGROUND AND OBJECTIVE

To ensure coherent and effective implementation of the Union’s obligations under the Protocol on persistent organic pollutants to the (UNECE) Convention on Long Range Transboundary Air Pollution (CLRTAP) and the Stockholm Convention on Persistent Organic Pollutants (POPs), waste-related issues in annexes IV and V of Regulation (EC) 850/2004 will be reviewed in this study.

Both instruments, the CLRTAP and the Stockholm Convention, aim to reduce and eliminate the production, use and releases of persistent organic pollutants (POPs) in the territories of all participating parties. Amongst other issues these conventions contain provisions on environmentally sound management of wastes consisting of, containing or contaminated by POPs (hereafter called “POP waste”).

POPs give rise to concern as, per definition, they are persistent in the environment and in human tissues, bio accumulate in the food chain due to their lipophilic properties, have the potential for long-range environmental transport and show adverse effects ranging from acute to chronic toxicity. Due to their specific properties, POP substances have been disseminated into the environment all over the world and have entered the food chain.

Although substantial progress has been achieved in limiting the use of listed POPs and reduce their emissions there are ongoing releases into the environment as well as a constant cycling of substances released in former times.

For an optimised approach to elimination, all sectors in the life cycle of a chemical and of anthropogenic emission sources should be addressed. In this framework, waste management is an essential sector for the control of POP releases into the environment. Thus, the comprehensive regulation of POP contamination in waste might have the potential to reduce POP presence in the environment.

The Stockholm Convention was implemented into European Community law in 2004 by Regulation EC (No) 850/2004 (hereafter “POP Regulation”). This Regulation, besides others, contains an obligation to destroy or irreversible transform the POP content of waste above a certain concentration, but also contains the provision that in exceptional cases waste above the limits may be otherwise managed with specified operations for specified waste types if destruction or irreversible transformation do not represent the environmentally preferable option and waste are below another threshold.

As stipulated in part 1 of Annex V to the POP Regulation, for all wastes exceeding this concentration limit, only the following disposal and recovery operations are permitted when applied in such a way as to ensure that the POP content is destroyed or irreversibly transformed:

- D9 (physico-chemical treatment),
- D10 (incineration on land)
- R1 (use as a fuel)
- R4 (reclamation of metals and metal compounds).

Permanent storage (D12) in safe, deep hard rock formations, salt mines or landfill sites for hazardous waste may be permitted in exceptional cases for wastes listed in part 2 of Annex V to the POP Regulation.

In 2017, the conference of the parties to the Stockholm Convention decided to list Decabromodiphenylether (decaBDE) in annex A, Short-chain chlorinated paraffins (SCCPs) in annex A and Hexachlorobutadiene HCBD in annex C to the Convention. Therefore, concentration limits for waste containing Decabromodiphenylether (decaBDE) must be established in annexes IV and V to the POP Regulation.
Furthermore, additional substances are expected to be added to the list of POP substances during the next years. Therefore, concentration limits shall also be derived for the so called ‘candidate POPs’ i.e. dicofol, PFOA, its salts and PFOA-related compounds as well as PFHxS, its salts and PFHxS-related compounds.

Polychlorinated biphenyls (PCBs) and polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs) were already listed in the POP Regulation in 2006 and HBCDD in 2016. With a view to ensuring the adaptation to scientific and technical progress, the Commission intends to examine the existing concentration limits for these POPs.
2. **APPROACH**

2.1 **Legal Analysis**

The commitments and decisions subscribed at the Conference of the Parties of the Stockholm Convention and of the Basel Convention may entail changes in relevant EU legislation.

The legal analysis aims to assess the extent to which provisions dealing with POPs waste in relevant pieces of EU legislation are affected by the results of the last Conference of the Parties of the Stockholm and Basel Conventions which took place in April and May 2017. The assessment is carried out for the new POPs which have recently been added to the Annexes of the Stockholm convention (i.e. decaBDE, SCCPs and HCBD), as well as for the new candidate POPs (i.e. Dicofol, PFOA, PFHxS). With regard to the latter, the extent was assessed to which provisions dealing with POPs waste in relevant pieces of EU legislation would be affected if the substances concerned were incorporated to the relevant Annexes of the Stockholm Convention.

The project team evaluated the above with two consecutive questions:

1. Is the relevant EU legislation affected by the decisions taken at the last Conference of the Parties of the Stockholm?

2. Is it considered necessary to amend relevant EU legislation as a result of the last Conference of the Parties of the Stockholm and Basel Conventions?

2.2 **Informal Stakeholder Consultation**

For obtaining information from Member States (MS) and other relevant stakeholders a non-formal consultation was carried out via a questionnaire. The questionnaire is attached to this report in Annex 0.

The stakeholder consultation started on 27.02.2018 with COM sending out the questionnaire accompanied by a recommendation letter provided by the Commission. Responses are received by COM and Ramboll via an established project correspondence email address (POPWaste2018@ramboll.com).2018. Follow-ups with stakeholders were carried out as appropriate. The documentation of replies can be found in Annex 10.2. Non-confidential stakeholder contributions were published on the project website.

**Dissemination of project related information**

A dedicated project website was set up to share information about the study to support the review of waste-related issues in Annexes IV and V of Regulation (EC) 850/2004. It invites a wide range of potentially interested stakeholders including MS’ authorities, industry associations and NGOs to participate (further details can be found at: http://pops-and-waste.bipro.de).

The website gives an overview on current activities related to the study in the “News” section. This section was updated on regular basis in accordance with the progress of the project. Activities such

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9 The following Decisions of the COP of the Stockholm Convention have been analysed: UNEP/POPS/COP.8/SC-8/4; UNEP/POPS/COP.8/SC-8/7; UNEP/POPS/COP.8/SC-8/10; UNEP/POPS/COP.8/SC-8/11; UNEP/POPS/COP.8/SC-8/12

10 The following Decisions of the COP of the Basel Convention have been analysed: UNEP/CHW/COP.13/BC-13/4; UNEP/CHW.13/6/Add.1/Rev.1;

as the project start, and the launch of the non-formal stakeholder consultation and the workshop were announced on the website.

Workshop related information and available project reports are or will be available at the project website.

2.3 Deriving limit values in accordance with the POP Regulation

In the following the applied methodology to derive limit values in accordance with the POP Regulation is explained.

Limit values in accordance with Annex IV of the POP Regulation as well as for specific waste treatment operations, ensure pollutant removal to the largest extent possible on the one hand and environmentally sound recycling processes on the other hand.

A distinction is made between “low POP concentration limits” (LPCLs) in accordance with Art. 7(4)a of the POP Regulation and “maximum POP concentration limits” (MPCLs) of substances listed in Annex IV of the POP Regulation.

Waste with a POP content exceeding the LPCL is subject to the regulations of the Stockholm Convention on destruction or irreversible transformation as stipulated in Article 6. 1. (d) (ii). Waste with a POP content ranging below the LPCL is subject to other relevant legal regulations on the disposal or recovery of waste.

For waste containing POPs in concentrations equal to or exceeding the LPCL, only certain disposal or recovery operations (D9, D10, R1, R4) are permissible in accordance with Annexes I and II of the Regulation 2008/98/EC.

Three concentration ranges (A, B and C), which are relevant under the POP Regulation, are generally possible for POP-containing waste. The consequences associated with different POP concentrations in waste depending on LPCL and MPCL are graphically illustrated in Figure 1.

![Figure 1: Consequences regarding POP concentrations in waste depending on LPCL and MPCL](image)

2.3.1 BASIC PRINCIPLES OF THE METHOD FOR THE DERIVATION OF LPCLs

The basic principle of the method is based on the delimitation of the concentration range of a possible limit value for each of the relevant substances by means of a set of different lower and upper limitation criteria which considered economic feasibility with regards to waste management, and associated risks to name but a few. These criteria indicate concentrations below which limit values should not be set, others indicate concentrations above which limit values should not be
Figure 2 illustrates the delimitation of a concentration range for a possible LPCL through lower and upper limitation criteria:

![Figure 2: Delimitation of the concentration range for a limit value through lower and upper limitation criteria](image)

To determine the range of possible limit values for every substance, four lower and two upper limitation criteria are applied. Options can be derived from the aggregation of the results for the individual criteria.

**Lower limitation criteria:**

(A) **Analytical potential:** It must be possible to control limit values analytically.

(B) **Background contamination:** Limit values should be above existing environmental background contaminations.

(DR) **Disposal and recovery capacities:** Limit values should be established in a way that the (new) required capacities for waste recovery and disposal are realistically available.

(E) **Economic feasibility:** Limit values should be established in a way that required additional waste management costs are economically reasonable.

**Upper limitation criteria:**

(LV) **Limit values:** Limit values should not conflict with existing limit values.

(R) **Risks (possible adverse effects on human health and the environment):** Limit values should be established in a way that adverse effects on human health and the environment are avoided.

This approach ensures on the one hand that the limit value can be implemented realistically in the light of the available data, and on the other hand, that human health and the environment are protected from POPs to the largest extent possible. Moreover, the method allows proposing differentiated limit values for specific waste streams, in case they are linked to a specific risk potential. A POP substance can be contained in residues resulting from waste incineration, for instance, and can therefore be landfilled underground as hazardous waste (→ minor risk potential). It can also be contained in sewage sludge and can possibly be applied directly on agricultural soil (→ high risk potential). In such cases, a (higher) limit value for the waste stream “incineration residues” and a (lower) limit value for the waste stream “sewage sludge” could be proposed.

The methodology to derive LPCLs does not contain a full socio-economic impact assessment. The criteria (DR) related to the availability of disposal and recovery capacities and (E) on economic feasibility which takes specifically changes in waste treatment costs into account are related to economic impacts. However, a full assessment of the socio-economic impacts goes beyond the scope of the present project. Also, positive cost impacts, such as reduced environmental or health costs are not specifically taken into account. It is generally considered that a significant decrease of releases of POPs to the environment contributes to reducing environmental or health costs which can be significant (see e.g. [Trasande et al. 2016; Grandjean and Bellanger 2017; Lam et al. 2017]). Based on the available information, relevant possible economic impacts of the proposed
measures are, however, discussed in the recommendations related to each substance (see section 8).

2.3.2 BASIC PRINCIPLES OF THE METHOD FOR THE DERIVATION OF MPCLs

MPCLs cannot be established below the LPCLs. The derivation of MPCLs is reduced to the following question/criterion: Above which POP concentration can risks occur in a worst-case scenario to humans or the environment that do not allow application of an environmental preferable solution other than destruction or irreversible transformation? The limit value must be below concentrations where corresponding risks may occur according to a worst-case scenario. For a detailed explanation of the methodology to derive MPCLs see [BiPRO 2005] and [BiPRO 2011].

2.3.3 REASSESSMENT OF LIMIT VALUES FOR ALREADY LISTED SUBSTANCES

The described methodology for the formulation of limit values is flexible to up-coming knowledge and technical development and thus enables to take new information into consideration. Notable developments with regard to the limitation criteria might influence the outcome of applying the methodology for the derivation of limit values.

In order to assess whether notable developments have occurred, a screening whether there is information available which indicates a possible need to adjust already established LPCLs or MPCLs has been performed.

The main conclusions were that there was a particular need to review the LPCL for HBCDD and PCDD/Fs and to possibly particularly consider the dioxin-like polychlorinated biphenyls (dl-PCBs) on a total dioxin toxic equivalence (TEQ) basis.

HBCDD was listed in the EU POP Regulation in 2016. The LPCL is established at 1,000 mg/kg and is subject to review by the Commission by 20.4.2019. [BiPRO 2015] considers a reduction of the LPCL to 100 mg/kg after a period of about 5 years possibly feasible. A lower LPCL has further been proposed in a current submission by International POPs Elimination Network (IPEN) (i.e. 100 mg/kg). Therefore, it should be evaluated, if an adjustment of the POP content limit values for HBCDD is appropriate at present.

With regard to PCDD/Fs, information from studies prepared by the Swedish Environmental Protection Agency (EPA) in 2011 as well as information provided by IPEN indicates the need for reviewing and possibly adjusting the currently established limit values for PCDD/Fs. Further, possible impact / contribution of dl-PCBs in waste pointed out by the Swedish EPA and IPEN and therewith associated implications on PCDD/F limit values. According to the Swedish EPA, as long as dl-PCBs are not included in the LPCL for PCDD/Fs, the estimated risk associated to food chain transfer of pollutants in waste may be underestimated [SE EPA 2011]. It could be considered to include the dl-PCBs in the TEQ-based POP-contents levels of PCDD/Fs and/or to lower the LPCL for PCDD/Fs.

Furthermore, the questionnaire as described under WP 2, includes questions on any notable developments known to the respondents.

If new literature sources or questionnaire responses indicate a notable development, the relevant information was considered to assess whether an adjustment of already established waste related POP concentration limits is appropriate.

12 See chapter 4.2 of the interim report from 14.3.2018
3. LEGAL ANALYSIS

3.1 Identification of actions to be taken to modify relevant provisions of the POPs Regulation and other EU legislation

Under the legal analysis, the project team assessed the extent to which provisions dealing with POPs waste in relevant pieces of EU legislation are affected by the results of the last Conference of the Parties of the Stockholm\(^\text{13}\) Convention which took place in April and May 2017. The assessment was carried out for the new POPs which have recently been added to the Annexes of the Stockholm Convention (i.e. decaBDE, SCCPs and HCBD), as well as for the new candidate POPs (i.e. Dicofol, PFOA, PFHxS). With regard to the latter, the project team assessed the extent to which provisions dealing with POPs waste in relevant pieces of EU legislation would be affected if the substances concerned were added to the relevant Annexes of the Stockholm Convention.

The project team evaluated the above, taking two questions into account:

1. Is the relevant EU legislation affected by the decisions taken at the last Conference of the Parties of the Stockholm Convention?

2. Is it considered necessary to amend relevant EU legislation as a result of the last Conference of the Parties of the Stockholm Convention?

As a starting point, the project team carried out a mapping of the main EU legislation which incorporates provisions on POPs waste or which regulates matters related to or relevant for one of the POPs covered by this study. Subsequently, an assessment was carried out per piece of legislation, through the two questions described above. A more specific analysis was in some instances based on available preliminary information on the occurrence of a specific new or candidate POP in waste streams (e.g. PFOA or SCCP in sewage sludge).

As a final step, an overview of relevant EU legislation that is affected and where amendments could or should be considered was prepared (see Table 1). Required or possible amendments, in order to align such legislation with the amended annexes of the Stockholm Convention are also suggested in the overview.

3.2 Mapping relevant EU provisions and assessing necessary actions

3.2.1 POP REGULATION (EC) NO 850/2004

The project team notes that the following analysis takes as a starting point the listing of the new and candidate POP under the Annexes I, II and III of the POP Regulation, in correspondence with their listing under the Stockholm Convention. In addition, the analysis takes as a starting point the listing of the new and candidate POPs under the relevant Annexes IV and V to the POP Regulation, accompanied by concentration limits under which waste containing a POP may be otherwise disposed of or recovered in accordance with the relevant EU legislation.

Disposal and recovery operations

Part 1 of Annex V of the POP Regulation lists the disposal and recovery operations with which POP waste shall be disposed of or recovered in accordance with Article 7. Decision BC 13/4 of the

\(^{13}\) The following Decisions of the COP of the Stockholm Convention have been analysed: UNEP/POPS/COP.8/SC-8/4; UNEP/POPS/COP.8/SC-8/7; UNEP/POPS/COP.8/SC-8/10; UNEP/POPS/COP.8/SC-8/11; UNEP/POPS/COP.8/SC-8/12
Conference of the Parties (COP) of the Basel Convention on Technical guidelines on the environmentally sound management of wastes consisting of, containing or contaminated with persistent organic pollutants adopts general technical guidelines\(^\text{14}\) on the environmentally sound management of wastes consisting of, containing or contaminated with persistent organic pollutants.

The project team has reviewed whether these technical guidelines require the adaptation of part 1 of Annex V of the POP Regulation, by addition or removal of disposal and recovery operations.

*The project team has not found indications in the technical guidelines under the Basel Convention that addition or removal of disposal and recovery operations under Annex V of the POP Regulation is necessary.*

### 3.2.2 REACH REGULATION (EC) NO 1907/2006 (REACH)

Firstly, the project team notes that Article 2(2) of REACH states that waste as defined in Waste Framework Directive 2006/12/EC (now Directive 2008/98/EC) of the European Parliament and of the Council is not a substance, mixture or article within the meaning of Article 3 of this Regulation. As the definition of waste as of Directive 2008/98/EC and the one used under EU POP Regulation are identical (see Article 2 (e) POP Regulation, Article 3(1) Was), POP waste in the sense of the POP Regulation is also excluded from the definitions of substance, mixture and article in REACH. As such, the COP decisions of Stockholm convention specifically related to POP waste do not directly influence provisions of REACH.

As such, REACH does regulate the restriction of the placing on the market of substances. Consequently, restrictions do cover the recycling of these substances, as this operation, although a waste recovery operation, can result in the placing on the market of a product containing a certain substance. DecaBDE is listed under Annex XVII of REACH with exemptions. For example, REACH restrictions on production of DecaBDE, or its placing on the market in an article, or any part thereof, do not apply to electrical and electronic equipment (EEE) within the scope of Directive 2011/65/EU (RoHS), as the RoHS Directive restricts the use of PBDEs (as a group) in EEEs with a maximum concentration value of 0,1% by weight in homogenous material.

The restrictions on the use in the production of or placement on the market in an article of DecaBDE and PFOA, as well as the exemptions to these restrictions, are influenced by the COP decisions of the Stockholm Convention. To avoid overlap and inconsistencies between Regulations, the listing of these substances in the Annexes of the Stockholm Convention and the subsequent implementation in the corresponding Annexes of the POP Regulation, should lead to the removal of the restrictions regarding decaBDE\(^\text{15}\) from Annex XVII of REACH. Subsequently, the restrictions and their exemptions will have to be replaced by the prohibitions (Annex I) or restrictions (Annex II) of the POP Regulation. Furthermore, the existing exemptions under the restrictions for DecaBDE and PFOA will become obsolete and may be replaced by exemptions under the Stockholm Convention for which the EU has registered after the listing. With regard to further implications of this last point, we would like to refer to the paragraph regarding the RoHS Directive.

The project team has analysed the potential relevance of REACH for all new and candidate POPs but has not found any indication of relevance, other than the above.

\(^{14}\) UNEP/CHW.13/6/Add.1/Rev.1

\(^{15}\) PFOA is also listed under Annex XVII of REACH. As such, removal of its listing may also become necessary upon its listing as a POP under the Stockholm Convention and subsequently the POP Regulation.
The project team therefore concludes that REACH is affected by the COP decisions of the Basel and Stockholm conventions and that amendment of Annex XVII of that Regulation is necessary as described above.

3.2.3 CLP-REGULATION (EC) NO 1272/2008 (CLP)

Article 1(1) of the CLP-Regulation states that the purpose of this Regulation is to ensure a high level of protection of human health and the environment as well as the free movement of substances, mixtures and articles.

Article 1(3) of the CLP-Regulation states that waste as defined in Directive 2006/12/EC of the European Parliament and of the Council of 5 April 2006 on waste (2) is not a substance, mixture or article within the meaning of Article 2 of this Regulation.

As the definition of Directive 2008/98/EC includes all categories of waste, POP waste is also not regulated by the CLP-Regulation. As such, the COP decisions of the Stockholm Convention affecting POP waste do not directly influence provisions of the CLP-Regulation.

No influence of the COP decisions of the Stockholm convention has been identified by the project team.

3.2.4 RESTRICTION OF HAZARDOUS SUBSTANCES DIRECTIVE 2011/65/EU (ROHS)

RoHS is relevant for the POP Regulation provisions related to POP waste, since recycling of POP waste, although a waste recovery operation, can result in the production of a product containing POPs. In this regard, the project team notes that recycling of waste streams for use in EEE is likely to be covered by article 7 of the POP Regulation. This Article has the effect that waste containing POPs above a specified limit value can only be recovered, if this operation is in accordance with Annex V part 1 of the POP Regulation and ensures that the POP is destroyed or irreversibly transformed. As such, in practice the use of recycled POPs waste in the production of EEE is prohibited, unless the POP is removed from the waste and is destroyed or irreversibly transformed. RoHS allows use of PBDEs as long as it complies with the 1000 mg/kg limit for PBDEs by weight in homogeneous material.

Article 4(1) of RoHS states that Member States shall ensure that EEE placed on the market, including cables and spare parts for its repair, its reuse, updating of its functionalities or upgrading of its capacity, does not contain the substances listed in Annex II. Annex II of RoHS lists various restricted substances, including polybrominated diphenyl ethers (PBDE). The maximum concentration value tolerated by weight in homogeneous materials for PBDE is 0.1%.

The project team notes that PBDE, as a general category for BFR, also covers DecaBDE. However, as discussed under the paragraph concerning REACH, the exemptions which are to be listed for DecaBDE under Annex I of the POP Regulation should be in line with the exemptions for DecaBDE under the Stockholm convention.

Annex A to the Stockholm Convention does not provide for an exemption possibility for decaBDE similar to the recycling exemptions applicable to formerly listed BFR (see e.g. SC, Annex A part IV, 1). In a strict interpretation of the Stockholm Convention, DecaBDE would only be allowed if it is contained in articles and preparations produced partially or fully from recycled materials or materials from waste prepared for re-use in a concentration lower than the unintentional trace contamination (UTC) which remains to be established in Annex I of the POP Regulation. If the UTC will be lower than the concentration limit in homogeneous materials for PBDE under RoHS, DecaBDE could be explicitly excluded from the entry for PBDE under Annex II of RoHS in order to avoid recycling of decaBDE in concentrations above the UTC (since, following this strict interpretation, it would not be allowed to recycle decaBDE according to the Stockholm Convention). As a consequence, the prohibition on the use of decaBDE, as laid down in Article 3(1)
of the POP Regulation would be applicable. This would mean that decaBDE can only be present in EEE in levels under the UTC which is to be established. In this context it should be considered that there is already a RoHS exemption for PBDEs in spare parts recovered from and used for the repair or refurbishment of medical devices (see RoHS, Annex IV, 31a).

With regard to electronic devices, COP Decision SC-8/10 only allows Parties to the Convention to list for an exemption for the following:

"Additives in plastic housings and parts used for heating home appliances, irons, fans, immersion heaters that contain or are in direct contact with electrical parts or are required to comply with fire retardancy standards, at concentrations lower than 10 per cent by weight of the part."

If the EU were to apply for this exemption, such an exemption, as limited to the categories of EEE listed under SC-8/10, will have to be reflected in Annex II of RoHS. Any limit values with which such an exemption is qualifed on EU-level, should also be reflected in Annex II of RoHS.

The project team has analysed the potential relevance of RoHS for all new and candidate POPs but has not found any indication of relevance, other than the above.

Therefore, the project team concludes that RoHS is possibly affected by the COP decisions of the Stockholm convention affecting POP waste. Article 7 of the POP Regulation already prohibits the recycling of POPs waste. However, depending on the UTC for decaBDE, the exemption which exists for PBDE under 0.1% by weight may not be fully applicable to DecaBDE. This is because the Stockholm convention does not explicitly allow for a recycling exemption for decaBDE. In such case, it could be reflected in the wording of Annex II of the RoHS Directive.

### 3.2.5 WEEE DIRECTIVE 2012/19/EU (WEEE DIRECTIVE)

Article 8(1) states that Member States shall ensure that all separately collected waste from electrical & electronic equipment (WEEE) undergoes proper treatment. Article 8(2) states that proper treatment, other than preparing for re-use, and recovery or recycling operations shall, as a minimum, include the removal of all fluids and a selective treatment in accordance with Annex VII.

Annex VII of the WEEE Directive describes the required elements of selective treatment for materials and components of WEEE. Paragraph 1 of the annex lists that plastic containing BFR has to be removed from any separately collected WEEE. The project team notes that the requirement of removing components containing certain substances from WEEE is not directly linked to any hard obligation under the POP Regulation pertaining to POPs waste. The obligation under Article 7 of the POPs Regulation regulates disposal, recovery and recycling. Separation of components can be considered a pre-treatment to these operations. However, the project team notes that Article 8(1) of the WEEE Directive, in combination with its Annex VII, provides a mechanism to increase the volume of waste streams that do not contain POPs and therefore can be recycled without falling under the prohibition of Article 7 of the POP Regulation.

In this regard, it is of interest that the broad requirement removing of any plastic containing BFR covers DecaBDE, which is a BFR, as this might facilitate the removal of this new POP from WEEE components. However, the project team notes that, in practice, the detection and removal of components containing specifically decaBDE can be challenging. Nevertheless, developments regarding the removal of BFR from post-shreddering plastics from WEEE may provide new possibilities. Such developments might, at least in part, be stimulated by legal mechanisms such as article 8(1) of the WEEE Directive in combination with its Annex VII.

The project team has analysed the potential relevance of the WEEE Directive for all new and candidate POPs but has not found any indication of relevance, other than the above.

In light of the above, the project team notes that the requirement of removing components containing a certain substance from WEEE can be applied with regard to other POPs. For example,
if recycling of waste containing PFOA where to be forbidden on the basis of article 7 of the POP Regulation, a requirement of removing components of medical devices (or WEEE in general) containing PFOA could be an option. However, the project team notes that a requirement of removing components containing certain POPs requires that such removal be technically and economically feasible. Furthermore, the project team notes that extending such a requirement to other POPs could decrease the quantity of WEEE which can be recycled. This in turn, might make it difficult for Member States to meet the minimum recovery targets of article 11 in combination with Annex V of the WEEE Directive.

The project team concludes that the WEEE Directive is not directly affected by the COP decisions of the Stockholm convention affecting POP waste. However, the project team notes that Article 8(1) of the WEEE Directive, in combination with its Annex VII (requirement of removing components containing certain substances) provides a mechanism to increase the volume of waste streams that do not contain POPs and therefore can be recycled without falling under the prohibition of Article 7 of the POP Regulation.

3.2.6 END OF LIFE VEHICLES DIRECTIVE 2000/53/EC (ELV DIRECTIVE)

Article 6(1) of the ELV Directive states that Member States shall take the necessary measures to ensure that all end-of life vehicles are stored (even temporarily) and treated in accordance with the general requirements laid down in Article 4 of Directive 75/442/EEC, and in compliance with the minimum technical requirements set out in Annex I to this Directive, without prejudice to national regulations on health and environment.

Furthermore, Article 6(3) states that Member States shall take the necessary measures to ensure that any establishment or undertaking carrying out treatment operations fulfils certain obligations in accordance with Annex I. Paragraphs 3 and 4 of Annex I require removal of certain components from ELVs. Paragraph 3 concerns Treatment operations for depollution of end-of-life vehicle. Paragraph 4 concerns treatment operations in order to promote recycling.

The project team notes that the requirement of removing components containing certain substances from ELV is not directly linked to any hard obligation under the POP Regulation pertaining to POPs waste. The obligation under Article 7 POPs Regulation regulates disposal, recovery and recycling. Separation of components can be considered a pre-treatment to these operations. However, the project team notes that Article 6(1) of the ELV Directive, in combination with its Annex I, provides a mechanism to increase the volume of ELV that do not contain POPs and therefore can be recycled.

The components to be removed according to Paragraphs 3 and 4 of Annex I are described in such a broad manner that it is hard to assess whether the treatment operations cover components containing POPs therefore the new and candidate POPs.

Explicit treatment requirement on removal of components containing POPs (or designated POPs) could be added to point 3 of Annex I. However, the project team notes that such a requirement might be difficult to fulfil in practice. As with WEEE, developments regarding the removal of BFR from post-shredding ELV may provide new possibilities. However, the project team also notes that a requirement of removing components containing a certain POP requires that such removal be technically and economically feasible. Furthermore, such a requirement decreases the quantity of ELVs which can be recycled. This in turn, might make it difficult for MS to meet recovery targets of article 7 the ELV Directive.

The project team has analysed the potential relevance of the ELV Directive for all new and candidate POPs but has not found any indication of relevance, other than the indicated above (coverage by article 7 POP Regulation).
The classification of waste as hazardous is carried out by application of the Waste Framework Directive 2008/98/EC (WFD) definition of hazardous waste: waste which displays one or more of the hazardous properties listed in Annex III of the WFD.

The methodology for identification of waste as hazardous has been linked to entries under the European List of Waste (COM Decision 2014/955/EU) (LoW). The entries under the LoW categorise waste streams according to the source and type. Of central importance is the fact that certain entries on the list of waste are marked with an asterisk (*). These waste types can be considered hazardous pursuant to the definition of the WFD. Hence the list of waste distinguishes between so-called absolute entries and mirror-entries. For absolute entries (hazardous or non-hazardous), the waste producer/holder can assume that they display one or more of the hazardous properties listed in Annex III to the WFD or respectively do not display these properties. By contrast, so-called mirror entries exist, which are a pair of related entries where one is hazardous and the other is not. Sometimes a member of a mirror entry can be the pair of several possible related alternative entries. Only for mirror entries, the waste producer/holder specifically needs to assess whether the waste displays any of the 15 hazardous properties mentioned in Annex III of the WFD. This Annex further describes 15 hazardous properties (HPs), which are largely aligned (but not identical) with Regulation 1272/2008 (CLP Regulation). With regard to POPs, the List of Waste contains a description of the method of assessing hazardousness of waste. Bullet 3 of this part mentions specific POPs which, if contained in a waste stream, lead to classification of the waste as hazardous.

"Wastes containing polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/PCDF), DDT (1,1,1-trichloro-2,2- bis (4-chlorophenyl)ethane), chlordane, hexachlorocyclohexanes (including lindane), dieldrin, endrin, heptachlor, hexachlorobenzene, chlordcone, aldrine, pentachlorobenzene, mirex, toxaphene hexabromobiphenyl and/or PCB exceeding the concentration limits indicated in Annex IV to Regulation (EC) No 850/2004 of the European Parliament and of the Council (1) ) shall be classified as hazardous."

Independent from this specific reference to POPs by the List of Waste, any waste containing a POP, which is classified as hazardous substance under the CLP Regulation, may lead to a classification of the POP-containing waste as hazardous due to the assessment of hazardous properties as described in Annex III to the WFD. The following example further explains this aspect.

If a waste contains, e.g. chlorinated paraffins C10-13, and an assessment of hazardous properties is required, the definition of HP14 in Annex III to the WFD will provide the most stringent threshold for this assessment, as chlorinated paraffins C10-13 are classified, among others, as aquatic chronic 1 (H410) under the CLP Regulation. This means in practice that any waste containing chlorinated paraffins C10-13 equal or above 2,500 mg/kg would be classified as hazardous, independent from the fact that chlorinated paraffins are considered to be a POP.

In addition to general provisions applying to all types of waste (e.g. the polluter pays principle, the waste hierarchy, the principles of proximity and self-sufficiency, etc.), the WFD contains specific provisions related to the environmentally safe management of hazardous waste. Hazardous wastes pose a greater risk to the environment and human health compared to non-hazardous waste and thus require a stricter control regime. This is for instance laid down in Articles 17 to 20 of Directive 2008/98/EC, providing additional labelling and identification documents, record keeping, monitoring and control obligations, i.e. from the waste production to the final disposal or recovery. In addition, mixing of hazardous waste is banned (Article 18) to prevent risks for the environment and human health. Moreover, permit exemptions that may be granted to hazardous waste installations are more restrictive (Article 25) than those for installations dealing with other wastes (Article 23). At the same time, treatment-related legal documents such as the Landfill Directive or
the Industrial Emissions Directive 2010/75/EU (IED), in its provisions on waste incineration facilities, make important distinction whether the waste to be treated is hazardous or not; obviously, the requirements for treating hazardous waste are stricter.

With regard to the new and candidate POPs, the project team notes that a choice exists to add any of these to this part of the List of Waste, in order to enable the classification of a waste stream containing that specific POP as hazardous. The result would be that such a waste stream would be subject to obligations under the WFD and other regulations, pertaining to hazardous waste, most notably, obligations on transportation, storage and record keeping.

However, the project team notes that if the new and candidate POPs are not added to the mentioned part of the List of Waste, the assessment based on hazardous properties as described in Annex III to the WFD will apply. It should be noted that adjustments of the LPCLs for the POPs specifically mentioned in Bullet 3 of section 2 may lead to changes in waste classification in practice (e.g. a lower LPCL may lead to a higher quantity of waste to be classified as hazardous).

3.2.8 SEWAGE SLUDGE DIRECTIVE 86/278/EEC

Regulation covering application of sewage sludge for agricultural purposes is relevant for this assessment because the current information indicates that such sludge might contain POPs such SCCP and PFOA in significant concentrations.

In principle, the application of sewage sludge for agricultural purposes is likely to be covered by article 7 of the POP regulation. Sludge is generally considered to be a waste stream. Consequently, sludge containing POPs cannot be recovered (so also applied to land directly for agricultural purposes) as this application does not lead to the POPs being destroyed or irreversibly transformed.

However, it should be noted that, in practice, individuals offering and applying the sewage sludge might not have POPs fully within their scope of consideration. Furthermore, the competent authorities with regard to the application of sewage sludge might not be fully aware of the working of the POP regulation or its most recent entries into the annexes.

The Sewage Sludge Directive does not explicitly refer to POPs at the moment. Article 4 refers to values for concentrations of heavy metals in soil to which sludge is applied, concentrations of heavy metals in sludge and the maximum annual quantities of such heavy metals which may be introduced into soil intended for agriculture are given in Annexes I A, I B and I C. Consequently, Article 5 states that Member States shall prohibit the use of sludge where the concentration of one or more heavy metals in the soil exceeds the limit values which they lay down in accordance with Annex I A and shall take the necessary steps to ensure that those limit values are not exceeded as a result of the use of sludge.

Bearing in mind the analysis above, it might be advisable to create a link in the sewage sludge directive with the annexes of the POP regulation. An alternative form could be entries in annex I A of the Sewage Sludge directive on the POPs which are most commonly found in Sewage Sludge.

3.3 Stakeholder input related to adjustments to EU legal framework

The listing of “new POPs” under the Stockholm Convention possibly induces further adjustments to EU legislation, beyond the EU POP Regulation. Thus, information was requested during the informal stakeholder consultation (see chapter 2.2) on this matter. In total, nine opinions could be obtained from the submission answering the question “…if, beyond the EU POP Regulation, there are any adjustments to EU legislation needed, resulting from the listing of the “new POPs” decaBDE, SCCPs and HCBD under the Stockholm Convention”. The response received are summarized and further described in section 0 of the Annex to this report.
One stakeholder suggested legal obligations related to eco-design. Recyclers have to deal with end-of-life product containing decaBDE, so manufacturers benefiting from exemptions under the Stockholm Convention who will continue to include decaBDE in articles should be obliged to inform recyclers of the decaBDE content and location of the parts containing decaBDE in the products. Moreover, the stakeholder argues that the recyclers had to bear the added cost of treatment at end-of-life linked to decaBDE.

Remarks Finish Environment Institute

According to the Finish Environment Institute, all EC waste legislation that sets specific recycling targets (ELV-directive 2000/53/EC, waste framework directive 98/2008/EU, WEEE-directive 2012/19/EU, etc.) should have a general derogation stating that all recycling targets are calculated from the material left after the removal of materials that are required to be disposed of in accordance with the POPs regulation. Further, this amendment should apply to all POP substances and not just the new POPs [FI Env. Institute Sub. 2018].

With regard to Finland’s submission, it should be noted that waste legislation laying down targets for recovery and recycling defines such targets in terms of percentage of the weight of a collected waste stream which enters a recovery or recycling operation. As such, the targets currently do not explicitly take into account the possibility of a waste stream containing POPs equal to or above a given limit value laid down in Annex IV to the POP Regulation. Depending on the type and content of POPs, the originally collected waste could be reduced in terms of volume. As such, the quantity of waste which can be recovered and recycled and therefore counted towards the established targets is reduced.

One advantage of the proposed approach would be that a potential incentive of waste management actors to dilute and recycle waste streams containing POPs in a content higher than an established limit value under Annex IV POP regulation would be decreased. However, one disadvantage of such an approach would be that waste management operators would be incentivised to simply dispose of such waste streams, as they do not count towards achieving recycling targets. There would be no incentive to develop separation and recycling methods which leave out or extract POPs from a waste stream.

As the issue raised by Finland is, to a large extent, beyond the scope of this study, it is proposed that the issue is analysed and discussed further as part of future amendments to the POP regulation.

Remarks Danish Environmental Protection Agency (EPA)

The Danish EPA states, that the classification of POP-waste as hazardous waste should be amended, so that POP waste is classified as hazardous waste when the concentration of POPs in waste is greater or the same as the Annex IV limit value and not at concentrations greater than the Annex IV limit value, as is the case today (cf. Commission Decision of 18 December 2014 amending Decision 2000/532/EC on the list of waste). The reason being, that it will align the classification as hazardous waste with the requirement for destruction [Danish EPA Sub. 2018].

With regard to The Danish EPA´s submission, it should be noted that indeed Article 7(4)(a) determines that waste containing or contaminated by any substance listed in Annex IV may be

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17 https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52018XC0409(01)&from=EN
otherwise disposed of or recovered in accordance with the relevant Community legislation, provided that the content of the listed substances in the waste is below the concentration limits to be specified in Annex IV. Conversely Commission decision 2014/955/EU (List of Waste) determines that waste containing specified POPs exceeding the concentration limits indicated in Annex IV to Regulation (EC) No 850/2004 of the European Parliament and of the Council (1) shall be classified as hazardous. The combination of both provisions will lead to a legal situation under which:

- waste streams with a POP content equal to exactly the limit value established for that specific POP under Annex IV POP regulation will not be classified as hazardous waste; while
- this stream is not subject to the requirement of being disposed of or recovered, without undue delay and in accordance with Annex V, part 1 in such a way as to ensure that the persistent organic pollutant content is destroyed or irreversibly transformed (Article 7(2) POP regulation).

Notwithstanding the question how commonly such situations might arise, it would be legally appropriate to align the List of Waste provision with the POP regulation, as the latter is the legal document more specifically providing the legal framework for POPs.

Remarks French Ministry for an ecological and solidary transition

The French Ministry for an ecological and solidary transition referred to an issue raised by the European Electronics Recyclers Association (EERA) at the stakeholder workshop. The issue concerned the link between the POPs regulation and Basel Convention on the control of transboundary movements of hazardous wastes and their disposal (the Basel Convention), and therefore the moment when waste can cross borders [FR Ministry for an Ecological and Solidary Transition Sub. 2018]. The French Ministry for an ecological and solidary transition submitted that the question could be asked in the following way: can we sort POPs waste during the recycling process, after the export of waste when recycling is done in another country, or is it only allowed to export the waste after removing the contaminated waste by a substance regulated by the Regulation.

The French Ministry submitted that this point seems worthy of further reflection, since the Basel Convention allows the competent authority of a country not to permit the export of waste if their conditions of treatment will not be satisfactory. Moreover, as soon as the „ban amendment” comes into effect, the Convention will prevent the export of hazardous waste to developing countries. The EU has already implemented this export ban. To enable the exporting country to control that POPs wastes are treated in accordance with the POPs Regulation and to prevent waste containing POP compounds to be sent to developing countries that do not have the ability to deal with them, it would seem useful to consider whether the Basel Convention allows controlling wastes containing POPs even when they do not meet the hazard properties described in Annex III of the Convention.

With regard to the French Ministry’s submission, it should firstly be noted that EU regulation (EC) No 1013/2006 on shipments of waste (WSR) provides the legal framework for shipments of waste from the EU to third countries. The WSR implements the Basel Convention into the EU’s legal acquis. As such, any issues concerning the export of waste containing POPs to third countries should be primarily analysed within the light of the WSR.

In addition, it should be noted that waste which requires separation in order to remove POPs before recycling is likely to contain a concentration of such POPs above the limit values laid down in Annex IV to the POP regulation (i.e. POP waste). Article 7(2) POP regulation determines that waste consisting of, containing or contaminated by any substance listed in Annex IV shall be disposed of or recovered, without undue delay and in accordance with Annex V part 1 to the Regulation in such a way as to ensure that the persistent organic pollutant content is destroyed or...
irreversibly transformed so that the remaining waste and releases do not exhibit the characteristics of persistent organic pollutants.

Article 34(1) WSR determines that export of waste from the EU to third countries (except for EFTA countries) destined for disposal shall be prohibited. Therefore, any POP waste which is shipped to non-EU or non-EFTA countries for the purpose of disposal operations D9 (Physico-chemical treatment) and D10 (incineration on land) in accordance with part 1 of Annex V to the POP regulation should be prohibited per se. The POP regulation determines that a pre-treatment operation prior to destruction or irreversible transformation pursuant to part 1 of Annex V may be performed, provided that a substance listed in Annex IV that is isolated from the waste during the pre-treatment is subsequently disposed of in accordance with this Part of this Annex. Nevertheless, it seems unlikely that the WSR would allow the export of POP waste to a third country for pre-treatment with the aim of subsequent destruction, as the aim of such a shipment still includes the disposal of the POP-containing fraction of the waste. An elaborate discussion on this issue goes beyond the scope of this study and requires more elaborate legal analysis, taking into account the structure and system of both the POP regulation and the WSR.

With regard to recovery (which includes recycling), it is relevant to note that such operations are only possible if, in accordance with Article 7(4)(a) POP regulation, the content of the POPs in the waste is below the concentration limits specified in Annex IV. In such a case, separation of the POP-containing fractions of the waste stream in question might not be necessary. However, it must be noted that Article 36(1) WSR determines that exports from the EU of specified types of waste destined for recovery (which includes recycling) in countries to which the Organisation for Economic Co-operation and Development (OECD) Decision does not apply are prohibited. The Article lists categories subject to the export ban which could be of relevance to the issue raised by the French Ministry. An elaborate discussion on this issue goes beyond the scope of this study.

### Conclusion from the legal analysis

The overall conclusion of the project team is that a few EU regulations which are linked to the POP Regulation provisions on POP waste are concretely affected by the results of the last COP of the Stockholm Convention. The table below provides an overview of the regulations which are linked to the POP Regulation provisions on POP waste and an indication of required or possible amendments in the light of the results of the last COP of the Stockholm Convention. Indications of amendments can concern

- required amendments to ensure correct coverage by relevant EU legislation of new and candidate POPs; or
- possibilities for amendments which could enhance or strengthen the protective function of relevant EU regulation with regard to the new and candidate POPs.

According to the analysis carried out, proposed measures are briefly summarised in Table 1.

<table>
<thead>
<tr>
<th>Regulation Covered</th>
<th>Partly covered</th>
<th>Not covered</th>
<th>Reference to POP</th>
<th>Proposed measure</th>
</tr>
</thead>
</table>

18 This is also the case if the POP waste is disposed of in accordance with Article 7(4)(b) POP regulation

19 The following Decisions of the COP of the Stockholm Convention have been analyzed: UNEP/POPS/COP.8/SC-8/4; UNEP/POPS/COP.8/SC-8/7; UNEP/POPS/COP.8/SC-8/10; UNEP/POPS/COP.8/SC-8/11; UNEP/POPS/COP.8/SC-8/12
<table>
<thead>
<tr>
<th>Regulation</th>
<th>Covered</th>
<th>Partially covered</th>
<th>Not covered</th>
<th>Reference to POP</th>
<th>Proposed measure</th>
</tr>
</thead>
<tbody>
<tr>
<td>POP Regulation No 850/2004</td>
<td>x</td>
<td></td>
<td></td>
<td>All new POPs to be listed under Annexes</td>
<td>None</td>
</tr>
<tr>
<td>REACH Regulation (EC) No 1907/2006</td>
<td></td>
<td></td>
<td>DecaBDE, PFOA</td>
<td>Required</td>
<td>The listing of DecaBDE in the Annexes of the Stockholm convention and the subsequent implementation in the corresponding Annexes of the POP Regulation, will lead to the removal of the restrictions regarding DecaBDE from Annex XVII of REACH. As such, the restrictions and their exemptions will become obsolete and may be replaced by prohibitions (Annex I) or restrictions (Annex II) of the POP Regulation.</td>
</tr>
<tr>
<td>CLP- Regulation (EC) No 1272/2008</td>
<td>None</td>
<td></td>
<td></td>
<td>None</td>
<td>None</td>
</tr>
<tr>
<td>RoHS Directive 2011/65/EU</td>
<td></td>
<td></td>
<td>Indirect: DecaBDE (under PBDE)</td>
<td>Possible</td>
<td>RoHS is affected by the COP decisions of the Stockholm convention affecting POP waste. Article 7 of the POP Regulation already prohibits the recycling of POPs waste. However, depending on the UTC for decaBDE, the exemption which exists for PBDE under 0.1% by weight in homogeneous materials may not be fully applicable to DecaBDE. This is because the Stockholm convention does not explicitly allow for a recycling exemption for decaBDE. This could be reflected in the wording of Annex II of the RoHS Directive.</td>
</tr>
<tr>
<td>WEEE Directive 2012/19/EU</td>
<td></td>
<td>Indirect: DecaBDE</td>
<td></td>
<td>Possible</td>
<td>The requirement of removing components containing a certain substance from WEEE could be applied with regard to other POPs..</td>
</tr>
<tr>
<td>ELV Directive</td>
<td></td>
<td></td>
<td>None</td>
<td>Possible</td>
<td></td>
</tr>
</tbody>
</table>

PFOA is also listed under Annex XVII of REACH. As such, removal of its listing may also become necessary upon its listing as a POP under the Stockholm Convention and subsequently the POP Regulation.
<table>
<thead>
<tr>
<th>Regulation</th>
<th>Covered</th>
<th>Partly covered</th>
<th>Not covered</th>
<th>Reference to POP</th>
<th>Proposed measure</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000/53/EC</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Explicit treatment requirement on removal of components containing POPs (or designated POPs) could be added to point 3 of Annex I</td>
</tr>
<tr>
<td>List of Waste Commission Decision 2014/955/EU</td>
<td>x</td>
<td></td>
<td></td>
<td>None</td>
<td>Possible</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>A choice exists to add any of the new or candidate POPs to the List of Waste, in order to enable the classification of a waste stream containing that specific POP as hazardous.</td>
</tr>
<tr>
<td>Sewage Sludge Directive 86/278/EEC</td>
<td></td>
<td></td>
<td></td>
<td>None</td>
<td>Possible</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Create a link in the sewage sludge directive with the annexes of the POP regulation. An alternative form could be entries in annex I A of the Sewage Sludge directive on the POPs which are most commonly found in Sewage Sludge.</td>
</tr>
</tbody>
</table>
4. MATERIAL FLOWS

This section focuses on the compilation and analysis of relevant data for the establishment of substance and mass flows for the newly listed POPs and the candidate POPs for the most relevant waste streams. The term “substance” refers to the corresponding POP contained in the “mass” of a specific waste stream. Most relevant available information to develop the material flows is compiled according to the following structure for each POP substance or substance group:

- Production, use and trade
- Chemical characteristics
- Legal background
- Occurrence in products, wastes and recyclates
- Selection of most relevant waste streams in the EU

Further, for each of the selected waste streams the following information is compiled:

- Background information
- Processes, inputs and outputs
- Concentrations in products, waste and recyclates
- Activity data
- Waste management
- Substance and mass flow

4.1 Decabromodiphenyl ether (decaBDE)

DecaBDE belongs to the group of polybrominated diphenyl ethers (PBDEs). PBDEs are additive flame retardants [FRs] used in polymer resins and plastics and, to a lesser extent, adhesives, sealants and coatings to slow the ignition and spread of fire. Besides that, decaBDE has been used in textiles. PBDEs contain an identical base chemical structure but differ in the number of attached bromine atoms (ranging from 1 to 10). According to the number of chlorine atoms they are allocated to BDE congener groups, such as e.g. tetraBDEs, pentaBDEs, hexaBDEs, etc. PBDEs are or were used in three commercial mixtures: pentabromodiphenyl ether (hereafter referred to as c-pentaBDE), octabromodiphenyl ether (hereafter referred to as c-octaBDE) and decaBDE (hereafter referred to as decaBDE or c-decaBDE\(^\text{21}\)). Table 2 provides information on typical compositions of the PBDE commercial mixtures [Environment Canada 2013].

<table>
<thead>
<tr>
<th>Commercial mixtures</th>
<th>PBDE Groups</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>tetraBDE</td>
</tr>
<tr>
<td></td>
<td>BDE-47, etc.</td>
</tr>
<tr>
<td>c-pentaBDE</td>
<td>24-38</td>
</tr>
</tbody>
</table>

\(^{21}\) Depending on the information source, it is not always possible to unequivocally differentiate whether the information relates to the congener decaBDE (BDE-209) or the commercial mixture. In the present document, it is however specified in the text as far as possible or clear from the context whether the term “decaBDE” refers to the commercial mixture c-decaBDE or the individual congener decaBDE (BDE-209).
Release of PBDEs into the environment may occur during manufacture, processing, transportation, use, handling, storage or containment, product usage and disposal of the substance or products containing the substance [Environment Canada 2013].

4.1.1 PRODUCTION, USE AND TRADE

Production

The total production of c-decaBDE in the period 1970-2005 ranged between 1.1 and 1.25 mio. t [UNEP/POPS/POPRC.10/10/Add.2 2014; UNEP/POPS/POPRC.11/10/Add.1 2015]. C-decaBDE production accounts for approximately 75 % of PBDEs worldwide production and it is only produced in a few countries (China, India, Japan) [UNEP/POPS/POPRC.10/3 2014]. According to BSEF, in Europe, there is currently no production of decaBDE [BSEF Sub. 2018].

Use

Global c-decaBDE consumption reached its peak in the early 2000s. Many countries and regions have already adopted restrictions or voluntary programs to end the use of c-decaBDE [UNEP/POPS/POPRC.10/3 2014].

[Earnshaw et al. 2013] gathered data on the European decaBDE consumption and used different interpolations to construct four consumption scenarios (see Figure 3).

<table>
<thead>
<tr>
<th></th>
<th>0.5</th>
<th>12</th>
<th>45</th>
<th>33</th>
<th>10</th>
<th>0.7</th>
</tr>
</thead>
<tbody>
<tr>
<td>c-octaBDE</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>c-decaBDE</td>
<td></td>
<td></td>
<td></td>
<td>Trace</td>
<td></td>
<td>97-98</td>
</tr>
</tbody>
</table>

Figure 3: Estimated historical European decaBDE consumption under different scenarios [Earnshaw et al. 2013]

While the consumption volumes are zero before 1970 in all scenarios, the interpolations differ especially for the time from 1970 to 1990. Scenarios C0, C1 and C2 basically used available data points and based different interpolations on them, C3 is derived from global bromine production. It
uses world bromine production data, % BFR production, % individual PBDE production and c-DecaBDE % share of PBDEs to estimate world c-DecaBDE production. For the EU market, scenario C3 uses a constant share of 14% of the global market.

Scenario C2 is based on BSEF and Voluntary Emissions Control Action Programme (VECAP) European consumption data post-1991 and can be considered comparatively realistic. The scenario uses interpolation for missing data. Scenario C2 uses linear interpolation from 0 tonnes/year in 1969 to 4,200 tonnes/year in 1991, calculated from the BSEF global figure for 1991 (30,000 tonnes/year) and assuming European consumption was equal to a constant 14% of the global market. Total European consumption for 1970–2010 is estimated at 200,000 tonnes for scenario C2. According to scenario C2, decaBDE consumption in the EU is increasing from 1970 (387 tonnes) to 1992 (8,900 tonnes). For the time from 1992 to 2010 the consumption varies in the range from 6,000 to 9,000 tonnes per year (see [Earnshaw et al. 2013]).

The main uses of c-decaBDE were in textiles and plastics which cover various products and commercial sectors. Textile applications contribute to around 10% of the total use. The remaining 90% of c-decaBDE was used in plastics, whereby the focus is on plastics used for EEE (housings of computers and TVs, wires and cables) [UNEP/CHW.13/INF/14 2017]. The relevance of the decaBDE use for different applications in EEE, vehicles and construction products shows regional variation and changes over time. The share of c-decaBDE used for plastics in Europe was 81.7% in 2002 and decreased to 66% in 2010 and below 50% in 2014 [VECAP 2014].

C-decaBDE is a general-purpose additive FR, that is physically combined with the material in which it is used to inhibit the ignition and slow the rate at which flames spread. It is compatible with a wide variety of materials. It is used on its own or in combination with other FRs [UNEP/POPS/POPRC.10/10/Add.2 2014].

Trade

Over the last decade, a decreasing trend of c-decaBDE imports into the EU can be observed. In 2007 11,484 t of c-decaBDE were imported in the EU27. In 2012, only 4,133 t c-decaBDE were imported into the EU27 [UNEP/CHW.13/INF/14 2017].

Table 3 provides an overview of selected production and consumption levels of c-decaBDE [UNEP/CHW.13/INF/14 2017].

<table>
<thead>
<tr>
<th>Year</th>
<th>Comment</th>
</tr>
</thead>
<tbody>
<tr>
<td>1994</td>
<td>Worldwide consumption</td>
</tr>
<tr>
<td>2001</td>
<td>Worldwide consumption</td>
</tr>
<tr>
<td>2001</td>
<td>Market demand of c-decaBDE in 2001 in America (24,500 t), Europe (7,600 t) and Asia (24,050 t).</td>
</tr>
<tr>
<td>2007</td>
<td>Imports into the EU27</td>
</tr>
<tr>
<td>2009</td>
<td>Entry into the Canadian market in 2009 from all sources (substance and in imported products); some PBDEs are exported from Canada in finished products.</td>
</tr>
<tr>
<td>2011</td>
<td>Worldwide consumption</td>
</tr>
<tr>
<td>2012</td>
<td>Imports into the EU27 (decreasing trend from 2007)</td>
</tr>
<tr>
<td>2012</td>
<td>National production volume of the USA including domestic production and import</td>
</tr>
<tr>
<td>2013</td>
<td>Japanese consumption</td>
</tr>
<tr>
<td>2013</td>
<td>c-decaBDE was manufactured and imported less than 1,000 tons in Japan (production and import still underway, 2014).</td>
</tr>
<tr>
<td>2014</td>
<td>Volume sold in Europe by the member companies of the VECAP and the European Flame Retardants Association (EFRA) in 2014</td>
</tr>
</tbody>
</table>
Imports of products or waste containing decaBDE into the EU is possible. DecaBDE may be present in various imported articles e.g. cars and other means of transportation [Danish EPA Sub. 2018].

### 4.1.2 CHEMICAL CHARACTERISTICS

C-decaBDE consists predominantly of the congener BDE-209 (≥ 97%), with low levels of other PBDE congeners (see Table 2) [UNEP/POPS/POPRC.11/10/Add.1 2015]. Below an overview of the generic chemical formula and the CAS-number is given. Table 4 highlights the synonyms and previous trade-names of decaBDE.

Chemical formula (generic): $\text{C}_{12}\text{Br}_{10}\text{O}$

CAS registry numbers (mixture): 1163-19-5 (109945-70-2, 145538-74-5 and 1201677-32-8 were used in the past)

<table>
<thead>
<tr>
<th>Name</th>
<th>decaBDE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Synonyms</td>
<td>decabromodiphenyl ether, decabromodiphenyl oxide, bis(pentabromophenyl) oxide, decabromo biphenyl oxide, decabromo phenoxybenzene, benzene 1,1’ oxybis-, decabromo derivative, decaBDE, DBDPE2, DBBE, DDBO, DBDO.</td>
</tr>
<tr>
<td>Previous trade names</td>
<td>DE-83R, DE-83, Bromkal 82-ODE, Bromkal 70-5, Saytex 102 E, FR1210, Flamecut 110R, FR-300-BA (no longer commercially available).</td>
</tr>
</tbody>
</table>

**Figure 4:** Chemical structure decaBDE (BDE-209)

**Table 5:** Chemical-physical properties decaBDE and its main constituent BDE-209 [UNEP/POPS/POPRC.11/10/Add.1 2015]

<table>
<thead>
<tr>
<th>Chemical-physical properties decaBDE and its main constituent BDE-209</th>
</tr>
</thead>
<tbody>
<tr>
<td>Physical State at 20°C and 101.3 kPa</td>
</tr>
<tr>
<td>Boiling point</td>
</tr>
<tr>
<td>Vapour pressure (at 100°C)</td>
</tr>
<tr>
<td>Log Kow</td>
</tr>
<tr>
<td>Water Solubility</td>
</tr>
</tbody>
</table>

Detailed information on the chemical characterization of BDE-209 is available in the Risk Profile prepared under the Stockholm Convention (see [UNEP/POPS/POPRC.10/10/Add.2 2014]).

### 4.1.3 LEGAL BACKGROUND

**Stockholm Convention**

In 2017 it was decided to list decaBDE (BDE-209) present in c-decaBDE in Annex A to the Stockholm Convention with specific exemptions [UNEP/POPS/COP.8/SC-8/10 2017].
Annex A, part I of the Stockholm Convention establishes the specific exemptions for the production and use of decaBDE as follows:

“Production: As allowed for the parties listed in the Register”

“Use: In accordance with Part IX of this Annex:

- Parts for use in vehicles specified in paragraph 2 of Part IX of this Annex
- Aircraft for which type approval has been applied for before December 2018 and has been received before December 2022 and spare parts for those aircraft
- Textile products that require anti-flammable characteristics, excluding clothing and toys
- Additives in plastic housings and parts used for heating home appliances, irons, fans, immersion heaters that contain or are in direct contact with electrical parts or are required to comply with fire retardancy standards, at concentrations lower than 10 per cent by weight of the part
- Polyurethane foam for building insulation”

Annex A, part IX of the Stockholm Convention outlines specific requirements for decaBDE as follows:

1. The production and use of decaBDE shall be eliminated except for Parties that have notified the Secretariat of their intention to produce and/or use it in accordance with Article 4.

2. Specific exemptions for parts for use in vehicles may be available for the production and use of commercial decaBDE limited to the following:
   (a) Parts for use in legacy vehicles, defined as vehicles that have ceased mass production, and with such parts falling into one or more of the following categories:
      (i) Powertrain and under-hood applications such as battery mass wires, battery interconnection wires, mobile air-conditioning (MAC) pipes, powertrains, exhaust manifold bushings, under-hood insulation, wiring and harness under hood (engine wiring, etc.), speed sensors, hoses, fan modules and knock sensors;
      (ii) Fuel system applications such as fuel hoses, fuel tanks and fuel tanks under body;
      (iii) Pyrotechnical devices and applications affected by pyrotechnical devices such as air bag ignition cables, seat covers/fabrics (only if airbag relevant) and airbags (front and side);
      (iv) Suspension and interior applications such as trim components, acoustic material and seat belts;
   (b) Parts in vehicles specified in paragraphs 2 (a) (i)–(iv) above and those falling into one or more of the following categories:
      (i) Reinforced plastics (instrument panels and interior trim);
      (ii) Under the hood or dash (terminal/fuse blocks, higher-amperage wires and cable jacketing (spark plug wires));
      (iii) Electric and electronic equipment (battery cases and battery trays, engine control electrical connectors, components of radio disks, navigation satellite systems, global positioning systems and computer systems);
      (iv) Fabric such as rear decks, upholstery, headliners, automobile seats, head rests, sun visors, trim panels, carpets.

3. The specific exemptions for parts specified in paragraph 2 (a) above shall expire at the end of the service life of legacy vehicles or in 2036, whichever comes earlier.
4. The specific exemptions for parts specified in paragraph 2 (b) above shall expire at the end of the service life of vehicles or in 2036, whichever comes earlier.

5. The specific exemptions for spare parts for aircraft for which type approval has been applied for before December 2018 and has been received before December 2022 shall expire at the end of the service life of those aircraft.”

Under the Stockholm Convention, currently one party (Iran) has registered for a specific exemption for textile products that require anti-flammable characteristics, excluding clothing and toys.22

REACH

The production, placing on the market or use of decaBDE as a substance or as an ingredient of other substances, in mixtures and in products, is set out in Annex XVII (List of Limitations), entry 67 to Regulation (EC) No 1907/2006 of the European Parliament and of the Council (REACH), which prohibits the production and placing on the market of decaBDE as a pure substance from 2 March 2019.

For use as an ingredient of other substances, in mixtures and in products, there will be a 0.1% (weight)23 limit from 2 March 2019 with the following derogations:

(a) in the production of an aircraft before 2 March 2027.

(b) in the production of spare parts for either of the following:

   (i) an aircraft produced before 2 March 2027;


DecaBDE is identified as a PBT/ vPvB (Persistent, Bioaccumulative & Toxic/ very Persistent & very Bioaccumulative) substance under Article 57 (d) and 57 (e)24. This results in certain obligations under the REACH Regulation. Article 7.2 (notifying ECHA), Article 7.3 (appropriate instructions to the consumer), Article 31.1 (providing a safety data sheet) and Article 33 (Duty to disclose information on substances in articles).

RoHS

Directive 2011/65/EU (also known as the RoHS Directive) restricts the use of certain hazardous substances in EEE. As this directive restricts PBDEs as a substance group, it also applies to decaBDE. Member states are required to ensure that any EEE placed on the market, including cables and spare parts for repair, reuse, update of functions or performance enhancement, does not contain any of the substances listed in Annex II. The maximum concentration value tolerated for PBDEs is 0.1 % (1,000 mg/kg) by weight in homogeneous materials.

---


23 equal to 1,000 mg/kg

24 Nr. 1907/2006
According to Annex VII of Directive 2012/19/EU of the European Union and the Council on waste electrical and electronic equipment (WEEE) on the selective treatment of materials and components of EEE, referred to in Article 8 (2), plastics containing BFR have to be removed from any separately collected WEEE.

The WEEE Directive sets minimum standards for the treatment of WEEE. The current version of the voluntary standard of the European Committee for Electrotechnical Standardization (CENELEC) for the treatment of WEEE defines a concentration limit value in CENELEC TS 50625-3-1 2016. According to this standard, treatment operators shall ensure the segregation of BFR of all WEEE categories other than large appliances and refrigerators and freezers, that can contain a total BFR

- if the total concentration of Br exceeds 2,000 mg/kg, or
- if it is assumed to be above 2,000 mg/kg, or
- if there is no information on the content of BFR.

The segregated plastic fraction that contains BFR has to be treated according to relevant legislation. If the plastic fraction contains POP-BFRs above the LPCL, the POP content shall be destroyed or irreversibly transformed without undue delay in accordance with the EU POP legislation. To this end, the plastic fraction can e.g. be incinerated under controlled conditions to destroy the POP content. Another possibility would be to remove the POP content from the plastic fraction (e.g. by solvolyse) prior to its destruction of irreversible transformation. Any plastic fraction which is not segregated, as above, shall be considered as a BFR-containing fraction and shall be management accordingly. If the total bromine concentration is below 2,000 mg/kg, it is considered that the treatment operator complies with the de-pollution requirements for BFR according to the WEEE Directive.

### 4.1.4 OCCURRENCE IN PRODUCTS, WASTES AND RECYCLED PRODUCTS

**Products**

DecaBDE has been and is being used in products that are traded globally [NO EA 2015]. The applications of decaBDE can roughly be divided into two categories - in plastics and in textiles [NO EA 2015]. On a smaller scale, decaBDE has also been used in adhesives, sealants, coatings and inks. As discussed in detail in the risk profile (see [UNEP/POPS/POPRC.10/10/Add.2 2014]), the use of decaBDE within these applications varies in national and supraregional context. Typically, c-decaBDE is used in plastics/polymers at loadings of 10-15% by weight, though in some cases loadings as high as 20% or higher are possible. Available information indicates that decaBDE treated textiles contain up to 12% of decaBDE [NO EA 2016].

Table 6 summarises reported concentration levels in products from relevant sectors based on data provided by [UNEP/CHW.13/INF/14 2017]

<table>
<thead>
<tr>
<th>Sector</th>
<th>Levels</th>
<th>Remark</th>
</tr>
</thead>
<tbody>
<tr>
<td>EEE products</td>
<td>0 to 15% (0 to 150,000 mg/kg)</td>
<td>In EEE c-decaBDE is typically used in concentrations ranging from 10 to 15% (up to 20%). A number of studies have measured c-decaBDE/ BDE-209 levels in products and materials commonly used in EEE such as plastics used for EEE, computer casings, TV casings, PC boards, rice cookers, plastic parts from small EEE, white goods, small domestic appliances, plastics from cathode ray tube screens. Based on a literature review, Table 3 in [Norway 2015] provides an overview of reported concentrations of decaBDE in EE and the specific information sources (see Annex to the present document).</td>
</tr>
</tbody>
</table>
According to data submitted in the informal stakeholder consultation (see sections 10.6 and of the Annex), decaBDE concentrations in articles in use range from 0.2 – 300,000 mg/kg decaBDE.

Wastes

Specific information on the occurrence of POP-BDEs, including decaBDE, in wastes, is contained in section B.4 of the Revised technical guidelines on the environmentally sound management of wastes consisting of, containing or contaminated with hexabromodiphenyl ether and heptabromodiphenyl ether, or tetrabromodiphenyl ether and pentabromodiphenyl ether or decaBDE.25

Wastes may contain variable concentrations of decaBDE, depending on the quantities in which decaBDE was originally present in specific products and the quantities released during product use and end-of-life management.

To assess how relevant the occurrence of decaBDE in wastes in the EU is for waste it is relevant, which decaBDE-containing products are on the market and in which quantities. Further, it is relevant in what concentration they contain decaBDE and when these products will become waste [NO EA 2015].

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Sections 10.7 and 10.8 of the Annex give an overview of the occurrence and concentrations of decaBDE in waste provided by stakeholders and found in literature during the informal stakeholder consultation and found in literature. DecaBDE concentrations found in waste range from 0.021 – 200,000 mg/kg.

The product lifetime is of crucial importance when estimating the presence of decaBDE in products, wastes and recyclates. After the product’s lifetime, the products are disposed of as waste and cause contamination of waste streams and possibly recyclates. The average product lifetime depends on the product itself. Average product lifetimes for most relevant product categories related to the use of decaBDE are indicated in Table 7.

<table>
<thead>
<tr>
<th>Application</th>
<th>EEE products</th>
<th>Cars</th>
<th>Construction material</th>
<th>Textiles and furniture</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average product lifetimes (in years; range in brackets)</td>
<td>9 (+/-5)</td>
<td>17 - 20</td>
<td>50 (+/-25)</td>
<td>10 (+/- 3)</td>
</tr>
</tbody>
</table>

Specific and reliable estimates concerning cumulated quantities of decaBDE becoming available in the technosphere in specific waste streams are not feasible due to limited quantitative information on consumption history in the EU. However, material flows associated with main applications can be summarized qualitatively:

- Declining decaBDE concentrations are expected in WEEE plastics. It can be estimated that currently more than 600 tonnes per year of decaBDE are contained in plastics resulting from separately collected WEEE in the EU. A majority of decaBDE content is constantly being destroyed due to established end-of-life practices (incineration of high Br plastic fraction from WEEE). However, a relevant share is recycled and will enter the waste streams after subsequent product cycles. Based on the finding that consumption of decaBDE declined from 2010 onwards, it can be expected that decaBDE levels will start to decline in the late 2010s and that they will continue to decline throughout the 2020s. Only low levels of decaBDE in WEEE are expected after 2030 in the EU.

- DecaBDE quantities in car manufacturing are decreasing. EU manufacturers will phase out the use by mid-2018 at the latest. The current freight of decaBDE in shredder residues from cars is around 100 tonnes per year. A significant share of decaBDE is not destroyed during current waste treatment and is partly recycled and will enter the waste streams after subsequent product cycles. Considering the typical lifetimes of cars it can be expected that decaBDE levels in ELV waste streams will particularly be relevant until the end of the 2030s. It is expected that they will significantly decline afterward.

- Considering a typical lifetime for buildings of 50 +/- years, decaBDE will start to occur in construction and demolition (C&D) waste around 2020 and will reach a plateau from 2040 to 2060 when possibly approximately 1,500 tonnes of decaBDE may enter the C&D waste stream each year. Recycling rates for plastic-based building products containing decaBDE currently remain low. Yet, future recycling rates will differ between EU MS and are highly speculative. Consequently, treatment practices for plastic-based building materials deserve attention until 2060 and beyond.

- Only a limited share of decaBDE was used in textiles (~10% globally and <20% at EU level). A relevant share of decaBDE use for textiles was in the transport sector and is e.g. covered by the ELV waste flow. DecaBDE used in textiles for cars will reach the waste phase via ELVs and will thus continue to occur until the end of the 2030s (see above). The use of decaBDE in other textile and furniture applications was already largely phased out.
around 2014. Owing to the designation of decaBDE as a SVHC, most textile finishers already had switched to alternatives at that time. Considering the typical lifetime of textiles and furniture it can be therefore be expected that the occurrence of decaBDE in textile waste will decline throughout the 2020s. After 2030 only insignificant amounts of decaBDE in textiles and furniture are expected.

In summary, the decaBDE content in future waste streams from WEEE, ELVs and from construction & demolition may require special attention throughout the 2020s (WEEE), until the end of the 2030s (ELVs) and until 2060 and beyond (C&D waste).

Recyclates

Despite the phase out of the use of decaBDE in the EU, it can still occur in recyclates and new products due to a number of reasons:

- The UTC for PBDEs in Annex I EU POP Regulation (UTC for decaBDE yet to be defined)
- Imports of decaBDE-containing products from outside the EU
- Recycling of products containing decaBDE below the LPCL according to Annex IV EU POP Regulation (LPCL yet to be defined)
- Unintentional recycling of products containing decaBDE above the LPCL according to Annex IV EU POP Regulation (LPCL yet to be defined)

Plastics from WEEE and ELVs are often recycled at the end of the use phase. For the Netherlands, it is estimated that, in the WEEE sector, 22% of the POP-BDE is likely to end up in recycled plastics. In the automotive sector, 14% is expected to end up in recycled plastics, while an additional 19% is expected to end up in second-hand parts (reuse) (see [IVM 2013]; Leslie et al. 2016]).

It follows that products made from recycled plastics can contain decaBDE. In particularly where manufacturers are not selective about the purity of the recycled plastic pellets they purchase decaBDE concentrations can be expected [NO EA 2016].

All carpeting and insulation materials made from recyclates studied in [Leslie et al. 2016] contained either BDE-209 (up to 0.8 mg/kg) or both BDE-209 and POP-BDEs (up to 0.4 mg/kg). A quarter of the toys sampled contained POP-BDEs (up to 44 mg/kg) and BDE209 (up to 800 mg/kg) [Leslie et al. 2016].

According to the EERA, WEEE plastics containing decaBDE has a recycling rate of 60%. New articles produced from recycled material include electronics, automotive parts and office equipment [EERA Sub. 2018a]. In some EU countries WEEE plastic containing PBDE including decaBDE are not recycled (e.g. Sweden) [SE EPA Sub. 2018].

Detected concentration values of recycled products range between 0 and 9,225 mg/kg, with the highest value found in a car launcher (toy) [IPEN & Arnika Sub. 2018a]. In a study published by [Straková et al. 2018a], 92% of the tested samples of toys, hair accessories and kitchen utensils contained decaBDE.

An overview of reported concentrations of decaBDE in articles made from recycle in a number of different publications is given in the Annex in section 10.6.

4.1.5 SELECTION OF MOST RELEVANT WASTE STREAMS IN THE EU

The occurrence of decaBDE in waste streams is directly connected to its historic and/or current use and use levels of decaBDE in articles. For "Substances of Very High Concern" (SVHCs), detailed data on the use of decaBDE were collected in the REACH context on identified uses and respective article categories. The contained data is based both on the notifications that companies have
submitted to the European Chemicals Agency (ECHA) as well as on the information contained in the registration dossiers [ECHA 2017a]. The covered article categories are:

- Fabrics, textiles and apparel
- Leather articles
- Machinery, mechanical appliances, electrical/ electronic articles
- Metal articles
- Paper articles
- Plastic articles
- Rubber articles
- Vehicles
- Wood articles [ECHA 2017a].

As outlined above, the main uses of decaBDE were in textiles and plastics which cover various products and commercial sectors. Textile applications contribute to around 10 % of the total use globally. The remaining 90 % of decaBDE was used in plastics, whereby the focus is on plastics used for EEE (housings of computers and TVs, wires and cables)

To get an overview of the amounts and concentrations of decaBDE in products, their wastes and in recycled materials, the subsequent sections focus on the waste streams WEEE, ELV, textiles and furniture, C&D waste and other waste streams. These waste streams were identified as major uses as part of the stakeholder consultation and from previous studies (see e.g. [UNEP/CHW.13/INF/14 2017]).

### 4.1.6 WEEE PLASTICS

#### 4.1.6.1 Background information

Plastics used in EEE appliances include plastics not containing c-decaBDE. In accordance with the typical c-decaBDE loadings, the levels in EEE plastics range from zero to 15%. Usually, c-decaBDE is not intentionally used at concentrations below 2% [NO EA 2015].

According to EERA, based on waste flows in WEEE collected in Switzerland, deca-BDE, octaBDE and pentaBDE account for 22%, 7% and 0% respectively of the BFRs in WEEE plastics in 2011. This number is continuously decreasing [EERA UTC Opinion 2018]. In the average mix of solid plastics wastes coming out of the WEEE recycling processes, typically 5 – 10 % consist of plastics with FRs [EERA UTC Opinion 2018]. According to the "Bromine Science and Environmental" forum, decaBDE was used in the following products of the EEE sector [BSEF Factsheet 2006]:

- Housing and internal components of televisions
- Mobile phones and fax machines
- Audio and video devices
- Remote controls
- Communication cables
- Plastic film capacitors
- Building cables
- Cables, e.g. heat shrink tubing
- Connections in electrical appliances
- Circuit breakers
- Coils of bobbins
- Printer and photocopier components - toner cartridges and connectors
- Scanner components.
The EU RoHS Directive 2011/65/EU limits the use of PBDEs (including decaBDE) in EEE. PBDEs in quantities of more than 0.1% (1,000 mg/kg) by weight in homogenous material are not permitted. The RoHS Directive covers the following eleven categories of EEE:

- Large household appliances (LHA).
- Small household appliances (SHA).
- IT and telecommunications equipment.
- Consumer equipment.
- Lighting equipment.
- Electrical and electronic tools.
- Toys, leisure and sports equipment.
- Medical devices.
- Monitoring and control instruments including industrial monitoring and control instruments.
- Automatic dispensers.
- Other EEE not covered by any of the categories above.

4.1.6.2 Processes, inputs and outputs

DecaBDE is incorporated into the plastic either prior to, during, or more frequently following polymerisation as additive type FR. Additive FRs are only physically bonded to the polymer as monomer molecules, and if they are exposed to the environment there will be a possible impact depending on the original characteristics of this particular molecule [Kemi 2005].

![Diagram showing inputs and outputs of decaBDE in the production of plastics for WEEE]

4.1.6.3 Concentration in products, wastes and recyclates

Products

In EEE c-decaBDE is typically used in plastic polymers in concentrations ranging from 10 to 15% (up to 20%) [UNEP/CHW.13/INF/14 2017].

The EEE categories which contain most frequently BFRs are, according to [Hennebert and Filella 2017]:

- LHA
- Toys, leisure and sports equipment
- Electrical and electronic tools
- SHA
- Lighting equipment

In the Annex in section 10.7 data on occurrence in products is provided.
Waste

At the end of their service life, EEE appliances contribute to WEEE. Average concentrations of EEE products and of WEEE range typically between zero and 0.3% with typical levels in the higher end of the range in CRT TVs and monitors and office equipment and levels in the lower range for other EEE products or WEEE (results from Europe) [NO EA 2015].

DecaBDE occurs in shredder residues from WEEE, ELVs and other waste materials which are often treated together in the same shredder plants. Several sources provide information on occurrence levels of decaBDE and other PBDEs in different types of shredder residues (SR) from WEEE (WEEE SR), automotive applications, and mixed input from WEEE and ELV and other input sources (mixed SR). DecaBDE levels reported in automotive shredder residues (ASR), mixed SR and in WEEE SR range from zero to 13.8% (or 138,000 mg/kg) [NO EA 2015].

According to a recent study from [Swerea 2018], a total of 25 samples were analysed from various European countries for decaBDE, but also other FRs (TBBPA, HBCDD, DBDPE). Before analysis, the samples were not subject to a selection process such as density separation or similar. The highest concentration was found in a mixture of polystyrene (PS)/ Acrylonitrile-Butadiene-Styrene (ABS) plastic (900 mg/kg), which was originally used in TV housings as measured by GC-MS analysis.

The concentration of decaBDE in a sample sorted by density separation before analyzation was found in a sample from Sweden (280 mg/kg). Again, this was a mixture of PS/ABS plastic derived from small household appliances [Swerea 2018].

A study published in 2017 from Switzerland comparing concentrations from 2011, found that concentrations of decaBDE in WEEE show a decreasing trend compared to samples taken in 2003. This is attributed to the regulatory measures in place [BAFU 2017].

In summary, the various literature sources for decaBDE in WEEE plastics demonstrate levels in products that are in many cases higher than the concentration limit of 1,000 mg/kg defined in the RoHS Directive.

The concentration levels described in the literature for decaBDE in WEEE plastics are very inhomogeneous, with a concentration range from a few to a hundred thousand mg/kg. Nevertheless, in the context of this project, a concentration is estimated through which estimations of substance and mass flows for the EU can be established.

We compiled literature values data from various sources and that way established a data set, which is used as basis for further calculations. To estimate the level of decaBDE in WEEE plastics, known concentrations from literature and the formal stakeholder consultation were used to calculate a median decaBDE concentration. To this end data from literature [UBA 2017a], [Hennebert and Filella 2017], [UNEP/CHW.13/INF/14 2017], [NO EA 2016], [Swerea 2018], [UBA 2018b], [Sharkey et al. 2018], [NO EA 2018] and from the informal stakeholder consultation ([AT Ministry of Sustain. and Tourism Sub. 2018a; DEFRA Sub. 2018; Plastic Recyc. Europe Sub. 2018; UBA Sub. 2018; ACEA Sub. 2018]) were used. The total data set consists of 132 literature values26. It is important to note in this context that in many sources the given concentrations were described in ranges. In accordance with the precautionary principle, in these cases the highest concentration was included in the further calculation. Only data from the European economic area was considered. A distribution of the individual concentration values was established with the following mg/kg ranges: ≥100,000, ≥10,000, ≥1,000, ≥100, ≥10, ≥1, ≥0.1. The majority of the

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26 Reported decaBDE concentrations in Europe within a given waste stream from several sources are used as a baseline to derive the median value of concentration. The median is used as it is less affected by outliers and skewed data.
concentrations mentioned in literature and from the stakeholder consultation are in the range from 100 to 1,000 mg/kg. Therefore, a realistic median concentration value for the material flow should be within this range.

All values were divided into the fractions: (1) 35 values had a decaBDE concentration >2,000 mg/kg and are therefore in the high-Br-fraction, 7 of the data points represent functional concentrations and were therefore dismissed for the further calculations. (2) 97 values belong to the low-Br-fraction (decaBDE <2,000 mg/kg). This division was used to calculate the median value within both fractions which are for the high-Br and the low-Br fraction 5,410 and 100 mg/kg, respectively.

[Hennebert and Filella 2017] indicate a decaBDE concentration within the high- and low-Br-fraction in WEEE collected in France. They analysed three high bromine WEEE fractions: SHA, cathode ray tubes (CRT) and flat screens with an average decaBDE concentration in the inflow of 1,673 mg/kg. The average concentration in both fractions after sorting by on-line X-ray transmission were 5,577 for the high-Br-fraction and 335 mg/kg for the low-Br-fraction. The low-Br-concentration value of the sample by [Hennebert and Filella 2017] is three times higher, which is not surprising as their samples, i.e. SHS, CRT screens and flat screens, are in general expected to contain high levels of Br. [Hennebert and Filella 2017] state, that the sorting mechanism in the sites in France need to be improved as the bromine concentration in the low-Br-fraction did not reach concentrations <2,000 mg/kg. Therefore, regarding the setting of the experiment by [Hennebert and Fillela 2017], our values (i.e. median values for the WEEE high-Br and the low-Br fraction 5,410 and 100 mg/kg respectively) are considered plausible and in accordance with the results of [Hennebert and Fillela 2017].

Distribution of decaBDE within WEEE plastics samples

As a first step of the waste management process, WEEE plastics is shredded into small plastic particles. According to [French MDD 2018], a large part of those plastic shreds has the quality to be recycled (decaBDE concentration <2,000 mg/kg) and only a few shreds are very highly brominated and contaminate the sample. Preliminary results from a research with a sample consisting of 289 pieces and an estimated Br concentration of 632 mg/kg provided by [French Ministry for an ecological and solidary transition Sub. 2018] state that in 91 % of the shreds within the sample, the Br concentration was smaller than 100 mg/kg. In 9 shreds, representing 3.1% of the sample, the Br concentration laid above 2,000 mg/kg. The authors of that research estimate that those 9 shreds are responsible for almost 90% of the total Br within the sample.

Distribution over WEEE streams

According to the European Recycling Industries’ Confederation (EuRIC), the concentration of decaBDE in WEEE plastics vary greatly depending on the types of appliances. Appliances having a potential to heat up during use usually contain higher concentrations [EuRIC Sub. 2018b]. This was also confirmed by a study from the German Environment Agency where six types of old electrical appliances were analysed. Two samples per device type were obtained and showed particularly high decaBDE levels for consumer electronics and small appliances which need to be heat resistant [UBA 2017b].

According to [NL Ministry of Infrastructure and the Environment Sub. 2018], the distribution of decaBDE concentrations over various waste streams should be taken into account when discussing low POP content limits. Plastics of WEEE articles such as fridges and freezers are in general less brominated.

The median decaBDE concentration of our data set for waste fridges and freezers lays at zero, with only 4 out of 13 values in the data set being larger than 5 mg/kg (n=13, see Table 8). In accordance with this result, [Hennebert 2018] emphasises this WEEE category (together with other LHAs) to have significantly lower decaBDE concentrations than other categories. In contrast, a
larger decaBDE concentration of 1,300 mg/kg (median) is calculated for the stream "screens and TVs" (including CRTs) (n=28).

Table 8: Median decaBDE concentrations distributed over WEEE streams, based on own calculations

<table>
<thead>
<tr>
<th>WEEE category</th>
<th>Median value</th>
<th>n of data points</th>
</tr>
</thead>
<tbody>
<tr>
<td>LHA</td>
<td>1,300 mg/kg</td>
<td>4</td>
</tr>
<tr>
<td>Fridges and freezers</td>
<td>0 mg/kg</td>
<td>13</td>
</tr>
<tr>
<td>Flat Screens, TV, CRT Monitors, Displays</td>
<td>3,100 mg/kg</td>
<td>27</td>
</tr>
<tr>
<td>SHA, IT &amp; Telecommunication, Consumer Electronics</td>
<td>210 mg/kg</td>
<td>49</td>
</tr>
</tbody>
</table>

These results are in accordance with information from the literature. According to the Norwegian Environmental Agency old TV and computer monitors have comparatively high decaBDE levels (up to 237,000 mg/kg) and plastic contents between 10 and 40% by weight and therefore provide two of the richest sources of plastic [NO EA 2015; NO EA 2016].

The report from the Norwegian Environment Agency provides an overview of decaBDE levels in EEE articles and waste (maximum level 237,000 mg/kg in TV CRT screens; see [NO EA 2016]). A British study detected high decaBDE concentrations in waste in televisions, IT equipment, printed circuit boards, digiboxes, large and small household appliances, coat hangers, refrigerators and PC monitors. A decaBDE concentration of 138,000 mg/kg was found in a television [NO EA 2016].

Comparatively high concentrations were also observed in mother boards (11,583 mg/kg) [Kumari et al. 2014]. The concentrations reported are further summarised in the Annex in section 10.7.

The German Environmental Agency (UBA) is currently working on a so called “negative list” which lists all WEEE components that possibly contain BFRs. Those components can then be manually removed from the waste stream and that way lower the decaBDE concentration of the shredded WEEE plastics [UBA Sub. 2018].

Recyclates

DecaBDE has been detected in recycled plastics and products made thereof (including toys) in concentrations ranging from a few mg/kg up to several thousand mg/kg. An overview is given in section 10.6 in the Annex of this document.

4.1.6.4 Activity data

Electrical appliances are manufactured and traded globally. In 2015, about 9.8 million tonnes of electrical appliances were placed on the European market [Eurostat 2018a].

A total of 3.9 million tons of WEEE was collected in the EU-28 in 2015 [Eurostat 2018a]. According to the German Environmental Agency (UBA) [UBA 2018b] the share of plastic in WEEE is in the range of 5 and 20 %, depending on the type. According to Austria the plastic content of WEEE is 33 % [AT Ministry of Sustain. and Tourism Sub. 2018a]. [EERA UTC Opinion 2018] states that 1.2 million tonnes of WEEE plastics are separately collected per year. Huisman et al. 2008 gave an estimation of 2.3 million tonnes of plastic in European WEEE in 2008 [Hennebert and Filella 2017]. For the calculations of the mass flow the quantity of 1.2 million tonnes per year of plastics from separately collected WEEE was used.

The EEE industry accounted for 6.2 % of the total European plastic demand in 2016 [PlasticsEurope 2018; WEEE Forum 2017].

The average product lifetime for EEE is considered around 9 years [BiPRO 2011].
4.1.6.5 Waste management

The typical decaBDE-containing EEE products are usually classified as electrical waste (e.g. waste codes 200123*, 200135*, 200136*) or on a small scale may be regarded as plastics in the construction waste (e.g. 17023). A certain proportion of EEE ends up in the household waste. In general, the plastic components in the construction waste and household waste are recovered energetically.

The usual treatment of plastics from the electronic waste includes the manual disassembly or the mechanical treatment in shredders. The shredding is done by large metal shredders or shredders for the particular electronic waste categories. The manual disassembly allows the manual separation of the individual plastic parts. Shredder processes are often associated with an automatic sorting process.

According to Annex VII of the Directive 2012/19/EU of the European Parliament and the Council on WEEE on the selective treatment for materials and components of EEE in accordance with Article 8 (2), plastics containing BFR have to be removed from any separately collected waste including EEE. This obligation includes, among other things, plastics containing decaBDE as BFR. Since in practice not all brominated plastics are separated, decaBDE occurs in recyclates.

The European Committee for Electrotechnical Standardization (CENELEC) for the treatment of WEEE defines a concentration limit value for bromine which is used as an indicator for the content of BFRs (see CENELEC TS 50625 2016). This concentration limit value requires the separation of plastic of all WEEE categories other than large appliances and refrigerators and freezers, that contain a total bromine concentration exceeding 2,000 mg/kg from the plastic recycling stream. If there is no information on the content of BFR or if the concentration is greater than 2,000 mg/kg, the plastic fraction has to be treated according to relevant legislation (see section 4.1.3).

DecaBDE (C_{12}B_{10}O) consists of 83.3% Br (w/w). Therefore, a Br content of 2,000 mg/kg corresponds to a decaBDE content of 2,401 mg/kg\(^{27}\), under the assumption that decaBDE is the only brominated substance within the plastic. As several different BFRs are statistically found in WEEE plastics (TBBPA (36%), decaBDE (22%), DBDPE (20%), BTBPE (9%) and octaBDE (7%) (see [BAFU 2017]), it can be concluded that usually, considering the different BFRs statistically found, the content of decaBDE and of POP-BDEs is well below 1,000 mg/kg for a Br content below 2,000 mg/kg (average content < 530 mg/kg\(^{28}\) for decaBDE and < 700 mg/kg\(^{29}\) for decaBDE and octaBDE).

Separately collected old electrical appliances are examined as to whether individual components or the appliance itself can be reused. WEEE is usually further disassembled manually or crushed in a shredder. If a manual disassembly is carried out, pure fractions can be achieved. In the case of mechanical comminution, mechanical processes are required to produce separate fractions which are subsequently fed to further material recycling. The resulting plastic fractions are thereafter treated separately [UBA 2016].

A first step in recycling WEEE plastics is the mechanical reduction of particle-size through e.g. shredding, grinding, and/or micronisation. Afterwards, the brominated material is separated from the non-brominated material through separation processes which will be described in detail below.

\(^{27}\) 83.3% decaBDE correspond to 100% Br, hence 2,000 mg/kg Br correspond to 2,401 mg/kg decaBDE

\(^{28}\) Calculation: 2410 mg/kg x 22% corresponds to ~ 530 mg/kg decaBDE

\(^{29}\) Calculation: 2410 mg/kg x (22%+7%) corresponds to ~ 700 mg/kg POP-BDEs
The non-contaminated plastic is melted and homogenised in an extruder and pellets are produced [Plastic Recyc. Europe Sub. 2018].

According to Stakeholder submission the recycling rate of the infeed mixed WEEE plastic is around 50-60% [Plastic Recyc. Europe Sub. 2018; DEFRA Sub. 2018; EuRIC Sub. 2018b]. New plastics produced from recycled materials are regrind, pellets or compounds of Polypropylene (PP)/PE ABS and PS [Plastic Recyc. Europe Sub. 2018; DEFRA Sub. 2018; EuRIC Sub. 2018b]. A wide variety of new articles is produced from recycled WEEE plastic such as vacuum cleaners, plant-trays, other electronics, and office equipment [Plastic Recyc. Europe Sub. 2018].

The recycling process of plastics in WEEE is described by [EERA UTC Opinion 2018] and illustrated in Figure 6. According to EERA, annually 1.2 million tonnes of WEEE plastics result from separately collected WEEE in the EU. Only a minor share of WEEE [EERA 2018] is manually dismantled and the plastic components are manually checked for the Br content. In case the test (described in the EN 50625-1 General treatment and depollution standard) shows a Br content >2,000 mg/kg, the plastic is sent to a specialized recycling facility for WEEE plastics. Plastics with Br content <2,000 mg/kg enter the low-Br-fraction. The major share of WEEE is not dismantled manually and is directly brought to a specialized recycling facility for WEEE plastics. There, advanced identification and separation methods are applied. Plastics containing >2,000 mg/kg enter the “high-Br-fraction”. According to [EERA UTC Opinion 2018], this concerns 8% of the total WEEE plastics. 92% are entering the “low-Br-fraction” (Br content <2,000 mg/kg). Plastics within the high-Br-fraction are typically incinerated as waste. Plastics within the low-Br-fraction are used in recycling processes. The first step is the separation of the individual polymers. The separated polymers from WEEE will then be extruded and if needed compounded. Non-recyclable fractions within the low-Br-fraction are typically incinerated as waste [EERA UTC Opinion 2018].
In order to achieve a material recycling of WEEE plastics, decaBDE-containing waste must be separated from decaBDE-free waste. Several different test methods exist to distinguish between those two fractions. They can be categorized along their underlying physical principles, namely density and optical identification. Based on these physical principles, plastics containing BFRs can be identified and separated. Methods for the identification of bromine or decaBDE within plastics (identification methods) are described in detail in chapter 6.2.1.1.

According to EERA, density separation is the widest used separation process. In general, it does not only separate the bromine material. Due to the fact, that it separates BFR-containing plastics which have a higher density, the separated material will also include other heavier plastics not containing BFRs [EERA Sub. 2018]. One specific technique based on the density separation is the float-sink-separation. According to UBA, one can distinguish between decaBDE contaminated and non-contaminated plastic waste by using this method [UBA Sub. 2018]. IPEN & Arnika agree and state that it is probably the simplest method to distinguish between BFR-containing and BFR-free waste [IPEN & Arnika Sub. 2018a].

The main technique used for separation is based on an optical method namely X-Ray Fluorescence Spectroscopy (XRF) to identify BFR-containing waste. Handheld and benchtop X-ray fluorescence (XRF) instruments are commercially available for the quantification of heavy atoms commonly present in WEEE plastic samples, such as bromine, lead, cadmium, mercury, and chromium.
According to Plastic Recyc. Europe, the analysis of a plastic sample is completed in approximately five minutes [Plastic Recyc. Europe Sub. 2018].

XRF technology allows the detection of unintentional BFRs in plastics as well as those having cross-contaminated the plastics during previous recycling processes. According to Plastic Recyclers Europe, in-line sorting of compounds cannot be obtained through this technology due to analytical technologies and physical limitations. XRF can be used on a batch to provide guarantee to customers on the average levels, and to exclude the possibility that substances are present above a certain limit value. Naturally, the lower the limit value is, the more frequent testing needs to be performed and the more frequent batches of material need to be discarded to waste operators [Plastic Recyc. Europe Sub. 2018]. According to EuRIC, the only way to distinguish contaminated and non-contaminated waste is to screen each plastic piece by XRF, which is unrealistic and impractical in an industrial setting [EuRIC Sub. 2018b].

Practical problems with XRF technology are mentioned by the Ministry of Environment of Estonia. According the them, waste management operators are counselled on reasonable and feasible methods that they can use in identifying and separating potential decaBDE contaminated waste, but the practice has shown that even when the operators uses all knowledge available to them and separate the materials, as is the case e.g. with WEEE plastic, the XRF screening method still shows high bromine levels in both materials - materials deemed non-contaminated, and materials presumably contaminated. These discoveries create high levels of uncertainty of the actual content of certain POPs in the waste. It was further stated that it is not feasible and never applicable in practice to send the entire incoming waste flow to be analysed in a laboratory [EE MoE Sub 2018].

To distinguish their samples in low-Br- and high-Br-fractions, [Hennebert and Filella 2017] used portable XRF in one study and an automatically sorting machine based on XRF technology plus a further density sorting in another study.

According to stakeholder submissions, Gas Chromatography – Mass Spectrometry (GC-MS) (DIN EN ISO 22032) is the up-to-date measurement method for identifying the presence and levels of decaBDE in waste [ARN Sub. 2018; IPEN & Arnika Sub. 2018a]. Using the GC-MS is the only identification process which distinguishes BFRs from one another. All other available technologies cannot make this distinction. This is a problem as many of the BFRs are not restricted and can still be recycled [EERA Sub. 2018].

According to the Netherlands Ministry of Infrastructure and Water Management, in practical daily operations it is not possible to distinguish between contaminated and non-contaminated waste. For the recycling of compound, either all bromine-containing articles are separated from other articles and incinerated or none. According to [NL Ministry of Infrastructure and the Environment Sub. 2018], two complicating factors have to be mentioned: (1) half of the European MS have no or by far insufficient incineration capacity and (2) in practice BFR-containing plastics are often not separated from BFR-free plastics, which both can lead to recycling of decaBDE-containing components.

In addition to the identification methods described above, the recycler can also manually separate contaminated from non-contaminated decaBDE waste. Using such an approach, the separator can work with a so called “negative lists” for WEEE components that possibly contain BFRs. These components can then be manually removed from the waste stream. The German Environmental Agency (UBA) is currently working on such a list, but no results can be published yet [UBA Sub. 2018]. According to Plastic Recyclers Europe, it is possible to identify streams where FRs are present (e.g. in hair dryers, as there is significant heat production). However, economically it is not

30 https://www.beuth.de/de/norm/din-en-iso-22032/116496877
feasible to determine in each sub-stream and for each batch whether FRs were used [Plastic Recyc. Europe Sub. 2018]. EERA in contrast to the previous statement, indicated that although it is known that some categories of WEEE are more likely to contain decaBDE than others, it is not possible to assume that a certain type of waste does not contain decaBDE [EERA Sub. 2018]. The Ministry of the Environment of Estonia expressed the associated difficulty for waste management operators to visually distinguish POP-contaminated from non-contaminated waste. Moreover, necessary information from producers on what materials might be contaminated is usually rare [EE MoE Sub. 2018].

After separation, plastics in the high-Br-fraction are usually incinerated. Incineration of decaBDE e.g. under conditions of an advanced solid waste incinerator or in cement kiln co-incineration is considered appropriate to destroy decaBDE [UNEP/CHW.12/5/Add.6/Rev.1 2015].

Plastics in the low-Br-fraction undergo a polymer separation after which 5.6% of plastics is landfilled (D1-D7, D12), further, 13.9 % are incinerated (D10+R1) and 80.4 % are recycled (R2-R11) (numbers for 2014 by [Eurostat 2018a]).

4.1.6.6 Substance and mass flow

A total of 3.9 million tons of WEEE was collected in the EU-28 in 2015 [Eurostat 2018a]. The amount of separately collected plastics from WEEE in Europe is 1.2 million tonnes annually [EERA UTC Opinion 2018]. As according to [EERA UTC Opinion 2018], 8% of the total collected WEEE plastics are related to the high-Br fraction and 92% to the low-Br fraction, we calculated the amounts of WEEE plastics within both fractions (96,000 and 1,104,000 tonnes respectively). Those amounts were multiplied with the decaBDE concentrations of the median concentration levels calculated from literature (i.e. 5,410 mg/kg for the high-Br fraction and 100 mg/kg for the low-Br fraction). This results in an estimate of the total amount of decaBDE within both fractions: 519 tonnes decaBDE are estimated within the high-Br fraction and 110 tonnes within the low-Br fraction. This corresponds to a total of 630 tonnes of decaBDE within WEEE plastics per year. Based on this estimate we can calculate the estimated average decaBDE concentration of 525 mg/kg in unsorted WEEE plastics (630 tonnes divided by 1.2 Mio tonnes). This estimated concentration level is lower compared to the average value of 860±170 mg/kg reported in [BAFU 2017] but it is in a similar range.

[Hennebert 2018] gives an overview on decaBDE concentration data from literature (25 references with a total of 736 single concentration data and 25 range concentration data, representing 639 data). He does not give a median decaBDE concentration value for the high- and low-Br-fraction but only a median of the total fraction which is 190 mg/kg. It has to be said that the median of all data sets in this context is by far smaller than the average. This statistical divergence is due to the fact that some data points are very high (up to 100,000 mg/kg). Those values heavily influence the average value of a data set and much less its median value.

For our mass and substance flow we divided all literature values into two fractions, the high-Br- and the low-Br-fraction. We calculated the median (100 and 5,410 mg/kg respectively) for both fraction which we consider as best statistical value for the compiled data set.
Figure 7: Estimated material flow of plastics in WEEE in the EU in 2015 (own illustration based on [Eurostat 2018a])
The mass flow demonstrates that the major share of decaBDE contained in WEEE plastics is destroyed (~85% or 534 t destroyed in incineration). The bulk of the decaBDE which is destroyed, is contained in the high Br fraction which is separated during WEEE recycling (~82.5% or 519 t).

However, a relevant share is not destroyed. Approximately ~15% or 95 t of decaBDE is not destroyed since it is recycled (~14% or 89 t) or landfilled (1%).

4.1.7 PLASTICS AND TEXTILES IN END OF LIFE VEHICLES (ELV)

4.1.7.1 Background information

FRs are especially found in plastics (e.g. ABS and high-impact polystyrene (HIPS)) inside vehicles and in the textiles of the seats [UBA 2018b]. The following automotive parts were identified containing decaBDE:

- Seats
- Rear deck
- Upholstery
- Visors
- Head rest
- Interior Lining
- Dash board
- Underneath the engine bonnet or the instrument panel
  - Clamp/fuse block
  - High current cables & cable casing (ignition plug)
- EEE
  - Battery compartment
  - Motor control
  - Electrical connections
  - Components in radio-, GPS und computer systems

DecaBDE can particularly be contained in ELV components such as electrical and electronic components, fuels system applications, pyrotechnical devices and applications, suspension and interior applications, reinforced plastics and fabrics as further specified in the text of the Stockholm convention in Annex A part IX Paragraph 2 (see also chapter 4.1.3., legal background)

4.1.7.2 Processes, inputs and outputs

![Diagram](image)

**Figure 8:** Inputs and outputs of decaBDE in the production of textiles and plastics for ELV
4.1.7.3  Concentration in products, wastes and recyclates

The use of plastic components of vehicles is increasing with efforts to make cars more fuel efficient. On average plastic accounts for around 9 to 12% of a vehicle’s weight or around 150 to 180kg [WRc Addendum 2012] quoting [EC 2011] of which about 20% is composite material i.e. plastic mixed with other materials. As with WEEE a single vehicle can contain a range of plastic components which contain varying levels of commercial PBDE formulations [WRc Addendum 2012].

Section 10.8 in the Annex provides a comprising overview of decaBDE levels in ELV articles and waste. The decaBDE levels in ELV materials range from 0 to 27,000 mg/kg. The highest levels in specific materials were found in seat cover material from cars manufactured in the 1990s. The reported results demonstrate that many of the investigated car parts/materials did not contain decaBDE or only in some cases.

According to [NO EA 2016], decaBDE concentrations in ASR found in literature amount up to 3,915 mg/kg (mean 2,163 mg/kg). Measured concentrations of decaBDE are higher in the mixed medium and high-density plastic ASR fractions compared to the mixed light plastic ASR fraction (2,163 mg/kg (range from 644 to 3,915 mg/kg; 5 samples) in mixed medium density plastic and 1,357 mg/kg (1 sample only) in mixed high-density plastic) [WRc addendum 2012]. However, average decaBDE concentrations in ASR are usually in a range between 2 and 406 mg/kg. The decaBDE levels in ASR and ASR fine of mixed samples, including 201 ELVs from Ireland, were 3.5 and 2.55 mg/kg respectively [ELVES 2016].

As reported by a Japanese study there is a notable gap of decaBDE concentrations in ASR from ELVs from the Japanese market between cars having been manufactured before 1996 and cars having been manufactured after 2000. Whilst ASR from ELVs before 1996 contain a mean value of 406 mg/kg and a maximum value of 590 mg/kg, the ASR from ELVs after 2000 show a mean value of 123 mg/kg and a maximum level of 180 mg/kg. Due to legislation the amount of used decaBDE has decreased. The data indicate a decreasing time trend for decaBDE in ASR from ELVs. ([NO EA 2016] referring to [Japanese MoE survey 2011]).

Recent measurements confirm that post-shredder technologies can separate output fractions with high PBDE content and fractions with low PBDE content (including decaBDE) by reprocessing of shredder residues by means of density separation possibly paired with electrostatic separation. PST allows to separate the different shredder fractions and direct fractions with higher Br content e.g. shredder granulate with a density of > 1.3 g/cm³ to appropriate treatment operations. According to the authors, fractions with higher content are either used as a reducing agent for blast furnaces (equivalent to feed stock recycling) or sent to waste incineration plants. Both options are considered possible treatment operations for destroying decaBDE [Öko-Institut 2018].

The following table shows the bromine contents (note: not specific for decaBDE) that were found by X-ray fluorescence (XRF)-measurements in fractions resulting from post shredder treatment.

Table 9: Bromine content found by XRF-measurements in fractions from post shredder treatment [Öko-Institut 2018]

<table>
<thead>
<tr>
<th>No. of samples</th>
<th>Min</th>
<th>Max</th>
<th>Median</th>
<th>Average</th>
<th>Values &lt; 1,000 mg/kg Br</th>
<th>Values &lt; 500 mg/kg Br</th>
</tr>
</thead>
<tbody>
<tr>
<td>Br in shredder sand</td>
<td>72</td>
<td>30</td>
<td>515</td>
<td>197</td>
<td>211</td>
<td>100%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>99%</td>
</tr>
<tr>
<td>Br in shredder fluff</td>
<td>58</td>
<td>154</td>
<td>3,594</td>
<td>527</td>
<td>729</td>
<td>83%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>47%</td>
</tr>
</tbody>
</table>
The data demonstrate that the bromine rich fractions accumulate in the shredder residues with a density above 1.1 g/cm$^3$ and confirm the findings of [WRc addendum 2012].

In addition, chemical analysis showed that decaBDE levels in granulate <1.1 g/cm$^3$ and in shredder fluff are below 50 mg/kg and the sum level of PBDEs (including decaBDE) is below 200 mg/kg [Öko-Institut 2018].

**Table 10:** Levels of Br, decaBDE and PBDE in granulate and shredder fluff [Öko-Institut 2018]

<table>
<thead>
<tr>
<th>Date (dd.mm.yyyy)</th>
<th>Bromine (HUK)$^1$ mg/kg</th>
<th>Bromine (ARN)$^2$ mg/kg</th>
<th>decaBDE$^3$ mg/kg</th>
<th>Sum PBDE$^3$ mg/kg</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Granulate &lt;1.1 g/cm$^3$</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>13.12.2017</td>
<td>341</td>
<td>27</td>
<td>&lt;50</td>
<td>&lt;200</td>
</tr>
<tr>
<td>27.12.2017</td>
<td>336</td>
<td>426</td>
<td>&lt;50</td>
<td>&lt;200</td>
</tr>
<tr>
<td>10.01.2018</td>
<td>&lt;100</td>
<td>30</td>
<td>11</td>
<td>&lt;100</td>
</tr>
<tr>
<td>24.01.2018</td>
<td>136</td>
<td>277</td>
<td>&lt;10</td>
<td>&lt;100</td>
</tr>
<tr>
<td>07.02.2018</td>
<td>&lt;100</td>
<td>566</td>
<td>12</td>
<td>&lt;100</td>
</tr>
<tr>
<td><strong>Fluff</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>13.12.2017</td>
<td>536</td>
<td>434</td>
<td>&lt;50</td>
<td>&lt;200</td>
</tr>
<tr>
<td>27.12.2017</td>
<td>340</td>
<td>448</td>
<td>&lt;50</td>
<td>&lt;200</td>
</tr>
<tr>
<td>10.01.2018</td>
<td>316</td>
<td>949</td>
<td>24</td>
<td>&lt;100</td>
</tr>
<tr>
<td>24.01.2018</td>
<td>117</td>
<td>272</td>
<td>27</td>
<td>&lt;100</td>
</tr>
<tr>
<td>07.02.2018</td>
<td>313</td>
<td>475</td>
<td>48</td>
<td>&lt;100</td>
</tr>
</tbody>
</table>

$^1$ DIN EN 14582
$^2$ With XRF after sample comminution
$^3$ And $^2$ not exactly the same sample but from the same batch only
$^3$ Measured according to DIN EN ISO 22032
Strååt and Nilsson analysed different fractions from Shredders / Post Shredder Technologies/Treatment (PST). All analysed plastic types are of a density of < 1.1 g/cm³ and all have decaBDE content of less than 200 mg/kg (as cited in [Öko-Institut 2018]. [ACEA Sub. 2018b] reports decaBDE content for plastics below 1.1 g/cm³ in ASR of 0 to 12 mg/kg.

4.1.7.4 Activity data

The automotive industry has made progress towards the substitution and elimination of decaBDE. While car manufacturers in some regions, among them the European Automobile Manufacturers Association (ACEA) member companies, have already phased-out decaBDE, others are still in the process of substituting decaBDE. At POPRC-11, the 15 European car manufacturers of ACEA indicated that they will phase-out decaBDE at the latest by mid-2018 for cars already in production and for new models under development. The substitution in progress does not extend to legacy spare parts i.e. parts for cars that are no longer in mass production. Although a phaseout of decaBDE has been in progress in Canada and the United States for some years. The members of the Canadian Vehicle Manufacturers’ Association (CVMA) have indicated that they need a minimum of five years i.e. a full design cycle to be able to eliminate a substance for all applications in new vehicle models [UNEP POPS/POPRC.12/INF/9 2016].

4.1.7.5 Waste management

Depollution and dismantling

The ELV Directive 2000/53/EC regulates the dismantling and depollution of ELVs particularly in Article 6(3). Hazardous materials and components shall be removed and segregated in a selective way so as not to contaminate subsequent shredder waste. Specific components including batteries and liquefied gas tanks, explosive components, specific fluids and components containing mercury shall be removed.

Dismantling is the process of taking a car apart so that valuable and reusable parts may be removed for manufacture or re-manufacture. According to Annex 2 of the EC Study to Examine the Costs and Benefits of the ELV Directive, the most commonly reused items at dismantlers were wheels (steel/alloy), engines, gearboxes, spare parts (e.g. carburettors), tyres and radiators [GHK 2006]. The selective dismantling of decaBDE-containing components is not considered feasible for two reasons: (1) usually the dismantlers cannot be provided with detailed information on parts which contain decaBDE; (2) it would be a too complex and time-consuming process [Öko-Institut 2018].

Shredding and sorting

The materials that cannot be readily dismantled and reused enter the shredding process where it is beaten into small pieces. The aim of this process is to get flakes of a size of 150 mm or less which can be sorted [Martens 2011]. In this report, the output of this shredding process is called automotive shredding residue (ASR)³¹.

After shredding, the ASR is separated into three different fractions:

- Ferrous fraction
- Non-ferrous metallic fraction (shredder heavy fraction, SHF)
- Shredder light fraction (SLF)

³¹ In some publications, ASR and shredder light fraction (SLF) is used synonymously.
The mixture containing non-ferrous metals is called shredder heavy fraction (SHF), the one containing non-metallic components (such glass, textiles, fibres, polyurethane foam) shredder light fraction (SLF) or car fluff [Sinha and Taylor 2015]. The technical process of separating the metallic from the non-metallic fraction is performed by a wind sifter, a technology which uses the different material densities of the shreds to separate the light material from the heavier one [Martens 2011]. After the ferrous and non-ferrous metals are separated from the shred, the SLF represents about 20 to 25% of the ELV weights [Kanari et al. 2003]. As the components of the SHF are not magnetic, they are separated from the ferrous (and magnetic) fraction by using a drum-type magnetic separator [Martens 2011].

The SHF as well as the SLF contain plastic components. The plastic particles within both fractions are assumed to contain the brominated compounds.

Mechanical treatment of waste in shredders including post shredder technology is described in section 3.1 of the Final Draft Best Available Techniques (BAT) Reference Document for Waste Treatment from October 2017 [BREF WT FD 2017].

*Post shredder treatment (PST)*

The ferrous fraction has an iron share of more than 95%. As iron and steel are materials which are easily recyclable, this fraction is reintroduced in the steel cycle [Martens 2011].

PST includes the separation of the SHF into several fractions for further processing. [Martens 2011] give the following fractions: metals (directly reusable, iron, aluminium, copper, steel), metallic composites, plastics, substitute fuels. According to Eurostat data, SHF is typically recycled.

In terms of waste management, the SLF is the most demanding fraction of ASR. Due to its high organic share and toxicity, it is classified as hazardous waste in many countries [Kanari et al. 2013]. Landfilling of SLF without preparatory treatment is illegal in some EU MS, e.g. in Germany and the Netherlands. In general, shredder residues may not be landfilled prior to certain processing steps. Due to its high plastic fraction of approximately 30%, material or energetic recovery are the preferred treatment options for the SLF [Martens 2011].

To separate the variety of materials within the SLF, different technologies are available. Simple versions of PST include metal separation and screening, producing a mineral-enriched fine-grained material that is landfilled in some MS while the coarse-grained fraction is introduced into waste incineration plants. Metal concentrates are forwarded to metallurgical plants. State-of-the-art technologies of PST split the SLF into fractions of higher quality to be used in more efficient recovery processes. Only in very few cases fractions are won for potential material recycling. The largest part is used for feedstock recycling (e.g. as reducing agent in blast furnaces) or energy recovery (e.g. in cement kilns or state of the art waste incineration plants [Öko-Institut 2018].

[Martens 2011] and [Öko-Institut 2018] describe the VW-Si-Con technology which has a plastic raw granulate as end product of a two-stage wet density sorting process. The main objective is to sort out heavier types of plastic (rigid plastic fractions such as PVC) for further material recycling steps. The technology is applied in several EU MS [Öko-Institut 2018].

Output fractions of this process are listed below [Öko-Institut 2018]

- Shredder granulate, used as substitute for heavy oil or coal dust in a blast furnace
- PVC-rich plastic, used for material recycling
- Shredder fluff, used in sewage sludge dewatering processes
- Fibres, used as additives in cement kilns
- Shredder sand, after further processing steps potentially usable as feed for specific building materials, ferrous and non-ferrous metallurgy and for backfilling applications
At a plant in the Netherlands of the company ARN recycling, SLF is separated into fractions that are subject to recycling and recovery (plastics with a density < 1.1 g/cm³ and fibers) [ARN Sub. 2018].

As Br has a larger density than plastic like PE (3.12 g/cm³ compared to 0.91-0.97 g/cm³) [Martens 2011], brominated plastic is heavier than unbrominated one. Therefore, plastics with a density above 1.1 g/cm³ are suspected to be brominated.

Post shredder technologies based on density separation therefore allow to accumulate bromine rich fractions in the shredder residues with higher density (e.g. above 1.1 g/cm³). These high-density fractions can e.g. be incinerated at appropriate conditions to destroy the PBDE content (including the decaBDE content).

4.1.7.6 Substance and mass flow

The decaBDE concentrations described in the literature for ASR from ELV are inhomogeneous, ranging from 0.01 to 2,163 mg/kg. We compiled literature values data from various sources and that way established a data set, which is used as basis for further calculations. To estimate the level of decaBDE in ASR, concentrations from literature were used to calculate a median decaBDE concentrations. In detail, this includes concentrations from studies [Swerea 2018; Petreas and Oros, 2009; WRc addendum 2012; Sinkkonen et al. 2004; NO EA 2016; MoE survey 2011; ELVES; IVM 2013] and from the informal stakeholder consultation [DEFRA Sub. 2018].

For the estimation of an average decaBDE-content of the Shredder light fraction and the Shredder heavy fraction, the median value of concentration levels of decaBDE reported in relevant auto shredder residues was used (where indicated based on mean values, where only ranges were indicated, the upper value was used). The resulting median of the concentrations from this data set is 109 mg/kg. This value is considered in a plausible range. The values the concentration is based on can be found in section 10.9 in the Annex. In accordance with the data provided by [ACEA Sub. 2018b] and [Öko-Institut 2018] given in Table 11, it can be considered that the concentration in light SLF fractions tends to be lower, whereas it tends to be higher in medium or high density SLF fractions (≥ 1.1 g/cm³).
Table 11: State-of-the-art Post-Shredder Technology: Grained fractions, Bromine content and usage based on [ACEA Sub. 2018b; Öko-Institut 2018]

<table>
<thead>
<tr>
<th>Grained Fraction (ARN 2017)</th>
<th>Share in ASR [%]</th>
<th>Bromine content* in 2016/2017 [mg/kg]</th>
<th>DecaBDE content [mg/kg]</th>
<th>Treatment path</th>
</tr>
</thead>
</table>
| Fibres (fluff)              | 30.4              | 527 (median)                         | 0 – 48 (2018)           | Recycling and Recovery  
Outliers can be separated and are directed to recovery operations |
| Minerals (shredder sand)    | 20.6              | 197 (median)                         | DecaBDE concentration above 1,000 mg/kg not likely | Recycling |
| Plastics < 1.1 g/cm³        | 8.2               | 86 (median)                          | 0 – 12 (2018)           | Recycling |
| Plastics 1.1 – 1.3 g/cm³    | 5.2               | 2,220 (median)                       |                         | Incineration (reducing agent in blast furnaces). Advanced sorting technologies might enable separation of POP-containing particles and serve as an option for material recycling in the future. |
| Plastics > 1.3 g/cm³        | 13.5              | 2,600 (median)                       |                         | High PVC concentrations and therefore not suitable for use as a reducing agent in blast furnaces or cement kilns. Treated by means of incineration and chemical recycling processes |
| Rubber/Wood/plastics        | 8.0               | 42 (median)                          |                         | Recovery |
| Misc. Heavy parts           | 5.7               | Not measured                         |                         | Recycling, recovery and landfill |
| Dust                        | 4.6               | 489 (median)                         |                         | Recovery |
| Metals                      | 3.8               | Not measured                         |                         | Recycling |

*Measured at ARN, Tiel with XRF
Based on Eurostat data for 2015 (17 MS) and 2014 (5 MS) the output from shredding of ELVs amounted to approximately 3.95 Mio tonnes in 2014/15 (see Table 12).

### Table 12: Output from ELV shredding for 2015 (17 MS) and 2014 (5 MS) based on [Eurostat 2018c, Eurostat 2018d]

<table>
<thead>
<tr>
<th>22 EU MS</th>
<th>Ferrous scrap</th>
<th>Other materials</th>
<th>Non-ferrous materials</th>
<th>SLF</th>
<th>Total shredding</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quantity (t)</td>
<td>2,900,498</td>
<td>144,360</td>
<td>188,037</td>
<td>712,842</td>
<td>3,945,739</td>
<td>[ACEA Sub. 2018b];</td>
</tr>
<tr>
<td>Share (%)</td>
<td>73.5</td>
<td>3.7</td>
<td>4.8</td>
<td>18.1</td>
<td>100.0</td>
<td>[ACEA Sub. 2018b];</td>
</tr>
<tr>
<td>decaBDE conc. [mg/kg]</td>
<td>No decaBDE</td>
<td>No decaBDE</td>
<td>109</td>
<td>109</td>
<td>Own data set, as explained above</td>
<td></td>
</tr>
</tbody>
</table>

### Ferrous fraction (steel) from shredding

The quantity of the ferrous fraction amounted to 2,900,498 t/y (73.5%). This fraction does not contain relevant quantities of plastics and is therefore considered almost free of decaBDE.

### Other materials arising from shredding

The quantity of other materials arising from shredding amounted to 144,360 t/y (3.7%). This fraction does not contain relevant quantities of plastics and is therefore considered almost free of decaBDE.

### Non-ferrous fraction (also referred to as heavy shredder fraction)

The quantity of the non-ferrous fraction amounted to 188,037 t/y (4.8%). Primarily, aluminium, copper and zinc are found in this fraction. The residues of this fraction include, for example, rubber, plastics, glass or stones [Martens 2011]. The share of plastics in the non-ferrous fraction is unknown. Assuming decaBDE is contained in the non-ferrous fraction with a similar concentration as in the SLF, i.e. with an average concentration of 109 mg/kg (specific information is not available) this fraction contains 20.5 tonnes of decaBDE. According to Eurostat data, this fraction is typically recycled (approximately 3 % (5,594 t) where incinerated (R1: 3,889 t) or landfilled (1,705 t) in 2014/15).

### Shredder light fraction (SLF)

Based on Eurostat data for 2015 (17 MS) and 2014 (5 MS) the relevance of the waste treatment of the shredder light fraction (SLF) is as follows:

### Table 13: Waste treatment of the SLF for 2017 (17 MS) and 2014 (5 MS)

<table>
<thead>
<tr>
<th>22 Member States</th>
<th>Total SLF</th>
<th>Recovery - energy recovery (R1)</th>
<th>Disposal</th>
<th>Recycling</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quantity (t)</td>
<td>712,842</td>
<td>236,295</td>
<td>263,500</td>
<td>213,047</td>
</tr>
<tr>
<td>Share (%)</td>
<td>100</td>
<td>33.1</td>
<td>37.0</td>
<td>29.9</td>
</tr>
</tbody>
</table>

According to these data, a quantity of 712,842 tonnes of SLF was generated. In 2014/15, 37.0 % of it was disposed of, 33.1 % energetically recovered and 29.9 % recycled. It can be considered...
that these figures for 22 MS underestimate the quantities for the whole EU 28 to a certain degree. However, data from larger MS are all included.

The substance and material flow for decaBDE in shredder residues of ELVs in the EU in 2014/15 based on this data is presented in Figure 9.

In summary, according to the estimations, the major share of the decaBDE quantity from the ELVs is treated in the shredder light fraction (77.7 t). From the SLF, about one third (33.1 %) is incinerated whereas the major share is recycled (29.9 %) or landfilled (37.0 %).

Figure 9: Substance and Material Flow for decaBDE in plastics from ELVs in the EU-28 in 2014/15

The mass flow demonstrates that only one third of the SLF including its corresponding decaBDE freight is destroyed whereas significant shares are landfilled (37.0 %) or recycled (29.9 %).

Possible impacts on the substance and mass flow of applying state-of-the-art PST

As described above, advanced post shredder technologies enable to accumulate bromine rich fractions in the shredder residues with a density above 1.1 g/cm³. High density fractions could be and are, according to [ACEA Sub. 2018b], incinerated to destroy the PBDE content (including the decaBDE content). According to [Öko-Institut 2018] more than one third of the SLF is currently not subject to onsite or off-site post shredder treatment. Eurostat data do not enable to differentiate whether PST is done or not. In the mass flow in Figure 9 a separation of the high-density fraction is not specifically considered.

The presented mass flow is based on Eurostat data from 2014 and 2015. Taking the information provided by [ACEA Sub. 2018b] and the results of [Öko-Institut 2018] into account (see Table 11) enables to increase the rate of destruction of decaBDE in SLF. To this end, by applying state of the art PST, high density fractions (18.7% of the SLF which contain high bromine levels) can be directed to feedstock recycling, incineration or chemical recycling processes.

Currently, as stated by [Öko-Institut 2018] several shredders across Europe do not use on- or off-site PST. At least 260,000 tonnes/y of SLF from ELVs is disposed without further treatment.
4.1.8 CONSTRUCTION MATERIALS AND CONSTRUCTION WASTE

4.1.8.1 Background information

In addition to the use in textiles and plastics decaBDE is used in some applications in buildings and construction. DecaBDE is used in wall and roof panels, which are typically made from unsaturated polyester (UPE) glass composites; floor tiles; and commercial grade carpeting. DecaBDE is also used in e.g. in insulation materials, and in roofing materials such as membranes and films for use under roofs to protect building areas. It can also be found in ducting elements such as the duct covering or insulation [UNEP/POPS/POPRC.11/10/Add.1 2015].

Materials, in which decaBDE may be present, can be used in a wide range of building elements. In 2006, the BSEF published a list of potential uses of decaBDE in the building sector. Building and construction applications of decaBDE include [RPA 2014].

- Pipes
- Lamp holders
- Stadium seats (mentioned in a Danish EPA report but without further clarification [Danish EPA, 2007])
- Switches and connectors
- Facing laminates for insulation panels
- Films for use under the roof and to protect building areas
- Electrical ducts and fittings
- Components in analytical equipment in industrial and medical laboratories
- Air ducts for ventilation systems
- Pillars for telephone and communication cables [RPA 2014].

Uses related to buildings and construction could be classified as follows:

- Structural elements (roofing, wall and floor coverings)
- Insulation (thermal)
- Cables and wiring [RPA 2014].

In a study from [RPA 2014], the relevant uses of decaBDE in the construction sector are identified at EU level. Further a study from the US Environmental Protection Agency [US EPA 2017] gives information on these uses. Relevant uses identified from both studies are:

- Roofing (opaque roofs made from unsaturated PES and polyolefin-based roofs; cross-linked elastomer compositions based on polyvinyl chloride (PVC) / nitrile rubber or PE)
- Seals and adhesives (Adhesives imported until 2010 contained c-decaBDE (identified use by Norway from the registration dossier, industry denies this use)
- Coatings (possibly in protective coatings of buildings)
- Insulation (possible): decaBDE is used in PVC-nitrile rubber compounds for heating tube insulation or A/C systems and in PE foams for thermal and acoustic insulation. PU-Europe, the voice of polyurethane insulation industry, denies the use of decaBDE in PU-based thermal insulation.
- Epoxy resins.

The occurrence of decaBDE in plastics of the waste streams WEEE (see chapter 4.1.6) and ELV (see chapter 4.1.7) are widely known, and significant amounts of research are dedicated to those streams (see chapter 4.1.1). On the contrary, data availability on decaBDE in C&D waste is less good [Hennebert 2018].

Specific information on decaBDE levels in construction and demolition waste has not been identified in [UNEP/CHW.13/INF/14 2017].
4.1.8.2 Processes, inputs and outputs

![Diagram of Processes, inputs and outputs]

**Figure 10:** Inputs and outputs of decaBDE in the production of construction material

4.1.8.3 Concentration in products, wastes and recyclates

There is limited data on decaBDE concentrations in products, wastes and recyclates of the construction sector. The limited data found on concentration ranges in construction waste from [RPA 2014] is summarised in Table 14.

Typical decaBDE concentrations in electrical insulation range from 10-30% and in epoxy adhesives it is below 30%. Specific information on decaBDE levels in construction materials is very sparse. In PVC flooring it was not detected. In damp proof membrane filter it was detected below 0.1% and in rubber insulation board it was contained at a level of 6%. Further, Luxembourg submitted as part of the stakeholder consultation (see chapter 2.2) information on isolation plates, which is included in the table. According to Luxembourg, in 2017 different products were tested for PBDE. No PDBE was detected in samples from isolation plates (2x) [LU Ministry of Sust. Dev. and Infrastructure Sub. 2018].

<table>
<thead>
<tr>
<th>Country</th>
<th>Product/ product category</th>
<th>DecaBDE concentration (mg/kg)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>UK</td>
<td>Electrical insulation</td>
<td>100,000 – 300,000</td>
<td>[RPA 2014]</td>
</tr>
<tr>
<td>UK</td>
<td>Epoxy resin adhesive</td>
<td>&lt; 300,000</td>
<td>[RPA 2014]</td>
</tr>
<tr>
<td>India</td>
<td>PVC floor coverings</td>
<td>Not defined</td>
<td>[Kumari et al. 2014]</td>
</tr>
<tr>
<td>Switzerland</td>
<td>Moisture resistant membrane / foil (Vapor barrier)</td>
<td>&gt; 1,000</td>
<td>[Kant. Lab BS 2009]</td>
</tr>
<tr>
<td>Germany</td>
<td>Vulcanised synthetic rubber insulation / insulating panel</td>
<td>60,000</td>
<td>[Kemmlein et al. 2005]</td>
</tr>
<tr>
<td>Luxembourg</td>
<td>Isolation plates</td>
<td>0</td>
<td>[LU Ministry of Sust. Dev. and Infrastructure Sub. 2018]</td>
</tr>
</tbody>
</table>

It should be noted that, considering the main use phase of decaBDE from the 1990s and 2000s, data on concentration levels of decaBDE in construction products or waste construction should
particularly relate to products which were used during that time. Investigations on decaBDE content in the production sector should therefore focus on products or waste from that time frame.

A preliminary report from the [US EPA 2017] is dealing with the manufacture, distribution, use and disposal of decaBDE. The report includes a summary of materials or products with decaBDE concentrations, which were obtained in most cases from the information from the technical data sheets. To what extent these products are also used in the EU is unclear. The decaBDE concentrations mentioned in connection with building products are summarised in Table 15. The concentrations given there must be considered as functional concentrations. Therefore, it is no surprise that they are higher than the decaBDE concentrations within waste products given in Table 14.

Table 15: Concentration ranges of decaBDE in construction materials [US EPA 2017]

<table>
<thead>
<tr>
<th>Product</th>
<th>decaBDE concentration in % w/w and (mg/kg)</th>
<th>Trade name</th>
</tr>
</thead>
<tbody>
<tr>
<td>Polypropylene glass fibre product</td>
<td>20-50 (200,000-500,000)</td>
<td>RTP PP 30 GF FR0 BLK</td>
</tr>
<tr>
<td>Coating product</td>
<td>1-10 (10,000-100,000)</td>
<td>Superseal® Edge Treatment; Superseal® HV</td>
</tr>
<tr>
<td>Facing for fiberglass batt insulation, duct wrap or duct board products.</td>
<td>7-11 (70,000-110,000)</td>
<td>GL-10; GL-30; GL-PD; GL-VR</td>
</tr>
<tr>
<td>Polyurethane polyol blend with additives</td>
<td>1-20 (10,000-200,000)</td>
<td>CHEMTHANE 7001F B FR</td>
</tr>
<tr>
<td>Industrial applications. Plastics</td>
<td>25 – 50 (250,000 – 500,000)</td>
<td>V0044G Natural</td>
</tr>
<tr>
<td>Industrial applications. Plastics.</td>
<td>5 – 10 (50,000-100,000)</td>
<td>XCPE 30-2 D5 BLACK</td>
</tr>
<tr>
<td>Epoxy resin paste</td>
<td>10-30 (100,000-300,000)</td>
<td>Araldite 252 Resin</td>
</tr>
</tbody>
</table>

4.1.8.4 Activity data

Construction products are manufactured and distributed globally. In the context of this project plastic-based and decaBDE flame retarded building products are relevant. Specific data on quantities of decaBDE used in the building sector in the EU is not available.

Though decaBDE is mainly known to occur in WEEE and ELV streams, data from other countries than the EU suggest a significant share of decaBDE was used within the construction sector. Compile data on decaBDE usage in different countries is indicated in the table below.

Table 16: Share of decaBDE usage in construction material within different countries [UNEP/POPS/POPRC.10/10/Add.2 2014]

<table>
<thead>
<tr>
<th>Country</th>
<th>decaBDE consumption in construction materials</th>
</tr>
</thead>
<tbody>
<tr>
<td>Japan</td>
<td>19%</td>
</tr>
<tr>
<td>Switzerland</td>
<td>25%</td>
</tr>
<tr>
<td>USA</td>
<td>26%</td>
</tr>
</tbody>
</table>

Though specific data for the EU is not available one can assume a similar distribution than in those three countries and therefore a rough estimation is made that 20% of the total decaBDE was consumed within the construction sector.
4.1.8.5  Waste management

In the "EU Construction & Demolition Waste Management Protocol" [EC 2017] it is stated, that almost 50% of C&D waste is recycled (including backfilling), whereby the WFD’s target lies at 70% in 2020.

The main fraction of C&D waste is the inert fraction. Detailed information on that topic is given in chapter 4.7. Though some EU countries attain high recycling rates for the inert fraction of C&D waste, recycling rates for other C&D waste fractions and in particular for plastic waste remain low. Established techniques for recycling the inert fraction exist and where they are applied high recycling rates are reached. The steel fraction is also well recycled. On the contrary, plastics are poorly collected and recycled. They are usually landfilled or incinerated [BIPRO 2011]. This may not apply for MS with landfill restrictions of recyclable and recoverable waste. On average, these countries have higher incineration and recycling rates of plastic post-consumer waste, since landfilling is not an option. These MS are namely Austria, Germany, the Netherlands, Sweden, Denmark, Luxembourg, Belgium and Finland [PlasticsEurope 2018].

The total waste quantity generated per year in construction and demolition waste is estimated to amount to around 860 million tonnes. The plastic content of the C&D waste in the EU amounts to 930,000 tonnes per year which means a share of around 0.11% (Eurostat 2018b).

Based on the assumption that at EU level 20% of the total decaBDE was consumed within the construction sector, C&D waste is considered a relevant path of decaBDE. As the consumption of decaBDE in Europe started approximately 50 years ago (see chapter 4.1.1) and buildings have lifetimes of typically 50 years and more, the occurrence of decaBDE in significant quantities within C&D waste starts now, around 2020 (related to the use around 1970), and will increase in the future (due to rising consumptions of decaBDE in the 1970s).

To what extent end-products containing decaBDE are landfilled, incinerated, left in the environment or recycled in the EU will largely depend on the waste management strategy chosen by different MS. EU-wide data on treatment of plastics within the C&D waste stream was not available.

Although the recycling of waste containing decaBDE above the LPCL (to be defined) will not be permitted, in management practice for C&D waste, no distinction is made e.g. between streams of brominated and non-brominated material.

4.1.8.6  Substance and mass flow

The plastic content of the C&D waste in the EU amounts to 930,000 tonnes per year [Eurostat 2018e]. As data on the decaBDE volume in this waste is not available, a substance and mass flow cannot be derived. However, based on the following data and assumptions, we can provide a theoretical calculation which gives a rough estimation of how much decaBDE will arise in C&D waste in the future. Starting point are the data given by [Earnshaw et al. 2013] (see Figure 3).

Considering a typical lifetime of a building of 50 +/-25 years and scenario C2 of [Earnshaw et al. 2013], it can be deduced that decaBDE in construction waste related to its use around 1970 starts to occur around now (2020) in C&D waste and will increase constantly over the following 20 years. As about 20% of the consumed decaBDE was used in the construction sector, we can expect a plateau of approximately 1,500 tonnes decaBDE per year to become waste via the C&D waste stream for the period from 2040 to 2060.

Table 17:  Overview on assumptions used for the calculation and its results

<table>
<thead>
<tr>
<th>Value</th>
<th>Comments</th>
<th>Source / Remarks</th>
</tr>
</thead>
<tbody>
<tr>
<td>7,500 t</td>
<td>Mean consumption of decaBDE in EU for the years 1990 to 2040</td>
<td>Estimation based on [Earnshaw et al. 2013]</td>
</tr>
<tr>
<td>2010</td>
<td>al. 2013</td>
<td></td>
</tr>
<tr>
<td>------</td>
<td>---------</td>
<td></td>
</tr>
<tr>
<td>20%</td>
<td>Share of decaBDE consumption for construction material in EU</td>
<td>Estimation based on [UNEP/POPS/POPRC.10/10/Add 2 2014]</td>
</tr>
<tr>
<td>1,500 t</td>
<td>Estimated volume of decaBDE within C&amp;D waste in the years 2040 to 2060</td>
<td>Result from calculation (20% of A1)</td>
</tr>
<tr>
<td>1,600 mg/kg</td>
<td>Estimated concentration of decaBDE in plastics within C&amp;D waste (in the years 2040 to 2060)</td>
<td>Result from calculation (A3/930,000 t)</td>
</tr>
</tbody>
</table>

The estimations are based on Scenario C2 of [Earnshaw et al. 2013], who interpolate from several literature sources. The mean value in the years 1990 to 2010 of 7,500 tonnes per year are in accordance with the numbers on import of decaBDE in the EU of [UNEP/POPS/POPRC.10/10/Add 2 2014].

20% of decaBDE is estimated to be used in construction sector. This estimation is justified by the fact, the approximately this share is given for Japan, Switzerland and the USA.

Considering a quantity of 1,500 t of decaBDE in C&D waste per year with the tonnage of plastic within C&D waste in EU per year (assuming that it stays constant at 930,000 tonnes/y) results in an expected hypothetical decaBDE concentration within plastics from C&D waste of approximately 1,600 mg/kg.
4.1.9 TEXTILES AND FURNITURE

4.1.9.1 Background information

According to [RPA 2014], decaBDE is a very common FR for increasing the fire retardancy of textile fibres, especially those in upholsteries for furniture and seating. It has been reported that around 10% of the produced decaBDE is used in the textile sector. In addition to the use of decaBDE flame retarded textiles in vehicles and other means of transportation. Typical products which have been reported in the literature are coated textiles, upholstery, window blinds, curtains, mattresses and carpets for public and domestic buildings, and tents as well as textiles used in the transportation sector [UNEP/POPS/POPRC.11/10/Add.1 2015].

The use of decaBDE in furniture and textiles was confirmed by the VECAP report for 2012, and also from the consultation with the European Flame Retardant Association (EFRA), where it is stated that decaBDE was used as a FR in textiles, in the transport sector and in construction and building [RPA 2014].

There are no general European requirements regarding fire protection applying to wholesale textiles. However, a few countries, e.g. England, have specific rules regarding fire protection [Danish EPA 2007]. The UK has the strictest regulatory framework in the form of the Furniture and Furnishings Fire Safety Regulations (FFFRS), which require testing of upholstery separately from other components. The textile must be tested for fire resistance over untreated foam or filling, which means that it has to compensate for a more flammable substrate as well [RPA 2014].

DecaBDE which is present in textiles used in the automotive sector is managed together with ELVs and contributes to the occurrence of decaBDE in ASR [UNEP/CHW.13/INF/14 2017]. Waste textiles related to their former use in cars arise therefore in ELVs. DecaBDE in textiles in ELVs and their waste management and related mass flows are part of the considerations in section 4.1.7 on plastics and textiles in ELVs.

Furniture textiles

Textiles for furniture can be used as upholstery in the domestic sector for furniture that includes sofas, cushions, carpets and armchairs, and possibly curtains and mattresses. In the contract sector, they can be used as upholstery in office furniture (such as chairs), in drapes or curtains, seats and other articles for public buildings. Furniture may include FRs in the following components [RPA 2014]:

- Covering materials, which can be made from a variety of fibres, such as wool, cotton, viscose, polyacrylic, polyester, polypropylene or polyamide but also natural or imitation (PVC or polyurethane) leather
- Filling materials, mainly from polyurethane foam, flexible polyurethane or latex
- Interlining, which may be from non-woven materials, non-flammable material (such as glass fibres) or aramid fibres
- Plastic parts of furniture (such as desk-chair handles).

The use of decaBDE in flexible polyurethane (PU) foams has been mentioned in the past. In the USA decaBDE has been identified in the fabric covering of mattress foam. According to the European Flame Retardant Association, decaBDE is not used for increasing fire resistance in PU foams in mattresses. Recent consultation with the flexible PU foam industry has shown decaBDE is not used in mattresses fillings. Where standards are strict, such as in the UK and Ireland, TCPP is predominantly used. DecaBDE could be present in polyethylene terephthalate (PET). In the Annex XV dossier, it has been mentioned that DecaBDE may be used in synthetic rubber (latex) foam for mattresses, according to consultation with undisclosed stakeholders [RPA 2014].
According to a stakeholder from the furniture industry, around nine million upholstered items (with an estimated value of £1.58 billion) were sold in the UK market per year, with a lifetime of roughly 9 years, but, owing to the designation of decaBDE as a SVHC, most if not all textile finishers had already switched to alternatives [RPA 2014].

A manufacturer of textiles for contract applications has commented that they are using decaBDE (in the range of < 100 t/y) but are currently in the process of evaluating potential alternatives. In backcoated textiles, the flame retardant (FR) coating is carried in relatively low loadings. However, in other applications, the FR mixture can act as a carrier to the textile, as, for example, in window blinds [RPA 2014].

It has also been mentioned that until recently (2013), decaBDE was used in the manufacture of covers for mattresses to be used in medical beds. The quantities of decaBDE used for this were in the range of < 100 t/y [RPA 2014].

**Interliners**

Interliner fabrics are commonly used as a fire barrier, delaying the spread of fire to the more flammable fillings. They are usually made from fabrics that have been treated with a FR and find use in seats for public venues, such as cinemas and theatres. DecaBDE is not mentioned explicitly in this context and it is possible that it does not find applications at all in interliners [RPA 2014].

**Consumer and protective clothing**

The use of decaBDE in consumer clothing has not been identified in the EU. However, the presence of decaBDE in protective clothing has been reported by the Norwegian Environmental Agency, during the public consultation for the identification of decaBDE as a SVHC. Here, decaBDE was not used on the fabric itself, but rather on the adhesive part of the reflective tapes [RPA 2014].

DecaBDE was used in textiles of heat resistant uniforms for workers exposed to extremely high temperatures with very small relevant quantities, in the order of < 10 tonnes per year. Since 2012, an alternative substance is used [RPA 2014].

**Tents**

The use of decaBDE in tents seems to be important, but it was not possible to verify this use through the consultation process already performed by RPA (2014). Military tents are usually coated with a PVC layer for water-proofing and it is possible that the canvas is also backcoated with decaBDE. PVC canvas is inherently fire retardant, and so it is usually preferred over materials that must be made fire retardant. In addition, it is possible for decaBDE to be used for manufacture of disaster relief tents in Italy. These tents need to be both waterproof but also FR because people might cook inside them. A study from the US, from 2014 reported FRs present in tents for general use with the highest FR concentration of 3.8 %, but the relevant FR is not mentioned in the abstract. It is not known whether such tents are also used in the EU [Keller et al. 2014].

Further, in a study from the Danish Environmental Agency from 2007 [Danish EPA 2007], the use of decaBDE in (camping) tents has been identified. The tents were produced in China using decaBDE in quantities of about 2 g per tent and imported into Denmark, from where the majority were exported to other EU countries. Based on the projected import of 160,000 tents in 2007, about 320 kg of decaBDE was expected to be imported through this route [RPA 2014].
4.1.9.2 Processes, inputs and outputs

According to RPA (2014), there are different techniques to improve textile flame retardancy; FRs may be incorporated into fibres, applied to the surface of textiles or applied to the back of textiles in the form of a polymeric coating.

DecaBDE has traditionally been applied to textiles as a back-coating in combination with antimony oxide as a synergist [UNEP/POPS/POPRC.11/10/Add.1 2015].

![Diagram](image)

**Figure 11:** Inputs and outputs of decaBDE in the production of textiles

4.1.9.3 Concentration in products, wastes and recyclates

DecaBDE is incorporated in the plastic at a loading of roughly 12% in a ratio of three parts decaBDE to one-part antimony trioxide.

Information from consultation suggests that the proposed content of decaBDE in plastic materials depends on the polymer. In general, the FR mixture amounts ranged between 15–25% of the plastic compound [RPA 2014]. The following table summarises the concentrations found in articles in use in textiles and furniture in the EU as identified in literature and in stakeholder submission.

<table>
<thead>
<tr>
<th>Country/region</th>
<th>Product/Product category</th>
<th>Relevant wastestream/waste category</th>
<th>decaBDE concentration (range)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Norway</td>
<td>Tents</td>
<td>Textile waste /ELV</td>
<td>2 g/tent</td>
<td>[Danish EPA 2007]</td>
</tr>
<tr>
<td>UK</td>
<td>Velour pile fabrics (70-80 g/m2)</td>
<td>Textile waste /ELV</td>
<td>21-32 g/m2</td>
<td>[RPA 2014]</td>
</tr>
<tr>
<td>UK</td>
<td>Cotton (30-40 g/m2)</td>
<td>Textile waste /ELV</td>
<td>9-16 g/m2</td>
<td>[RPA 2014]</td>
</tr>
<tr>
<td>UK</td>
<td>Flat woven (30-80 g/m2)</td>
<td>Textile waste /ELV</td>
<td>9-32 g/m2</td>
<td>[RPA 2014]</td>
</tr>
<tr>
<td>UK</td>
<td>Tent</td>
<td>Textile waste /ELV</td>
<td>19,000 mg/kg</td>
<td>[NO EA 2018]</td>
</tr>
<tr>
<td>UK</td>
<td>Upholstery</td>
<td>Textile waste /ELV</td>
<td>70 – 22,000 mg/kg</td>
<td>[SE EPA Sub. 2018]</td>
</tr>
<tr>
<td>UK</td>
<td>Textiles (in vehicles, furniture)</td>
<td>Textile waste /ELV</td>
<td>200,000-300,000 mg/kg</td>
<td>[SE EPA Sub. 2018]</td>
</tr>
<tr>
<td>UK</td>
<td>Upholstered furniture</td>
<td>Textile waste /ELV</td>
<td>60-100 g/m²</td>
<td>[BSEF Sub. 2018]</td>
</tr>
<tr>
<td>UK</td>
<td>Adhesive layer of reflective tape</td>
<td>Textile waste /ELV</td>
<td>10,000 – 50,000 mg/kg</td>
<td>[RPA 2014]</td>
</tr>
<tr>
<td>UK</td>
<td>Various textiles</td>
<td>Textile waste /ELV</td>
<td>15,500 – 64,200 mg/kg</td>
<td>[RPA 2014]</td>
</tr>
</tbody>
</table>
Specific information on concentration levels of decaBDE in textile waste was not identified.

4.1.9.4 Activity data

The amount of decaBDE used in plastics and textiles globally varies but up to about 90% of decaBDE ends up in plastic and plastics used in electronics while the remainder is used in coated textiles, upholstered furniture and mattresses [UNEP/POPS/POPRC.11/10/Add.1 2015].

Fire safety requirements for furniture vary across Europe, with UK and Irish standards being the most stringent. While various MS have regulations in place that require the furniture be tested for flammability, the UK regulations require that the components (fillings, upholstery, loose fillings and cover) need to pass similar tests. This means that especially furniture fillings, upholstery, loose fillings and covers from the UK will be contaminated with decaBDE [RPA 2014].

4.1.9.5 Waste management

In general, textiles are disposed together with municipal solid waste from households, which might be collected and reused. It is however expected that a EU level, incineration and landfilling are the most common disposal routes.

According to [FOEE 2013], within the EU 25% of discarded textile waste is recycled. The remaining share goes to landfill or is burnt in municipal waste incinerators.

However, the situation regarding the management of decaBDE-containing textile waste in Europe is unclear. The European Chemicals Agency (ECHA) stipulates that decaBDE-containing textiles are currently not recycled in the EU [UNEP/POPS/POPRC.11/10/Add.1 2015]. There are uncertainties in the level of recycling of textile within parties and in terms of decaBDE content. Thus, it is difficult to define whether a restriction on recycling of textiles containing decaBDE will have economic implications for the textile recycling industry [UNEP/POPS/POPRC.11/10/Add.1 2015].

Relevant quantities of textile waste are exported. The USA is the largest exporter of used textiles (18% share of total mass exported globally) followed by Germany (12%) and the UK (8%) [ECAP 2017].

4.1.9.6 Substance and mass flow

It is expected that decaBDE from the application on textile may continue to appear in the waste streams and recycled materials due to the assumed lifetime of textiles (10 +/- 3 years) and from imports. However, from the total gathered waste in EU, textile waste represents 0.2 % [Ernantez et al. 2017]. This accounts for approximately 4.6 million tonnes of textiles in 2014. Since only certain textiles and furniture textiles are contaminated with decaBDE and no information is available on quantities or shares of the total textile and furniture waste and their treatment paths, no mass flow can be established. Moreover, relevant shared of treated textiles are already included within the ELV waste flow and the majority of textiles not treated within the ELV waste flow are either landfilled or incinerated within Europe.

The use of decaBDE in textile and furniture applications was in general comparatively low and already largely phased out around 2014. Considering typical lifetimes of textiles and furniture, the occurrence of decaBDE in textile waste has already now comparatively low relevance and will decline throughout the 2020s. After 2030 only insignificant amounts of decaBDE in textiles and furniture are expected.
4.1.10 OTHER WASTE STREAMS

4.1.10.1 Background information

Another potentially relevant waste stream in which decaBDE can be found is sewage sludge, a residual material that is produced as a by-product during sewage treatment of industrial or municipal wastewater in wastewater treatment plants (hereafter referred to as WWTP) [UNEP/CHW.13/INF/14 2017].

4.1.10.2 Processes, inputs and outputs

PBDEs are not removed during waste water treatment. The PBDEs either pass directly through into the downstream aquatic environment or else are contained in the sludge [UNEP/CHW.13/INF/14 2017].

PBDEs largely end up in the solid sludge fraction within the WWTP, e.g. according to [Ricklund et al. 2009] for more than 99% of decaBDE is retained in the sludge after waste water treatment in a large WWTP in California which is comparable to plants in Henriksdal [Danish EPA 2012].

Dewatered sludge can be reused for fertilisation of industrial crops if it complies with agricultural standards, otherwise it is disposed of [UBA 2013].

4.1.10.3 Concentration in products, wastes and recyclates

DecaBDE levels in sewage sludge from sewage treatment plants from different countries are in the range of several hundred µg/kg. Reported levels of decaBDE in sewage sludge samples range from 0.0026 to 17 mg/kg.

A Danish study found a concentration of 0.467 mg/kg for the sum of eight PBDE congeners, and a concentration of decaBDE of 0.248 mg/kg. The number of samples was rather limited in this study and only from one WWTP. The largest single survey covered 116 Swedish sludge samples from 22 WWTPs. Here, a mean decaBDE concentration was reported at a markedly lower level than the Danish study, i.e. 0.011 mg/kg wet weight, whereas the maximum level was 0.39 mg/kg wet weight (Jensen et al. 2012 as cited in [NO EA 2016]). Compared to the levels found in other European countries, the levels in Ireland and the UK are a factor of ten higher [RPA 2014].
4.1.10.4 Activity data

In total, 3.06 million t of sewage sludge arises from WWTPs in the EU in 2014 [Eurostat 2018i].

4.1.10.5 Waste management

Sewage sludge leaving the WWTPs are consecutively treated. Treatment options for sewage sludge are:

- Landfill (D1-D7, D12)
- Incineration (D10)
- Thermal recovery (R1)
- Recycling.

Around 50 % of sewage sludge is landfilled, approximately 16 % are incinerated (with and without thermal recovery) and 33 % are recycled [Eurostat 2018i].

Recycling includes both recycling and “backfilling”. Backfilling means a recovery operation where suitable waste is used for reclamation purposes in excavated areas or for engineering purposes in landscaping and where the waste is a substitute for non-waste materials” [EC n.d.a].

4.1.10.6 Substance and mass flow

The associated substance and material flow is presented below.
As shown in Figure 13 approximately 3 million tons of sewage sludge from waste water treatment was generated in the EU-28 in 2014. With the estimated decaBDE concentration in sewage sludge of 0.3 mg/kg, 0.92 tonnes of decaBDE were subject to further management from sludges and liquid waste from waste treatment in 2014. A total of 0.3 tonnes of decaBDE were subject to recycling, resulting from approx. 1 million tonnes of sewage sludge. Approx. 510 tonnes of sewage sludge were incinerated, and 1.5 million tonnes were landfilled related to 0.16 tonnes and 0.46 tonnes decaBDE, respectively.
4.2 Short chain chlorinated paraffins (SCCP)

SCCP are chlorinated paraffin mixtures that are viscous, colourless or yellowish dense oils. They are primarily used in metalworking application and in polyvinyl chloride plastics. Other uses include the use as paints, adhesives and sealants, leather fat liquors, plastics and FRs in rubber, textiles and polymeric material [UNEP/POPS/POPRC.11/10/Add.2 2015].

4.2.1 PRODUCTION, TRADE AND USE

SCCP can be used in a wide range of applications. Recent literature reports use of SCCP as plasticizers, binders, FRs in plastics, coatings and paints, rubber products, paper, textiles, joints and sealants and adhesives. Other sources also report SCCPs contents in everyday products such as microwave dishes, lamps, electronic items such as cables, adapters, keyboards, memory media, photo frames, headphones, detergent, etc (see [BiPRO 2015]).

The manufactured and used SCCP volumes declined significantly since 1998 and especially after 2002 due to the introduction of EU regulations (see Figure 14), with the coming into force of Directive 2002/45/EC, which banned the two most important uses of SCCP at that time in the metalworking and for fat liquoring of leather. Figure 14 summarizes the historic annual sales of SCCP in the EU from 1994 to 2009 [RIVM 2010].

![Figure 14: Total annual sales of SCCP in the EU [RIVM 2010]](image-url)

In 2010, the use of SCCPs in the EU 27 was estimated to be about 530 t [RIVM 2010]. According to the Risk Management Evaluation dossier on SCCP from 2016, Canada, Norway, the US as well as the EU have completely transitioned away from SCCPs. In addition, parties have not identified any uses where alternatives are not available, or any technical challenges associated with the transition to alternative chemicals and processes [UNEP/POPS/POPRC.12/11/Add.3 2016].

Parties to the Stockholm Convention shall prohibit and/or eliminate the production of SCCPs, unless they have notified the Secretariat of their intention to produce it for the time-limited specific exemptions listed in Annex A (i.e. additives in transmission belts, rubber conveyor belts, leather, lubricant additives, tubes for outdoor decoration bulbs, paints, adhesives, metal processing and plasticizers). Parties for which the amendment did not enter into force automatically may continue to produce SCCPs for any purpose until they have ratified the amendment through which the chemical was listed in Annex A [UNEP/CHW/OEWG.11/INF/10]
According to information provided in Annex E, Annex F, and several other information sources, SCCPs were reported to be produced in Brazil, and were reported to be imported by Albania, Australia, Republic of Korea, Croatia, Argentina, Dominican Republic, Ecuador and Mexico [UNEP/POPS/POPRC.12/11/Add.3 2016]. In general, available information on production often includes chlorinated paraffins (CP) of various chain-lengths and degrees of chlorination and it may therefore not be possible to specifically address the production of SCCPs. According to recent information, China has been reported to be the major producer of CP. Further, it is pointed out that depending on raw materials, some of the technical mixtures made by Chinese manufacturers may have been mixtures of SCCP, medium chain chlorinated paraffin (MCCP), and long chain chlorinated paraffin (LCCP) [UNEP/CHW/OEWG.11/INF/10 2018]. In summary, currently produced volumes of SCCPs are not known, but it can be expected that the production decreased on the global scale.

Regarding the production within the EU, Slovakia reported historic production quantities of 560 t, 354 t, 480 t and 410 t for the years 2004 to 2007, respectively. In Germany the production of SCCPs by the companies Clariant, Hoechst and Huels ceased already in 1995. Hoechst reported on production quantities between 9,300 t to 19,300 t between 1993 and 1995, respectively [UNEP/POPS/POPRC.6/11 2010]. In June 2012, the only known entity that registered manufacture of SCCPs under Regulation (EC) No 1907/2006 of the European Parliament and of the Council (REACH) declared that they had stopped and do not intend to restart manufacture of the substances 33.

Further, SCCPs were added to Annex I of the EU POP Regulation (EC Regulation No. 850/2004) allowing the use of SCCPs in conveyor belts in the mining industry and dam sealants. The Regulation was amended by Commission Regulation (EU) 2015/2030 to remove these exemptions. This change entered into force on 4 December 2015 and consequently all uses of SCCPs are prohibited above the specified limit values. The production, placing on the market and use of substances or preparations containing SCCPs in concentrations lower than 1 % by weight or articles containing SCCPs in concentrations lower than 0.15 % by weight shall be allowed. The Regulations stipulates that articles that contain SCCPs in concentrations lower than 0.15 % by weight are allowed to be placed on the market and used, as this is the amount of SCCPs that may be present as an impurity in an article produced with MCCPs. Besides, it is clarified that the use of conveyor belts in the mining industry and dam sealants containing SCCPs already in use on or before 4 December 2015, and articles containing SCCPs already in use on or before 10 July 2012 is allowed 33. Initially the Regulation EC 850/2004 allowed the use of SCCPs in conveyor belts and dam sealants without setting any concentration limits. The amendment entered into force in 2015 and subsequently all uses of SCCPs are prohibited above the previously mentioned limit values.

In this connection, in 2013 the Commission consulted with the relevant stakeholders in the mining industry. The consultation indicated that conveyor belts containing SCCPs are no longer used by the industry for mining purposes. Regarding dam sealants containing SCCPs, it was reported that they do not appear to be manufactured, placed on the market or used in the EU. Already in 2008, the relevant stakeholders indicated to the European Chemicals Agency that SCCPs are not


considered to be in use or are in the process of being phased out in sealants (including dam sealants) in Europe.

Based on the available information, it can be concluded that SCCPs are currently neither produced nor used in the EU (at least not above the limit values defined in Annex I of Regulation EU 2015/2030). As the production in the EU ceased some years ago, remaining stocks are not likely to occur in the EU. Nevertheless, some of the historic uses, especially in products with typically long life-times, may still be relevant also in the EU from the waste management perspective (including conveyor belts and sealants and adhesives). Besides, possible imports of new products and/or recycled products contaminated with or containing SCCPs (also in concentrations above the set limit values in Regulation EU 2015/2030) cannot be excluded and will be further discussed in this report. In addition, a possible contribution from other CPs (e.g. MCCPs) to SCCP concentrations detected in various products (including consumer products) placed on the EU market will be further investigated.

### 4.2.2 CHEMICAL CHARACTERISTICS

CPs are common products of the chlorine chemistry. Chemically, CPs are saturated linear hydrocarbons with varying chlorine contents and chain lengths. CPs are divided in accordance to their chain length in short, medium and long chain CPs. Short chain CPs are saturated, linear hydrocarbons with 10-13 carbon atoms and an average degree of chlorination of 40-70% [HDU 2007; BAuA 2011].

- **Chemical formula:** $C_{x}(2x-y+2)Cl_y$ ($x=10-13; y=3-12$)
- **CAS registry number:** 85535-84-8
- **EC List number:** 287-476-5
- **Molecular mass:** 320-500 g/mol [ERA 2000]

![Chemical structure SCCP](image)

**Figure 15:** Chemical structure SCCP

e.g.: two SCCP compounds ($C_{10}H_{17}Cl_5$ and $C_{13}H_{22}Cl_6$)

Table 19 summarizes the main chemical and physical properties of SCCP.

### Table 19: Chemical-physical properties SCCP

<table>
<thead>
<tr>
<th>Chemical-physical properties SCCP</th>
<th>Sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>Appearance</td>
<td>liquid</td>
</tr>
<tr>
<td>Density</td>
<td>1.18-1.59 g/cm³ (20°C)</td>
</tr>
<tr>
<td>Boiling point</td>
<td>&gt;200°C</td>
</tr>
<tr>
<td>Vapour pressure (at 100°C)</td>
<td>0.028-2.8 x 10⁻⁷ Pa; 0.0213 Pa</td>
</tr>
<tr>
<td>Log Kow</td>
<td>4.48-8.69</td>
</tr>
</tbody>
</table>
In Annex I to the Stockholm convention only short chain CPs with chain lengths ranging from C10 to C13 and a content of chlorine greater than 48 % by weight are listed. Further information related to the chemical identity is provided in the next chapter on the legal background.

### 4.2.3 LEGAL BACKGROUND

#### International/ EU

**(Stockholm Convention)**

SCCP are currently listed under the Stockholm Convention, Annex A (elimination) with specific exemptions. The COP amended Part I of Annex A to list SCCPs therein, with specific exemptions, by inserting the following:

**Table 20: Listing of short-chain chlorinated paraffins [UNEP/POPS/COP.8/SC-8/11 2017]**

<table>
<thead>
<tr>
<th>Chemical</th>
<th>Activity</th>
<th>Specific Exemption</th>
</tr>
</thead>
<tbody>
<tr>
<td>Short-chain chlorinated paraffins (Alkanes, C10-13, chloro)+: straight-chain chlorinated hydrocarbons with chain lengths ranging from C10 to C13 and a content of chlorine greater than 48 per cent by weight</td>
<td>Production</td>
<td>As allowed for the Parties listed in the Register</td>
</tr>
<tr>
<td>For example, the substances with the following CAS numbers may contain short-chain chlorinated paraffins: CAS No. 85535-84-8; CAS No. 68920-70-7; CAS No. 71011-12-6; CAS No. 85536-22-7; CAS No. 85681-73-8; CAS No. 108171-26-2.</td>
<td>Use</td>
<td>• Additives in the production of transmission belts in the natural and synthetic rubber industry • Spare parts of rubber conveyor belts in the mining and forestry industries • Leather industry, in particular fatliquoring in leather • Lubricant additives, in particular for engines of automobiles, electric generators and wind power facilities, and for drilling in oil and gas exploration, petroleum refinery to produce diesel oil • Tubes for outdoor decoration bulbs • Waterproofing and fire-retardant paints • Adhesives • Metal processing • Secondary plasticizers in flexible polyvinyl chloride, except in toys and children's products</td>
</tr>
</tbody>
</table>

Consequently, Parties to the Stockholm Convention must prohibit and/or eliminate the production of SCCPs, unless they have notified the Secretariat of their intention to produce it for the time-limited specific exemptions listed in Annex A to the Convention. Furthermore, Annex A specifies in Note (vii) that quantities of SCCPs that occur in mixtures at concentrations greater than or equal to 1 per cent by weight, cannot be considered unintentional trace contaminants, as outlined for other chemicals in Note (i). This is flagged by a plus sign (“+”) in the definition above.

According to information from the SCCP risk management evaluation (2016), no party or observer submitted information to propose or justify the need for a specific exemption or acceptable purpose in the listing of SCCPs to the Convention. No party has identified a specific use where flexibility in the recommended control measure is required.

 Parties listed in the Register of specific exemptions, as well as observers, have been invited to provide to the Secretariat, by December 2019, information (among other on production, use, control measures, etc.) to justify its need for the registration of exemptions (if relevant). Besides, parties and observers are invited to provide information on progress made in building the capacity of countries to transfer safely to reliance on alternatives to SCCPs. The Secretariat is requested to...
compile all relevant information provided and to make information available to the POPRC by 2020 for further discussions on international level.

EU POP Regulation

SCCPs are listed under the EU POP Regulation. The concentration limit in Annex IV of the EU POP regulation is currently set with an LPCL of 10,000 mg/kg. An MPCL of 10,000 mg/kg is specified in Annex V for waste management.

The EU POP Regulation addresses SCCPs with the CAS number 85535-84-8. Within this CAS number "the commercial SCCP product that is produced by the chlorination of a single hydrocarbon fraction consisting of n-alkanes that have a carbon chain length distribution consisting of 10, 11, 12 and 13 carbon atoms is represented" [UNEP/CHW.13/CRP.24/Add.3 2017]. The CAS number, however, does not specify the degree of chlorination of the SCCPs and thus may include substances not included under the Stockholm Convention [UNEP/CHW/OEWG.11/INF/10 2018].

The annexes of the EU POP Regulation are updated in accordance to international agreements and the technical development, which means that as soon as new information on details of uses and safer alternative substances or technologies become available, the Commission shall review the derogations so that the uses of SCCP can be phased out. In this connection, a dossier has been submitted as a basis for discussion by the Netherlands in advance to the "Meeting of the Competent Authorities under Regulation EC 850/2004" in Brussels on 2 October 2013 ("Evaluation of Possible Restrictions on Short Chain Chlorinated Paraffins"). The contained information from the dossier confirms that there are suitable alternatives to SCCP in all areas of application. Besides, it was highlighted that several companies have already made the switch to alternatives without major implications (mostly to MCCP and LCCP).

Regarding the two applications exempt under Regulation (EC) No 850/2004, a major manufacturer of conveyor belts indicated that transition to alternatives was smooth and incurred low costs. At the same time of the consultation, two other companies were working on alternatives. Further, in 2013 relevant stakeholders in the mining industry confirmed that conveyer belts containing SCCPs are no longer used by the industry for mining purposes. Regarding dam sealants containing SCCPs, it was expressed that they do not appear to be manufactured, placed on the market or used in the European Union. Already in 2008, the relevant stakeholders indicated to the European Chemicals Agency that SCCPs are either not used or in the process of being phased out in sealants (including dam sealants) in Europe.

In June 2012, the only known entity that registered manufacture of SCCPs under Regulation (EC) No 1907/2006 of the European Parliament and of the Council declared that they had stopped and do not intend to restart manufacture of the substances (see Regulation EU 2015/2030).

Consequently, Annex I to Regulation (EC) No 850/2004 was amended as follows:

According to Annex I of the POP Regulation, the following specific exemptions on intermediate use or other specifications are defined for SCCPs:

1. "By way of derogation, the production, placing on the market and use of substances or preparations containing SCCPs in concentrations lower than 1 % by weight or articles containing SCCPs in concentrations lower than 0.15 % by weight shall be allowed.
2. Use shall be allowed in respect of:
   a. conveyor belts in the mining industry and dam sealants containing SCCPs already in use before or on 4 December 2015; and
   b. articles containing SCCPs other than those referred to in (a) already in use before or on 10 July 2012.
3. Article 4(2) third and fourth subparagraphs shall apply to the articles referred to in point 2 above.
It is further stipulated that articles containing SCCP in concentrations lower than 0.15 % by weight are allowed to be placed on the market and used, as this is the amount of SCCP that may be present as an impurity in an article produced with MCCPs.

**REACH Regulation (EC) No 1907/2006**

Alkanes, C10-C13, chloro are registered under REACH. The relevant information (chemical properties, uses, etc.) have been submitted to ECHA in a registration dossier. According to Annex XVII, Number 42, Alkanes, C10-C13, chloro (short-chain chlorinated paraffins) shall not be placed on the market for use as substances or as constituents of other substances or preparations in concentrations higher than 1 % in metalworking and for fat liquoring of leather. The entry 42 in Annex XVII to Regulation (EC) No 1907/2006, which restricts two uses of SCCPs that are now banned under EU POP Regulation has become superfluous and is deleted.

Besides, SCCP have been included to the candidate list of substances of very high concern in 2008. Any producer or importer of articles shall notify ECHA about substances included in the candidate list, in case the following conditions are met:

- the substance is present in those articles above a concentration of 0.1 % (w/w);
- the substance is present in those articles in quantities totalling over one tonne per producer or importer per year.

Thus, only the obligations related to Article 7 (notification) and Article 33 (communication) apply regarding articles containing > 0.1% SCCP.


The classification into hazardous and non-hazardous waste is based on the system for the classification and labelling of dangerous substances and preparations, which ensures the application of similar principles over the whole life cycle of materials. The properties / conditions which render waste hazardous are laid down in Annex III of Directive 2008/98/EC.

According to Directive 2008/98/EC, “hazardous waste” means waste which displays one or more of the hazardous properties listed in Annex III of the Directive (e.g. HP14). Chlorinated paraffins C10-13 are classified, among others, as aquatic chronic 1 (H410) under the CLP Regulation, which means that any waste containing ≥ 2,500 mg/kg would be classified as hazardous.

In addition to general provisions applying to all types of waste, the WFD contains specific provisions related to the environmentally safe management of hazardous waste. Hazardous wastes pose a greater risk to the environment and human health compared to non-hazardous waste and thus require a stricter control regime. This is for instance laid down in Articles 17 to 20 of Directive 2008/98/EC, providing additional labelling and identification documents, record keeping, monitoring and control obligations, i.e. from the waste production to the final disposal or recovery. In addition, mixing of hazardous waste is banned (Article 18) to prevent risks for the environment and human health. Moreover, permit exemptions that may be granted to hazardous waste installations are more restrictive (Article 25) than those for installations dealing with other wastes (Article 23). At the same time, treatment-related legal documents such as the Landfill Directive or the Industrial Emissions Directive 2010/75/EU (IED), in its provisions on waste incineration facilities, make important distinction whether the waste to be treated is hazardous or not; obviously, the requirements for treating hazardous waste are stricter.

34 ‘Ecotoxic:’ waste which presents or may present immediate or delayed risks for one or more sectors of the environment.
4.2.4 OCCURRENCE IN PRODUCTS, WASTES AND RECYCLES

[RIVM 2010] estimates that around 1,500 t SCCP were produced, of which around 530 t were consumed within the EU27 in 2009. The remaining amount of 970 t was probably exported. As indicated, SCCPs are currently neither produced nor used in the EU (at least not above the limit values set in Regulation EU 2015/2030). As the production in the EU already ceased in 2012, remaining stocks of SCCPs are also not expected within the EU. Some historic uses with typically long life-times, may still be relevant from the waste management perspective as the corresponding waste streams may also be managed in the EU. Imports of new products and/or recycled products / materials contaminated with or containing SCCPs, also in concentrations above the allowed limit values, cannot be excluded and are relevant for the EU. In addition, a possible contribution from other CPs (e.g. MCCPs) to SCCP concentrations detected in various products (including consumer products) placed on the EU market will be further discussed.

4.2.5 SELECTION OF MOST RELEVANT WASTE STREAMS IN THE EU

SCCPs are currently neither produced nor used in the EU (at least not above the limit values set in Regulation EU 2015/2030). Regarding the two applications initially exempt under Regulation (EC) No 850/2004, a major manufacturer of conveyor belts indicated transition to alternatives and at the same time two other companies were working on alternatives. Further, already in 2013 relevant stakeholders in the mining industry confirmed that conveyor belts containing SCCPs are no longer used by the industry for mining purposes. Regarding dam sealants containing SCCPs, it was expressed that they do not appear to be manufactured, placed on the market or used in the EU. Already in 2008, the relevant stakeholders indicated to the European Chemicals Agency that SCCPs are not used or in the process of being phased out in sealants (including dam sealants) in Europe.

There are several options for reusing end-of-life conveyor belts (e.g. as protection against wear, squat or bounce protection, weather protection curtains, floor mats, etc.). This was also confirmed by a mining company. The company also stated, that the used conveyor belts from underground mining are sold to individuals and are possibly also reused in above ground applications. Two companies, specialized in rubber recycling, stated that production wastes and disposed conveyor belts (also from underground mining) can be used to produce granulates for further use in other applications [BiPRO 2015].

SCCP can be used as plasticizers in sealants and adhesives to provide for the desired hardness and elasticity of products. In combination with antimony trioxide or aluminium hydroxide, they are also effective FRs. The areas of application include polymers such as polyacrylates, polyurethanes and polysulfides for the construction industry and automotive as well as joint seals. In these applications, SCCP were typically used to substitute PCB [BiPRO 2015].

Based on the historic uses of SCCPs in the EU and considering the relatively long life-time of conveyer belts and sealants and the fact that rubber waste and construction and demolitions waste is also recycled in the EU, SCCP-containing conveyor belts and dam sealants are still expected to be relevant from the waste management perspective (including recyclates for further use). Extensive discussions on the relevance of other historic uses of SCCPs in the EU (e.g. in leather and metalworking fluids, textiles, PVC and other plastics, paints and varnishes, specialty paper, etc.) also including related substance and mass flows are available in [BiPRO 2011] and [BiPRO 2015] reports. Considering the relatively short life cycles of most of other products and the fact that the use of SCCPs has now either been banned or voluntarily ended several years ago, it can be expected that most of the SCCP-containing waste has been already disposed of and/or subject to controlled waste management in the EU (e.g. waste incineration with energy recovery).

In this connection it should also be mentioned that a technical guideline on the environmentally sound management of wastes consisting of, containing or contaminated with SCCPs is currently
being developed under the Basel Convention framework. The guidance contains in-depth information on production and use as well as on environmentally sound management of various SCCP-containing waste streams [UNEP/CHW/OEWG.11/INF/10 2018].

Even though SCCPs are currently not used in the EU, further input of SCCPs-contaminated or -containing products into the EU is to be expected from imports of new products, recycled products and possibly also from recycled materials for further processing (e.g. as granulates). According to legislation, the production, placing on the market and use of substances or preparations containing SCCPs in concentrations lower than 1 % by weight or articles containing SCCPs in concentrations lower than 0.15 % by weight is allowed. However, SCCP contents above the allowed concentration limits are regularly detected / reported in various kinds of products (including consumer products). There is recent evidence (data available for 2013-2017) on consumer products including toys, sports accessories, electric cables of kitchen equipment containing several thousands of mg/kg SCCPs which are commonly found on the EU market [UNEP/CHW/OEWG.11/INF/10 2018]. In the following, SCCP concentrations detected in various kinds of consumer products placed on the EU market are shown for the years 2016 and 2017.

Table 21: Consumer products containing SCCPs on the EU market (adopted from RAPEX database

<table>
<thead>
<tr>
<th>Year</th>
<th>Product</th>
<th>SCCP content %</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017</td>
<td>Sports equipment: Boxing gloves</td>
<td>0.44</td>
</tr>
<tr>
<td></td>
<td>Sports equipment: Gym ball</td>
<td>0.85</td>
</tr>
<tr>
<td></td>
<td>Toy pistol (plastic cord)</td>
<td>0.70</td>
</tr>
<tr>
<td></td>
<td>Toy rifle (cords)</td>
<td>0.20</td>
</tr>
<tr>
<td></td>
<td>Toy keyboard (cord)</td>
<td>1.30</td>
</tr>
<tr>
<td></td>
<td>Toy Pony</td>
<td>0.25</td>
</tr>
<tr>
<td></td>
<td>Power cord</td>
<td>2.60</td>
</tr>
<tr>
<td></td>
<td>Shoulder Bag (synthetic leather)</td>
<td>1.60</td>
</tr>
<tr>
<td></td>
<td>Training baton</td>
<td>0.37</td>
</tr>
<tr>
<td></td>
<td>Bathtub pillow</td>
<td>1.70</td>
</tr>
<tr>
<td></td>
<td>Electric shaver (cable)</td>
<td>0.98</td>
</tr>
<tr>
<td></td>
<td>Hobby/sports equipment: Hot pack</td>
<td>0.40</td>
</tr>
<tr>
<td></td>
<td>Hobby/sports equipment: Exercise tube</td>
<td>9.00 (Handles)</td>
</tr>
<tr>
<td></td>
<td>Speaker (cord)</td>
<td>1.00</td>
</tr>
<tr>
<td></td>
<td>Radio controlled car (tyres)</td>
<td>1.70</td>
</tr>
<tr>
<td></td>
<td>Claw hammer (Handle)</td>
<td>0.70</td>
</tr>
<tr>
<td></td>
<td>In-ear headphones (USB cord)</td>
<td>0.30</td>
</tr>
<tr>
<td></td>
<td>LED candle (cord)</td>
<td>1.30</td>
</tr>
<tr>
<td></td>
<td>Power cord</td>
<td>2.60</td>
</tr>
<tr>
<td></td>
<td>Table cloth</td>
<td>0.60</td>
</tr>
<tr>
<td></td>
<td>Selfie stick (cord)</td>
<td>4.57</td>
</tr>
<tr>
<td></td>
<td>USB cable</td>
<td>1.60</td>
</tr>
<tr>
<td></td>
<td>Bath toy</td>
<td>1.34</td>
</tr>
<tr>
<td></td>
<td>Game controller</td>
<td>4.30</td>
</tr>
<tr>
<td></td>
<td>Plastic doll</td>
<td>0.86</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Year</th>
<th>Product</th>
<th>SCCP content %</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Babies' sleeping bag/footmuff (packaging)</td>
<td>4.00</td>
</tr>
<tr>
<td></td>
<td>Babies' sleeping bag (anti-slip knobs)</td>
<td>1.80</td>
</tr>
<tr>
<td></td>
<td>Handle (cycle parts)</td>
<td>0.97</td>
</tr>
<tr>
<td></td>
<td>Breastfeeding pillow (packaging)</td>
<td>6.00</td>
</tr>
<tr>
<td></td>
<td>Hammer (handle)</td>
<td>0.77</td>
</tr>
<tr>
<td></td>
<td>Sports equipment: Yoga mat</td>
<td>0.80</td>
</tr>
<tr>
<td></td>
<td>Erotic article</td>
<td>0.44</td>
</tr>
<tr>
<td></td>
<td>Door gym bar</td>
<td>2.20</td>
</tr>
<tr>
<td>2016</td>
<td>Lighting chain (cord)</td>
<td>0.70</td>
</tr>
<tr>
<td></td>
<td>Sports equipment: Yoga mat</td>
<td>0.23</td>
</tr>
<tr>
<td></td>
<td>Sports equipment: Abs trainer</td>
<td>0.40</td>
</tr>
<tr>
<td></td>
<td>Steering wheel cover</td>
<td>0.46</td>
</tr>
<tr>
<td></td>
<td>Long sleeved sweater (print)</td>
<td>0.23</td>
</tr>
<tr>
<td></td>
<td>Steering wheel cover</td>
<td>0.30</td>
</tr>
<tr>
<td></td>
<td>Motor vehicle sidelight (cable)</td>
<td>0.26</td>
</tr>
<tr>
<td></td>
<td>USB-cord</td>
<td>0.26</td>
</tr>
<tr>
<td></td>
<td>Selfie Stick</td>
<td>0.16</td>
</tr>
<tr>
<td></td>
<td>Digital thermometer (cable)</td>
<td>1.10</td>
</tr>
<tr>
<td></td>
<td>Stickers (toys)</td>
<td>0.90 and 1.40 (2 Stickers)</td>
</tr>
<tr>
<td></td>
<td>Mobile phone case</td>
<td>0.44</td>
</tr>
<tr>
<td></td>
<td>Sports equipment: Baseball glove</td>
<td>1.36</td>
</tr>
<tr>
<td></td>
<td>All-purpose mat</td>
<td>0.36</td>
</tr>
<tr>
<td></td>
<td>Sports equipment: Yoga mats</td>
<td>0.35 – 6.90 (5 Yoga mats tested)</td>
</tr>
<tr>
<td></td>
<td>Sports equipment: Fitness gloves</td>
<td>0.18</td>
</tr>
<tr>
<td></td>
<td>Rain cover for pushchair</td>
<td>0.73</td>
</tr>
<tr>
<td></td>
<td>Extension lead</td>
<td>1.70 and 4.70 (2 Extension leads)</td>
</tr>
</tbody>
</table>

As shown in Table 21, the products range from sports equipment like yoga mats and boxing gloves, to electric devices, to children’s toys. The data is made available by the European Rapid Alert System (RAPEX) for dangerous non-food products. Amongst other substances, non-food products are published for which irregular SCCP levels have been reported, including a description of the level of risk, i.e. pollution, and the measures adopted by the stakeholders. The time of measurement should date to the respective year, the origin country of the products is given and if applicable, also in which of the EU MS the product was sampled and analysed. However, the analytical methods are not indicated and there is no information on the quantity of the related products that have reached the EU market. All products contained SCCP concentrations above the threshold of 0.15 % by weight for the European market, ranging from 0.18 % for boxing gloves, up to 9.0 % in the handles of exercise tubes (see Table 21).

Further, during the stakeholder consultation for this study (conducted in April 2018), information and data on SCCP concentrations in articles in use has been provided by several stakeholders including several EU MS and environmental NGOs. The most relevant information is summarized in the following, however, excluding outdated information or information already covered in previous sections of this report.

According to information submitted by the Dutch Ministry of Infrastructure and Waste management, there is no information on potential stockpiles, quantities and concentrations of SCCPs in products and wastes. SCCPs are no longer used for many years [NL MoIW Sub. 2018].
The Luxembourg Ministry of Sustainable Development and Infrastructure assumes that no stockpiles of SCCPs remained within the country. No data is however available on SCCP concentrations in articles [LU Ministry of Sust. Dev. and Infrastructure Sub. 2018].

The Lithuanian EPA states that the inventory for newly listed POPs, including SCCP, has not been completed, yet. Therefore, Lithuania could not provide data on potential SCCP concentrations in articles. Nonetheless, the agency states that according to IS AIVIKS, SCCPs have not been placed on the Lithuanian market, imported, produced, used and distributed since 2007. Neither are Lithuanian companies using such substances. IS AIVIKS, the Integrated Computerized Information System for Environmental Management, is a national database, administered by the Lithuanian Environmental Protection Agency [LT EPA Sub. 2018].

The Swedish EPA provided information on SCCP concentrations in different article categories, such as caulks and sealants, paints, fire retardants, textiles, toys, accessories and stick blenders. The data has been extracted from a 2016 report 'Särskilt farliga ämnen, avfall och materialåtervinning' (available in Swedish), and is at least partially derived from international sources. Therefore, it has been pointed out that it does not specifically reflect Swedish conditions [SE EPA Sub. 2018]. For instance, for caulks and sealants, fire retardants, and gaskets (significant SCCP concentrations of up to 70 %), the Agency is referring to a report from the Danish EPA (i.e. Danish EPA 2013), however this data originated from safety data sheets from the UK and is rather outdated (i.e. from 2005). In general, it is rather difficult to comprehend from which original sources the reported data has been obtained. Even though the information source cited is from 2016, most of the information seems to be rather outdated. Regarding the prevalence of SCCP concentrations in wastes and recycled articles, no data is available. The same applies to stockpiles.

According to the Finish Environment Institute currently there is no industry in Finland using SCCPs. Therefore, it is also assumed that no stockpiles are available either. Further, it is stated that some articles have been seized from the market in market surveillance because of SCCP concentrations exceeded the allowed concentration limits. Reference has been provided to the concentrations reported by European Authorities in the RAPEX system. No information is available on SCCPs concentrations in waste and recycled articles.

According to the Austrian Federal Ministry of Sustainability and Tourism there are no SCCP stockpiles in Austria. The Ministry reports that imported gymnastic (fitness) mats of one producer contained SCCP between 370 – 40,500 mg/kg (average 19,000 mg/kg), [AT Ministry of Sustain. and Tourism Sub. 2018a]. Further information such as sampling location, date of the analysis, analytical methods applies was not disclosed. No information is available on SCCP concentrations in wastes and recycled articles.

The environmental NGOs IPEN & Arnika provided data on measured SCCP concentrations in articles from the Czech Republic. The data is derived from a 2017 report on contaminated children’s toys. Samples originated from Brazil, Canada, China, Czech Republic, India, Japan, Kenya, Netherlands, Russia, and the United States, and were taken in the Czech Republic. The concentrations ranged from 8.4 to 19,808 mg/kg SCCP (with the highest SCCP concentration measured in products from markets in countries outside of EU).

Table 22: Concentration of SCCPs in articles within the Czech Republic [IPEN & Anika Sub. 2018]

<table>
<thead>
<tr>
<th>SCCPs</th>
<th>Specification of article</th>
<th>Concentration [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Articles in use</td>
<td>Toy - Plastic spaghetti ropes from soft PVC (obtained in Czech Republic, made in China)</td>
<td>13</td>
</tr>
<tr>
<td></td>
<td>Wallpaper with pattern from PVC (obtained in the Czech Republic) Not labelled (country of origin: Germany)</td>
<td>56.6</td>
</tr>
<tr>
<td></td>
<td>Gym ball from PVC (Tesco) (obtained in the Czech Republic; made in China)</td>
<td>9,715</td>
</tr>
<tr>
<td></td>
<td>Different products from markets in countries outside of EU (toys and children care products by majority); (Miller and DiGangi 2017)</td>
<td>8.4 – 19,808</td>
</tr>
</tbody>
</table>
As shown in Table 22, PVC samples from children’s toys and other consumer goods were analyzed. The analysis was carried out by the Institute of Chemical Technology, an accredited laboratory in the Czech Republic. SCCPs were extracted by hexane and dichloromethane mix (1:4). The extract was transferred into cyclohexane and diluted. Identification and quantification of SCCPs was accessed via gas chromatography / time-of-flight high resolution mass spectrometry (GC/TOF-HRMS) in the mode of negative chemical ionization (NCI). The instrumental measurement performed on an Agilent 7890B gas chromatograph coupled with a 7200 QTOF mass spectrometer (both Agilent Technologies, USA) in negative chemical ionization (NCI).

Stockpiles are not expected to occur according to the IPEN & Arnika Sub. (2018).

The Norwegian Environment Agency provided quantitative information for articles in use showing SCCP concentrations ranging from 250 to 66,000 mg/kg. Besides, for most of the articles analysed, detected levels of MCCPs have been indicated as well (see Table 23).

<table>
<thead>
<tr>
<th>SCCPs</th>
<th>Specification of article</th>
<th>Concentration [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Articles in use</td>
<td>Pullover with print</td>
<td>2,300 (MCCP 9,100)</td>
</tr>
<tr>
<td></td>
<td>T-shirt with print</td>
<td>1,300 (MCCP 700)</td>
</tr>
<tr>
<td></td>
<td>Sports gloves</td>
<td>250 and 260 (two samples)</td>
</tr>
<tr>
<td></td>
<td>Sports textile</td>
<td>4,000 (MCCP 6,500)</td>
</tr>
<tr>
<td></td>
<td>Sports yoga mats</td>
<td>2,300 (MCCP 17,000)</td>
</tr>
<tr>
<td></td>
<td>Textile car</td>
<td>3,000 (MCCP 3,100)</td>
</tr>
<tr>
<td></td>
<td>Textile car</td>
<td>860 (MCCP 1,500)</td>
</tr>
<tr>
<td></td>
<td>Textile car</td>
<td>4,600 (MCCP 2,200)</td>
</tr>
<tr>
<td></td>
<td>Children product pillow packaging</td>
<td>60,000 (MCCP 27,000)</td>
</tr>
<tr>
<td></td>
<td>Children stroller bag plastic anti-slip mats</td>
<td>18,000 (MCCP 130,000)</td>
</tr>
<tr>
<td></td>
<td>Children stroller bag packaging</td>
<td>40,000 (MCCP 110,000)</td>
</tr>
<tr>
<td></td>
<td>Foam textile children car seat</td>
<td>4,300 (MCCP 12,000)</td>
</tr>
<tr>
<td></td>
<td>Reflective bands</td>
<td>6,700 (MCCP 38,000)</td>
</tr>
<tr>
<td></td>
<td>Raincoat hat</td>
<td>14,000 (MCCP 73,000)</td>
</tr>
<tr>
<td></td>
<td>USB-speaker wire</td>
<td>10,000 (MCCP 16,000)</td>
</tr>
<tr>
<td></td>
<td>Reflective bands</td>
<td>4,900</td>
</tr>
<tr>
<td></td>
<td>Reflective bands</td>
<td>8,700 (MCCP 7,600)</td>
</tr>
<tr>
<td></td>
<td>Reflective bands</td>
<td>2,500 (MCCP 40,000)</td>
</tr>
<tr>
<td></td>
<td>Duffy bag plastic</td>
<td>2,800 (MCCP 18,000)</td>
</tr>
<tr>
<td></td>
<td>Powerbank wire</td>
<td>32,000 (MCCP 72,000)</td>
</tr>
<tr>
<td></td>
<td>Childrens Jacket (plastic detail)</td>
<td>66,000</td>
</tr>
<tr>
<td></td>
<td>Childrens Jacket (plastic detail)</td>
<td>5,100 (MCCP 7,600)</td>
</tr>
<tr>
<td></td>
<td>Childrens pencil case</td>
<td>1,600 (MCCP 3,400)</td>
</tr>
<tr>
<td></td>
<td>Shoe</td>
<td>9,000</td>
</tr>
<tr>
<td></td>
<td>Plastic football goal</td>
<td>28,000</td>
</tr>
<tr>
<td></td>
<td>Rain hat</td>
<td>43,000</td>
</tr>
<tr>
<td></td>
<td>Purse</td>
<td>16,000</td>
</tr>
<tr>
<td></td>
<td>Door gym power trainer</td>
<td>22,000</td>
</tr>
<tr>
<td></td>
<td>Light chain</td>
<td>29,000 (MCCP 74,000)</td>
</tr>
<tr>
<td></td>
<td>Light chain</td>
<td>3,800 (MCCP 56,000)</td>
</tr>
<tr>
<td></td>
<td>Light chain</td>
<td>3,200 (MCCP 20,000)</td>
</tr>
<tr>
<td></td>
<td>Light chain</td>
<td>4,900 (MCCP 53,000)</td>
</tr>
<tr>
<td></td>
<td>Light chain</td>
<td>5,200 (MCCP 48,000)</td>
</tr>
<tr>
<td></td>
<td>Light chain</td>
<td>6,800 (MCCP 47,000)</td>
</tr>
<tr>
<td></td>
<td>Slippers</td>
<td>7,900 (MCCP 9,100)</td>
</tr>
<tr>
<td></td>
<td>Slippers</td>
<td>4,500</td>
</tr>
<tr>
<td></td>
<td>Sports wrist band</td>
<td>5,400 (MCCP 3,400)</td>
</tr>
</tbody>
</table>

The articles range from clothing, sports equipment, car parts and children’s articles to electric devices. The article origin and the exact date of sampling and measurements is not indicated, but it can be assumed that samples have been obtained from the Norwegian market. Further, as indicated by the Norwegian Environment Agency (2018), no data is available for SCCP concentrations in wastes and recycled articles. Furthermore, the Agency confirms that there are no known SCCP stockpiles within Norway.
It can be assumed that the Norwegian Environment Agency used analytical methods capable of distinguishing between SCCPs and MCCPs. However, the used analytical methods have not been indicated within the information provided by Norway.

Use of MCCPs may lead to high concentrations of SCCPs in the products and articles (the EU limit for UTC was set at 1,500 mg/kg in articles to allow for continued use of MCCPs). MCCPs containing up to 1% by weight SCCPs, are commonly used as alternatives to SCCPs [UNEP/CHW/OEWG.11/INF/10 2018]. Further, the SCCP Risk management evaluation (RME) Dossier (2016) point out that more recently in North America, manufacturers have further divided long-chain feedstocks (C18+) into those used to produce LCCPs (C18-20) and those used to produce very long-chain CPs (C20+). In other regions, the chain length composition of feedstocks can vary significantly, for example, China produces a CP mixture with chain lengths ranging from C10 to C20. As such, the feedstocks used to manufacture CP mixtures may contain other carbon chain lengths outside the defined ranges, which affect the composition of the CP mixture that is produced. A wide-ranging feedstock (i.e., C10 to C20) or a feedstock that contains trace amounts of short-chain lengths may result in CP mixtures that contain SCCPs [UNEP/POPS/POPRC.12/11/Add.3 2016].

This is partly in contrast with the information provided by the industry association Euro Chlor [Euro Chlor Sub. 2018] during the stakeholder consultation for this study, stating that the short carbon chain constituents in MCCPs are not likely SCCPs as defined in REACH Regulation and the detection of chains below C14 in an MCCP product does not mean that the product contains SCCPs. In addition, Euro Chlor doubts whether much of the material identified in waste comes from deliberate use of SCCPs in articles. According to Euro Chlor, it is most likely that wider C10-C13 fragments in C10-20 cuts (which is common outside of Europe) are detected. Hence, detected 'shorter' fragments must be there as part of a wider cut component and are not SCCP under the CAS definition [Euro Chlor Sub. 2018]. Further, Euro Chlor explains that for MCCPs (and MCCP products), when small amounts of chlorinated components, that are less than 14 carbon atoms long are found, they are not SCCP but rather <C14 constituents that are part of the registered MCCP product. SCCPs and MCCPs are not blended together, nor are C13 components added to MCCPs. MCCPs are not defined as a mixture under REACH but are defined as substances. It is further stipulated that when distinguishing between SCCP and short carbon chain components of other longer chain unknown or variable composition, complex reaction products or biological materials substances, regulatory action on (and definition of) SCCPs is on the substance as defined by CAS No 85535-84-8/EINECS 287-476-5 which have been tested fully. As mentioned before, this includes chain lengths from, predominantly, C10 to C13 and the concentration of longer chain chlorinated alkane products (such as MCCP or LCCP is always zero. MCCPs are also defined by particular CAS and EINECS Numbers (85535-85-9/287-477-0). These are described by chain lengths of, predominantly, C14 to C17. There will be some constituents of MCCPs which lie outside this range. However, they are part of the substance and will have been present when the substance was tested in regulatory required assessment [Euro Chlor Sub. 2018].

In this connection, Euro Chlor also states that many state laboratories use standards for comparison which are based on chlorinated end groups which may add to the detection problem. The laboratories detect matches for the C10-13 standard (which do not reflect historic, commercially produced SCCP) and conclude that SCCP has been detected. The most commonly

36 According to the European chlor-alkali industry association Euro Chlor, the short carbon chain constituents in MCCPs are not likely SCCPs as defined in REACH Regulation and the detection of chains below C14 in an MCCP product does not mean that the product contains SCCPs (http://www.eurochlor.org/chlorinated-alkanes-(casg)/education-spotlight.aspx, accessed 3 December, 2017)
used method of detection and quantification is either high or low definition gas chromatography followed by electron capture negative ion mass spectrometry (GC-ECNI-MS). Whilst popular, this method has difficulty in accurately separating different congeners with the same chlorine number, and the detection of congeners containing low numbers of chlorine atoms (∆Cl5). State-of-the-art CA detection uses 2-dimensional gas chromatography combined with electron capture detection (GCxGC-ECD). The GCxGC separation method can qualitatively identify groups of CA isomers by carbon chain length and chlorination level, although this is very difficult due to the complex nature of chlorinated alkanes. Advantages of this technique include the detection of lower chlorinated congeners, the high separation power of congeners with different chlorination levels and the ability to detect groups of congeners with equal chlorine levels.

Further information on analytical methods as well as recent developments including novel analytical methods is presented in the dedicated Chapter 6.2.1.1 (see section on SCCPs).

In the following, main findings are summarized related to SCCP stockpiles, relevant products, wastes and recyclates in the EU.

Stockpiles

Based on the available information, it can be concluded that SCCPs are currently neither produced nor used in the EU (at least not above the limit values defined in Annex I of Regulation EU 2015/2030). As the production in the EU ceased some years ago, remaining stocks are not likely to occur in the EU. This has been also confirmed by several stakeholders during the consultation phase for this study, including EU MS, industry representative (i.e. Euro Chlor) and Environmental NGOs (IPEN and Arnica), stating that remaining stockpiles of SCCPs are not expected.

Products (incl. potential contribution from MCCP-containing products)

As mentioned, SCCPs are currently not used in the EU, at least not above the limit values defined in Annex I of Regulation EU 2015/2030. The high concentrations reported in recent literature sources show data that often dates back up to twenty years. For instance, some of the data reported in [UNEP/CHW/OEWG.11/INF/10 2018] providing examples of concentration of SCCPs in materials and articles refers to data from other sources (i.e. ECB 2008) and is often subtracted from original but even older publications. Looking at the more recent report from the Danish EPA (2014) cited in [UNEP/CHW/OEWG.11/INF/10 2018], reporting SCCP concentrations up to 300,000 mg/kg in sealants and adhesives, it becomes apparent that the source quotes data from another report, which refers to information from safety data sheets from the UK from 2005.

Regarding the two applications initially exempt under Regulation (EC) No 850/2004, in 2013 relevant stakeholders in the mining industry confirmed that conveyor belts containing SCCPs are no longer used. Regarding dam sealants containing SCCPs, already in 2008, the relevant stakeholders indicated to the European Chemicals Agency that SCCPs are not used or in the process of being phased out in sealants (including dam sealants) in Europe33.

Considering that much of the data reported in recent literature is often outdated (representing application areas where SCCPs were typically used) and that the last two allowed uses in the EU ended around 2008 and 2013, it can be expected that currently SCCPs are not deliberately used in the EU in their former typical application areas where they were used in the past (including metal working fluids, in PVC, rubber, adhesives and sealants, leather, etc.). It has been demonstrated that technically feasible alternatives are commercially available for all known uses of SCCPs.

However, inspection and enforcement activities carried out in EU MS where SCCPs are banned have found the continued presence of SCCPs in articles (including consumer products). These findings demonstrate that new products continue to be a source of SCCPs and contribute to human and environmental exposure. Considering that currently SCCPs are expected not to be deliberately used in the EU, the relevant products and/or recycled products contaminated with or containing
SCCPs (also above permitted levels) are probably associated with imported articles and materials into the EU market.

In addition, a possible contribution from other CPs (e.g. MCCPs) to SCCP concentrations detected in various products cannot be generally excluded. Stakeholders’ opinion on this topic differ largely. This is mainly due to the acknowledged complexity of the mixtures, making the chemical analysis challenging. In the absence of more complete characterizations of the mixtures and suitable individual standards, quantification is usually based on a technical product, introducing major uncertainties if compositions of the sample and the standard do not match [UNEP/CHW/OEWG.11/INF/10 2018].

Regulation EC 2015/2030 defines concentration limits on the presence of SCCPs in products (1.0 %) and articles (0.15%). The Regulations specifically states that articles that contain SCCPs in concentrations lower than 0.15 % by weight can be placed on the market and used, as this is the amount of SCCPs that may be present as an impurity in an article produced with MCCPs. Therefore, a possible contribution of MCCPs to SCCP concentration in articles, wastes and recyclates cannot be excluded. Further development of reliable analytical method would allow to quantify possible contribution of MCCPs to SCCP concentrations in wastes and recyclates.

Waste

In general, information and data on SCCP concentrations in waste and recyclates is limited to information / measurements contained in [BiPRO 2015]. All uses of SCCPs have been phased out in EU MS for years and more recently, the remaining uses in rubber conveyor belts and sealants and adhesives have been also replaced with alternatives, indicating that technically feasible alternatives exist, are accessible and available [UNEP/POPS/POPRC.12/11/Add.3 2016].

Nevertheless, based on the historic uses of SCCPs in the EU, the relatively long life-time of conveyor belts and sealants and the fact that rubber waste and construction and demolitions waste can be recycled, SCCP-containing conveyor belts and sealants and adhesives are still expected to be relevant in the EU from the waste management perspective (including recyclates for further use). Extensive discussions on the relevance of other historic uses of SCCPs in the EU (e.g. in leather and metalworking fluids, textiles, PVC and other plastics, paints and varnishes, specialty paper, etc.) also including detailed substance and mass flows are available in [BiPRO 2011] and [BiPRO 2015] reports. Considering the relatively short life cycles of most of other articles and the fact that the use of SCCPs has either been banned or voluntarily ended several years ago, it is expected that most of the SCCP-containing waste has already disposed of and/or subject to controlled waste management in the EU (e.g. incineration with energy recovery). Therefore, in the following chapters, additional information will be provided related to conveyor belts and sealants and adhesives including the development of respective substance and mass flows. Due to their comparably limited relevance other historic uses are not further considered in this report.

In addition, possible imports of new products and/or recycled products contaminated with or containing SCCPs (also in concentrations above the set limit values in Regulation EU 2015/2030) cannot be excluded. It can be expected that most of those articles will be landfilled or incinerated together with municipal solid waste streams; however, a possible recycling of respective articles cannot be generally excluded. However, up-to-date information on recycling activities in EU MS and quantitative information on SCCP concentrations and amounts in waste and recyclates is not available. No information on SCCP concentrations in waste and recyclates could be obtained during the stakeholder consultation for this project (April 2018). In addition, considering the large diversity of imported new articles containing SCCPs and the fact that imports cannot be traced it is not possible to further investigate and develop substance and mass flows for the respective waste streams. Further development of reliable analytical methods and sampling and analysis of waste streams will be needed to obtain relevant data on EU level.
Recyclates

Rubber waste and construction and demolitions waste is being managed in the EU (including recycling). Therefore, recyclates from SCCP-containing conveyor belts and sealants and adhesives are still expected to be relevant as recyclates for further use in other application areas. Additional information on recyclates from conveyor belts and sealants and adhesives will be provided in subsequent chapters. Additionally, as already mentioned, new products entering the EU market continue to be a source of SCCPs. A possible recycling of respective articles cannot be excluded, however, up-to-date information on recycling activities in the EU and data on SCCP concentrations and quantities is not available. Considering the large diversity of imported articles, the fact that imports cannot be traced and the absence of quantitative data, it is not possible to further investigate and develop substance and mass flows for the respective recyclates. In addition, a potential contribution of MCCPs to SCCP concentrations in recyclates cannot be generally excluded but is currently not quantifiable. Further development of reliable analytical methods and sampling and analysis of recyclates streams will be needed to obtain relevant data on EU level.

4.2.6 RUBBER (CONVEYOR BELTS)

4.2.6.1 Background information

SCCP were mainly used for the manufacture of flame retarded conveyor belts for the use in underground mining (75-90 \% of the total SCCP use in the rubber industry). Different rubbers can be used for this purpose (e.g. styrene, butadiene, PVC). For the use in underground mining, the conveyor belts are mainly made with a textile or steel carcass [RIVM 2010]. In the following, the use of SCCP in conveyor belts for underground mining if further elaborated. Other possible areas of applications within the rubber industry (e.g. acoustic insulation material, gasket and hoses) are not further considered, due to their wide dispersive areas of application and the comparably low volumes.

4.2.6.2 Processes, inputs and outputs

![Figure 16: Inputs and outputs of SCCP in the production of conveyor belts]

SCCP are additive FRs and therefore physically incorporated into the rubber matrix. SCCP can also be used with other FRs. The carcass (mostly metal of textile carcass) is another input to the production process. The main output streams are the flame retarded conveyor belts and process wastes generated during the production. It is assumed, that process waste is recovered in the production.
4.2.6.3 Concentration in products, wastes and recyclates

The SCCP content in rubber is expected to be about 10 %, which again has a total share of around 33 % within the conveyor belt [RIVM 2010]. More up-to-date information on SCCP concentrations in wastes and recyclates is not available. No information on SCCP concentrations in wastes and recyclates could be obtained during the consultation phase for this project (April 2018).

4.2.6.4 Activity data

Already in 2013 relevant stakeholders in the mining industry confirmed that conveyer belts containing SCCPs are no longer used in Europe. [BiPRO 2011] estimates that around 1,100 t SCCP used in conveyor belts were generated and managed in the EU in 2013. Recent data on the generation and waste management of used conveyor belts in the EU is not available.

Information regarding the average product life time of conveyor belts varies. ECHA estimated an average life time of around 10 years, whereas other literature sources estimate it to be about 15 years. Conveyor belt manufacturers indicate a service life between 2 and 30 years [BiPRO 2011]. In [BiPRO 2011] a lognormal distribution with an average life time of 12.5 years was used with a standard deviation of 2.5 years for the estimations of the waste generation.

Under the assumption, that since 2013 SCCPs were not used in new conveyor belts it can be expected that also the remaining associated waste amounts dropped during the last years. However, the exact waste amounts to be treated in 2018 are not known. In the absence of newer data, it is assumed that the estimated SCCP tonnage bond in conveyor belts remained constant during the last 5 years in the EU (~1,100 t SCCP contained in waste from used conveyor belts). With an average SCCP concentration of 10 % in rubber, consequently around 11,000 t of SCCP-containing rubber waste from conveyor belts is expected to be subject to waste management in the EU in 2018.

4.2.6.5 Waste management

According to [BiPRO 2015] end of life rubber conveyor belts can be used for various purposes (as wear protection, start-up or bounce protection, weather protection curtains, floor mats, etc.). This was also reconfirmed by a mining company during the research work. In addition, the company stated that the used conveyor belts are also sold to individuals and may probably also be re-used.

Two companies, who specialized in rubber recycling also confirmed, that production waste as well as disposed conveyor belts from underground mining are used to produce granulates. The granulates can for instance be used to produce floors for sports fields, to produce mats, etc. When choosing the proper recovery techniques for rubber waste, the degree of contamination of rubber is crucial. Problems may arise when rubber waste with a high content of metals is processed. The recycling is only possible if a small number of metal impurities are contained in the rubber. Larger metal parts would damage the blades of shredders and grinders. Although the steel fraction can be separated during the recycling process, the presence of steel in conveyor belts can cause problems [BiPRO 2015]. In the absence of more detailed information on the management of used conveyor belts it is assumed that the waste from conveyor belts is managed together with other rubber wastes (e.g. used tires, etc.).

According to the newest data available from [Eurostat 2018f] (newest data available from 2014), rubber waste in the EU is treated as follows (see Table 24).

Table 24: Waste management of rubber waste in the EU (adopted from [Eurostat 2018f])

<table>
<thead>
<tr>
<th>Waste management operations for rubber waste in the EU</th>
<th>Waste amounts managed in the EU [t]</th>
<th>[%]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Disposal - landfill and other (D1-D7, D12)</td>
<td>20,000</td>
<td>0.76</td>
</tr>
</tbody>
</table>
### 4.2.6.6 Substance and mass flow

Around 53.61% of the rubber waste is recovered (without R1), around 44.87% is incinerated with energy recovery (R1), around 1% is incinerated without energy recovery (D10) and approximately the same waste amount is disposed (landfill and other) in the EU in 2014 (newest data available from [Eurostat 2018]).

Assuming that the quantities of treated rubber waste have remained constant since 2014, and that the applied recovery and disposal management operations are similar, it is estimated that around 589.7 t of SCCPs is recovered, around 493.6 t incinerated with energy recovery (R1) and only around 8.4 t incinerated without energy recovery (D10) and landfilled, respectively (see Figure 17).

![Substance and mass flow diagram (SCCP in rubber waste)](image)

In relation to waste streams, around 11,000 t rubber waste from SCCP-containing conveyor belts is expected to be managed in the EU in 2018. When considering that the SCCP-containing rubber waste stream is managed together with all other rubber wastes (i.e. 2,619,000 t), the theoretical SCCP concentration in the mixed waste stream decreases to around 420 mg/kg.

Around 1,409,943 t (~53.61%) of the relevant rubber waste is recovered, around 1,180,081 t (~44.87%) is incinerated with energy recovery (R1) and only comparably low waste amounts of around 8.36 t each (~0.76%) is incinerated without energy recovery and landfilled, respectively.
Due to high incineration temperatures during waste incineration with and without energy recovery, it can be expected that the SCCP content in rubber waste will be destroyed. During the recovery, it cannot be excluded that SCCPs enter recycle streams and accumulate in recyclates and new products.

### 4.2.7 SEALANTS AND ADHESIVES

#### 4.2.7.1 Background information

SCCP were used as plasticisers in sealants and adhesives. In combination with antimony trioxide or aluminium hydroxide, they are also effective FRs. The application areas include polymers such as polyacrylate, polyurethanes and polysulphides in the construction industry and the automotive sector as well as joint sealing. SCCPs were also used as a substitute for PCB [BiPRO 2015].

#### 4.2.7.2 Processes, inputs and outputs

![Diagram of Inputs and outputs in the production of SCCP-containing sealants and adhesives]

SCCP were used to produce sealants and adhesives, especially for the application in the construction sector. The polymer (e.g. polyacrylate, polyurethanes) is a further input variable to the production process. The main outputs are the SCCP-containing sealants and adhesives and process wastes, which may arise during production (see Figure 18).

#### 4.2.7.3 Concentration in products, wastes and recyclates

SCCP were typically used in a concentration ranges of about 5-14 % in sealants and adhesives. However, the concentrations may also be higher (e.g. 20 %). [BiPRO 2011] considers an average SCCP concentration of 20 % for the estimation of the relevant waste streams in the EU which will be also used in the following to estimate related waste amounts in the EU.

#### 4.2.7.4 Activity data

[BiPRO 2011] assumes an average product life cycle time of 17 years for the estimation of corresponding waste amounts in the EU. The SCCP amount in waste disposed in the EU27 was estimated to be about 412 t in 2010. Under the assumption, that since 2008, SCCPs were not used in new sealants and adhesives, it can be expected that also the remaining associated waste amounts dropped. However, the exact waste amounts to be managed in 2018 are not known. In the absence of more up-to-date information, it is assumed that the estimated SCCP tonnage in waste sealants and adhesives remained constant during the last 10 years in the EU. With an average SCCP concentration of 20 % in sealants and adhesives, consequently around 2,060 t of...
SCCP containing sealants and adhesives is expected to be subject to waste management in the EU in 2018.

4.2.7.5 Waste management

In the absence of precise information regarding the waste management of sealants and adhesives waste in the EU, and considering their properties, it is assumed that a certain proportion of sealants and adhesives used in the construction sector adheres to the surface of building materials (e.g. on concrete, tiles, bricks). In practice, it cannot be expected that the sealants and adhesives can be completely separated and managed separately from other construction and demolition waste. For further estimations, it is assumed that around 2/3 of the SCCP-containing sealants and adhesives are treated together with other construction and demolition wastes. In case the sealants and adhesives can be successfully separated, it is assumed that this waste will be subsequently incinerated (e.g. in cases of suspected PCB-containing joint sealants). Therefore, it is assumed that the rest of SCCP-containing sealants and adhesives (approximately 1/3) is separately managed.

According to the newest data available from [Eurostat 2018e] (newest data available from 2014), mineral waste from construction and demolition in the EU is treated as follows (see Table 25).

Table 25: Waste management of mineral waste from construction and demolition in the EU (adopted from [Eurostat 2018e])

<table>
<thead>
<tr>
<th>Waste management operations for rubber waste in the EU</th>
<th>Waste amounts managed in the EU</th>
</tr>
</thead>
<tbody>
<tr>
<td>Disposal - landfill and other (D1-D7, D12)</td>
<td>32,420,000 t</td>
</tr>
<tr>
<td>Disposal - incineration (D10)</td>
<td>140,000 t</td>
</tr>
<tr>
<td>Recovery - energy recovery (R1)</td>
<td>1,440,000 t</td>
</tr>
<tr>
<td>Recovery - recycling and backfilling (R2-R11)</td>
<td>242,910,000 t</td>
</tr>
<tr>
<td>Total waste managed</td>
<td>276,910,000 t</td>
</tr>
</tbody>
</table>

4.2.7.6 Substance and mass flow

Assuming that the amount of mineral waste treated from construction and demolition remained constant since 2014 and that the applied waste management operations have not changed, it can be estimated that around 240.95 t SCCPs are recovered (incl. recycling and backfilling), approximately 32.17 t SCCPs landfilled (also including other disposal operations), about 1.43 t SCCPs incinerated with energy recovery and only 0.14 t SCCPs incinerated without energy recovery. Besides, the successfully separated SCCP-containing waste stream is disposed of in hazardous waste incinerators (~137.3 t SCCPs).

In relation to waste streams, approximately 277 million tonnes of mineral waste from construction and demolition is expected to be managed in 2018. A total of about 243 million tonnes is recovered, about 32.4 million tonnes disposed off (incl. landfilling), 1.4 million tonnes is incinerated with energy recovery and about 140 kt is incinerated without energy recovery.

When considering that around two thirds of the SCCP-containing waste is managed together with wastes such as concrete, tiles, bricks and ceramics (~277 mio. t), the theoretical SCCP concentration in the mixed waste stream decreases considerably (~1.0 mg/kg). For the separately managed waste fraction, it can be considered that, due to high incineration temperatures (>1,000 °C) during hazardous waste incineration, the high SCCP content in waste (~200,000 mg/kg) is destroyed. This also applies for the mixed waste stream which is incinerated with and without energy recovery.
With regard to the recycling and landfilling, it cannot be excluded that SCCPs enter recycle streams or are released to the environment. The theoretically concerned waste amount would be significant (~275 mio. t), while the SCCP concentration within the mixed waste stream rather low (~1.0 mg/kg) (see Figure 25).

Figure 19: Substance and mass flow diagram (SCCP in sealants and adhesives)
4.3 Hexachlorobutadiene (HCBD)

Hexachlorobutadiene (HCBD), as a halogenated aliphatic compound, is mainly generated as a by-product in the manufacture of chlorinated solvents or hexachlorocyclopentadiene. Anthropogenic emission sources of HCBD are intentional production, unintentional production and waste from historical application and related potentially contaminated sites [UNEP/POPS/COP.8/INF/18 2017; UNEP/CHW.13/6/Add.2/Rev.1 2017].

4.3.1 PRODUCTION, TRADE AND USE

Production

Anthropogenic production sources of HCBD are the targeted production and the unintentional generation. There are no natural sources of HCBD in the environment [UNEP/CHW.13/6/Add.2/Rev.1 2017].

HCBD was first prepared in 1877 by chlorination of hexyl oxide. For intentional production, HCBD is separated from the heavy fraction of the production of these solvents and was, and possibly is marketed for the various uses [UNEP/POPS/COP.8/INF/18 2017].

In Europe the commercial production of HCBD ended in the late 1970s [UNEP/CHW.13/6/Add.2/Rev.1 2017; UNEP/POPS/POPRC.8/16/Add.2 2012].

HCBD is neither reported as a High Production Volume Chemical (HPVC) nor as a Low Production Volume Chemical (LPVC) by the EU Industry within the European chemical Substances Information System (ESIS) [UNEP/POPS/COP.8/INF/18 2017].

There are suspicions, that it has been produced in the former USSR. Reported common trade names were Dolen-Pur; C-46, UN2279 and GP-40-66:120. Possible remaining intentional production (particularly in quantities below the limits for high-production volumes) in other regions of the world cannot be excluded [UNEP/CHW.13/6/Add.2/Rev.1 2017].

Intentional production of HCBD is prohibited in Canada, the European Union, Mexico and in Japan [UNEP/CHW.13/6/Add.2/Rev.1 2017]. HCBD is not known to currently being intentionally produced in Europe, Canada, Japan and the USA and it has never been manufactured as a commercial product in the United States of America (USA) or Canada [UNEP/CHW.13/6/Add.2/Rev.1 2017; UNEP/CHW.13/6/Add.2/Rev.1 2018]. The UNEP POPRC Committee estimated that there is no intentional production of HCBD anymore in the entire UNECE region, even though the production and use, especially of small amounts, cannot be excluded [UNEP/POPS/POPRC.8/16/Add.2 2012].


Table 26 summarises data on intentional HCBD production and important milestones in the EU.

Table 26: Intentional HCBD production and important milestones in the EU

<table>
<thead>
<tr>
<th>Year</th>
<th>Intentional production</th>
<th>Milestones</th>
<th>Source</th>
</tr>
</thead>
</table>

37 Substance with a production or import volume in excess of 1,000 tonnes/year

38 Substance with a production or import volume between 10 and 1,000 tonne/year
Unintentional Production

Information on unintentional HCBD production is scarce. HCBD is unintentionally formed and released from industrial processes and other sources. These include the production of certain chlorinated hydrocarbons, the production of magnesium, and incineration processes [UNEP/POPS/POPRC.12/6 2016].

High volumes were produced unintentionally in chlorination processes involving organic compounds during the 1970s and 1980s. The worldwide unintentional production or manufacture as a by-product in industrial processes was 10,000 t in 1982 [UBA 2006].

In the USA, the estimated annual HCBD unintentional generation was 3,600 tonnes in 1975, and 12,000 tonnes in 1982 [US EPA 2003]. In 2000, 15,000 tonnes of HCBD was produced unintentionally in the USA [Euro Chlor 2004].

In the European Union in 1980 (EU-10), about 10,000 tonnes of HCBD were generated unintentionally [BUA 1991; UNEP/CHW.13/6/Add.2/Rev.1 2017]. In 1990, an unintentional HCBD formation quantity of 2,000 to 49,900 tonnes was estimated based on the production volumes of perchloroethylene and tetrachloromethane in Western Europe [BUA 1991; BUA 2006]. Table 27 summarises the unintentional HCBD production and important milestones in the EU.

Table 27: Unintentional HCBD production and important milestones in the EU [BiPRO 2015]

<table>
<thead>
<tr>
<th>Year</th>
<th>Unintentional production [t]</th>
<th>Milestones</th>
</tr>
</thead>
<tbody>
<tr>
<td>1980</td>
<td>10,000</td>
<td>Unintentional production in the EU (EU-10)</td>
</tr>
<tr>
<td>1990</td>
<td>2,000 - 49,900</td>
<td>Estimation based on the production volumes of perchloroethylene and tetrachloromethane in Western Europe</td>
</tr>
</tbody>
</table>

More recent estimates, considering releases of HCBD during production of chlorinated processes, show drastic decreases in possible releases compared to 1997 in Europe [UNEP/POPS/POPRC.12/6 2016].

No publications on levels unintentionally produced HCBD in air emissions from industrial magnesium production could be identified. The related unintentionally produced HCBD quantities are thus unknown [UNEP/POPS/POPRC.12/INF/12 2016].

For the unintentional secondary production of HCBD during waste incineration neither total numbers of waste sites affected, nor their releases could be identified [UNEP/POPS/POPRC.12/INF/12 2016].

Trade

In accordance with Regulation EC No 850/2004 of the European Parliament the trading of HCBD is prohibited in the European Union and thus since 2012 in Germany.

Use
The use of HCBD is prohibited within the European Union since 2012 [UNEP/POPS/POPRC.9/13/Add.2 2013], thus all applications of HCBD have ceased [UNEP/CHW.13/6/Add.2/Rev.1 2017]. The same applies for all Parties to the Stockholm Convention and the same provision, of prohibition and/or elimination, also applies to the use of unintentionally produced HCBD [UNEP/CHW.13/6/Add.2/Rev.1 2017].

In the past, HCBD was mainly used as a chemical intermediate in the manufacture of rubber compounds and lower quantities were used as solvent, fluid for gyroscopes, heat transfer liquid, hydraulic fluid, chemical intermediate in the production of chlorofluorocarbons and lubricants, laboratory reagent and a non-flammable insulating liquid in transformers [UNEP/CHW.13/6/Add.2/Rev.1 2017].

Prior to 1975 the largest use of HCBD in the USA was as an absorbent to retain gas impurities in chlorine-containing gas or to remove volatile organic components from gas in the chlorine manufacturing process (gas scrubber) [UNEP/CHW.13/6/Add.2/Rev.1 2017].

Other uses of HCBD include the use as a seed-dressing fungicide or insecticide in vineyards in the former USSR (application rate of 100-350 kg/ha), in Mediterranean European countries (France, Italy, Greece and Spain) and in Argentina. In France the use of HCBD as fumigant was extensive and discontinued in 2003. It is unclear whether HCBD is still used as a plant protection product anywhere [UNEP/CHW.13/6/Add.2/Rev.1 2017].

According to [UBA 2006] HCBD has also been applied in the past as biocide for prevention of algae production in industrial reservoirs and cooling water systems. However, this use is not mentioned in the technical guidelines on the environmentally sound management of waste consisting of, containing or contaminated with HCBD [UNEP/CHW.13/6/Add.2 Rev.1 2017].

A method using HCBD to synthesize graphite sheets has been developed relatively recently. Graphite flakes are used as electronically conducting fillers in the production of conducting polymer composites in various fields such as fuel cell electrodes, corrosion resistant materials, batteries etc. However, there is no information on whether HCBD is actually used for this purpose anywhere [UNEP/CHW.13/6/Add.2 Rev.1 2017].

Table 28 summarises the HCBD uses.

<table>
<thead>
<tr>
<th>Use</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chemical intermediate in the production of rubber compounds</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>Solvent</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>Fluid for gyroscopes</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>Heat transfer fluid</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>Hydraulic fluid</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>Chemical intermediate in the production of chlorofluorocarbons and</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>lubricants</td>
<td></td>
</tr>
<tr>
<td>Laboratory reagent</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>Non-flammable insulating liquid in transformers</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>Absorbent to retain gas impurities in chlorine-containing gas or</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>to remove volatile organic components from gas in the chlorine</td>
<td></td>
</tr>
<tr>
<td>manufacturing process (gas scrubber)</td>
<td></td>
</tr>
<tr>
<td>Plant protection product in viticulture (France, Italy, Greece and</td>
<td>[UNEP/CHW.13/6/Add.2 Rev.1 2017]</td>
</tr>
<tr>
<td>Argentina)</td>
<td></td>
</tr>
</tbody>
</table>
4.3.2 CHEMICAL CHARACTERISTICS

HCBD is a toxic and bioaccumulative organic compound belonging to the halogenated aliphatic unsaturated hydrocarbons. It appears as a clear, colourless, oily liquid with a mild turpentine-like odour, which is non-inflammable, poorly soluble in water as well as marginally volatile [UBA 2006; Euro Chlor 2004]. Below an overview of the chemical formula, the CAS-number and the EINECS number is given. Table 4 highlights the synonyms and trade-names of HCBD.

Common name: Hexachlorobutadiene
IUPAC name: \(1,1,2,3,4,4\)-hexachlorobuta-1,3-diene
Chemical formula: \(C_4Cl_6\)
CAS registry number: 87-68-3
EINECS Number: 201-765-5
Molecular mass: 260.76 g/mol [UNEP/POPS/POPRC.8/3 2012]

Table 29: Names, CAS numbers and trade names for decaBDE

<table>
<thead>
<tr>
<th>Name</th>
<th>decaBDE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Synonyms</td>
<td>HCDB, perchloro-1,3-butadiene, perchlorobutadiene, 1,3 hexachlorobutadiene, 1,1,2,3,4,4 hexachlorobuta-1,3-diene, hexachlorobuta-1,3-diene, tripen [UNEP/POPS/POPRC.8/16/Add.2 2012]</td>
</tr>
<tr>
<td>Trade names</td>
<td>C-46, Dolen-pur, GP40-66:120, UN2279 [UNECE 2007; UNEP/POPS/POPRC.8/16/Add.2 2012].</td>
</tr>
</tbody>
</table>

![Chemical structure of HCBD](image)

Figure 20: Chemical structure of HCBD

Table 30: Physical-chemical properties of HCBD [BiPRO 2015]

<table>
<thead>
<tr>
<th>Physical-chemical properties</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Melting point</td>
<td>-21 °C</td>
</tr>
<tr>
<td>Boiling point</td>
<td>215 °C</td>
</tr>
<tr>
<td>Density (at 20 °C)</td>
<td>1.68 g/cm(^3)</td>
</tr>
<tr>
<td></td>
<td>1.55 g/cm(^3)</td>
</tr>
<tr>
<td>Vapour pressure (at 20, 30, 50 °C)</td>
<td>0.36, 0.75, 2.80 mbar</td>
</tr>
<tr>
<td>Vapour pressure (at 20 °C)</td>
<td>0.2 mbar</td>
</tr>
</tbody>
</table>
### Physical-chemical properties

<table>
<thead>
<tr>
<th>Property</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vapour pressure (at 100°C)</td>
<td>29.26 mbar</td>
</tr>
<tr>
<td>Ignition temperature</td>
<td>610 °C</td>
</tr>
<tr>
<td>Partition coefficient Log Kow</td>
<td>4.78</td>
</tr>
<tr>
<td>Water solubility (at 20 °C)</td>
<td>0.50 g/L</td>
</tr>
<tr>
<td>Water solubility (at 25°)</td>
<td>2.55 mg/L</td>
</tr>
<tr>
<td>Water solubility (at 30°)</td>
<td>3.20 mg/L</td>
</tr>
<tr>
<td>Henry’s law constant</td>
<td>1044 Pa m³/mol</td>
</tr>
</tbody>
</table>

### 4.3.3 LEGAL BACKGROUND


As of 2012, in accordance with the Regulation EC No 850/2004 on persistent organic pollutants the production, trade and use of HCBD is prohibited in the European Union [BiPRO 2015]. HCBD is already listed as a POP in the EU POP regulation. The waste related concentration limits in Annexes IV and V of the EU POP regulation are set at 100 mg/kg (LPCL) and 1,000 mg/kg (MPCL) respectively.


In the context of the EU Water Framework Directive 2000/60/EC HCBD is classified as priority hazardous.

### 4.3.4 OCCURRENCE IN PRODUCTS, WASTES AND RECYCLATES

#### Products

HCBD is prohibited in the European Union and all applications have ceased [UNEP/CHW.13/6/Add.2/Rev.1 2017]. Therefore, no products containing HCBD are on the market in the EU. Relevant products are only from historical uses that enter the waste stream.

At present HCBD is mainly formed as unintentional by-product during several industrial processes, but the available information basis is very scarce.
Wastes

Specific information on the occurrence of HCBD in wastes is contained in section B.4 of the revised technical guidelines on the environmentally sound management of wastes consisting of, containing or contaminated with HCBD.

HCBD releases can arise from the disposal of old HCBD-containing products such as hydraulic, cooling and absorbent liquids. Some HCBD applications have a long service-life and despite the use having ceased, HCBD may still enter the waste management stage. The UBA however, as already stated in 2002, suspects that these releases “play a minor role in quantitative terms” [UBA 2002].

Recyclates

HCBD if at all, only occurs in recyclates in negligible concentration. Specific information has not been identified.

4.3.5 SELECTION OF MOST RELEVANT WASTE STREAMS IN THE EU

In Europe the commercial production of HCBD ended in the late 1970s. As of 2012 all applications of HCBD in the European Union have ceased. HCBD is not known to be currently intentionally produced or used [UNEP/POPS/POPRC.9/13/Add.2 2013]. The intentional production of HCBD in Europe ended in the late 1970s [UNEP/POPS/POPRC.8/16/Add.2]. Intentional production is therefore not considered of relevance as a potential waste stream.

The remaining generation of HCBD is unintentional during the production of chlorinated solvents, plastics and magnesium as well as residues (ashes and slag) from incineration of municipal, clinical and hazardous waste and sludge from municipal and industrial sewage treatment. The relevance of such unintentional production for potential waste stream will be discussed in the following.

Based on the known uses, wastes consisting of, containing or contaminated with HCBD may potentially be found in:

- Unintentional HCBD chemical production during
  - the manufacture of chlorinated chemicals,
  - the manufacturing of plastics, and
  - electrolysis in magnesium manufacturing;
- Residues (ashes and slags) from incineration of unintentional produced HCBD;
- Electrical transformers, heat exchangers, electrical hydraulic fluids, cooling and absorbent liquids;
- Other industrial electrical equipment, including removed lining (ebonite) and graphite electrodes from chlorine electrolysis cells;
- Rubber compounds;
- Agricultural insecticides, fungicides and biocides;
- Residues (ashes and slag) from incineration of municipal, clinical and hazardous waste;
- Sludge from municipal and industrial sewage treatment.

Unintentional HCBD chemical production

HCBD can be unintentionally produced during the manufacture of chlorinated chemicals [UNEP/POPS/COP.8/INF/18 2017]. For example, HCBD is still unintentionally generated during the
production of chlorinated hydrocarbons, particularly of perchloroethylene and trichloroethylene [UNEP/POPS/POPRC.9/13/Add.2 2013]. Manufacturing plants are able to largely destroy or recover HCBD in the process [UNEP/POPS/POPRC.12/INF/12 2016].

Most trace amounts of HCBD are derivatives from low pressure chlorolysis in the processing of chlorinated solvents such as perchloroethylene, tetrachloroethylene, trichloroethylene and tetrachloromethane/carbon tetrachloride [UNEP/POPS/COP.8/INF/18 2017].

Tetrachloroethylene is a solvent which is used in the textile, movie, optical and metal sector. Because of its highly degreasing power it is being applied as degreasing agent. The optical manufacturing uses tetrachloroethylene to clean lenses and prisms prior to connecting these to optical elements. Because of its carcinogenic and toxic properties, as well as its ozone depleting potential tetrachloromethane may not be used anymore. The application as fire-extinguishing, degreasing, cleaning, solving and diluting agent is only allowed for research purposes due to its toxicity and unfavourable environmental properties [BiPRO 2015].

Trichloroethene is being used as starting material for the manufacture of fluorocarbons or as solvent for high precision surface cleaning and degreasing [BiPRO 2015]. Furthermore, it is applied in laboratories for asphalt density detection [BG RCI 2012]. In Germany, in 1990, the output of trichloroethene was around 58,000 t/year [GeoDZ 2010]. Due to stricter environmental regulations and substitution through other chemicals the production in Europe has decreased to 25,000 t/year in 2006. Meanwhile the only European companies still producing trichloroethene are DOW EUROPE and another Romanian producer (Chimcomplex Borzesti). Specific figures for the current production output are not available. Control measures and regulations over the reuse of chemical by-products of chlorinated solvents such as HCBD, along with the substitution and/or reduction of chlorinated solvents containing HCBD, has been effective in reducing total chemical emissions from unintentional HCBD production and recovery. For example, it has been observed that most perchloroethylene and trichloroethylene in manufacturing has been in decline to recommended levels as the result of more efficient methods to reduce its usage [UNEP/POPS/COP.8/INF/18 2017; UNEP/POPS/POPRC.12/6 2016].

Table 31 gives an overview of the reported volumes of HCBD in waste by a producer of chlorinated solvents including perchloroethylene in the Czech Republic.

<table>
<thead>
<tr>
<th>Year</th>
<th>2004</th>
<th>2005</th>
<th>2006</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
</tr>
</thead>
<tbody>
<tr>
<td>HCBD [t]</td>
<td>161</td>
<td>178</td>
<td>194</td>
<td>175</td>
<td>140</td>
<td>66</td>
<td>162</td>
</tr>
</tbody>
</table>

It was noted in [UNEP/POPS/POPRC.9/13/Add.2 2013] that emissive uses of perchloroethylene, trichloroethylene, and carbon tetrachloride have been phased out in several applications and that for the remaining industrial uses, processes have been improved in a way that less product is consumed in the process. Moreover, the production volumes of perchloroethylene and trichloroethylene are declining and under BAT/BEP conditions releases from production of relevant chlorinated solvents are negligible [UNEP/POPS/POPRC.12/INF/12 2016]
Regarding the generation of HCBD in the manufacture of plastics, [WWF 2005] refers to a preliminary document from Environment Canada\(^4\). The final document [Environment Canada and Health Canada 2000] doesn’t mention the manufacture of plastics anymore. Apart from this, there is no specific indication that HCBD occurs during the manufacture of plastics. Accordingly, the recent Risk Management Evaluation of the Stockholm Convention plastics manufacture is not quoted as potential source of unintentional HCBD production anymore [UNEP/POPS/POPRC.9/13/Add. 2 2013]. Thus, plastics manufacture is not considered as relevant source of HCBD.

The release of HCBD from the production of magnesium can occur during electrolysis in magnesium manufacturing [UNEP/POPS/POPRC.12/INF/12 2016].

The relevance of HCBD release as a by-product from the production of magnesium is considered low. This is due to the fact that global production of magnesium is nowadays primarily carried out by reduction of oxide with silicone at high temperatures. This process is not known to lead to unintentional production of HCBD. Moreover, emissions of HCBD from the production by means of electrolysis can potentially be controlled by measures based on BAT [UNEP/POPS/POPRC.12/INF/12 2016].

Considering the recent developments and the declining importance unintentional production of HCBD during manufacturing is not deemed relevant anymore within Europe.

Residues (ashes and slag) from incineration of unintentionally produced HCBD

Dow produced tetrachloroethene and tetrachloromethane at two plants in Germany. Production residues were directly incinerated on-site. Ashes didn’t accrue during this process since they were being redirected to incineration. The only solid residue from incineration is slag [UNEP/POPS/POPRC.9/13/Add.2 2013; UNEP/POPS/POPRC.12/6 2016]. Routine tests are conducted to check on halogenated organic compounds, no relevant amount is detectable. Specific measurements of HCBD concentration in the incineration residues are not available. [BiPRO 2015] found no HCBD concentration above the detection limits (detection limits 8.87 µg/kg and 9.24 µg/kg) in two slag samples from Germany.

HCBD formation as a by-product in incineration processes, however, is unlikely as there is a relation to PCDD/PCDF and other unintentional POPs releases formed by combustion. The measures taken to reduce such POPs releases will also lead to a significant reduction of releases of HCBD. Considering the required BAT and BEP methods relevant for unintentionally produced POPs for various types of incinerators and other thermal sources, relevant amounts of HCBD to residues are negligible [UNEP/POPS/POPRC.9/13/Add.2 2013; UNEP/POPS/POPRC.12/6 2016].

According to these results, it can be concluded that there are no relevant amounts of HCBD in incineration residues from the incineration of production waste generated in the manufacture of chlorinated solvents. There is no indication that, if at all, relevant amounts of HCBD enter the waste stream via this pathway.

Electrical transformers, heat exchangers, electrical hydraulic fluids, cooling and absorbent liquids

Due to past application of HCBD as chemical in transformer fluids, heat-transfer liquids, hydraulic fluids and cooling and absorbent liquids in the 1970s and 1980s, their presence may still persist.

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with the use of these articles from this period or from their disposal [UNEP/POPS/COP.8/INF/18 2017]. Quantities of HCBD used in the past in these liquids and fluids are unknown.

From measurements of the chlorine or organochlorine content, there are no cases known of HCBD found within transformer fluids. Waste flows from historical transformer articles in the present time are considered relatively small. Furthermore, compared to PCBs the concentration of HCBD in transformer oils was much lower [UNEP/POPS/COP.8/INF/18 2017]. However, it is possible that articles containing HCBD may have been or are improperly disposed [UNEP/POPS/COP.8/INF/18 2017], thus HCBD releases from old HCBD-containing products such as transformer, heat-transfer and hydraulic fluids are possible. The UBA however suspects, as already stated in 2002, that these releases “play a minor role in quantitative terms” [UBA 2002].

In conclusion, current input of HCBD into waste streams or related releases due to the historic intentional use in transformer fluids, heat-transfer liquids, hydraulic fluids and cooling and absorbent liquids are considered negligible.

Other industrial electrical equipment, including removed lining (ebonite) and graphite electrodes from chlorine electrolysis cells

HCBD was unintentionally formed during the production of chlorine with graphite electrodes from chlorine electrolysis cells in tonnes scale, while the formation using other technologies is considered low. However, the need to collect numerical data for ‘other industrial electrical equipment’ was expressed within the document, indicating a lack of available quantitative data. It is thus possible that small quantities of HCBD could exist from their disposal of waste and scraps in landfills as e.g. chlorine production is a reported process where HCBD recovery is unlikely [UNEP/POPS/COP.8/INF/18 2017].

In general, it can be stated that old chemicals industry sites may be contaminated by HCBD. Examples of such contamination can also be found in Europe (Barnes et al. 2002 as cited in [UNEP/CHW.13/6/Add.2/Rev.1 2017]. Historical landfilling of heavy fractions from the production of chlorinated organic substances and perchloroethylene use can also lead to secondary HCDB emission [UNEP/CHW.13/6/Add.2/Rev.1 2017].

Rubber compounds

In the past, HCBD was mainly used as a chemical to synthesize rubber compounds and as solvent for elastomeric plastics and chloroprene rubber, the amounts of which present today are marginal according to the national association on rubber and polymers [UNEP/POPS/COP.8/INF/18 2017].

Agricultural insecticides, fungicides and biocides

In the past, HCBD was used for pesticidal purposes including insecticide uses to combat pests, as pesticide, fungicide and as biocide. Presently, no use of HCBD as pesticide has been reported, stockpiles containing or contaminated with HCBD however might still exist and improperly disposal is possible [UNEP/POPS/COP.8/INF/18 2017]. Biocides were e.g. used for the prevention of algal formation in industrial water reservoirs and cooling water systems, use however has ceased in Europe [UBA 2006].

Residues (ashes and slag) from incineration of municipal, clinical and hazardous waste

Unintentional generation of HCBD such as during the incineration of acetylene has been reported in the literature. Acetylene is a component of all incinerations. Other sources have also reported release of HCBD from incineration processes without defining them in more detail [UNEP/POPS/POPRC.12/INF/12 2016].
HCBD is suspected to be developed under similar incineration conditions like dioxins and furans, thus it can occur during waste incineration (e.g. incineration of municipal waste, clinical waste and hazardous waste) and might enter the waste stream through incineration residues (ashes and slag). HCBD is not a standard parameter for the analysis of solid residues [UNEP/POPS/POPRC.12/INF/12 2016].

Minor emissions are possibly released during refuse combustion [UNECE 2007]. Several reports mentioned that HCBD has been found in emissions from refuse combustion in Canada and Europe [BiPRO 2011; Canadian EPA 2001]. According to [BiPRO 2015] HCBD is not present in incineration residues from municipal waste incineration in relevant quantities in Germany. This conclusion was drawn on the basis of five random samples which were analysed for HCBD concentrations in order to estimate the potential relevance of waste incineration processes. No concentrations above the detection limits were found (detection limits ranging from about 9 to 10 µg/kg).

Based on the possibility of occurrence of HCBD in residues from incineration a mass flow was developed. The concentration in waste is based on the study conducted by [BiPRO 2015], where no HCBD concentrations above the detection limits were found in samples from German municipal and hazardous waste incineration plants.

Values reported for China by [Zhang et al. 2018] show drastically higher concentrations detected in fly ash samples with a mean of 49.5 µg/kg and are thus not comparable for values that might occur within Europe.

Based on the available information, taking into account the precautionary principle, 0.8 µg/kg was chosen as concentration of HCBD in ashes and slag from waste incineration for further calculations in the mass flow. 0.8 µg/kg (0.0008 mg/kg) is below the detection limit, but the highest value below the detection limit.

Incineration residues from municipal waste (hazardous and non-hazardous) are mainly recycled or disposed of [Eurostat 2018]. This is due to the fact, that the largest amount of incineration residues from municipal waste is non-hazardous. In 2014, 59,960,000 tonnes of ashes and residues from the incineration of municipal waste (hazardous and non-hazardous) were generated in the EU [Eurostat 2018]. Only a fraction (9%) of the incineration residues from municipal waste are hazardous with 5,230,000 t, the rest is non-hazardous. The derived mass flow is based on Eurostat data on incineration residue from hazardous municipal incineration as no information is available on ashes and slag from hazardous or even clinical waste incineration. The following figure gives an overview of the treatment paths and quantities treated within the EU in 2014 for hazardous waste only.
Taking into account the values from hazardous waste incineration, 43 % of the HCBD content in ashes and slag from incineration would be recycled and 56 % disposed of.

Assuming, that 0.00008 mg/kg is the concentration of HCBD in ashes and slag from waste incineration – even though this concentration is below the limit of detection (LOD) – HCBD content recycled is assumed to be in the range of 0.004 tonnes. Under the assumptions that current calculations are correct, this waste stream is deemed irrelevant in terms of concentration quantity.

These results indicate that HCBD is not present in incineration residues from municipal waste incineration plants operated in accordance with Directive 2010/75/EU on industrial emissions. It is therefore assumed that HCBD is not present in significant quantities in incineration residues from municipal waste incineration in Europe. Moreover, [UNEP/POPS/POPRC.12/INF/12 2016] stated that releases of HCBD from municipal and hospital waste incineration processes can be negligible under Best Available Techniques (BAT)/Best Environmental Practices (BEP) conditions. They are thus not deemed relevant.

**Sludge from municipal and industrial sewage treatment**

Wastewater treatment plants are mentioned as a possible secondary source of HCBD according to [UNEP/POPS/POPRC.12/INF/12 2016; UNEP/POPS/POPRC.9/13/Add.2 2013]. Where HCBD enters the wastewater system, HCBD can possibly be released to water and via sewage sludge to soil. Moreover, literature reports occurrence of HCBD in both, industrial and municipal waste water treatment plants [BiPRO 2015]. The process of WWTP and the water input is outline in section 4.1.10.2.

The European Pollutant Release and Transfer Register (E-PRTR)\(^{41}\) of 2010 contains information about one European company for the production and treatment of metals that releases an HCBD amount of 17.0 kg in 2010 into water. Another company from the chemical sector (industrial production of basic organic chemicals or basic plastics) reported the release of 2.35 kg HCBD into

\(^{41}\) [http://prtr.ec.europa.eu/]
water. Nine municipal waste water treatment facilities reported the release of 69.5 kg HCBD into water. The reporting facilities are located in Portugal, France, Italy, Belgium, Poland, Slovakia and Czech Republic.

The amount of HCBD estimated to end up in the sewage sludge in the EU 27 is approximately 6 kg/year, based on sewage sludge contamination data from China (European Commission 2012 as cited in [UNEP/POPS/POPRC.12/INF/12 2016]).

In the context of a German research project regarding investigations on releases and elimination of hazardous substances in municipal sewage treatment plants, HCBD measurements were performed for several German waste water treatment facilities. Only in one influent sample (n=38) HCBD could be detected in a concentration of 0.2 µg/l [UBA 2006]. In examinations on drains of a German municipal sewage treatment plants in 2002 and 2003, no sample exceeded the detection limit of 0.1 µg/l [UBA 2006]. [BiPRO 2015] did not find HCBD in any of the analysed sewage sludge samples from municipal sewage treatment plants above the detection limit (0.54 µg/kg).

The derived concentration is based on the following information identified within literature in Europe and outside.

A German information portal for waste assessment (ABANDA) provides data on measured HCBD content of sewage sludge from municipal and industrial waste water treatment plants, with values of 0.01 mg/kg and 0.193 mg/kg respectively. An investigation of the Irish EPA didn’t measure HCBD in sewage sludge samples above the detection limit (0.1 mg/kg).

According to BiPRO HCBD doesn’t occur in relevant amounts in municipal sewage sludge in Germany. German sewage sludge samples from one municipal sewage treatment plant were analysed and HCBD was not measured above the detection limit (0.54 µg/kg) [BiPRO 2015].

Individual information and ranges on HCBD contents in sewage sludge are available for China (average 39 µg/kg dm; [Cai et al. 2007]; median 0.30 µg/kg dw [Zhang et al. 2014]) and the US (0.6 µg/kg; [IPCS 1993]). Showing comparable concentrations as reported for Europe.

The following table summarises the collected concentration values and derived the median used to derive the yearly HCBD that is not destroyed within Europe.

<table>
<thead>
<tr>
<th>Location</th>
<th>Mean/Median [mg/kg]</th>
<th>Type of incinerator</th>
<th>Primary Source</th>
<th>Cited in</th>
</tr>
</thead>
<tbody>
<tr>
<td>DE</td>
<td>0.01</td>
<td>Municipal</td>
<td>ABANDA</td>
<td>[BiPRO 2015]</td>
</tr>
<tr>
<td>DE</td>
<td>0.193</td>
<td>Industrial</td>
<td>ABANDA</td>
<td>[BiPRO 2015]</td>
</tr>
<tr>
<td>IE</td>
<td>&lt; 0.1</td>
<td>Irish EPA</td>
<td>[BiPRO 2015]</td>
<td></td>
</tr>
<tr>
<td>DE</td>
<td>0.0005</td>
<td>Municipal</td>
<td>BIPRO</td>
<td>[BiPRO 2015]</td>
</tr>
<tr>
<td>US</td>
<td>0.0006</td>
<td></td>
<td>[IPCS 1993]</td>
<td></td>
</tr>
<tr>
<td>China</td>
<td>0.039</td>
<td></td>
<td>Cai et al. 2007</td>
<td></td>
</tr>
<tr>
<td>China</td>
<td>0.0003</td>
<td></td>
<td>[Zhang et al. 2014]</td>
<td></td>
</tr>
</tbody>
</table>

Median: 0.01 mg/kg

In total, 3.06 million t of sewage sludge arises from WWTPs in the EU in 2014 [Eurostat 2018i]. A median concentration of 0.01 mg/kg is derived from the presented concentrations and taken as basis for Europe. It can thus be assessed that the overall amount of sewage sludge (3.06 million
tons in 2014) contains an estimated HCBD freight of about 0.031 t. This freight can be assigned to the different disposal and recovery pathways as set out in the material and substance flow below.

**Sludges and liquid wastes from waste water treatment**

<table>
<thead>
<tr>
<th></th>
<th>Total amount</th>
<th>HCBD conc.</th>
<th>HCBD in sludge</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>3,060,000 t</td>
<td>0.01 mg/kg</td>
<td>0.031 t</td>
</tr>
</tbody>
</table>

**Figure 22: Substance and mass flow sludges from WWT**

In Europe annually (values from 2014) around 50 % of sludge occurring are landfilled, 33 % are recycled and around 16 % are thermally recovered or incinerated.

Recycling includes both recycling and ‘backfilling’. Backfilling means the recovery operation where suitable waste is used for reclamation purposes in excavated areas or for engineering purposes in landscaping and where the waste is a substitute for non-waste materials” [EC n.d.a].

According to the developed mass flows, no relevant amounts of HCBD are estimated within this waste streams. Unintentional releases however might be relevant considering the lack of data. Thus, more information should be obtained.
4.4 Dicofol

Dicofol is an organochlorine pesticide which is chemically related to dichlorodiphenyltrichloroethane. It is used as a miticidal pesticide and acaricide in many countries worldwide on a variety of fruits, vegetables, ornamental and field crops [UNEP/POPS/POPRC.9/3 2013]. In recent times, however the use has been largely restricted or prohibited in many countries and the use as a pesticide has markedly decreased.

4.4.1 PRODUCTION, TRADE AND USE

Dicofol as an organochlorine insecticide and acaricide was introduced commercially in 1955 [WHO 1996]. It can be manufactured by hydroxylation of dichlorodiphenyltrichloroethane (DDT), or directly, without isolation of DDT by the reaction of chloral (trichloroacetaldehyde) with monochlorobenzene in the presence of oleum (SO$_3$ and H$_2$SO$_4$) followed by dehydrochlorination, chlorination and hydrolysis. Dicofol product size varied significantly from as small as 1 litre containers (AK-20 HC Free, Adama) to 200 kg containers (Hindustan Insecticide Limited) [UNEP/POPS/POPRC.13/7/Add.1 2017].

Between 2000 and 2007, the global annual production of dicofol was estimated in a range between 2,700-5,500 t but production has declined sharply since 2007 as several countries initiated phase-out of their production and use [UNEP/POPS/POPRC.13/7/Add.1 2017]. According to estimations, 28,200 t dicofol was used globally from 2000 - 2012, with 1,745 t produced in Europe [UNEP/POPS/POPRC.11/INF/17 2015].

Production of dicofol is now limited to companies in a small number of countries. In India this includes one manufacturer (Hindustan Insecticides Limited), while a second (Indofil Industries Limited) has registered for the production but is currently not producing dicofol [UNEP/POPS/POPRC.13/7/Add.1 2017]. In Israel the company Adama (formerly Makhteshim Agan) has registered a product containing dicofol (i.e. Acarin T 285), [UNEP/POPS/POPRC.13/7/Add.1 2017]. Based on the information provided through Annex F responses (under the Stockholm Convention), production is predominantly limited to the facility based in India, although no further information on the production facility in Israel has been given. In 2015-2016, the production at the facility based in India was 93 t [UNEP/POPS/POPRC.13/7/Add.1 2017].

The expiry date for the production and use of DDT as a closed-system site-limited intermediate in the production of dicofol has been extended until 2024 by decision SC-7/1 [UNEP/POPS/COP.7/36 2015; UNEP/POPS/COP.7/SC-7/1 2015]. China was one of the major producers of technical DDT and dicofol. It is estimated that 97,000 t of technical DDT was produced in China between 1988 and 2002, with approximately 54,000 t for the manufacture of dicofol (40,000 t of dicofol produced). In 2014, it was reported that the last remaining technical dicofol producer in China ceased production. Further, until 2010, Brazil manufactured annually around 90 t of dicofol, but has also ceased production completely in 2014. According to the addendum of the risk management evaluation of dicofol (of the Stockholm Convention), remaining stocks of dicofol in Brazil are expected to have been fully used/destroyed by 2015. Production of dicofol in the US was estimated at 160 t per year for the years 1999 to 2004 [UNEP/POPS/POPRC.13/7/Add.1 2017].

Dicofol is no longer produced in EU MS [UNEP/POPS/POPRC.13/7/Add.1 2017]. In the EU, dicofol was initially allowed to be produced from DDT as a closed-system site-limited intermediate until January 2014 [EC DG ENV 2011]. However, this specific exemption was later withdrawn from the EU POP Regulation and no inclusion of dicofol to Annex I of Council Directive 91/414/EEC and on the withdrawal of authorisations for plant protection products containing that substance
(2008/764/EC\textsuperscript{42}) was agreed in 2008 [EC 2014]. All existing authorizations for dicofol in plant protection products had to be withdrawn before 30 March 2009. National registration of dicofol was no longer possible after March 2009 and any transitional period granted by the MS expired by 30 March 2010 [EC 2014].

Until 2006, Spain was the main manufacturer and consumer of dicofol in Europe. Dicofol was only produced by the company Montecinca, S.A. under the contract to Dow Agro Sciences [UNEP/POPS/POPRC.13/7/Add.1 2017]. According to another source, Montecinca, S.A. continued production until 2008 [EC 2014]. In 2000, Spain produced about 1,500 t dicofol and the use was reported to be between 100 – 150 t per year. In 2006, Spain produced 90 t of dicofol [UNEP/POPS/POPRC.11/INF/17 2015].

Table 33 summarizes the historic dicofol volumes produced in the EU. Further, the table also contains important milestones of dicofol production.

\textbf{Table 33: Dicofol production and important milestones in the EU}

<table>
<thead>
<tr>
<th>Year</th>
<th>Dicofol production in [t]</th>
<th>Additional information / Milestones</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000 – 2012</td>
<td>1,745 (accumulated)</td>
<td>Accumulated production in Europe (from 2000-2012)</td>
</tr>
<tr>
<td>2000</td>
<td>1,500</td>
<td>Produced in Spain (one production facility)</td>
</tr>
<tr>
<td>2006</td>
<td>90</td>
<td>Produced in Spain (one production facility)</td>
</tr>
<tr>
<td>2006 (2008/09)</td>
<td>-</td>
<td>Production in Spain ceased after 2006 (possibly around 2008/09)</td>
</tr>
</tbody>
</table>

\textit{Use}

Dicofol is restricted or prohibited in many countries and globally the use as a pesticide has markedly decreased in the last decade and was reported to be below 1,000 t in 2012 (approximate decrease of 80 % since 2000) [UNEP/POPS/POPRC.11/INF/17, 2015; UNEP/POPS/POPRC.12/11/Add.1, 2016]. The use of dicofol has been banned in Benin, Brazil, Colombia, the 28 EU MS, Guinea, Japan, Mauritania, Oman, Saudi Arabia, Switzerland as well as Indonesia and Sri Lanka. Furthermore, it has been voluntarily stopped in Canada and the USA [UNEP/POPS/POPRC.13/7/Add.1 2017].

In general, intended uses of dicofol cover the use on fruits including apple, citrus, litchi, longan, and pear as well as vegetables, ornamentals such as orchids and rosebushes, field crops such as cotton and tea and Christmas tree plantations (US EPA 1998 and Li et al. 2015 as cited in [UNEP/POPS/POPRC.11/INF/17 2015] and [UNEP/POPS/POPRC.13/7/Add.1, 2017]).

Dicofol restrictions vary in different countries. In China, dicofol is not permitted on vegetables or fruit (Wang et al. 2015 in [UNEP/POPS/POPRC.11/INF/17 2015]). In Senegal, dicofol is used on


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onion, watermelon, potato and pimento crops. In Mexico, there are 17 registrations for dicofol, which is authorised for the application on aubergine, chilli, strawberry, lime, apple, orange, pear, watermelon, mandarin, grapefruit, vine, citrus fruits, ornamental shrubs, ornamental plants and nursery gardens [UNEP/POPS/POPRC.11/INF/17 2015]. Dicofol is used in pesticides in Australia, with four products registered in concentrations of 150 g/litre [IPEN & Arnika Sub. 2018]. According to the addendum of the risk management evaluation [UNEP/POPS/POPRC.13/7/Add.1 2017], no critical uses of dicofol have been identified and the Parties and observers submitting information under the Stockholm Convention (Annex F) did not provide any information on critical uses.

In 2000, the major EU consumers were Spain, Italy, Romania and France [UNEP/POPS/POPRC.13/7/Add.1 2017]. In EU countries, the use of dicofol for plant protection products expired by 2010 at the latest according to Decision 2008/764/EC. In addition, in the EU all non-agricultural uses are prohibited according to the Biocidal Products Regulation No (EC) 528/2012 [UNEP/POPS/POPRC.11/INF/17 2015]).

The following table summarizes the historic dicofol volumes used in the EU. Further, the table also contains important milestones of dicofol use.

Table 34: Dicofol consumption and important milestones in the EU

<table>
<thead>
<tr>
<th>Year</th>
<th>Dicofol usage in [t]</th>
<th>Additional information / Milestones</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000</td>
<td>317</td>
<td>Annual consumption (all uses/application areas) in Europe decreased from 2000 until 2009 (see below); Major European consumers include among other EU MS, Spain, Italy, Romania and France.</td>
</tr>
<tr>
<td>2009</td>
<td>32</td>
<td>Significant decrease of dicofol consumption in Europe since 2000 (from 317 t – 32 t); Figures also include non-EU countries. Major European consumers include among other EU MS, Spain, Italy, Romania and France.</td>
</tr>
<tr>
<td>2010</td>
<td>-</td>
<td>In EU countries, the use of dicofol for plant protection products expired by 2010, at the latest. In addition, all non-agricultural uses are prohibited according to the Biocidal Products Regulation.</td>
</tr>
</tbody>
</table>

4.4.2 CHEMICAL CHARACTERISTICS

Dicofol is comprised of two isomers: p,p'-dicofol and o,p'-dicofol. The technical product (95 % pure) is composed of 80-85 % p,p'-dicofol and 15-20 % o,p'-dicofol with up to 18 reported impurities. The purer form is generally >95 % dicofol which contains less than 0.1 % DDT and related compounds (i.e. DDE and DDD) [WHO 1996].

Chemical formula: Dicofol C_{14}H_{9}Cl_{5}O

CAS registry numbers:
- p,p'-dicofol (CAS No. 115-32-2)
- o,p'-dicofol (CAS No. 10606-46-9)

Molecular mass: Dicofol 370.49 g/mol

![Chemical structure of p,p'-dicofol (left) and o,p'-dicofol (right)](image)

Figure 23: Chemical structure of p,p'-dicofol (left) and o,p'-dicofol (right)

Table 35 summarizes the main chemical and physical properties of dicofol.

<table>
<thead>
<tr>
<th>Chemical-physical properties of Dicofol</th>
</tr>
</thead>
<tbody>
<tr>
<td>Appearance</td>
</tr>
<tr>
<td>Melting point</td>
</tr>
<tr>
<td>Density</td>
</tr>
<tr>
<td>Boiling point</td>
</tr>
<tr>
<td>Vapour pressure</td>
</tr>
<tr>
<td>Log Kow</td>
</tr>
<tr>
<td>Solubility (in water)</td>
</tr>
<tr>
<td>Henry’s law constant</td>
</tr>
</tbody>
</table>

4.4.3 LEGAL BACKGROUND

Stockholm Convention

In May 2013, the EU and its MS, being Parties to the Stockholm Convention, submitted to the 9th POPRC meeting a proposal to list dicofol in Annex A, B and/or C of the Convention. The proposal was further evaluated by the Committee at its 10th and 11th POPRC meetings. Having examined the proposal, the Committee adopted a decision that dicofol meets the criteria and established an intersessional working group to further review the proposal and prepare a draft risk profile.

At the 12th meeting of the POPRC in September 2016 the Committee, having reviewed the risk profile on dicofol, decided in accordance with paragraph 7(a) of Article 8 of the Convention, that dicofol is likely as a result of its long range environmental transport to lead to significant adverse human health and environmental effects such that global action is warranted. The Committee also established an intersessional working group to prepare a risk management evaluation that includes
an analysis of possible control measures for dicofol. Parties and observers were invited to submit information specified in Annex F before 9 December 2016. The risk management evaluation in accordance with paragraph 9 of Article 8 of the Convention recommends to the COP to consider listing dicofol in Annex A to the Stockholm Convention without specific exemptions [UNEP/POPS/POPRC.13/POPRC-13/1 2017].

The substance DDT is listed in Annex B of the Convention. Since 2009 the specific exemptions for DDT as an intermediate in the production process of dicofol have expired and no new registrations may be made with respect to such exemptions. However, after a request from India, the expiry date for the production and use of DDT as a closed-system site-limited intermediate that is chemically transformed in the manufacture of other chemicals that, taking into consideration the criteria in paragraph 1 of Annex D, do not exhibit the characteristics of persistent organic pollutants has been extended from June 2014 to May 2024 by decision SC-7/1 [UNEP/POPS/COP.7/36 2015; UNEP/POPS/POPRC.13/7/Add.1 2017]. In March 2014, India submitted a notification to the Secretariat relating to the production and use of 150 t DDT. To date, this is the only submission of notification to the Secretariat. The exemption for use of DDT as a closed-system site-limited intermediate to produce dicofol expired for Brazil in 2014 and China withdrew their exemption for this use the same year [UNEP/POPS/POPRC.13/7/Add.1 2017].

**EU Regulation**

In 2008, Dicofol has not been included in Annex I to Council Directive 91/414/EEC concerning the non-inclusion of dicofol and the withdrawal of authorisations for plant protection products containing that substance. In the Commission Decision on the non-inclusion (2008/764/EC) it is stated that: "Any period of grace granted by a MS for the disposal, storage, placing on the market and use of existing stocks of plant protection products containing dicofol should be limited to 12 months in order to allow existing stocks to be used in one further growing season, which ensures that plant protection products containing dicofol remain available for 18 months from the adoption of the Commission Decision (2008/764/EC)".

In 2008, based on the Risk Profile and Summary Report for Dicofol, the EU nominated dicofol for listing as a POP substance under the Protocol. Dicofol is also listed under Annex III to the REACH Regulation (criteria for registration of substances with limited information) and in Annex VI of the CLP Regulation. The evaluation of the PBT/vPvB properties of dicofol resulted in its status of fulfilling POP screening criteria (PBT WG). Moreover, dicofol is listed under Part 2 of Annex V of the PIC Regulation (chemicals and articles subject to export ban other than persistent organic pollutants), [ECHA 2018b]. Between 2007 and 2014, 40 export notifications of dicofol-containing material are listed under PIC. During this period, exporting EU MS were mainly France and Spain with importers being e.g. Morocco, the Dominican Republic, Niger and Senegal [ECHA 2018b].

### 4.4.4 OCCURRENCE IN PRODUCTS, WASTES AND RECYCLES

As summarized in previous sections, dicofol is no longer produced in EU MS since 2006. The withdrawal of dicofol of the authorisations for plant protection products containing that substance was agreed in 2008. National registration of dicofol was no longer possible after March 2009 and any transitional period granted by EU MS expired by 30 March 2010 [EC 2014]. It can therefore be expected that all the remaining stockpiles have been either exported (mainly to developing countries; see export notifications under PIC in Chapter 4.4.3 and/or already been subject to controlled waste management (e.g. waste incineration). The absence of remaining stockpiles in the

EU has been additionally confirmed by several MS, who submitted information on dicofol in response to a questionnaire conducted during this project. The European Crop Care Association (ECCA) stated that 16 of their 20 members confirmed that they hold no stockpile of products containing dicofol and no chemical waste containing dicofol [ECCA Sub. 2018]. Further, no EU imports and exports of dicofol and dicofol-containing products has been indicated within the MS responses. Besides, there is no known occurrence of dicofol in the EU as well as a decreasing relevance and occurrence on the global scale, with bans being in place in various countries around the globe (see Chapter 4.4.1). Insufficient data has been identified regarding critical uses, both from the Annex F survey and supporting literature search and no critical uses have been specified by Parties and observers [UNEP/POPS/POPRC.13/7/Add.1 2017].

4.4.5 SELECTION OF MOST RELEVANT WASTE STREAMS IN THE EU

As highlighted in previous sections, the main global use of dicofol has been as insecticide/acaricide in the production of agricultural products [UNEP/POPS/POPRC.13/7/Add.1 2017]. In general, global intended uses of dicofol cover application on fruits including apple, citrus, litchi, longan, and pear as well as vegetables, ornamentals such as orchids and rosebushes, field crops such as cotton and tea, Christmas tree plantations and others [UNEP/POPS/POPRC.11/INF/17 2015; UNEP/POPS/POPRC.13/7/Add.1 2017]. In general, the use of dicofol in agriculture has markedly decreased in the last decade [UNEP/POPS/POPRC.11/INF/17 2015; UNEP/POPS/POPRC.12/11/Add.1 2016]. In the EU, Spain was the main manufacturer and consumer of dicofol mainly for production of agricultural products [UNEP/POPS/POPRC.13/7/Add.1 2017]. At the present, there is no known production and use of dicofol in the agricultural sector in the EU. In addition, the US EPA reported that dicofol was also used for non-agricultural applications such as outdoor buildings and structures in the US [US EPA 1998]. However, this application could not be confirmed for the EU.

In conclusion, based on historic uses of dicofol in the EU, two waste streams which might still be relevant for the EU are discussed in the following (i.e. obsolete stock piles and C&D waste). Former production sites which may be contaminated with dicofol, e.g. at the former dicofol manufacturing sites in Spain, can be a subject of concern, but do not fall under the scope of this project and are not further discussed in this report.

Obsolete Stockpiles

To investigate on this matter, information was requested via a questionnaire from relevant stakeholders. Several stakeholders submitted information in response to the questionnaire. The submissions have generally confirmed that dicofol stocks are no longer available in the EU. Some respondents have indicated a lack of information and data related to stockpiles. Sweden and the Netherlands reported that no data is available on dicofol stockpiles [SE EPA Sub. 2018] and [NL MoIW Sub. 2018]. According to the Swedish chemicals agency´s products register, there is no current process using dicofol [SE EPA Sub. 2018]. The Netherlands have further reported that there is no production of dicofol-containing waste in the Netherlands [NL MoIW Sub. 2018]. Luxembourg, stated that no stockpiles of dicofol are expected to be present in the country [LU Ministry of Sust. Dev. and Infrastructure Sub. 2018]. The same has also been indicted in the Finish and the Norwegian submissions [FI Env. Institute Sub. 2018].

In addition, several associations and NGOs have supplied information on dicofol. The ECCA, stated that 16 of their 20 members confirmed that they hold no stockpile of products and no chemical waste containing dicofol, and that they do not manufacture, import or export any products containing dicofol [ECCA Sub. 2018]. Further, the Italian fine and specialty chemicals association (AISPEC) reported that they hold no stockpiles since dicofol is not used in any of their production processes [AISPEC Sub., 2018]. IPEN & Arniika state that waste pesticides with unknown amounts
remain [IPEN & Arnika Sub. 2018a], however no additional information was provided to support this statement.

Based on the available information, it can be expected that the obsolete dicofol stockpiles in the EU had been exported and been subject to controlled waste management. No present stockpiles of dicofol have been identified during further research or reported by any stakeholders consulted including EU MS. Therefore, at the present, obsolete dicofol stock are not expected in the EU and will not be further investigated in the present study.

Contaminated C&D Waste

The past production and use of dicofol in the EU was limited to agricultural products. The US EPA reported that dicofol was also used in non-agricultural applications to treat outdoor buildings and structures [US EPA 1998]. Consequently, dicofol-containing waste streams may be expected from contaminated C&D waste from outdoor buildings and structures.

The application of dicofol (e.g. under the product name Kelthane®) on agricultural and non-agricultural outdoor buildings and structures in the US can be confirmed by two further information sources. In a study conducted by Rodrigues and Riehl (1958) a spray mixture containing Kelthane® was applied to hen houses for mite control. Dicofol was applied to the wooden surfaces to analyse mite control efficiency [Rodriguez and Riehl 1958]. Further, the use of Kelthane® on non-agricultural outdoor buildings and structures was proven by a guide published by the Institute of Agriculture and Natural Resources. The document describes the use of dicofol on clover mites on homes and other properties in Nebraska [Kamble et al. 1993]. According to [Kamble et a. 1993] clover mites invade homes or other buildings in large numbers and that mite control can be achieved through several insecticides/miticides available to the public. Furthermore, [Kamble et al. 1993] describe the application method as: “Insecticide sprays should be applied in a band of 10-15 feet wide around the building foundation and on the lower portions of the outside walls”.

Both literature sources indicate that dicofol was used on agricultural as well as on non-agricultural applications on outdoor buildings and structures in the US. However, as already indicated in previous chapters, this specific use could not be confirmed in the EU by any information source reviewed. Therefore, this specific use and associated waste streams will not be further considered in the present study.
### 4.5 Pentadecafluorooctanoic acid (PFOA, perfluorooctanoic acid), its salts and PFOA-related compounds

PFOA, its salts and PFOA-related compounds belong to the family of perfluoroalkyl and polyfluoroalkyl substances (PFASs). PFASs, which can be degraded to PFOA are referred to as PFOA-related substances. PFOA, its salts and PFOA-related compounds have been used in many different applications and consumer products across sectors due to their physico-chemical properties. In recent years they have become subject to a number of regulations with bans, restrictions and phase-out dates. PFOA and its salts are mostly used as processing aids for the production of fluoroelastomers and fluoropolymers. PFOA-related compounds are widely used as surfactants and surface treatment agents e.g. in textiles, paper and paints and fire-fighting foams, and for the manufacturing of side-chain fluorinated polymers [UNEP/POPS/POPRC.12/11/Add.2 2016].

#### 4.5.1 PRODUCTION, TRADE AND USE

**Production**

In 2005 the producers of PFOA and its salts were located in U.S., South America, Japan, China and Italy. All production of PFOA and its salts in the EU has now ceased, since the only known producer within the EU (Miteni in Italy) has phased-out production in 2010. [UNEP/POPS/POPRC.12/11/Add.2 2016; ECHA 2015a]. In addition, the production should also have stopped in the U.S. and Japan, which committed to phase out production by the end of 2015 [ECHA 2015a]. In contrast, the production of PFOA and its salts in China has tripped from around 30 t in 2004 to around 90 t in 2012 [Li et al. 2015]. For the period of 2004 to 2008, [RPS 2010] have estimated on behalf of the EU Commission that the average market volume of PFOA and ammonium perfluorooctanoate (APFO) in the EU was 100 t/a at most.

ECHA concludes that the EU demand for PFOA is mainly catered to through imports or in mixtures. ECHAs estimates from 2012 indicate that around 20 t/a enter the EU as import [ECHA 2015a]. Additional quantities of around 20 t/a enter in form of fluoropolymer dispersions amount and in products as indicated in Table 36. This leads to a total amount of PFOA and its salts being imported in the EU of 40 t/a [ECHA 2015a].

Table 36: Overview of production and import volumes of PFOA and its salts in the EU [RPS 2010; ECHA 2015a]

<table>
<thead>
<tr>
<th>Year/Period</th>
<th>Substance(s)</th>
<th>Description</th>
<th>Value [t/a]</th>
</tr>
</thead>
<tbody>
<tr>
<td>2012</td>
<td>PFOA and its salts</td>
<td>Import volumes into the EU</td>
<td>20</td>
</tr>
<tr>
<td>2004-2008</td>
<td>PFOA and its salts</td>
<td>Average market volume in the EU</td>
<td>&lt;100</td>
</tr>
<tr>
<td>2010</td>
<td>PFOA and its salts</td>
<td>Average market volume in the EU</td>
<td>&lt;50</td>
</tr>
<tr>
<td>2015</td>
<td>PFOA and its salts in imported fluoropolymer dispersions</td>
<td>Import volumes EU in fluoropolymer dispersions</td>
<td>10</td>
</tr>
<tr>
<td>2010</td>
<td>PFOA and its salts in imported products</td>
<td>EU volume in imported products</td>
<td>10</td>
</tr>
<tr>
<td>2015</td>
<td>PFOA and its salts, as substances, in mixtures and in articles</td>
<td>EU total import value in any form</td>
<td>40</td>
</tr>
</tbody>
</table>
No information is available on the global production quantities and production sites for PFOA-related substances. However, PFOA-related substances are still highly relevant in the EU. In 2014, production of PFOA-related compounds in the EU was reported by [ECHA 2015a] to be between 100 t and 1,000 t per year. Those production estimates, however, are derived from a search in the ECHA-database for four specific PFOA-related compounds registered under REACH. [ECHA 2015a] therefore concludes that the actual volume is likely to be higher. No other quantitative information concerning the production of PFOA-related compounds could be identified in literature. The quoted import range in 2014 was between 100 and 1,000 t/a for PFOA-related substances into the EU, with a decreasing trend. However, the authors conclude that this amount could have been higher as the stakeholder consultation only yielded a very limited amount of responses from the contacted companies.

### Table 37: Overview of production and import figures for PFOA-related substances [ECHA 2015a]

<table>
<thead>
<tr>
<th>Year/Period</th>
<th>Substance(s)</th>
<th>Description</th>
<th>Value/Range [t/a]</th>
<th>Comment</th>
</tr>
</thead>
<tbody>
<tr>
<td>2014</td>
<td>PFOA-related substances</td>
<td>EU manufacture volume (range)</td>
<td>100 – 1,000</td>
<td>ECHA 2015a, based on registrations</td>
</tr>
<tr>
<td>2014</td>
<td>PFOA-related substances</td>
<td>EU import volume (range)</td>
<td>100 – 1,000</td>
<td>ECHA 2015a, based on stakeholder consultation</td>
</tr>
</tbody>
</table>

### Use

The primary uses of PFOA, its salts and related compounds are in consumer products across sectors such as the automotive industry, in electronics, construction and aerospace industry especially due to their properties such as high friction resistance, resistance to heat and chemical agents, low surface energy, and being water, grease, oil and soil repellent [ECHA 2015a].

**PFOA and its salts** are most frequently used in fluoropolymer and fluoroelastomer production. In the form of APFO in aqueous solution, it is employed as an emulsifier and processing aid in the production of such polymers, including PTFE (polytetrafluoroethylene), FEP (fluorinated ethylene propylene), PFA (perfluoroalkoxy alkane) or PVDF (polyvinylidene fluoride). APFO is also used in the processing of paint, photographic film additives and in the textile coating industry and can be found in certain aqueous fire-fighting foams [UNEP/POPS/POPRC.12/11/Add.2 2016]. The produced fluoropolymers are in turn used in the manufacturing of hoses, cables and gaskets as well as non-stick coatings on cookware, amongst others [UNEP/POPS/PORC.12/11/Add.2 2016]. Data on quantities of PFOA and its salts used in different application areas is limited.

The use of **PFOA-related compounds** can be divided in non-polymeric uses and polymer uses. In non-polymeric uses, the substances are employed as surfactants for instance in fire-fighting foams, wetting agents and cleaners [ECHA 2015a; RPS 2010]. On the other hand, side-chain fluorinated polymers are used to repel water, grease and soil in textiles, products for apparels and leather, paper and cardboard as well as in paints, lacquers and non-woven medical garments or floor waxes (among others, for a more exhaustive list see UNEP 2016). According to the estimation by ECHA, the major uses of PFOA-related compounds identified in Europe are surface treated textiles, fire-fighting foam and surface treated paper and paints and inks [ECHA 2015a]. Data on quantities of PFOA-related compounds used in different application areas is limited.

The following table summarizes data on the use of PFOA, its salts and related compounds according to both [ECHA 2015a] and [UNEP/POPS/POPRC.12/11/Add.2 2016].
Table 38: Use of PFOA, its salts and PFOA-related compounds in different sectors in the EU [ECHA 2015a] as cited in [UNEP/POPS/POPRC.12/11/Add.2 2016]

<table>
<thead>
<tr>
<th>Use in the EU</th>
<th>Sector/Industry</th>
<th>Value/Range [in t/a]</th>
</tr>
</thead>
<tbody>
<tr>
<td>PFOA-related compounds</td>
<td>Textile and leather treatment</td>
<td>~1,000</td>
</tr>
<tr>
<td></td>
<td>Paper treatment</td>
<td>&gt;150 - 200</td>
</tr>
<tr>
<td></td>
<td>Firefighting agents</td>
<td>&gt;50 - 100</td>
</tr>
<tr>
<td></td>
<td>Paints and inks</td>
<td>&gt;50 - 100</td>
</tr>
<tr>
<td></td>
<td>Other uses</td>
<td>&gt;0.1 - 0.5</td>
</tr>
<tr>
<td>PFOA and its salts</td>
<td>Manufacture of fluoropolymers</td>
<td>&lt;20</td>
</tr>
<tr>
<td></td>
<td>Photo industry</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>Semiconductor industry</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td></td>
<td>Other uses 0.5-1.5 t</td>
<td>0.5 - 1.5</td>
</tr>
</tbody>
</table>

4.5.2 CHEMICAL CHARACTERISTICS

PFOA, its salts and PFOA-related compounds fall within a family of perfluoroalkyl and polyfluoroalkyl substances (PFASs). Perfluorinated acids, like PFOA, are not degradable in the environment and in biota. Certain polyfluorinated substances can be degraded to persistent perfluorinated substances like PFOA under environmental conditions and are therefore precursors. Those PFASs that can be degraded to PFOA in the environment and in biota are referred to as PFOA-related compounds [UNEP/POPS/POPRC.13/7/Add.2 2017]. A comprehensive list of substances covered is available in document [UNEP/POPS/POPRC.13/INF/6/Add.1 2017].

Chemical formula: PFOA $C_8H_{15}F_{15}O_2$

CAS registry numbers: PFOA 335-67-1

Molecular mass: PFOA 414.07 g/mol

Figure 24: Chemical structure of PFOA

Table 39 summarizes the main chemical and physical properties of PFOA.

Table 39: Chemical-physical properties of PFOA [UNEP/POPS/POPRC.13/7/Add.2 2017; Savu 1994]

<table>
<thead>
<tr>
<th>Chemical-physical properties of PFOA</th>
</tr>
</thead>
<tbody>
<tr>
<td>Appearance</td>
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<tr>
<td>Melting point</td>
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<tr>
<td>Density</td>
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<td>Boiling point</td>
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<td>Vapour pressure</td>
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<td>Log Kow</td>
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<td></td>
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<tr>
<td>Water Solubility</td>
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### 4.5.3 LEGAL BACKGROUND

PFOA is subject to regulatory measures under REACH and CLP. In 2013 PFOA was included in the REACH candidate list and will be restricted from 2020 due to an entry in REACH Annex XVII. PFOA has a harmonised classification under CLP. Further it is proposed for listing under the Stockholm Convention.

**Stockholm Convention**

In June 2015 the EU and its member States submitted a proposal to list pentadecafluorooctanoic acid, its salts and PFOA-related compounds in Annexes A, B, and/or C to the Stockholm Convention [UNEP/POPS/POPRC.11/5 2015].

At its 12th meeting in 2016, the Persistent Organic Pollutants Review Committee (POPRC) concluded that PFOA is persistent, bioaccumulative and toxic to animals including humans. There is widespread occurrence of PFOA and a number of PFOA-related compounds in environmental compartments and in biota and humans. Therefore, PFOA, its salts and PFOA-related compounds that degrade to PFOA 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 [UNEP/POPS/POPRC.12/11/Add.2 2016].

At its 13th meeting in 2017, the POPRC adopted the RME on PFOA, its salts and PFOA-related compounds [UNEP/POPS/POPRC.13/7/Add.2 2017] and recommended to the COP that it consider listing the chemicals in Annex A or B to the Convention with specific exemptions specified in decision POPRC-13/2 (also in [UNEP/POPS/POPRC.13/7/Add.2 2017]). However, the Committee was unable to reach conclusions on whether exemptions may be needed for specific uses. Furthermore, additional work was needed to consider the possibility of unintentional releases and specific issues related to substance identity.

The Committee established an intersessional work group to assess additional information to help further the discussion at the 14th meeting to define the need for possible specific exemptions and/or acceptable purposes for certain additional applications and to evaluate their unintentional releases in the view of strengthening its recommendation to the COP. Besides, the Committee invited Parties and observers, including the relevant industries, to provide information that would assist the possible defining by the Committee of specific exemptions for production and use of PFOA, its salts and PFOA-related compounds in particular in the following applications:

(a) Membranes intended for use in medical textiles, filtration in water treatment, production processes and effluent treatment: information on the scope of the applications, used amounts, availability of alternatives and socio-economic aspects;
(b) Transported isolated intermediates in order to enable reprocessing in another site than the production site: Information on the quantities used, extent of transport and risks, and use;
(c) Medical devices: information on specific applications/uses and timelines foreseen as needed for potential related exemptions;
(d) Implantable medical devices: information on the quantities used, extent of transport and risks, and use;
(e) Photo imaging sector: information on paper and printing, and information relevant for developing countries;
(f) Automotive industry: information on spare parts;
(g) Firefighting foams: information on chemical composition of mixtures and the volumes of pre-installed amount of firefighting foam mixtures.

At its 14th meeting in 2018, the POPRC adopted the addendum to the RME on PFOA, its salts and PFOA-related compounds. The POPRC also recommends to the COP that it consider listing PFOA, its salts, and related compounds in Annex A to the Convention with specific exemptions.

For five years from the date of entry into force of the amendment, the Committee recommends the following exemptions:

- Manufacture of semiconductors or related electronic devices for: equipment or fabrication plant-related infrastructure containing fluoropolymers and/or fluoroelastomers with PFOA residues; legacy equipment or legacy fabrication plant-related infrastructure maintenance; and photo-lithography or etch processes;
- photographic coatings applied to film;
- textiles for oil and water repellency for the protection of workers from dangerous liquids that comprise risks to their health and safety;
- invasive and implantable medical devices; and
- fire-fighting foam for liquid fuel vapor suppression and liquid fuel fires (Class B fire) already in installed systems, including both mobile and fixed systems, taking due account of the possible related control measures.

For ten years from the date of entry into force of the amendment, the Committee recommends the following exemptions for manufacture of semiconductors or related electronic devices: refurbishment parts containing fluoropolymers and/or fluoroelastomers with PFOA residues for legacy equipment or legacy refurbishment parts. Besides, the POPRC recommends a specific exemption for use of perfluorooctane iodide and production of perfluorooctane bromide for producing pharmaceutical products with a review of continued need for exemptions. The specific exemption should expire at the latest in 2036. Finally, the POPRC recommends to the COP that it consider encouraging parties not to replace fire-fighting foams that contain or may contain PFOA, its salts, and related compounds with short-chain PFAS due to their persistency and mobility, as well as potential negative environmental, health, and socio-economic impacts.

In April 2019, the COP will consider the recommendations made by the POPRC over the last two years. As a result, the COP could add two new chemicals to Annex A of the Stockholm Convention (i.e. dicofol and PFOA). While the dicofol decision may be relatively straightforward, since only India produces the chemical, Committee members may have to carefully explain their rationale for the PFOA recommendation given its widespread production and use.
**REACH**

PFOA was identified as a SVHC and included in the candidate list for authorisation in June 2013.\(^{45}\) PFOA, its salts and certain related substances\(^{46}\) will be restricted for most uses by July 4, 2020 (Entry 68 to Annex XVII of the REACH Regulation).\(^{47}\)

Annex XVII Entry 68 reads as follows:

1. PFOA ‘shall not be manufactured, or placed on the market as substances on their own from 4 July 2020’
2. PFOA ‘shall not, from 4 July 2020, be used in the production of, or placed on the market in:
   - (a) another substance, as a constituent;
   - (b) a mixture;
   - (c) an article,

   in a concentration equal to or above 25 ppb of PFOA including its salts or 1,000 ppb of one or a combination of PFOA-related substances.

3. Points 1 and 2 shall apply from:
   - (a) 4 July 2022 to: equipment used to manufacture semiconductors and latex printing inks
   - (b) 4 July 2023 to: textiles for the protection of workers from risks to their health and safety; membranes intended for use in medical textiles, filtration in water treatment, production processes and effluent treatment; and plasma nano-coatings.
   - (c) 4 July 2032 to medical devices other than implantable medical devices within the scope of Directive 93/42/EEC.

4. Points 1 and 2 shall not apply to any of the following:
   - (a) perfluorooctane sulfonic acid and its derivatives, which are listed in Part A of Annex I to Regulation (EC) No 850/2004;
   - (b) the manufacture of a substance where this occurs as an unavoidable by-product of the manufacture of fluorochemicals with a carbon chain equal to or shorter than 6 atoms;
   - (c) a substance that is to be used, or is used as a transported isolated intermediate, provided that the conditions in points (a) to (f) of Article 18(4) of this Regulation are met;
   - (d) a substance, constituent of another substance or mixture that is to be used, or is used: (i) in the production of implantable medical devices within the scope of Directive 93/42/EEC; (ii) in photographic coatings applied to films, papers or printing plates; (iii) in photolithography processes for semiconductors or in etching processes for compound semiconductors;

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\(^{45}\) ECHA Decision ED/69/2013, [https://echa.europa.eu/documents/10162/092663e6-b14a-4a06-aadf-fc0e56bc0a23](https://echa.europa.eu/documents/10162/092663e6-b14a-4a06-aadf-fc0e56bc0a23).

\(^{46}\) PFOA-related substances with a perfluoroheptyl group (C\(_7\)F\(_{15}\)-) directly attached to another carbon atom or perfluorooctyl group (C\(_8\)F\(_{17}\)-)* as one of the structural elements. The following substances are excluded from this designation:

- C\(_8\)F\(_{17}\)-X where X = F, Cl or Br
- C\(_8\)F\(_{17}\)-C(O)OH, C\(_8\)F\(_{17}\)-C(O)O-X’ or C\(_8\)F\(_{17}\)-CF\(_2\)-X’ where X’ = any group, including salts

• (e) concentrated fire-fighting foam mixtures that were placed on the market before 4 July 2020 and are to be used, or are used in the production of other fire-fighting foam mixtures.

5. Point 2(b) shall not apply to fire-fighting foam mixtures which were:

• (a) placed on the market before 4 July 2020; or
• (b) produced in accordance with point 4(e), provided that, where they are used for training purposes, emissions to the environment are minimised and effluents collected are safely disposed of.

6. Point 2(c) shall not apply to:

• (a) articles placed on the market before 4 July 2020;
• (b) implantable medical devices produced in accordance with point 4(d)(i);
• (c) articles coated with the photographic coatings referred to in point 4(d)(ii);
• (d) semiconductors or compound semiconductors referred to in point 4(d)(iii).

In case PFOA is included into Annex A of the Stockholm Convention (April 2019), Annex XVII of the REACH Regulation will have to be adjusted accordingly. This may result in exclusion of certain exemptions at EU level.

4.5.4 OCCURRENCE IN PRODUCTS, WASTES AND RECYCLATES

Literature findings indicate that manufacturers of PFOAs and its salts primarily takes place in companies in China and South America. Production of PFOA, and its salts within the EU as well as in the U.S. and Japan has phased out. The EU will further restrict the manufacturing, placing on the market and use, which includes imports, of PFOA, its salts and certain related compounds as well as articles containing these substances with an exemption for certain uses under REACH. [ECHA 2015a] estimates that production outside of PFOA and its salts has however increased, with China tripling its production between 2004 and 2012. Up-to-date information on production volumes of PFOA, its salts and PFOA-related compounds is not available.

Due to their numerous technical properties PFOA, its salts and related compounds have been widely used in many applications including consumer products across various sectors in the EU. However, quantitative information on used amounts in different application areas is limited. Therefore, as far as possible, the current relevance of PFOA, its salts and related compounds in products, wastes and recyclates in the EU is at least qualitatively evaluated / discussed in the following chapter.

4.5.5 SELECTION OF MOST RELEVANT WASTE STREAMS IN THE EU

Firefighting foams

Fire-fighting foams were identified as a dispersive use of PFOA in the RME resulting in direct release to the environment. Perfluorinated compounds within fire-fighting foams have been used because they proved effective against liquid fuel fires (Class B). In the past C8 based perfluro technologies were mostly used, including PFOS (POP under the Stockholm Convention). The primary use of PFOA within fire-fighting foams was as the ammonium salt (APFO), [UNEP/POPS/POPRC.14/3 2018].

The EU REACH restriction allows for the presence of PFOA and PFOA-related compounds up to a maximum concentration of 25 ppb for PFOA including its salts or 1,000 ppb of one or a combination of PFOA-related compounds, which is likely related to trace contamination within fluorotelomer based Aqueous Film Forming Foam (AFFF). This shall not apply to articles placed on the market before 4 July 2020. Besides, PFOA based fire-fighting foams mixtures placed on the market by or before 4 July 2020 would be permitted for use (for training purposes, emissions to
the environment are minimized and effluents collected are safely disposed of). Also concentrated fire-fighting foams mixtures place on the market before 4 July 2020 to be used to produce other fire-fighting foams mixtures are exempted.

Under the Stockholm Convention a 5-year exemption is currently proposed for fire-fighting foams for liquid fuel vapor suppression and liquid fuel fires (Class B fire) already in installed systems, including both mobile and fixed systems, taking due account of the possible related control measures. Besides, The POPRC recommends to the COP that it consider encouraging parties not to replace fire-fighting foams that contain or may contain PFOA, its salts, and related compounds with short-chain PFAS due to their persistency and mobility, as well as potential negative env., health, and socio-economic impacts.

Only limited information exists to quantify the PFOA-based fire-fighting foams already placed on the market. The addendum to the RME presents an estimation of APFO based concentrate remaining stockpiles (already placed on the market). A worst-case scenario has been applied, where all PFCA within the AFFF is PFOA/PFOA-related compound. Based on active concentrations of 0.1 to 1% APFO within AFFF fire-fighting foams, the global production estimates from Prevedouros et al. (2006), and assumption that all APFO produced is used in fire-fighting foams gives an estimate of global AFFF concentrates containing APFO produced between 1951–2004 as between 309 million litres and 4901 million litres. This would equate to an average annual production of between 6 and 96 million litres of APFO based AFFF concentrate per annum. Assuming the shelf life of AFFF is between 10–25 years, this would correspond to remaining global stockpiles of between 60 and 2,400 million litres of APFO based concentrate [UNEP/POPS/POPRC.14/3 2018].

IPEN and ACAT note reference to several fluorine free alternatives which have been tested as effective against Class B fires. Overall the costs associated with destruction and replacement can be perceived to be significant, one estimate quotes $1 AUD/ 1.5 Euro per litre of concentrate. However, costs associated with clean-up for sites contaminated by perfluorinated compounds are also significant, with examples quoted in the RME as millions of Euros per site. IPEN commented that the life span of foams varied depending on climate and storage and therefore a 20-year derogation would not be acceptable for the Stockholm Convention. The Fire-Fighting Foams Industry Coalition state that foams do not have an expiry date but have a shelf life of 10–25 years [UNEP/POPS/POPRC.14/3 2018].

Information / estimations related to stockpiles in use within the EU are not available. However, according to the addendum of the RME and based on additional information compiled and reviewed, the size of in-use stockpiles may be significant. Quantitative information related to wastes and recyclate volumes is not available. According to [BiPRO 2011] it can be expected that most AFFFs are typically incinerated in hazardous waste incinerators after their use stage. Recycling and/or recovery of AFFFs is not a common waste management operation applied in the EU and it is therefore not expected that PFOA accumulates in recycled products.

**Textiles**

PFOA can be used as a surfactant in the process to produce Polytetrafluoroethylene (PTFE), a linear polymer that consists of carbon and fluor atoms. As PTFE is highly fire resistant and repels hydrophobic materials (oil, stain, grease) but is also water resistant, it is utilized in various products. The use of PFOA-containing PTFE and PFOA in textiles can be generally divided in three categories: PFOA-containing PTFE in technical textiles, PFOA-containing PTFE in non-technical textiles and PFOA in surface-treated textiles.

The EU REACH Regulation allows production and use of PFOA, its salt and related compounds for textiles for the protection of workers from risks to their health and safety a time-limited derogation (until 4 July 2023) is given in the EU. Further, an exemption is in place for membranes intended for use in medical textiles, filtration in water treatment, production processes and effluent
treatment (also until 4 July 2023). All other relevant articles shall not, from 4 July 2020, be produced, placed on the market or used in a concentration equal to or above 25 µg/kg of PFOA including its salts or 1,000 µg/kg of one or a combination of PFOA-related substances.

Under the Stockholm Convention a 5-year exemption is currently proposed for textiles for oil and water repellency for the protection of workers from dangerous liquids that comprise risks to their health and safety.

Therefore, it can be expected that PFOA, its salts and PFOA-related compounds is still relevant in certain textile products and will remain relevant in specific waste streams within the EU for years. Quantitative information on the relevant amounts in products and wastes is not available. Even though textiles might be collected, reused (exported) and/or recycled, it can be expected that most textiles are incinerated and landfilled (usually together with municipal solid waste). Textile recycling is currently not a common waste management operation applied in the EU (might change in the future) and it is therefore not expected that PFOA accumulates in recycled products. The practical challenge to separate “POP-rich” from “POP-free” textiles remains.

**Semiconductors**

The Semiconductor Industry Association (SIA) surveyed its member companies and found that several companies continue to use PFOA and related chemicals in the photo-lithography process, a key step in the manufacturing process to produce advanced semiconductors. It has been reported, that the volume used in the sector is a minor part of the total volumes used in the EU and that the substances are used under strictly controlled conditions [UNEP/POPS/POPRC.13/7/Add.2 2017]. However, no quantitative information has been provided by the association. Further, information submitted by the sector tends to demonstrate that substitution is currently not possible, and that the timeframes for substitution are rather long (10 years). The EU public consultations confirmed that the costs incurred would be high. [UNEP/POPS/POPRC.13/7/Add.2 2017].

According to [ECHA 2015a], less than 0.05 t of PFOA and its salts were used by the EU semiconductor industry. Information on the current use within the sector is not available. Considering that semiconductors are manufactured and traded on a global level, it cannot be excluded that PFOA-containing semiconductors or finished products containing respective semiconductors enter the EU market in relevant amounts. However, information on imported volumes is not available.

In the EU REACH restriction, a time limited derogation (until 4 July 2022) is in place for the equipment used to manufacture semiconductors. In addition, a derogation is granted for substances, constituent of another substance or mixture used in photo-lithography processes for semiconductors or in etching processes for compound semiconductors as well as for articles.

Information regarding wastes and recyclates is missing. However, the use of semiconductors in various products will result in various waste streams (e.g. in WEEE, ELV, municipal solid waste) which will need to be managed. Besides, as some of the waste streams may be recycled it cannot be generally excluded that PFOA its salts and PFOA-related compounds accumulate in recyclates and subsequently in new products.

**Medical devices**

The RME [UNEP/POPS/POPRC.13/7/Add.2 2017] highlights that alternatives to PFOA for manufacture of PTFE exist and have been commercialised. A report by [ECHA 2015a] as part of the European restriction estimates total quantities in use for medical devices amounts to <5 kg of PFOA globally. Based on the information compiled and discussed within the RME and further elaborated upon within the addendum of the RME, the phase-out is still ongoing for some uses and a time-limited exemption may be needed for medical devices to prevent loss of critical applications. Further, the addendum RME highlighted a need for more information about a
potential exemption for medical implantable devices due to possible presence as a by-product in PTFE. Quantities of PFOA and PFOA-related compounds used in the production of PTFE found in implantable medical devices are small. As an indicative estimate for order of magnitude a manufacturer commented that the EU total is 20 g in all devices put on the market during the period 2018–2025. According to ECHA this would lead to an estimation of 100 g worldwide. ECHA reported during the EU REACH restriction that during the manufacture of PTFE, concentrations of PFOA as a by-product range from 0.0001 to 0.5 % wt/wt PTFE. A further personal communication with ECHA (2018) highlighted that during the discussions with industry working concentrations of PFOA specifically within implantable medical devices were likely to be lower again. However, alternatives such as PFOA free PTFE products have undergone clinical testing and been approved for use in some geographies [UNEP/POPS/POPRC.14/3 2018].

For medical devices, the EU allows a time limited exemption for relevant medical devices (excluding implantable ones) until 4 July 2032 and a non-time limited exemption for implantable medical devices. Under the Stockholm Convention, a specific exemption of 5 years is recommended by the POPRC for invasive and implantable medical devices.

Considering the low amounts of PFOA, its salts and PFOA related compounds expected to be used and needed in the future in medical devices placed on the EU market, this use and respective waste streams are not considered relevant and will not be further investigated in this study.

**Certain printing inks**

Comments from industry representatives submitted during the EU consultations indicate that PFOA and related compounds are present in latex printing inks used in professional printers. This use only continues in printers that are no longer manufactured, and therefore a phase-out is already underway. A clear decreasing trend in the amounts used and related emissions has been reported. However, no quantitative information has been given. The company that had manufactured the printers and produces the inks stated that in absence of a transitional period of 5 years, there would be a need for premature replacement of the printers in use, and the costs would be high. The scientific committee of the EU concluded that it is justified to accept a transitional period of 5 years for latex printing inks so that a time limited derogation (until 4 July 2022) is given in the EU [UNEP/POPS/POPRC.13/7/Add.2 2017]. No specific exemption for printing inks is currently recommended by the POPRC under the Stockholm Convention.

[ ECHA 2015a ] indicated that >50-100 t of PFOA related compounds were used within the EU paints and inks industry. Current consumption volumes in the EU are not known but it can be expected that the consumption is decreasing. Besides, no information is available on wastes and recyclates.

**Photo-imaging**

PFOA and its related substances are bond as surfactants upon a dense matrix of coating and then covered by sub-layers of photographic materials in the manufacture of paper, film and printing plates. Specifically, PFOA related substances are a known input in silver halide photographic film which is still used by a small number of antiquarian hobbyists and photographers to capture traditional black and white images. With its anti-static properties, PFOA is relevant for grey scale images of high photosensitivity which requiring enhanced contrast for military surveillance images or for medical purposes such as x-ray scans. The concentration of PFOA per article is estimated to be 0.1 to 0.8 μg/cm² and less than 0.1 t PFOA and PFOA related compounds were applied in photographic coatings for final products within the EU in 2013. The consumption slightly increased to 0.12 t in 2015, followed by an estimated decline to 0.088 t in 2016 [ ECHA 2015a ].

At POPRC-13, representatives of the European photographic industry provided information for the RME that the suggested time-limited exemptions for photographic coatings applied to paper and
for use in printing plates are no longer needed. Non-perfluorinated alternatives have successfully replaced most uses in the imaging and printing industry and the move to digital imaging will replace the need for PFOA. The European Imaging and Printing Association (I&P Europe) comment in 2016 that use of PFOA compounds in photo imaging has decreased globally by 95% since 2000 and only limited critical applications (limited to photographic coatings applied to films) still use PFOA. Based on the existing and rapid transition towards digital imaging, the wide use of digital techniques in developing and transitional countries, and the further reduction in use of PFOA in this sector, no exemptions for photographic coatings applied to paper and printing plates are considered necessary [UNEP/POPS/POPRC.14/3 2018].

At the present, within the EU, an exemption is given for substances, constituent of another substance or mixtures that is to be used, or is used in photographic coatings applied to films, papers or printing plates. Besides, an exemption is in place for articles coated with the photographic coatings. Under the Stockholm Convention, a time limited specific exemption of 5 years is currently recommended for photographic coating applied to film only. It can therefore be expected that the exemptions for photographic paper and printing plates will be also removed in EU legislation.

Due to the comparably low and decreasing use amounts in very specific applications and the fact that this specific application will become obsolete, this use will not be further investigated in the present study. In addition, no significant waste amounts are expected to arise in the EU.

**Automotive industry (service and replacement parts and vehicles)**

The RME highlighted the need for more information about a potential PFOA exemption for automotive service and replacement parts. Among others, specification of relevant automotive service and replacement parts as well as sound justification for any exemption is required. Based on the evaluation of updated information it is concluded within the Draft addendum of the RME that a time limited exemption could be considered for automotive service and replacement parts and vehicles. However, so far, no conclusive information has been provided on specific service and replacement parts and on the quantities of relevant substances used in different applications. Information such as amounts of PFOA and related compounds used in different parts is considered supportive to justify a consideration for an exemption [UNEP/POPS/POPRC.14/3 2018].

The automotive industry has been proactively phasing out PFOA. However, service and replacement parts might still contain PFOA, its salts and related compounds. According to industry information, these parts represent a small percentage of PFOA use and will decrease naturally over time as the vehicle fleet turns-over [UNEP/POPS/POPRC.13/7/Add.2 2017]. However, no indication of quantities has been provided by the industry.

At the present, according to Annex XVII of the REACH Regulation, PFOA, its salts and certain related substances shall not, from 4 July 2020, be used in the production of, or placed on the market in concentrations equal to or above 25 μg/kg of PFOA including its salts or 1,000 μg/kg of one or a combination of PFOA-related substances. This however, does not apply to articles placed on the market before 4 July 2020.

The amounts of PFOA and PFOA-related compounds in use as well as the respective waste amounts might be relevant. However, in the absence of further information (e.g. relevant application areas) and quantitative data on the amounts used, it is not possible to further investigate this use.

**Transported intermediate use in the production of pharmaceutical products**

According to chemical industry, alternatives have not been developed for all pharmaceutical and some other highly specialized chemicals which use PFOA-related compounds as their raw material
and/or processing media and which have socio-economic benefit in particular performance standards [UNEP/POPS/POPRC.13/7/Add.2 2017].

Perfluoroctyl bromide (PFOB) is produced from perfluoroctyl iodide (PFOI). PFOI is produced at a single site in Japan during the production of 6:2 fluorotelomer-based substances, and then transported as isolated intermediate to another site in Japan to produce PFOB. All the wastes generated from this production of PFOI are collected and incinerated. Subsequently, PFOB is transported to two sites in the US and Sweden to produce relevant pharmaceutical products. PFOB is used as a processing aid in the manufacture of “microporous” particles for pharmaceutical applications. PFOB is not a PFOA related compound but it contains unintended trace levels of PFOI, a PFOA related compound. The residual PFOB in the finished “microporous” pharmaceutical products is typically 0.1 %, which translates to residual PFOI at levels of 0.1 mg/kg. The detection limit for PFOB in the porous particles is 0.1 %. The PFOI residual in all currently produced pharmaceutical products totals to less than 2 g per year. Emission of PFOI to the environment from pharmaceutical production is currently less than 30 g total per year. PFOB in process waste is captured in serial carbon beds, which is the best available technology and it controls emissions to less than 1 % and typically to less than 0.1 % [UNEP/POPS/POPRC.13/7/Add.2 2017].

The POPRC recommends a specific exemption for use of perfluoroctane iodide and production of perfluoroctane bromide for producing pharmaceutical products with a review of continued need for exemptions. The specific exemption should expire at the latest in 2036. According to the addendum RME for PFOA and based on additional information provided by stakeholders a time limited exemption could be considered for perfluoroctane iodide (PFOI) as an isolated intermediate to enable reprocessing at another site than the production site [UNEP/POPS/POPRC.14/3 2018]. In case this use is exempted, PFOB may be continued to be used as processing aid in the manufacture of pharmaceutical products (also within the EU; i.e. Sweden).

No further information is available on the volumes produced and used in the EU. However, considering that PFOB only contain unintended trace levels of PFOI which is a PFOA-related compound, this application is also not expected to result in significant waste amounts and is therefore not further considered in this study.

Specific and reliable estimates concerning cumulated quantities of PFOA becoming available in waste streams are not feasible due to the limited quantitative information in the EU. However, material flows associated with main applications can be summarized qualitatively:

- It can be expected that most AFFFs are typically incinerated in hazardous waste incinerators after their use stage. Consequently, fire-fighting foams do not represent a major future source of PFOA accumulations, despite their typical shelf life of 10-25 years.
- Given existing derogations until 2023 for specific textiles and typical lifetimes of such textiles, PFOA-containing textiles are expected to be subject to waste management at least until the late 2030s.
- Looking at the semiconductor sector, the relevance for the EU is estimated to be rather low due to comparably low tonnages expected to be used. Semiconductors in imported products, however, might still play an important role from the waste management perspective but cannot be quantified.
- Consumption volumes of relevant printing inks in the EU are not known. However, professional printers using relevant latex printing inks are already being phased out and the existing derogation applies until 2022. Given the relatively short lifetimes of printing inks, significant accumulations are not to be expected from this source after 2030.
- No significant waste amounts are expected from photo-imaging applications.
- Waste amounts arising from automotive applications might be relevant, but will decrease over time. Noteworthy amounts may only be expected from automotive applications that
are placed on the market before 2020. Consequently, significant accumulations in ELV waste streams are unlikely after 2040.
- Despite existing derogations, waste streams associated with medical devices are not expected to become relevant in terms of significant accumulations.
- Pharmaceutical applications are not expected to result in significant waste amounts.

In conclusion, waste streams associated with specific textiles (e.g. membranes and textiles for the protection of workers from risks to their health and safety), ELVs and possibly also semiconductors deserve particular attention as to avoid PFOA accumulations in the future.

As it can be seen in the previous sections, public information on PFASs (Per- and Polyfluoroalkyl Substances) including PFOA, their hazardous properties, environmental fate and transport, exposure, and toxic effects is still very limited. In addition to long-chain PFASs, there are still many other overlooked, structurally similar PFASs on the market, including short-chain and novel PFASs. In response to this and to foster cooperative actions between science and policy to further address PFASs, a workshop was held in Zürich (CH) in November 2017 which resulted in the *Zürich Statement on Future Actions on PFASs*. The working group engaged over 50 academic scientists and regulators from across the globe [Ritscher et al. 2018]. From the discussions statements and recommendations emerged and are summarised in Table 40.  

**Table 40:** Statements and recommendations on Future Actions on PFASs [Ritscher et al. 2018]

<table>
<thead>
<tr>
<th>Current Scientific and Policy Needs Regarding PFASs</th>
<th>Coordination of scientific and regulatory efforts to avoid duplication and to ensure efficient use of resources. This could start from using the same terminology (including clear identifiers for substances in the PFAS family). Currently available knowledge on PFASs needs to be structured and made more easily accessible to the scientific and regulatory communities, e.g. through a centralized repository.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Implementation of a grouping approach for addressing PFASs rather than individual chemicals. The grouping approach needs to be scientifically sound and additional data is needed to support it.</td>
<td></td>
</tr>
<tr>
<td>Implementation of novel schemes to place specific focus on high (or very high) persistence of PFASs. The usefulness of current schemes that may trigger management actions on chemicals based on their persistent, bioaccumulative, and toxic (PBT) properties may not adequately reflect the high persistence of PFASs in the environment.</td>
<td></td>
</tr>
<tr>
<td>Further information on PFASs to fill in critical knowledge and data gaps including the development of analytical methods, including standardised methods for assessing PFASs in products and media in addition to environmental matrices. Further, information on chemical identity, uses, alternatives and current and future production and emission sites of PFASs and polymeric PFASs needs to be assessed.</td>
<td></td>
</tr>
<tr>
<td>Common Goals Regarding PFASs between Science and Policy</td>
<td>Determine a common chemical scope for cooperative actions. The chemical scope should address a wide range of PFASs, including not only long-chain PFCAs, PFSAs and their precursors, but also their shorter-chain homologues, other perfluoro alkyl acids (PFAAs) and precursors, fluoropolymers, perfluoropolyethers, and other newly identified PFASs such as side-chain fluorinated aromatics.</td>
</tr>
<tr>
<td>Exploring new approaches to assessing and managing highly persistent chemicals such as PFASs. They could encompass the development of new criteria that explicitly consider the special aspects related to the very high persistence, in addition to the existing PBT criteria. Further, socioeconomic analyses on the uses of PFASs and their alternatives need to be conducted.</td>
<td></td>
</tr>
</tbody>
</table>
including the reflection of the long-term societal costs associated with the very high persistence.

Working toward a phase out of nonessential uses of PFASs and developing safe alternatives. Collaborative efforts should be made to reduce, and eventually phase out, non-essential uses of PFASs. Given that there is no official or widely agreed-upon definition of essential uses, a first goal would be to define the term “essential use” with respect to the applications of PFASs.

Raising Awareness of PFAS-related issues among policy-makers and the general public including further empowerment of consumers, product manufacturers, and retailers in making informed decisions.

Collecting further information on PFASs availability and accessibility with regard to:

a) substance identities and compositions in products, including polymeric PFASs;
b) production, use, and disposal volumes of individual PFASs;
c) experimental data on physico chemical properties;
d) toxic effects of individual PFASs, in particular the so-far-overlooked PFASs, including a mechanistic understanding of their toxicokinetics and toxicodynamics and of their effects in humans and on the ecosystem level; and
e) the lifecycle of polymeric PFASs.

Developing and improving PFAS analytics in the following areas:

a) increasing availability and accessibility of analytical standards and certified standard reference materials;
b) continuously developing and improving targeted analytical methods, including approaches that reduce the costs and complexity of analytical methods; and
c) developing adequate PFAS databases to support and foster nontarget and suspect screening analyses and workflows.

Conducting joint assessments for groups of PFASs, which may range from hazard assessments (e.g., screening-level assessment of certain hazardous properties as a basis for grouping) to socio-economic impact assessments of the lifecycle of PFASs with a specific focus also on the end-of-life stages of the chemicals.

Future cooperative actions may also focus on the following areas to foster transitions to alternatives to PFASs:

a) continuous development of alternative assessment frameworks tailored to the specifics of PFASs;
b) development of guidance for transition processes from PFASs to safer alternatives; and
c) promotion of funding in the cross-cutting research area of “safe-by-design” and alternatives to PFASs. In addition, an overview of exemplary success stories and case studies on the implementation of alternatives to PFASs could be developed.

To promote public awareness of PFAS content in goods and products future cooperative actions may also focus on exploring existing mechanisms and developing new mechanisms that would allow consumers and other stakeholders to track PFAS content in goods and products throughout value chains.
The project team believes, that the approaches of the Zürich statement are important for future global action on PFASs and relevant in regard to POP waste management. Especially the grouping approach is important for the formulation of possible limit values (e.g. LPCL) for the PFAS families.

Further, the need of additional information addressed by the working group is strongly supported by the project team. This problem has been identified in the course of the research as part of this study. Information on uses, alternatives and current and future production and emission sites of PFASs is still missing for PFOA as well as the POP candidate PFHxS.

Therefore, based on the currently available information and data, it is not possible to investigate certain waste streams in detail as it has been done for other substances in this report. Nevertheless, lower and upper limitation criteria will be evaluated as far as possible and recommendations will be provided for possible limit values in this report.
4.6 Perfluorohexanoic acid (PFHxS), its salts and PFHxS-related compounds

PFHxS its salts and PFHxS-related compounds are short-chain PFAS. PFHxS its salts and PFHxS-related compounds have been used in many different applications, also to substitute PFOS and PFOA in some application areas. However, PFHxS has received less attention than other fluorinated compounds, also leading to major information and data gaps regarding its production and use volumes. In 2017, PFHxS got under regulatory pressure as it was included in the REACH candidate list of substances and is currently under review for inclusion to the Stockholm Convention.

PFHxS, its salts and PFHxS-related compounds have been widely used in fire-fighting foams, as surfactants, in metal plating as well as in cleaning, waxing, polish and other surface treatment products, and/or water- and stain-protective coatings for carpets, paper, leather and textiles and some other application areas also including consumer products [UNEP/POPS/POPRC.14/2 2018].

4.6.1 PRODUCTION, TRADE AND USE

Production

In general, information on the manufacture of PFHxS its salts and PFHxS-related compounds is limited and mostly qualitative rather than quantitative. PFHxS, its salts and PFHxS-related compounds may be produced from perfluorohexane sulfonyl fluoride (PFHxSF), which is also the parent compound of PFOS, its salts and PFOS-related compounds [UNEP/POPS/POPRC.14/2 2018].

Historically, 3M was most likely the main producer of PFHxS, its salts and PFHxS-related compounds with an annual production of about 227 t of PFHxSF in the US in 1997. Around 2000, 3M ceased its production of PFHxS, its salts and PFHxS-related compounds. In addition, there may have been some (historical) production by Dainippon Ink and Chemicals in Japan. Current manufacturers of PFHxS, its salts and PFHxS related compounds include and most likely several producers in China and at least one producer in the EU (i.e. Italy). In particular, historical and/or current manufacturers of PFHxS, its salts and PFHxS-related compounds include at least Miteni from Italy and Chinese manufacturers Hubei Hengxin, Wuhan Defu, Wuhan Yangtze River, Wuhan Fengfan, Shanghai Vatten and Time Chemical. OECD reported that <4,000 kg of PFHxSF, <1,500 kg of PFHxS, and <600 kg of FHxSA were produced in 2003 in Italy. Similarly, the subsequent two OECD surveys reported the production of some PFHxS, its salts and PFHxS-related compounds, but with no information on their respective production volumes and locations were disclosed, [UNEP/POPS/POPRC.14/2 2018].

Using market research reports for PFHxS (CAS No: 355-46-4) and PFHxSF (CAS No: 423-50-7), reviewing peer-reviewed literature, other publicly available information sources and conducting stakeholder consultations, BiPRO and ETH Zürich on behalf of the Norwegian Environment Agency performed a project to identify potential sources of PFHxS to the environment. Across all evaluated literature sources as well as from stakeholder consultations, it was concluded that there is a lack of publicly available information on the quantitative production levels and descriptions of product-specific uses of PFHxS and related compounds. There was also a lack of willingness from stakeholders to participate and provide relevant information. In addition, it was concluded that the quality of information sources reviewed, especially the market research reports was questionable and probably did not cover all global producers since only two Chinese companies were reflected in the reports [UNEP/POPS/POPRC.14/2 2018]. Nevertheless, some information and data is available on the production and consumption of PFHxSF (CAS: 423-50-7), the precursor for PFHxS related substances. According to the market research reports cited by [BiPRO and ETH Zürich 2018], the maximum consumption of PFHxSF is expected to be in the range of 100 Million US$ (2019) in USA and Canada, 70-90 Million US$ (2020) in Europe and more than 500 Million US$ (2020) in Asia. Based on the current price for commercially available PFHxSF of around 2-10 US$/g it was roughly estimated that about 10-60 t PFHxSF are expected to be consumed in US and Canada, about 10-
40 t in Europe and between 60-300 t in Asia. As mentioned, the reliability of the data from market research reports was highly questionable and could not be verified as no literature sources were disclosed to support the presented figures.

[Danish EPA 2013] reported that PFHxSF and its derivatives have an annual production volume of 20 t in China. The Chinese full production capacity of PFHxSF and its derivatives is up to 50 tons per year that is in line with production of PFOS. However, information about production and consumption of PFHxS cannot be directly derived since the compound is not only used to synthesise PFHxS and related compounds.

During the consultation phase for this project several stakeholders were contacted in order to obtain additional information related to production and use of PFHxS, its salts and related compounds. Information related to production was very limited and only a few stakeholders submitted information related to PFHxS use. The Swedish EPA reported that according to the Swedish Chemicals Agency’s products register, there is no current production process involving PFHxS in Sweden [SE EPA Sub. 2018]. The German Environmental Agency (UBA) informed in their submission that a study will be launched in 2019 on PFHxS in articles, wastes and recycyclates. The Netherlands informed in their submission that they consider preparing an inventory of production, use and waste issues of PFOA and PFHxS, following earlier inventories of SCCPs, PFOS, BDEs and HBCDD [NL MoIW Sub. 2018]. In addition, Norway intends to further investigate information on PFHxS (communicated during the final stakeholder workshop).

Despite being manufactured in a limited number of countries, PFHxS, its salts and PFHxS-related compounds have been distributed globally through the trade of products containing these substances.

**Use**

Due to the thermal and chemical stability as well as the hydro- and oleophobicity of the perfluoroalkyl moiety (CnF2n+1-), PFHxS, its salts and PFHxS-related compounds can be used as effective surfactants and/or surface protectors. Perfluorohexane sulfonic acid (PFHxS, PFHS), its salts and related substances have been widely used as surfactants, to make fluoropolymers and as water- and stain protective coatings for carpets, paper and textiles [UNEP/POPS/POPRC.13/4 2017].

Since the nomination of PFHxS, its salts and PFHxS-related compounds, the number of notifications to the EU C&L inventory has increased. Furthermore, it is noted that PFHxS, its salts and many PFHxS-related compounds have been listed on multiple national chemical registration inventories, indicating production, importation and/or uses of products containing these substances, historically and/or ongoing [UNEP/POPS/POPRC.14/2 2018]. In addition, several PFHxS-related substances have been pre-registered under REACH [UNEP/POPS/POPRC.13/4 2017] also indicating a potential import/use of these compounds in the EU.

In the light of information identified recently by the Norwegian Environment Agency and information from other publicly available sources, PFHxS, its salts and PFHxS-related compounds have been intentionally used at least in the following applications: firefighting foams, metal plating, textiles, leather and upholstery, polishing agents and cleaning/washing agents, coatings, impregnation/proofing (for protection from damp, fungus, etc.) and within the manufacturing of electronics and semiconductors [UNEP/POPS/POPRC.14/2 2018]. In addition, other potential uses mentioned in literature are papermaking, pesticides, agrichemicals, some synthetic materials, plasticisers, ski waxes and cookware [BiPRO and ETH Zürich 2018; IPEN & Arnika Sub. 2018a; UNEP/POPS/POPRC.14/2 2018].

Based on market research reports, [BiPRO and ETH Zürich 2018] reported that fire-fighting foams (66% of global consumption) and textile finishing (22% of global consumption) may represent the
main use areas for PFHxS and related substances. Other identified areas are intermediate feedstocks, electronics and semiconductors. However, the reliability of the information of the market report is highly questionable.

4.6.2 CHEMICAL CHARACTERISTICS

PFHxS, its salts and PFHxS-related compounds belong to the per- and poly-fluoroalkyl substances (PFAS) group. PFASs consist of carbon chains of different chain length, where the hydrogen atoms are completely (perfluorinated) or partly (polyfluorinated) substituted by fluorine atoms. The very stable bond between carbon and fluorine is only breakable with high energy input. However, PFHxS salts and other PFHxS-related compounds can degrade to PFHxS under environmental conditions and are therefore known as precursors [UNEP/POPS/POPRC.14/2 2018].

Chemical formula: \( \text{C}_6\text{F}_{13}\text{SO}_3\text{H} \)

CAS registry numbers: 335-46-4

Molecular mass: PFHxS 400.11 g/mol

The chemical structures of PFHxS, an example of its salt (perfluorohexanesulfonate ammonium salt), a PFHxS-related substance (perfluorohexane sulfonamide) and PFHxSF are illustrated in Figure 25 [UNEP/POPS/POPRC.13/4 2017].

![Chemical structures of PFHxS, its salt, and a PFHxS-related substance](image)

**Figure 25:** Examples of chemical structures of PFHxS its salt and a PFHxS-related substance [UNEP/POPS/POPRC.13/4 2017]

PFHxS is a strong acid with six fully fluorinated carbons, having both hydrophobic- and hydrophilic properties making it oil- and water-repellent. Experimental data on the physicochemical properties of PFHxS are limited, however, some studies have reported some empirical and estimated physicochemical properties of PFHxS and its related compounds [UNEP/POPS/POPRC.14/2 2018].

Table 41 summarizes the main chemical and physical properties of PFHxS.

**Table 41:** Chemical-physical properties of PFHxS (adapted from [UNEP/POPS/POPRC.14/2 2018])

<table>
<thead>
<tr>
<th>Property</th>
<th>Value</th>
<th>Source/Comment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Appearance (Physical state at 20°C and 101.3 kPa)</td>
<td>Solid powder at 20°C for PFHxS-K</td>
<td>[ECHA 2017a] (Company provided)</td>
</tr>
<tr>
<td>Melting point</td>
<td>320 K (41°C)</td>
<td>[Kim et al. 2015; ECHA 2017a] (Company provided)</td>
</tr>
<tr>
<td>Boiling point</td>
<td>238–239°C</td>
<td>[Kosswig 2000] (measured)</td>
</tr>
<tr>
<td>Vapour pressure</td>
<td>58.9 Pa (0.0046 mmHg)</td>
<td>[Wang et al. 2011] (Estimated for using COSMOTHERM)</td>
</tr>
<tr>
<td>Log Kow</td>
<td>5.17</td>
<td>[Wang et al. 2011] (estimated)</td>
</tr>
<tr>
<td>Water solubility</td>
<td>1.4 g/L (20–25°C)</td>
<td>[Campbell et al. 2009] (measured)</td>
</tr>
<tr>
<td></td>
<td>2.3 g/L (20–25°C)</td>
<td>[Wang et al. 2011] (Estimated for neutral (non-ionised))</td>
</tr>
</tbody>
</table>

151/390
The group of PFHxS, its salts and related substances comprises a large number of chemicals. In the nomination, the proposed compounds included PFHxS, its salts and PFHxS-related compounds defined in [UNEP/POPS/POPRC.13/4 2017] and in decision POPRC-13/3 (as cited in [UNEP/POPS/POPRC.13/7 2017]). In line with decision POPRC-13/3, the following apply:

(a) Perfluorohexane sulfonic acid (CAS No: 355-46-4, PFHxS);
(b) Any substance that contains the chemical moiety C6F13SO2- as one of its structural elements and that potentially degrades to PFHxS.

The OECD has identified 74 PFHxS-related substances and a study published by the Norwegian Environment Agency identified 80 commercially available compounds including PFHxS and PFHxSF. Both non-exhaustive lists of substances are contained in [UNEP/POPS/POPRC.14/INF/4 2018].

4.6.3 LEGAL BACKGROUND

Stockholm Convention

Perfluorohexane sulfonic acid, its salts and PFHxS-related compounds is currently reviewed under the Stockholm Convention. In May 2017, Norway proposed to list PFHxS, its salts and related compounds in Annexes A, B and/or C to the Convention [UNEP/POPS/POPRC.13/4 2017]. The POPs Review Committee at its 13th meeting in October 2017 concluded that PFHxS fulfil the screening criteria in Annex D and decided that issues related to the inclusion of PFHxS salts and PFHxS-related compounds that potentially degrade to PFHxS should be dealt with in developing the draft risk profile (decision POPRC-13/3). The substances covered in the risk profile include PFHxS (CAS No: 355-46-4), its salts and PFHxS-related compounds, defined as any substances that contain the chemical moiety C6F13SO2 as one of their structural elements and that potentially degrade to PFHxS [UNEP/POPS/POPRC.14/2 2018].

At the 14th POPRC meeting in 2018, the POPRC:

- adopts the risk profile for PFHxS, its salts and related compounds,
- decides that PFHxS, its salts, and related compounds are likely, as a result of their LRET, to lead to significant adverse human health and environmental effects such that global action is warranted, and to establish an intersessional working group to prepare a risk management evaluation,
- and invites parties and observers to submit information specified in Annex F (on socio-economic considerations) before 26 November 2018.

REACH Regulation

In 2017, PFHxS and its salts were identified as SVHC and added to the REACH Candidate List due to their persistent and bioaccumulative properties. Inclusion on this list means the substances can be subject to further review and only used for specific authorized purposes under strictly controlled conditions. Moreover, upon request industry is obliged to inform consumers on the occurrence of the listed substances in consumer articles [UNEP/POPS/POPRC.14/2 2018].

4.6.4 OCCURRENCE IN PRODUCTS, WASTES AND RECYCLATES

In summary, information on current production and use of PFHxS, its salts and related compounds in the EU is scarce and mostly qualitative rather than quantitative. It is quite certain that production is still ongoing and current manufacturers of PFHxS, its salts and PFHxS related compounds include most likely several companies in China and at least one producer form the EU
(i.e. Italy). It is not clear whether PFHxS, its salts and related compounds are currently also being produced in other countries around the world (possibly including other EU MS).

Quantitative information related to produced volumes in the EU is scarce and partly assessed as non-reliable. In 2003, the OECD reported that <1.5 t of PFHxS were produced in Italy. Two subsequent and more recent OECD surveys (i.e. OECD, 2006, 2010) reported the production of some PFHxS, its salts and PFHxS-related compounds, but did not disclose any information on their respective volumes [UNEP/POPS/POPRC.14/INF/4 2018]. Using market research reports for PFHxS and PFHxSF, review of peer-reviewed literature, other publicly available information sources and stakeholder consultations, BiPRO and ETH Zürich on behalf of the Norwegian Environment Agency performed a project to identify potential sources of PFHxS to the environment. Across all evaluated literature sources as well as from stakeholder consultations, it was concluded that there is a lack of publicly available information on the quantitative production levels. Nevertheless, some data is available on the production and consumption of PFHxSF (CAS: 423-50-7), the precursor for PFHxS related substances. According to the market research reports (cited by [BiPRO and ETH Zürich 2018]), the maximum consumption of PFHxSF is expected to be in the range of 70-90 Million US$ (2020) in Europe. Based on the current price for commercially available PFHxSF of around 2-10 US$/g it was roughly estimated that about 10-40 t PFHxSF are expected to be consumed in Europe. However, information related to production and consumption of PFHxS cannot be directly derived from this figure since the compound is not only used to synthesise PFHxS and related compounds. Besides, the reliability of the data from market research reports was assessed to be highly questionable.

4.6.1 SELECTION OF MOST RELEVANT WASTE STREAMS IN THE EU

As discussed in previous chapters, due to their numerous technical properties, PFHxS, its salts and related compounds have been used in many applications including consumer products. Further details on the typical uses identified in publicly available information source are presented below. Besides, as far as possible, the current relevance of PFHxS, its salts and related compounds in products, wastes and recyclates in the EU is evaluated.

Fire fighting foams (AFFFs)

Historically, 3M used PFHxS in the production of its AFFF formulations. In 1992, 3M indicated that PFHxS-related compounds and PFOS were likely used in the same AFFF formulations, i.e. previously known as “PFOS-based AFFFs” (e.g. FC-600). This is also in agreement with several measurements of legacy AFFF formulations conducted in several studies. Besides, this is also well supported by measurements of environmental media conducted at AFFF-impact sites and by measurements of firefighters’ serum levels, where similar or even higher levels of PFHxS than that of PFOS were detected. It is likely that PFHxS-related compounds containing “PFOS-based AFFFs” have been discontinued/decreased after 3M ceased its global production in 2000–2002. However, there may still be substantial stockpiles of such legacy AFFF formulations around the world. Furthermore, Shanghai Vatten has recently developed and commercialized at least one new PFHxS-related amphoteric surfactant for foam fire-extinguisher [UNEP/POPS/POPRC.14/2 2018].

In summary, it is certain that PFHxS has been used in production of AFFF formulations. This has been proven by several literature sources and measurements in environmental media and firefighters’ where PFHxS has been detected. It can be assumed that the global production of AFFFs containing PFHxS decreased after 3M (major producer) ceased its global production around 2000. However, it cannot be excluded that other producers continued to produce PFHxS-containing AFFF formulations. Therefore, the current global status of production and use of PFHxS-containing AFFFs is uncertain. However, recently at least one PFHxS-related surfactant for foam fire-extinguishers has been developed and commercialized indicating that the production and use is ongoing. Quantitative information to evaluate the current relevance of PFHxS -containing AFFFs in
the EU is not available. However, considering that formulations and/or products are marketed on a global level, it cannot be excluded that PFHxS-containing AFFFs also reach the EU market.

It can also be assumed that there may still be stockpiles of AFFFs containing PFHxS globally and potentially also in the EU. In relation to wastes, it can be assumed that AFFFs containing PFHxS have either been already disposed of and/or subject to controlled waste management in the EU or will enter the waste phase and be managed in the future. Further, according to [BiPRO 2011] it can be expected that most AFFFs are typically incinerated in hazardous waste incinerators. Recycling and/or recovery of AFFFs is not a common waste management operation applied in the EU. Quantitative information related to wastes and recyclates is not available.

**Metal plating**

Several patents suggest that PFHxS, its salts and various PFHxS-related compounds were used in metal plating as mist suppressants. It is likely that at least Hubei Hengxin from China has marketed the potassium salt of PFHxS for metal plating [UNEP/POPS/POPRC.14/2 2018]. It is however not clear if PFHxS is still used within the metal plating industry.

PFHxS has been detected in ground- and surface waters close to metal plating facilities in Sweden [UNEP/POPS/POPRC.14/2 2018] indicating potential use in the EU. However, there is no further information / indication on a deliberate use of PFHxS, its salts and related compounds in metal plating facilities in the EU.

Information related to wastes and recyclates is not available. Considering that PFHxS may be used to substitute PFOS, the following PFHxS-containing wastes might occur; regenerates of the cation exchangers in the chromium(VI) cycle, chromium hydroxide sludge from waste water treatment, ion exchange resins used to retain the mist suppressant, anion exchangers from rinse water cycles, lead chromate sludge forming in the chromium electrolyte, used porous lead anodes, electroplating racks (ca. 1x/a), tanks, pipes and waste air channels (at about 10-year intervals), barium sulfate sludge from bath maintenance measures of the chromium electrolyte (typical PFOS-containing wastes as summarized by [Blepp a t al. 2017]). It can be assumed that most of the listed waste streams will be collected by specialized companies and are subject to controlled waste management in the EU. It is not clear to what extent such waste streams are recycled and/or recovered in the EU.

**Textiles, leather and upholstery**

Historically, 3M used PFHxS-related compounds in some of its aftermarket (post-production) carpet protection products, e.g., those carpet and upholstery protector containing FC-228. This is in accordance with the measured higher serum PFHxS concentrations (range 27.5–423 ng/mL) than that of PFOS (15.2–108 ng/mL), as well as highly elevated levels of PFHxS in household dust and carpets of a Canadian family, whose household carpets were treated 8 times with Scotchgard formulations over 15 years. It is likely such aftermarket carpet and upholstery protector products have been discontinued after 3M ceased its global production in 2000–2002. However, it is reported that water-proofing textile finishes based on PFHxS-related compounds have recently been developed by at least Hubei Hengxin Chemical Co., Ltd. (CAS No: 68259-15-4, CAS No: 68555-75-9 and CAS No: 67584-57-0) and Wuhan Fengfan Surface Engineering Co., Ltd. from China as alternatives to PFOS-based compounds [UNEP/POPS/POPRC.14/2 2018].

In summary, PFHxS was used in stain protection products for carpets and upholstery. Carpets were often treated with PFHxS-containing products such as ScotchGard. It is assumed that most of these uses were phased out in 2002, however, recent information shows that water-proofing textile finishes based on PFHxS-related compounds have been developed by at least two Chinese companies. No relevant information / indication on a deliberate use of PFHxS, its salts and related compounds in products for carpets and upholstery could be identified in the EU.
Information related to wastes and recyclates containing or contaminated with PFHxS, its salts and related compounds is not available. It can be assumed that most of the relevant waste has already been or will be subject to controlled waste management in the EU (mostly landfilling and incineration). Even though textiles might be collected, reused (exported) and/or recycled, it can be expected that most textiles are incinerated and landfilled (usually together with municipal solid waste). Textile recycling is currently not a common waste management operation applied in the EU (might change in the future) and it is therefore not expected that PFHxS accumulates in recycled products. The practical challenge to separate “POP-rich” from “POP-free” textiles still remains.

The same also applies for leather applications. In general, leather can be recycled or reused (repurposed). However, most leather products are incinerated as recycling is often not economically viable. In most cases of leather recycling, shredded leather scraps and residues from the production process are treated with resin and glued. However, production waste is not considered to be relevant since PFHxS is used at a very late stage of the production process. Retailing and distribution waste as well as post-consumer leather waste could be a relevant source of PFHxS. However, material recycling for these two waste sources is more difficult and it is therefore also not expected that PFHxS accumulates in recycled products.

Use in the production of semiconductors

During the POPRC13 meeting in 2017, an industry representative noted that PFHxS, its salts and PFHxS-related compounds are used as replacements to PFOS, PFOA and their related compounds in the semiconductor industry. This is further strengthened by published information indicating that PFHxS was used in the semiconductor industry in Taiwan. PFHxS (133–330 ng/L), together with PFOS (128–670 ng/L), was one of the primary contaminants at a semiconductor fabrication plant waste water effluent site. Both PFSAs are present in the effluent in similar amounts showing that PFHxS is a primary substance in this process and are not unintentionally present at this site [UNEP/POPS/POPRC.14/2 2018].

It is not clear if PFHxS, its salts and related compounds have also been used in the EU in the production of semiconductors. However, considering that semiconductors are manufactured and traded on a global level, it cannot be excluded that PFHxS-containing semiconductors or finished products containing respective semiconductors enter the EU market. However, information on imported volumes is not available. Information regarding wastes and recyclates is also missing. However, the use of semiconductors in various products may result in various waste streams (e.g. in WEEE, ELV, municipal solid waste) which will need to be managed. Besides, as some of the waste streams may be recycled it cannot be generally excluded that PFHxS its salts and PFHxS-related compounds accumulate in recyclates and subsequently in new products.

Lubricants

The use of PFHxS in lubricants has been indicated during the recent stakeholder consultation for this project [IPEN and Arnika Sub. 2018]. However, no additional qualitative or any quantitative information on the use of PFHxS in lubricants has been identified.

According to the newest data from [Eurostat 2018g] (newest data available for 2014), a significant share of used oils was recovered/recycled in the EU. About 86% of waste oils were recovered/recycled, around 13% incinerated (with and without energy recovery) and only about 1% was disposed. Considering that the major share of spent oils is recovered/recycled in the EU, it cannot be excluded that PFHxS-containing or -contaminated waste oils enter recycling streams and consequently end up in new recovered/recycled oils. However, no qualitative or quantitative information on waste and recyclates containing or contaminated with PFHxS, its salts and related compounds is currently available.
Other uses reported in literature

Other uses reported in literature include polishing and cleaning/washing agents, coating and impregnation/proofing, FR, pesticide, plasticiser, etc.

One PFHxS-related compound (CAS No: 67584-53-6) was reportedly used in polishing agents and cleaning/washing agents at least between 2000 and 2015 in Denmark, Norway and Sweden with the use volumes claimed as confidential business information. For instance, the FCP102 Floor Sealer and FCP300 Duro Gloss Floor Sealer & Finish from Fritztille contain this compound. Another PFHxS-related compound (CAS No: 67584-61-6) was reportedly used in impregnation/proofing for protection from damp, fungus, etc. at least in four products between 2003 and 2009 in Denmark. Hubei Hengxin has marketed the potassium salt of PFHxS and PFHxS-related compounds (CAS No: 68259-15-4) for potential uses as a FR and in pesticides [UNEP/POPS/POPRC.14/2 2018].

In summary, PFHxS is one of the most frequently detected PFAS in the general population and has a very long half-life in humans. It has been detected in human blood, serum and breast milk. High concentrations of PFHxS have been detected in soil, ground and drinking water near airports or fire-fighting training sites, sludge and wastewater from waste water treatment plants, as well as near PFAS/PFHxS production/usage plants and in leachate from landfills.

Similar to PFOA, information about manufacture of PFHxS, its salts and PFHxS-related compounds is limited. Historical production was mainly carried out by 3M and possibly also by some producers in Japan. One European manufacturer and a number of producers in China have been identified, however, quantitative production data is not publicly available. PFHxS, its salts and many PFHxS-related compounds have been listed on various national chemical inventories (USA, Canada, Japan, Australia, Nordic countries and China) indicating historical/present production, import and/or use of products containing these substances.

Quantitative and qualitative information on PFHxS use is scarce. Important uses include the use in AFFFs, carpet and upholstery, waterproofing of textiles, in metal plating and lubricants. In addition, no information is available on wastes and recyclates. Therefore, based on the currently available information, it is not possible to investigate certain waste streams in detail as it has been done for other substances in this report and develop substance and mass flows. Nevertheless, lower and upper limitation criteria will be evaluated as far as possible and recommendations will be provided for potential limit values in this report. Besides, material flows associated with main applications may be summarized qualitatively as follows:

- PFHxS-containing AFFFs may reach the EU market and current production of AFFFs containing PFHxS cannot be excluded. However, most AFFFs are typically incinerated in hazardous waste incinerators and therefore possible PFHxS contents are usually destroyed.
- Use and stocks associated with metal plating applications are highly speculative. Presumably, associated waste streams will undergo controlled waste management in the EU.
- Most of the PFHxS use in carpets and textiles was presumably phased out in 2002. Therefore, significant amounts of relevant textile waste are not expected anymore. Moreover, a major share of textile waste is typically landfilled or incinerated. Textile recycling is currently not a common waste management operation applied in the EU (might change in the future) and it is therefore not expected that PFHxS accumulates in recycled products. The same also applies to lather products and related wastes.
- It cannot be excluded that PFHxS-containing or -contaminated waste oils enter recycling streams and consequently end up in new recovered/recycled oils.
- It cannot be avoided that semiconductors or finished products containing semiconductors enter the EU market in relevant amounts and are subject to versatile waste management practices (e.g. WEEE, ELV, municipal solid waste). Consequently, disperse accumulations cannot be ruled out in the future.
Also in the case of PFHxS, the project team believes, that the approaches described in the Zürich statement are important for future global action on PFASs and relevant with regard to POP waste management. Especially the grouping approach is important for the formulation of possible limit values for the PFAS families. Further, the need of additional information is essential and strongly supported by the project team. This problem has been identified in the course of the research as part of this study. Information on uses and current and future production is largely missing. Studies planned by Germany as well as Netherlands might help to improve the current situation regarding the data availability. Besides, Norway also foresees to continue the work on PFAS (including PFHxS).
4.7 Hexabromocyclododecane (HBCDD)

HBCDD belongs to the group of BFR. It is an additive type FR which is not chemically bound to the plastic matrix. HBCDD has long been the most important FR used in polystyrene insulation materials (expanded polystyrene (EPS) and extruded polystyrene (XPS)). Besides, HBCDD has been used in textiles, in packaging material (EPS), or in HIPS for electrical and electronic applications.

HBCDD is already listed as a POP in the EU POP regulation. An assessment with subsequent derivation of limit values for Annexes 4 and 5 of the EU POP regulation was already carried out in 2011. Details on relevant waste and substance streams are available in a corresponding report [BiPRO 2011]. The concentration limit in Annex IV of the EU POP regulation for HBCDD (LPCL = 1,000 mg/kg) is subject to review by 20 April 2019. The MPCL in Annex V of the EU POP regulation is also established at 1,000 mg/kg.

Against this background, the information in this and the following sections on HBCDD build upon existing knowledge and aim to provide new information which facilitates a revision of the existing LPCL of 1,000 mg/kg.

4.7.1 PRODUCTION, TRADE AND USE

HBCDD entered the world market in the late 1960s and since then was used in insulation boards in the construction sector. According to [BiPRO 2011] the main use of HBCDD in the EU (90 %) was in EPS and XPS for the building sector. Until (approximately) 2007 HBCDD was used for XPS in slightly higher amounts than for EPS (approximate ratio 52:48) [BiPRO 2011]. According to the HBCDD Industry Group the EPS/XPS-ratio in the last years of use of HBCDD was approximately 67:33 [European HBCDD IG & EUMEPS Sub. 2018a]. Approximately 6 % of HBCDD was used for EPS and XPS insulation panels used for transportation vehicles, packaging and other applications. About 2 % of the total consumption of HBCDD was in HIPS for electronic products and articles. Around 2 % was used in textile coating [BiPRO 2011].

Data on past consumption/demand for HBCDD in the EU can be found in [ECB 2008], [Nordic COM 2007], [IOM 2008] and [ICL-IP Europe BV 2014]. The consumption/demand of HBCDD in EPS/XPS outside the construction industry - in HIPS in electronic products and in polymer dispersions in textiles - stopped in 2014. Exemptions under REACH for the use of HBCDD until 2017 applied only to HBCDD for insulation material in the building and construction sector [ECHA n.d.]. HBCDD was used in the EU in EPS and XPS in the construction and building industry until 2016.

Data on the total demand of HBCDD in Europe demonstrate a significant decline after the year 2014 (see Table 42).

<table>
<thead>
<tr>
<th>Year</th>
<th>2011</th>
<th>2012</th>
<th>2013</th>
<th>2014</th>
<th>2015</th>
</tr>
</thead>
<tbody>
<tr>
<td>HBCDD demand EU (in tonnes)</td>
<td>12,300</td>
<td>11,560</td>
<td>10,800</td>
<td>10,800</td>
<td>2,800</td>
</tr>
</tbody>
</table>

Based on available data, Figure 26 shows past consumption of HBCDD in the main applications in Europe (project focus “EPS - construction” and “EPS - packaging” which is included in “EPS/XPS - others”) in tonnes.
Today in the EU, there is no more production, trade and use of the substance HBCDD as such. However, imports of products or waste containing HBCDD are still possible.

HBCDD was usually substituted by alternative chemical FRs. In EPS it was replaced by a brominated polymer (PolyFR), which is added to EPS insulation material in a proportion of about 1% (w/w). According to current knowledge, this replacement does not have the problematic environmental properties of HBCDD, since the plastic-like molecules are not bioavailable due to their size and are more firmly integrated into the plastic matrix [UBA 2017b].

Historically, HBCDD has been on the world market since the 1960s. HBCDD is solely used as an additive FR [IOM 2008]. HBCDD was used on its own or in combination with other FRs e.g. antimony trioxide and decaBDE. HBCDD is used in four principal product types, which are Expandable Polystyrene (EPS), Extruded Polystyrene (XPS), High Impact Polystyrene (HIPS) and in polymer dispersions for textiles [ECB 2008]. In PS foams, HBCDD was typically used alone and not with other additives [PS foam 2011].

Further details on historic use of HBCDD and the four main product categories are available in [BiPRO 2011].

4.7.2 CHEMICAL CHARACTERISTICS

Chemical formula: HBCDD $C_{12}H_{18}Br_6$
CAS registry number: HBCDD 25637-99-4\(^{48}\)

\(^{48}\) There is another CAS number for the same substance: 3194-55-6
There are 16 isomers of HBCDD which differ in the spatial arrangement of six covalent bonds of bromine substituents (see Figure 27). In technical products α-, β- and γ-isomers occur. A distinction is made between high and low melting HBCDD. The distribution of α-, β- and γ-isomers in technical HBCDD varies between 70-95 % of γ-HBCDD and 5-30 % of α- and β-HBCDD [Swerea 2010]. Low melting HBCDD contains 70-80% of γ-HBCDD and 20-30 % of α- and β-HBCDD. High melting HBCDD contains 90 % or more of γ-HBCDD. Commercial HBCDD is present as a white solid.

Detailed information on the chemical characterisation of HBCDD is available in the Stockholm Convention Risk Profiles (see [UNEP/POPS/POPRC.6/13/Add.2 2010], section 1.1).

4.7.3 LEGAL BACKGROUND

REACH

HBCDD is listed as a PBT substance in Annex XIV (List of substances subject to authorisation) of the REACH Regulation ((EC) No 1907/2006) coming into force on 21 February 2006 and expiring on 21 August 2015. After the expiration date, placing on the market and use of the substance is prohibited. In order to give manufacturers of EPS insulation materials enough time to replace HBCDD as a FR, an authorisation was granted by the European Commission in 2016 for the formulation of flame-retarded EPS to solid unexpanded pellets using HBCDD as the FR additive (for onward use in building applications) and for the production of flame-retarded EPS articles for use in building applications. The authorisation expired by 21 August 2017 [EC 2016].

It should be noted that under Article 56(6) (a) the authorisation is not required for PBT substances such as HBCDD in case it is present in preparations with a concentration limit of below 0.1 % by weight.

HBCDD and all relevant isomers (CAS No 25637-99-4, 3194-55-6, 134237- 50-6, 134237-51-7, 134237-52-8) are identified as SVHC due to their PBT properties and accordingly are added to the Candidate List from 28 October 2008 (Decision number ED/67/2008). Further this leads to certain obligations in the REACH Regulation laid down in Article 7.2 (inform ECHA), Article 7.3 (appropriate instructions to the recipient), Article 31.1 (provide a safety data sheet) and Article 33 (duty to communicate information on substances in articles).

Stockholm Convention

In May 2013, a global ban on HBCDD was decided at the COP to the Stockholm Convention [UNEP/POPS/COP.6/33 2013], Decision [UNEP/POPS/COP.6/SC-6/13 2013]. HBCDD was listed in Annex A with specific exemptions for production (As allowed for the parties listed in the Register)
and use (of EPS and extruded polystyrene in buildings) in accordance with the provisions of part VII of Annex A to the Convention (see Stockholm Convention Annex A part I and part VII).

**EU POP Regulation**

The EU has implemented this ban by listing HBCDD in part A of Annex I to Regulation (EC) No 850/2004 (POP regulation). As such, the production, placing on the market and use of HBCDD (except below an UTC limit of 100 mg/kg) is prohibited, in accordance with Article 3 par. 1 of the POP regulation. HBCDD was added to Annex IV by Regulation (EU) 2016/460 with a concentration limit of 1,000 mg/kg, subject to review by the Commission by 20 April 2019. Therefore, waste containing HBCDD with a concentration above 1,000 mg/kg shall be disposed of in accordance with Article 7 par. 2 of the POP regulation.

The EU POP regulation (Par. 4 sub a of Article 7) determines that, by way of derogation from paragraph 2:(a), waste containing or contaminated by any substance listed in Annex IV may be otherwise disposed of or recovered in accordance with the relevant Community legislation, provided that the content of the listed substances in the waste is below the concentration limits to be specified in Annex IV. Therefore, waste containing HBCDD with a concentration below 1,000 mg/kg may be treated in a way which does not destroy or irreversibly transform the HBCDD content. This means that recycling of HBCDD is principally allowed for waste containing HBCDD in a concentration below the concentration limit provided in Annex IV to the POP Regulation. However, the provisions regarding production, placing on the market and use of HBCDD as specified in Annex I to the EU POP regulation need to be respected.

**WEEE Directive**

According to Annex VII of 2012/19/EU of the European Union and the Council on WEEE on the selective treatment of materials and components of EEE, referred to in Article 8 (2), plastics containing BFR have to be removed from any separately collected WEEE.

**Classification in accordance with the CLP Regulation**

HBCDD has the classification Reproductive toxicity, Category 2; H361\(^{49}\). Additional category for effects on or via lactation; H362\(^{50}\) are in place. Furthermore, HBCDD is classified in the registration dossier that was submitted to the ECHA with the environmental hazard statement H410\(^{51}\) [BiPRO 2015; ECHA 2018c].

The classification of mixtures containing these substances results from Annex 1 of Regulation (EC) 1272/2008.

\(^{49}\) „Suspected of damaging fertility or the unborn child”

\(^{50}\) „May cause harm to breast-fed children”

\(^{51}\) „Very toxic to aquatic life with long lasting effects”
### 4.7.4 OCCURRENCE IN PRODUCTS, WASTES AND RECYCLATES

#### Products

HBCDD was added to the products to achieve specific flame retardancy according to the product requirements. HBCDD-containing products can be allocated to four categories. Table 43 shows typical HBCDD concentrations of relevant HBCDD-containing product categories [BiPRO 2015].

**Table 43:** Typical HBCDD concentration in HBCDD-containing product categories (adopted from [BiPRO 2015])

<table>
<thead>
<tr>
<th>Product category</th>
<th>HBCDD concentration (in % by weight)</th>
<th>Remark</th>
</tr>
</thead>
<tbody>
<tr>
<td>EPS products</td>
<td>0.7</td>
<td>Major use: construction</td>
</tr>
<tr>
<td>XPS products</td>
<td>1.5</td>
<td>Major use: construction</td>
</tr>
<tr>
<td>EPS/XPS others</td>
<td>0.7</td>
<td>Use: EPS products used for packaging are particularly relevant</td>
</tr>
<tr>
<td>HIPS products</td>
<td>4.0</td>
<td>Use: electric and electronic equipment</td>
</tr>
<tr>
<td>Textiles</td>
<td>25 (in coatings)</td>
<td>Use: coating of fabrics</td>
</tr>
<tr>
<td></td>
<td>8 (in treated textiles)</td>
<td></td>
</tr>
</tbody>
</table>

Detailed information regarding HBCDD related product categories is available in [BiPRO 2011] and [BiPRO 2015].

The relative importance of the consumption of HBCDD for the product categories changed over time. Figure 26 illustrates how the consumption of HBCDD for these product categories developed over time from 1988 until 2017 in the EU and when the different uses terminated. The last use occurred in 2016 for EPS and XPS in the construction sector. The illustration demonstrates that the dominant uses were for EPS and XPS products in the construction sector. Table 44 shows that historically these dominant uses in the construction sector amounted to 86.5% whereas the consumption for the other uses was less relevant (5.9% for textiles, 5.6% EPS/XPS particularly for packaging, 2.0% HIPS for electric products). Table 44 shows the shares of HBCDD consumed for the different product categories from 1988 to 2017 (cumulated consumption). The figures related to XPS products for construction are referred to the year 2015 since the use of HBCDD in XPS stopped in 2015, because no application for authorisation was submitted.

**Table 44:** Cumulated HBCDD consumption for the relevant product categories from 1988 to 2017

<table>
<thead>
<tr>
<th>Product category</th>
<th>Share of cumulated consumption from 1988 to 2017 (%)</th>
<th>Cumulated consumption from 1988 to 2017 (tonnes)</th>
</tr>
</thead>
<tbody>
<tr>
<td>EPS products for construction</td>
<td>45.0</td>
<td>100,639</td>
</tr>
<tr>
<td>XPS products for construction</td>
<td>41.5</td>
<td>92,701</td>
</tr>
<tr>
<td>EPS/XPS particularly for packaging</td>
<td>5.6</td>
<td>12,609</td>
</tr>
</tbody>
</table>
Waste

Specific information on the occurrence of HBCDD in wastes is contained in section B.4 of the Revised technical guidelines on the environmentally sound management of wastes consisting of, containing or contaminated with HBCDD.\(^{52}\)

The product lifetime is of crucial importance when estimating the presence of HBCDD in products, waste streams and recyclates. Various sources ([Morf et al. 2007; EBC 2008; PS foam 2011]) were considered in [BiPRO 2011] to estimate the average product lifetimes and ranges for the most relevant applications of HBCDD (details see [BiPRO 2011]). After the product’s lifetime, the products are disposed of as waste and cause contamination of waste streams and possibly recyclates. The product lifetimes of EPS/XPS in other applications relates to a broad range of relevant products (e.g. packaging, car seats, theatre props) and lifetimes (0 to 20 according to [BiPRO 2015]). However, these other applications are heavily dominated by EPS packaging, which is usually disposed of immediately after use and therefore contributes by far most to the generation of waste within one year after use [BiPRO 2015]. Table 45 gives an overview of the relevant average product lifetimes and ranges.

Table 45: Average product lifetimes and ranges in years for the most relevant applications of HBCDD (adopted from [BiPRO 2011; BiPRO 2015])

<table>
<thead>
<tr>
<th>Application</th>
<th>EPS/XPS in construction sector</th>
<th>EPS/XPS in other applications</th>
<th>HIPS for electrical equipment</th>
<th>Polymer dispersions for textiles</th>
</tr>
</thead>
<tbody>
<tr>
<td>Product lifetimes (in years; range in brackets)</td>
<td>50 (+/-25)</td>
<td>0.5 (+/- 0.5)</td>
<td>9 (+/-5)</td>
<td>10 (+/-3)</td>
</tr>
</tbody>
</table>

Table 46 shows the estimated cumulated quantities of HBCDD-containing waste and HBCDD contained in that waste from estimated use starting in 1988 until 2017, as illustrated in Figure 26. The waste quantities are calculated considering the concentrations indicated in Table 43. The amount of HBCDD which becomes waste by 2017 and after 2017 has been calculated using the product lifetimes and standard deviations shown in Table 45. The calculation takes into account that 1% of cut offs became waste in the year of production. The estimation does not take imports of products containing HBCDD (e.g. EPS packaging containing HBCDD) into account.

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Table 46: Cumulated quantities of waste and HBCDD in waste from estimated use from 1988 to 2017 in the EU (in tonnes)

<table>
<thead>
<tr>
<th>Use-category</th>
<th>Quantities</th>
<th>Will become waste after 2017</th>
<th>Already waste by 2017</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Tonnes waste</td>
<td>Tonnes HBCDD in waste</td>
<td>Tonnes HBCDD in waste</td>
</tr>
<tr>
<td>EPS</td>
<td>14,233,253</td>
<td>99,633</td>
<td>99,105</td>
</tr>
<tr>
<td>XPS</td>
<td>6,118,284</td>
<td>91,774</td>
<td>91,199</td>
</tr>
<tr>
<td>EPS&amp;XPS</td>
<td>1,801,338</td>
<td>12,609</td>
<td>1,483</td>
</tr>
<tr>
<td>HIPS</td>
<td>109,720</td>
<td>4,389</td>
<td>1,483</td>
</tr>
<tr>
<td>Textiles</td>
<td>166,268</td>
<td>13,301</td>
<td>1,619</td>
</tr>
<tr>
<td>Total</td>
<td>22,428,862</td>
<td>221,707</td>
<td>193,406</td>
</tr>
</tbody>
</table>

In total the cumulated historic use of HBCDD is related to the generation of more than 22,000 kt of waste which contains 221,707 tonnes of HBCDD. The bulk of the HBCDD-containing waste (87.2%) will arise in the future (i.e. past 2017, see Table 46 and Figure 28). Most relevant is EPS and XPS waste from the construction sector, mainly due to the long lifetime of the corresponding products and the important quantities used in this sector. More than 99% of HBCDD-containing EPS and XPS construction waste will arise in the future (peak expected around the 2050s; see [BiPRO 2011]) whereas HBCDD-containing EPS/XPS waste is already completely disposed of (according to the model which does not take into account imports and assumes that all packaging waste is disposed within one year). HBCDD-containing HIPS in electric equipment and HBCDD-containing textiles did already arise as waste in the past to a large degree (66.2% and 87.8% respectively).
Though the use of HBCDD in packaging has stopped in 2014 in the EU, HBCDD was found in 90% of 70 Irish and UK polystyrene packaging samples collected from 2015 to 2016. The concentration found ranged between <0.0007 mg/kg and 5,900 mg/kg. Four of the samples exceeded the LPCL level of 1,000 mg/kg. According to the authors, the isomer ratios found in EPS packaging suggest the source of HBCDD in PS packaging is recycled PS insulation foam [Abdallah et al. 2018].

In [BiPRO 2015] four relevant samples from recycling companies were procured and analysed (EPS mixed fraction packaging waste; grinded EPS packaging; EPS packaging waste; recyclates of EPS packaging). The results ranged between 0.165 mg/kg and 42 mg/kg. The results confirmed that waste fractions of EPS packaging usually contain HBCDD, however, levels of contamination are comparatively low. The results suggest that the average HBCDD contamination of packaging waste may be about 10 mg/kg [BiPRO 2015].

BFR were widely used within EPS and XPS and within the Netherlands between 1975 and 2015 all EPS and XPS used within the construction sector contained HBCDD. As the Stockholm Convention obligates the Netherlands to develop and execute the national implementation including to environmentally sound management of HBCDD in a first step an inventory of the volumes of various products and waste streams possibly contained HBCDD in the Netherlands was commissioned. The concentrations of HBCDD in those streams were not determined at this point (for more information see [Giraf Results 2016]).

In 2017 it was decided to investigate the HBCDD concentrations in the various streams which has been done in [Giraf Results 2018]. For the 2018 inventory a three-phase process was used to determine the occurrence of HCBD. Firstly, XRF spectrometry was used for initial screening of large samples, followed by XRF spectrometry with extraction use on samples containing Br to determine if it is soluble or insoluble and lastly LC-MS to accurately measure the HBCDD concentrations. Relevant concentrations of HCBD in waste were only found in packaging waste and construction waste, new packaging material did not contain HBCDD.
Based on the LC-MS method concentrations of HBCDD ranging from below 10 mg/kg to 17,100 mg/kg in construction waste was found. Of the samples tested above half contained concentrations above 1,000 mg/kg HBCDD. In packaging waste from household waste recycling centres, a small part contained HBCDD. Concentrations as high as 18,100 mg/kg were found within this fraction. Those high values are due to contamination with construction waste. This can be explained as the collection of packaging waste in the Netherlands is done via household waste recycling centres were packaging EPS waste is supposed to be collected as brought in by citizens. Average values of not contaminated packaging were calculated to be around 265 mg/kg assuming the brominated compounds in the remaining packaging EPS would all be HBCDD.

Recyclates

Production, placing on the market and use of HBCDD in the EU has been largely phased out between 2014 and 2016. The HBCDD content of waste containing HBCDD in concentrations above 1,000 mg/kg53 has to be destroyed or irreversibly transformed without undue delay. Therefore, for such waste, recycling of the HBCDD-containing waste is not an option. Although, HBCDD can still enter the waste streams in the EU today or will enter the waste stream in the future and consequently (will) occur in recyclates and new products. This is due to the following facts:

- Since waste containing HBCDD below 1,000 mg/kg is not defined as POP-containing waste, recycling would be allowed for such waste material. The maximum UTC for HBCDD (100 mg/kg54) must, however, be respected.
- The unintentional trace UTC in Annex I EU POP Regulation (100 mg/kg)
- Unintentional recycling of products containing HBCDD above the LPCL according to Annex IV EU POP Regulation (1,000 mg/kg)
- Imports of HBCDD-containing EPS products from outside the EU (e.g. EPS packaging)

Thus, according to [Abdallah et al. 2018] several studies raise concerns that new applications of recycled polymeric material that in its original application had been flame retarded, has led to the undesirable presence of restricted BFRs in goods such as children’s toys, food contact articles and PS packaging [Guzzonato et al. 2017; Kuang et al. 2018; Puype et al. 2015; Rani et al. 2014].

[Rani et al. 2014] report levels of HBCDD in PS based consumer products in Korea. The concentration of HBCDD was highest in EPS, with a mean value of 475 mg/kg. PS related to building construction and laboratory uses had a significantly higher concentration of HBCDD (3.3 to 905mg/kg), except XPS styroboard (0.191 mg/kg). Lower concentrations were measured in most food-related products (0.024–0.199 mg/kg). However, a relatively high concentration of HBCDD was detected in an ice box (960 mg/kg), aquaculture buoy (53.5mg/kg), and disposable tray (8.4 mg/kg) used in fish market. According to the authors, the data demonstrate a wide variation in the concentration of HBCDD, suggesting a lack of proper controls for the addition of HBCDD to PS products [Rani et al. 2014].

According to [DiGangi et al. 2017], HBCDD may be found in several consumer goods on the market. Rubik’s cubes and several other consumer goods were screened for bromine using a handheld XRF analyser to identify samples with significant bromine levels (hundreds of mg/kg). Positive samples were analysed for HBCDD (LOD 3 µg/kg). Results demonstrate that forty-five samples (43 %) contained HBCDD at concentrations ranging from 1 to 1,586 mg/kg. In products in which it was measured, seven samples (7%) contained HBCDD at concentrations higher than 100 mg/kg. Two samples exceeded the higher proposed level of 1,000 mg/kg [DiGangi et al. 53 Subject to review by 22 March 2019
54 Subject to review by 22 March 2019
A continued investigation by [Straková et al. 2018b] found HBCDD concentrations ranging from less than 0.01 mg/kg to 91.07 mg/kg in toys and concentrations of up to 7.71 mg/kg in hair accessories, both from sample products on the Czech market. [Straková et al. 2018a] found that 41% of the toys, hair accessories and kitchen utensils which were acquired for the study in 2018, contained HBCDD ranging from 1 to 207 mg/kg in. The concentration levels detected in the study in EU MS can be found in more detail in the Annex in section 10.10.

4.7.5 SELECTION OF MOST RELEVANT WASTE STREAMS IN THE EU

Considering the information from Table 44 and Table 46, i.e. from a HBCDD consumption and a waste perspective, the most relevant products and wastes are EPS and XPS in the construction sector, HIPS in the electronics sector and HBCDD-containing textiles related to the use of PD in the textile sector.

In other EPS and XPS relevant applications, which are dominated by EPS packaging, HBCDD is already no longer used in the EU. Due to the short product lifetime of EPS packaging HBCDD should not enter the waste streams anymore in relevant quantities via domestic sources. However, it is reported that HBCDD-containing EPS packaging enters the EU and the EPS packaging waste streams via imports [European HBCDD IG & EUMEPS Sub. 2018a; Plastic Recyc. Europe Sub. 2018]. In a study conducted by [Giraf Results 2018] concentrations above 100 mg/kg were only found in one sample tested and thus it was concluded that HBCDD does not occur in relevant concentration on imported packaging.

By far the largest HBCDD and waste quantities are expected in the coming decades and are expected to originate from construction and demolition (peak around the 2050s). HBCDD in waste streams stemming of HIPS from the electronic sector is still relevant at low levels but will decrease from around the beginning of the 2020s. The relevance of HBCDD in textile waste is already decreasing [BiPRO 2011; BiPRO 2015].

In the following sections additional information will be provided on

- EPS in the construction sector
- XPS in the construction sector
- EPS in the packaging sector
- HIPS in the electronic sector
- PD in the textile sector

4.7.6 EXPANDED POLYSTYRENE FOAM (EPS) IN CONSTRUCTION SECTOR

4.7.6.1 Background information

Detailed information on HBCDD and EPS in the construction sector, including background information, processing, releases, waste treatment and mass flows is provided in available reports. In this respect we refer to [BiPRO 2011] and [BiPRO 2015].

In the following section of the present report relevant new information is provided. In particular, the EU mass flows elaborated on in above mentioned BiPRO reports were based on specific assumptions, including waste quantities which may arise. The mass flows in the present report are based on recent information including data on post-consumer waste generation and management in EU countries in 2017 [Conversio 2018].

4.7.6.2 Processes, inputs and outputs

See [BiPRO 2011] for details.
4.7.6.3 **Concentration in products, wastes and recyclates**

A stakeholder survey conducted in 2017 indicates that concentrations of HBCDD in EPS are between 0.5 % and 1 % (5.100 mg/kg to 10,000 mg/kg) [Giraf Results 2016; European HBCDD IG and EUMEPS Sub. 2018b]. At EU, in 2009, on average, approximately 77% of EPS insulation boards for construction manufactured in the EU were flame retarded with HBCDD [BiPRO 2011]. A study conducted by [Giraf Results 2018] found that in the Netherlands around 70% of EPS/XPS from construction and demolition waste collected contained HBCDD in concentrations above 1,000 mg/kg whereof 96% were from EPS. It was put forward by [Giraf Results 2018] as the most likely conclusion, that some of the EPS/XPS in construction waste was packaging material.

For the calculation of the mass flows a concentration of 0.7% (typical functional concentration in EPS) is applied (see also section 4.8.4; for details see [BiPRO 2015]).

4.7.6.4 **Activity data**

Figure 26 shows the past consumption of HBCDD in the main applications in Europe. EPS for construction is related to 45% of the historic HBCDD consumption in the EU (see Table 44).

EPS/XPS boards are bulky products which are not likely to be transported over long distances. Insulation boards are mostly manufactured for the market where these are used [Swerea 2010].

Fire safety regulations can vary among individual MS. Therefore, regional differences in the consumption of HBCDD occur. In 2009, on average, approximately 77% of EPS/XPS insulation boards for construction manufactured in the EU were flame retarded with HBCDD. See [BiPRO 2011] for further details.

A lifetime of 50 years, with a standard deviation of 25 years is assumed for both EPS and XPS products for the construction sector.

4.7.6.5 **Waste management**

According to [EC 2017] construction and demolition (C&D) waste accounts for 1/3 of all waste generated in the EU. Latest data of the generation and treatment of C&D on [Eurostat 2018e] are available for 2014. In 2014 the EU MS generated 860 million tonnes of C&D waste, meaning 1.7 t/cap. The generation figures vary among EU MS. For instance, in Greece only around 0.04 tonnes of C&D waste are generated per capita, whereas in Luxembourg 10.8 tonnes have been reported. Figure 29 provides an overview of the C&D waste generation in the different EU MS.
In the "EU Construction & Demolition Waste Management Protocol" [EC 2017] it is stated, that almost 50% of C&D waste is recycled (including backfilling), whereby the WFD’s target lies at 70% in 2020. Already in the report from [Bio 2011] a recycling rate of 47% is indicated (as broad estimation). As with the generation figures, significant variations can be observed for the recycling rates of C&D waste in the EU-MS. Recycling rates thereby refer to reuse, recycling and other pathways of material recovery. It has been reported that Denmark, Estonia, Germany, Ireland and the Netherlands already fulfil recycling targets for C&D waste of 70%. Austria, Belgium, France, Lithuania and the United Kingdom achieve recycling rates between 60% and 70%. Latvia, Luxembourg and Slovenia attain lower recycling rates between 40% and 60%. Cyprus, Czech Republic, Finland, Greece, Hungary, Poland, Portugal and Spain reported recycling rates below 40% and for Bulgaria, Italy, Malta, Romania, Slovakia and Sweden no relevant data was available in order to allow estimation [Bio 2011].

Even though some EU countries attain high recycling rates for the inert fraction, which is the main fraction of C&D waste, recycling rates for other C&D waste fractions and in particular for plastic waste remain low. Established techniques for recycling the inert fraction exist and where they are applied high recycling rates are reached. The steel fraction is also well recycled. On the contrary, plastics are poorly collected and recycled. They are usually landfilled or incinerated [BiPRO 2011]. This may not apply for MS with landfill restrictions of recyclable and recoverable waste. On average, these countries have higher recycling rates of plastic post-consumer waste, since landfilling is not an option. These MS are namely Austria, Germany, the Netherlands, Sweden, Denmark, Luxembourg, Belgium and Finland [PlasticsEurope 2018].

Among the construction waste insulation material EPS / XPS is currently assigned the waste code 17 06 04 (insulation materials other than those mentioned in 17 06 01 and 17 06 03).

The total waste quantity generated in construction and demolition waste is estimated to amount to around 860 million tonnes. EPS and XPS are here not the primary target for the mechanical recycling. They are separated as far as possible from the mineral content and separately recovered or disposed. However, a clean separation in practice is very difficult due to the attachment to e.g. mineral components (concrete, brick, plaster, etc.). This leaves impurities in mineral fractions which enter the corresponding wastes and recyclates (because of the relatively small mass in very low concentrations).
An unquantifiable proportion of EPS waste is therefore recycled or disposed together with mineral construction and demolition waste, e.g. according to the German Landfill Ordinance, the loss of ignition may vary depending on the landfill class up to 3.5 or 10%. For the landfill class 0 for inert waste TOC can be up to 3%. Accordingly, it is possible to landfill mineral construction waste with organic contaminants such as EPS or XPS insulation up to a level of 3% [BiPRO 2015].

Quantities which remain in the environment after demolition are difficult to estimate and are not included in the calculation of mass flows. This does not exclude that these quantities may not be relevant.

The plastic content of the C&D waste in the EU amounts to 930,000 tonnes/y in 2014 (latest available data), which means a share of around 0.11% [Eurostat 2018e]. Only a small proportion of the C&D plastic waste is related to EPS. This indicates the small proportion of total waste related to EPS. EPS waste is mainly caused by the dismantling of buildings (insulation). The different fractions are either separated during the dismantling process or in a subsequent process.

In the future, an increasing amount of HBCDD-containing products will leave the use stage and enter the waste management sector. As a consequence, the amounts of HBCDD contained in C&D waste to be managed will increase. The bulk of this waste will be disposed of in the future with an expected peak around the 2050s or later [BiPRO 2011].

According to recent data, the quantity of post-consumer EPS construction waste in the EU amounted to 138,700 tonnes in 2017. Out of these, 98,600 tonnes consist of demolition waste [Conversio 2018].

EPS as insulation material can be recovered in three different ways (according to [Fraunhofer IRB 2015]):

- Material recycling
- Feedstock recycling
- Energetic recovery.

**Figure 30: Overview of suitable recovery processes for EPS insulation material [Fraunhofer IRB 2015]**

When treating HBCDD-containing EPS, according to Art. 7 (2) of the POP Regulation ((EC) No 850/2004) waste containing POPs must be disposed of or recovered in such a way as to ensure “that the persistent organic pollutant content is destroyed or irreversibly transformed”. The LPCL set for HBCDD of 1,000 mg/kg became effective on 30 September 2016. Destruction and irreversible transformation methods for the environmentally sound disposal of wastes with an
HBCDD content above the LPCL include at least cement kiln co-incineration, hazardous waste incineration and advanced solid waste incineration [UNEP/CHW.12/5/Add.7/Rev.1 2015].

Waste containing HBCDD is generated at each life cycle step. According to [ECB 2008], the so-called pre-consumer waste can be recycled into the process. The HBCDD-containing end products which become waste (post-consumer waste) are either incinerated, put on landfill, left in the environment or recycled. Waste ending up in the municipal waste streams is likely to be put on landfill or incinerated. HBCDD-containing waste could possibly also be left in the environment or used for road construction purposes or other purposes such as filling material. Another option is the recycling of larger pieces of EPS boards, where the EPS boards are ground and returned to the moulding process together with virgin EPS to form new boards. This has been indicated to occur in a number of European countries. It has also been reported that granulated EPS waste is used to improve the texture of agricultural and horticultural soil. To what extent end-products containing HBCDD are landfilled, incinerated, left in the environment or recycled in the EU will largely depend on the waste management strategy chosen by different MS [BiPRO 2011].

In the following, the three main recovery options for post-consumer EPS (insulation material) are discussed in more detail:

**Energetic recovery**

In 2013, a large-scale experiment for the combustion of EPS and XPS insulation panels in the state-of-the-art municipal solid waste incinerator (MSWI Würzburg) was carried out in Germany. According to the results, EPS and XPS insulation panels can be incinerated appropriately in state-of-the-art MSWI plants up to a share of 2 % of the total waste. HBCDD is largely destroyed (destruction rate > 99.999% independent from the amount of added PS-foam) and the incineration does not contribute to the generation of other POPs. Essential for the proper incineration is the preparation of a special blend in the bunker of incinerators. The incineration can be used without problems to manage commercially available insulation panels by using a fraction of 1 percent of weight (equivalent to about 15% per volume). Too high volumes could lead to disruptions in the operation of the waste incineration facility (e.g. clogging at the chute), eventually requiring the intervention by the operator. The co-feeding of 1 and 2 wt% of polystyrene foam had no influence on the operation of the plant [BiPRO 2015; Mark et al. 2015].

In another study by [Kajiwara et al. 2017], the factors influencing the destruction efficiency of HBCDD in co-incineration of XPS foam were tested at nine commercial-scale industrial waste incineration plants in Japan with various furnace types (rotary kiln, kiln stoker, stoker, fluidized bed, gasification melting), air pollution control systems and capacities. Source materials consisting of 0.34 to 3.3 weight-% of XPS insulation foam mixed with baseline industrial wastes were co-incinerated at temperatures of 880 to 1,210 °C, depending on the plant. The HBCDD destruction efficiency exceeded 99.999% at all the plants, with negligible discharge of brominated dioxins and furans. The type of furnace, the incineration temperature, the air pollution control system, and the XPS/waste mixing ratio had no obvious effect on the destruction efficiency. Usually approximately 80% of the total HBCDD output was found in the bottom ash, implying that undecomposed HBCDD is discharged unburned into the bottom ash. HBCDD concentrations in bottom ash ranged between <0.01 and 40 ng/g dry weight. The HBCDD concentration were the destruction efficiency was negatively correlated, suggesting that at XPS/waste mixing ratios of up to approximately 3 wt%, HBCDD residue concentrations in bottom ash may be a useful indicator that DE exceeds 99.999% [Kajiwara et al. 2017].

The results from [Mark et al. 2015] and [Kajiwara et al. 2017] indicate that EPS and XPS foams should not be co-incinerated in weight percentages above 2% and 3% respectively in order to ensure a high HBCDD destruction efficiency.
Advantages of energy recovery from EPS are (according to [Fraunhofer IRB 2015]):

- low transport distance when using municipal waste incineration plants
- low requirements for cleanliness and impurities content of the waste
- ecologically advantageous due to energy recovery and pollutant sink: EPS waste represents a valuable source of energy in solid waste recovery plants fully equipped to be compliant with air emissions requirements. Since EPS is made from oil, the stored energy can be recovered in modern incinerators and then used for local heating and for generation of electricity. In a modern incinerator, EPS releases most of its energy as heat. Pollution control equipment such as scrubbers and filters reduce pollutants release during incineration.

Mechanical (material) recycling:

HBCDD-free EPS used as insulation material can be easily recycled materially [UBA 2017b]. The actual use of the recycled products depends on the recycling process. On the one hand ("mechanical recycling as EPS"), EPS can be grounded and can be used as lightweight aggregates for mortar, concrete and insulating plaster. For the brick industry, the recycled EPS serves to form pores in clay material. On the other hand ("mechanical recycling as PS"), EPS can be melted/compressed and can be used as Polystyrene (PS) for the production of PS injection molded parts [IVH 2015].

Recovered foams can be reprocessed as they match the properties of virgin material foams. According to PS foam industry, PS is easily recycled and is one of the most recycled plastics. Since recycling of waste containing HBCDD above the LPCL of 1,000 mg/kg (HBCDD containing waste) is not permitted, for the purpose of recycling, HBCDD-free or low HBCDD-content EPS for recycling shall be identified and separated from HBCDD-containing waste. Identification and separation of HBCDD-containing material is therefore a key issue during collection and recycling. To distinguish between HBCDD-free and HBCDD-containing EPS materials different test methods exists. These are described in chapter 6.2.1.1.

In waste management practice, it is not clear to what extent, a distinction is made between streams of flame retarded and non-flame retarded material. There was no labelling/marking system so that the waste streams could be kept apart. In Germany, EPS insulation waste was sometimes mixed with EPS packaging waste. This leads to the mixing of HBCDD-free or low HBCDD content waste (EPS packaging waste) with HBCDD-containing waste (EPS insulation waste) and subsequently to relatively high content in PS re-granulates. It is not known how relevant such mixtures are in practice. In practice the mixing partly took place intentionally (insulation materials were deliberately mixed with packaging waste before recycling processes), and some mixing occurred due to the introduction of clean insulating materials (e.g. cut offs) in the packaging waste stream [BiPRO 2015]. In the Netherlands it was demonstrated that EPS from construction unintentionally enters the packaging waste stream during current waste collection [Giraf Results 2018]. There is no specific information on practices in other MS.

Theoretically, recycling of EPS/XPS-products can lead to an uncontrolled dilution and contamination of PS products, which do not need to be flame retarded, with HBCDD [BiPRO 2011].

Current obstacles for material recycling of EPS insulation material (according to [Fraunhofer IRB 2015]):

- Lack of logistics
- Lack of demand for recycled EPS insulation material
- Lack of economic efficiency
- The obligation, that HBCDD has to be removed from the environment [Fraunhofer IRB 2015].
Current drivers for material recycling of EPS insulation material (according to [Fraunhofer IRB 2015])

- Rising raw material prices and increasing EPS waste
- The regional distribution and technical equipment of the foaming producers, with their possibilities for reprocessing EPS

**Feedstock recycling**

An innovative process for the recycling of EPS containing HBCDD is CreaSolv®. The process consists of the CreaSolv® Technology and a Bromine Recovery Unit (BRU). In the process HBCDD is destroyed while PS and bromine can be recycled and used for further applications. The process is illustrated in Figure 31. A planned demonstration plant will be located in Terneuzen, Netherlands and start operation in 2019 [TÜV Rheinland 2018].

In preparation of the CreaSolv® process, shredding and compaction of EPS is necessary. Demolished External Thermal Insulation Composite Systems (ETICS) components are crushed and sieved and metals as well as inert material are separated. Finally, a compaction of EPS takes place. For the three separated streams different end of life treatment options are assumed (recycling for metal compounds; landfilling for amount of metal, which cannot be recycled; landfilling for amount of metal, which cannot be recycled;
incineration for inert material). After the compaction and shredding process the material is fed to the CreaSolv® Process and the BRU. The CreaSolv® Process is a selective extraction recycling process. First step is the dissolution of EPS using a selective solvent (other components in the waste fraction remain undissolved). After dissolution a separation of contaminants from the recovered polymer solution takes places. A precipitation of PS from the purified polymer solution follows. One component of polymer solution is the FR HBCDD. The HBCDD is then transported to the BRU plant (Terneuzen, Netherlands) where bromine is recovered [TÜV Rheinland 2018].

According to [TÜV Rheinland 2018], a Life Cycle Analysis shows that the CreaSolv® Process / BRU has a lower environmental impact in the impact categories climate change, eutrophication (freshwater), summer smog, resource depletion (fossil, elements), human toxicity (non-cancer, cancer), and freshwater ecotoxicity in comparison to the current status quo process (incineration with energy recovery). Effects for acidification and eutrophication (marine) are comparable for both alternatives [TÜV Rheinland 2018].

The selective extraction of EPS within this process is not yet a commercial process and it is difficult to assess how the process approach will develop over the coming years and decades under technical and economic conditions [TÜV Rheinland 2018; Fraunhofer IRB 2015].

**Waste management of EPS in the European Union**

Data regarding the waste management of EPS from the building and construction sector was provided by the PS foam industry for the purpose of the [BiPRO 2011] report according to which EPS waste from demolition and deconstruction have been sent to landfill (39.3 %), have been incinerated with energy recovery (R1; 53.1 %), and recycled (7.6 %; mainly recycled in the conversion process into new EPS insulation boards).

Recent data on EPS post-consumer waste generation and management in European Union (EU) countries in 2017 is available and can be used to refine the mass flows [Conversio 2018]. The total quantity of EPS construction waste in EU 28+2 in 2017 was 138.7 kt. Thereof 40 kt (29%) arise from installation of new (i.e. HBCDD-free) EPS products. 98.6 kt (71%) originate from demolition and contains HBCDD to a high extent in a typical concentration of 0.7% [Conversio 2018]. The waste originating from construction (98.6 kt) is relevant for the HBCDD mass flow. According to [Conversion 2018] the fate of this waste was in 2017 as follows:

- Mechanical recycling as EPS: 1.0 kt (1.0 %)
- Mechanical recycling as PS: 0.3 kt (0.3 %)
- Energy recovery: 68.6 kt (69.6 %)
- Landfill: 28.7 kt (29.1 %)

These shares are used to calculate the EPS and HBCDD flows related to EPS from C&D-waste.

Since recycling of EPS containing more than 1,000 mg/kg HBCDD is not allowed in the EU, we assume that HBCDD-containing EPS is incinerated (with or without energy recovery) and only EPS containing HBCDD below a concentration of 1,000 mg/kg is intentionally recycled.

EPS insulation has long lifetime in buildings. The use of EPS flame retarded with HBCDD in construction started in the late 1980’s and will need restoration or will be demolished in the future. Therefore, only a small share of HBCDD-containing EPS and XPS insulation materials from the C&D sector have been treated by existing waste management infrastructure in the EU so far (see Table 46).

**4.7.6.6 Substance and mass flow**

Figure 32 illustrates the resulting quantitative mass flows for HBCDD in EPS (substance flow) from collected EPS C&D waste (material flow). At EU level, 1.32% of the waste is recycled, 69.57% is incinerated with energy recovery and 29.11% is landfilled.
About 98.6 kt of EPS waste from C&D waste containing up to 960.2 tonnes of HBCDD was collected in 2017 and managed accordingly.

The mass flow is calculated based on a concentration of 0.7% HBCDD in EPS from C&D. However, in 2009, on average, approximately 77% of EPS insulation boards for construction manufactured in the EU were flame retarded with HBCDD. It is not clear to what extent the recycled EPS from C&D contains HBCDD.

4.7.7 EXTRUDED POLYSTYRENE FOAM (XPS) IN THE CONSTRUCTION SECTOR

4.7.7.1 Background information

XPS is a fine-pored foam or extruded polystyrene (XPS) which is mainly used in the construction sector as an insulating material. Due to the high flammability, XPS foam panels are flame retarded in the EU [BiPRO 2015].

Detailed information on HBCDD and XPS in the construction sector, including background information, processing, releases, waste treatment and mass flows is provided in available reports. In this respect we refer to [BiPRO 2011] and [BiPRO 2015].

In the following section of the present report relevant new information is given. In particular, the EU mass flows elaborated on in above mentioned BiPRO reports were based on specific assumptions, including waste quantities which arise. The mass flows in the present report are based on recent information including data on post-consumer waste generation and management in EU countries in 2017 [Conversio 2018]. For XPS from the construction sector data is based on [European HBCDD IG & EUMEPS Sub. 2018c].

4.7.7.2 Processes, inputs and outputs

See [BiPRO 2011] for details.

4.7.7.3 Concentration in products, wastes and recyclates

For XPS a typical concentration range as a FR was between 0.8% and 2.5% (8,000-25,000 mg/kg) [Giraf Results 2016].
For the calculation of the mass flows a typical functional concentration of 1.5% is applied (see also 4.7.4; for details see [BiPRO 2015]).

4.7.7.4 Activity data
Currently, the percentage of HBCDD used in insulation board in buildings is 31.5% for XPS as reported by the [European HBCDD IG & EUMEPS Sub. 2018c]. Figure 26 shows the past consumption of HBCDD in the main applications in Europe. XPS waste from construction is related to 41.5% of the historic HBCDD consumption in the EU (see Table 44). For further details see also section 4.7.6.4.

4.7.7.5 Waste management
Recent data on EPS post-consumer waste generation and management in EU countries in 2017 is available and is used to refine the mass flows for EPS waste [Conversio 2018]. However, the report does not contain specific information on the XPS waste quantities from demolition in 2017 or on the waste management practices.

[European HBCDD IG & EUMEPS Sub. 2018c] stated that recycling is not a treatment route for HBCDD-containing XPS since there is no recycling infrastructure in place. Only HBCDD free XPS is internally recycled. Specific data is largely missing for waste management of XPS and thus the substance and mass flow is largely based on data submitted by [European HBCDD IG & EUMEPS Sub. 2018c].

In accordance with the data provided by [European HBCDD IG & EUMEPS Sub. 2018c], 70.5% of the waste is incinerated with energy recovery and 29.5% are landfilled. There is no post-consumer recycling for XPS. XPS process foam waste is internally recycled.

XPS insulation has a long lifetime in buildings. The use of XPS flame retarded with HBCDD in construction started in the late 1980’s and will need restoration or will be demolished in the future. Therefore, only a small share of HBCDD-containing EPS and XPS insulation materials from the C&D sector have been treated by existing waste management infrastructure in the EU so far (see Table 46).

4.7.7.6 Substance and mass flow
Figure 33 illustrates the quantitative mass flows for HBCDD in XPS (substance flow) from collected XPS C&D waste (material flow). The collected waste quantity is estimated at approximately 32 kt. Based on the information provided from [European HBCDD IG & EUMEPS Sub. 2018c], at EU level 70.5% of XPS waste from construction is incinerated and 29.5% are landfilled. No recycling takes places.
Figure 33: Substance and mass flow HBCDD XPS

The mass flow is calculated based on a concentration of 1.5% HBCDD in XPS from C&D. However, in 2009, on average, approximately 77% of XPS insulation boards for construction manufactured in the EU were flame retarded with HBCDD.

4.7.8 USE OF EPS IN THE PACKAGING SECTOR

4.7.8.1 Background information

At European level, about 5.6 % of the HBCDD consumption for EPS and XPS products was used outside of the construction sector (see Table 44). Relevant information regarding the European situation is presented in [BiPRO 2011]. Further details are available in [BiPRO 2015].

According to industry data, the use of flame retarded PS foams outside the construction sector is essentially limited to EPS applications [BiPRO 2011]. XPS is of minor importance. Relevant are applications such as car seats, props for theatre, film and exhibitions. Additionally, possible applications are in rigid packing material for sensitive equipment or packaging material “chips” and moulded EPS parts. The [European HBCDD IG & EUMEPS Sub. 2018c] submitted information which indicated that packaging polystyrene foams typically do not contain HBCDD, as flame retarded properties are generally not required for this application.

Apart from packaging, the applications mentioned by the EU PS foam industry are considered to be of low relevance [PS foam 2011]. Therefore, the present section focuses on EPS packaging.

The use of HBCDD in EPS in packaging is banned in the EU and it is assumed that the last use in the EU was in 2014. Packaging is generally disposed of immediately after the use or within one year (see Table 45). Due to the short product lifetime of EPS packaging HBCDD should not enter the waste streams anymore in relevant quantities via domestic sources.

However, it is reported that HBCDD-containing EPS packaging enters the EU and the EPS packaging waste streams via imports and can enter the waste streams and thus recycles. However, it was pointed out that imports should have stopped since 22 February 2018, pursuant to EU Regulation 850/2004 [European HBCDD IG & EUMEPS Sub. 2018c].

HBCDD flame retarded products from the non-packaging sector do also enter the packaging waste stream as described by [Giraf Results 2018] who found that EPS packaging waste was unintentionally contaminated with EPS from construction waste.
4.7.8.2 Processes, inputs and outputs

See [BiPRO 2011] for details.

4.7.8.3 Concentrations in products, waste and recyclates

The HBCDD content of flame retarded EPS products is around 0.7% [BiPRO 2015]. For details see also section 4.7.4 and [BiPRO 2015].

Packaging polystyrene foams typically do not contain HBCDD, as flame retarded properties are generally not required in these applications. Levels of HBCDD in waste streams, due to eventual contamination, are expected to be far less than 1,000 mg/kg [European HBCDD IG & EUMEPS Sub. 2018c]. HBCDD should not be found in EPS food packaging in the light of compliance with the EC Regulation 1935/2004 on Food Contact Materials and EU Regulation 10/2011 on plastic materials and articles intended to come into contact with food [European HBCDD IG & EUMEPS Sub. 2018c].

Although the production and use of HBCDD for packaging material in the EU is prohibited, it must be considered, that small amounts of HBCDD-containing EPS packaging could still be imported (e.g. from countries which are not Parties to the Convention or have not ratified the HBCDD amendment) up until the beginning of 2018 and consequently enter the waste stream today. Plastic imports are mainly suggested to originate from the US, Korea, and China [PlasticsEurope 2018]. However, specific data is not available.

HBCDD may still be present in products, as an UTC of 100 mg/kg is still tolerated as well as the recycling of products with a concentration below 1,000 mg/kg.

The results from [BiPRO 2015] and [Abdallah et al. 2018] confirm that HBCDD is still present in packaging waste and recyclates made from packaging waste in the EU (see also section 4.7.4):

- In Germany four relevant samples from recycling companies were analysed. The results ranged between 0.165 mg/kg and 42 mg/kg (median 9.7 mg/kg; average 15.4 mg/kg) and confirm that waste fractions of EPS packaging usually contain HBCDD, however, levels of contamination are comparatively low. The results suggest that the average HBCDD contamination of packaging waste may be about 10 mg/kg [BiPRO 2015].
- Though the use of HBCDD in packaging has stopped in 2014 in the EU, HBCDD was found in 90% of 70 Irish and UK polystyrene packaging samples collected from 2015 to 2016. The concentration found ranged between <0.0007 mg/kg and 5,900 mg/kg. 4 of the samples exceeded the LPCL level of 1,000 mg/kg. The median HBCDD content in all EPS samples (from UK and Ireland) was at 11 mg/kg (average 190 mg/kg) [Abdallah et al. 2018].
- In the Netherlands HBCDD was found in the EPS packaging fraction collected in household waste recycling centres [Giraf Results 2018]. From the samples taken only a small part contained HBCDD with a range from 183 to 18,100 mg/kg as measured by LC-MS. It was however, found that the collected packaging from the household waste recycling centers did not only include packaging waste but were contaminated with construction EPS and XPS, explaining the high concentration. Average values for not contaminated packaging were calculated based on the assumption that the brominated compounds in the remaining packaging EPS would all be HBCDD and derived a value of 265 mg/kg. Due to the contamination of the reported concentrations for the EPS packaging fraction, reported concentrations within [Giraf Results 2018] are not considered in calculating the concentration found in EPS in packaging.

The reported data indicates that an median HBCDD content in EPS packaging waste around 10 mg/kg could be a realistic estimate at EU level. It however, needs to be considered, taking into account the findings from [Giraf Results 2018] that concentrations might fluctuate between MS depending on the waste collection operations in place and may for example also be in a range
between 200 and 300 mg/kg. It is assumed that the comparatively high HBCDD concentration in the packaging waste flows are particularly a result of cross-contamination stemming from EPS and C&D waste.

### 4.7.8.4 Activity data

Figure 26 shows past consumption of HBCDD in the main applications within Europe. EPS/XPS use for packaging is related to 5.6% of the historic HBCDD consumption in the EU (see Table 44).

### 4.7.8.5 Waste management

Usually the collection of EPS packaging waste is covered by the household and/or similar commercial waste collection. Although HBCDD-containing packaging could be sorted out before the actual waste treatment (e.g. with the XRF screening), this screening process is rarely undertaken. The collection of packaging waste in the Netherlands from households is done via household waste recycling centres where packaging EPS waste is collected as brought in by citizens. In 2011, a new promotional collection system, using large plastics bags, was introduced for a few locations and allowed the recycling centre employees to check the content. In 2015, 75 out of 400 Dutch municipalities were using the new collection system [Giraf Results 2016].

Theoretically HBCDD-free EPS packaging waste can be easily mechanically recycled as EPS or PS, if collected separately. Compared to other types of plastics the volumes of EPS in the packaging sector are relatively low and separate collection and further recycling is often not cost effective or difficult to realise. Nevertheless, some sea-countries, with high amounts of EPS fishboxes have already implemented separate collection systems for fishboxes. These boxes can then be compressed and mechanically recycled. Recycled fishboxes are then used to manufacture e.g. insulation panels for the construction sector [Conversio 2018].

In total 33% of EPS packaging waste has been mechanically recycled (15% as EPS, 18% as PS) in 2017 in the EU [Conversio 2018]. Whenever EPS packaging cannot be recycled, for example because it is too contaminated, the material must be treated according to the minimum standard for "other recovery" (e.g. incineration as a form of recovery) [Giraf Results 2018]. 35% of EPS packaging have been energetically recovered and 32% have been landfilled in 2017 in the EU [Conversio 2018].

### 4.7.8.6 Mass flow

Figure 34 illustrates the resulting quantitative mass flows for HBCDD in EPS packaging (substance flow) from collected EPS Packaging waste (material flow). The presented mass flow however does not reflect some uncertainties associated with the concentration of HBCDD in packaging as presented in [Giraf Results 2018] with higher concentrations measures. Data from [Giraf Results 2018] was not considered due to the contamination of EPS packing with construction waste.

At EU level, 32.55% of the waste is recycled, 34.79% is incinerated (assumption: usually with energy recovery) and 32.66% is landfilled. About 388.3 kt of EPS packaging waste containing an estimated quantity of 3.9 tonnes of HBCDD was collected in 2017 and managed accordingly.

For the estimation of the HBCDD quantity, an HBCDD concentration of 10 mg/kg was applied (see section 4.7.8.3). As mentioned above, considering cross contamination from EPS from construction, the average concentration may be higher with a correspondingly higher HBCDD freight.
According to existing legislation it can be assumed that HBCDD is no longer used in the EU packaging sector. The load of the waste fraction may derive from imports, waste packaging, cross contamination from construction products or use of HBCDD loaded recyclates.

**4.7.9 HIGH IMPACT POLYSTYRENE (HIPS)**

**4.7.9.1 Background information**

At European level, about 2.0 % of the HBCDD consumption was used for the manufacturing of HIPS. (see Table 44). Relevant information regarding the European situation is presented in BiPRO 2011. Further details are available in [BiPRO 2015].

Typical applications mentioned in the literature are in electrical appliances such as for audio and visual equipment, distribution boxes for electrical lines in the construction sector and refrigerator lining.

The use of HBCDD in HIPS is banned in the EU and it is assumed that the last use in the EU was in 2014 (see Figure 26). It may also be that the use at EU level was already phased out earlier. Already in 2012, the European Chemical Industry Council (CEFIC) stated that amounts of HBCDD used in HIPS are not relevant. Since there was no specific information on the specific end of use, it was estimated in [BiPRO 2015] that the last use of HBCDD in HIPS in the EU was in 2011 [BiPRO 2015].

Electrical equipment containing HBCDD-containing HIPS has a typical lifetime of 9 years (+/- 5 years) (see Table 45). HBCDD in waste streams stemming of HIPS from the electronic sector is still relevant but will decrease from around the beginning of the 2020s or earlier [BiPRO 2011; BiPRO 2015].

**4.7.9.2 Processes, inputs and outputs**

See [BiPRO 2011] for details.
4.7.9.3 Concentrations in products, waste and recyclates

The HBCDD content of flame retarded HIPS products ranges between 1 and 7 % and is typically around 4 % (see Table 43). Of the WEEE fraction that contains BFRs, HBCDD represents approximately 1 % in comparison to e.g. decaBDE with around 22 % [EERA UTC Opinion 2018].

HBCDD has been detected in articles made from recycled plastics in concentrations up to 207 mg/kg (for the analysis results see section 10.10 of the Annex).

4.7.9.4 Waste management

Typically, HBCDD-containing HIPS are treated in WEEE. The waste management of WEEE (related to decaBDE in WEEE) is described in section 4.1.6.5 and applies analogously to HBCDD in WEEE.

4.7.9.5 Mass flow

Considering that HBCDD is no longer used in HIPS, the decreasing relevance of the waste flow in the last years (see Table 46) and as HIPS are treated under WEEE no mass flow is developed.

For decaBDE we assume a total freight of 630 t/y in the total quantity of WEEE plastics. Considering that HBCDD represents approximately 1 % of the BFRs in WEEW in comparison to decaBDE which represents around 22 %, it can be assumed that the HBCDD quantity in the total of 1.2 million tonnes of WEEE plastics amount to around 30 t/y. This indicates that HBCDD in WEEE is still relevant at low levels. However, considering historic consumption of HBCDD for HIPS in EEE (see Figure 26) and typical product lifetimes of EEE (around 9 years), HBCDD in waste streams stemming of HIPS from the electronic sector will decrease from around the beginning of the 2020s.

Measures for environmentally sound management applied to other BFR in WEEE such as decaBDE will also apply to HBCDD in HIPS in WEEE.
4.7.10 POLYMER DISPERSIONS FOR TEXTILES

4.7.10.1 Background information

Polymer dispersions were processed in the textile industry and used especially for upholstered furniture, upholstered seats in transportation, curtains and drapes, mattress ticking, home textiles and automotive textiles [IOM 2008].

The use of HBCDD for textiles is banned in the EU and it is assumed that the last use in the EU was in 2014 (see Figure 26) with regional differences. For example, based on industry stakeholder information it can be assumed that the last use in Germany was in 2006 [BiPRO 2015].

4.7.10.2 Processes, inputs and outputs

HBCDD-containing polymer dispersions for textiles were used to comply with FR standards. The dispersion was processed in the textile finishing industry as back coating. For the detailed process, inputs and outputs of this historic use please refer to [BiPRO 2011].

4.7.10.3 Concentrations in products, waste and recyclates

The HBCDD content in the polymer dispersion was on average about 25 %, which leads to a content of about 7 to 9 % in the coated textiles (typical HBCDD content 8%).

4.7.10.4 Activity data

HBCDD flame retarded textiles were historically used for upholstery fabrics, curtains, wall coverings, etc. especially from the institutional sector (theatres, concert halls, cinemas, town halls, other public places, etc.) and in the automotive sector for seat covers, door panels and carpets.

At European level, about 5.9 % of the HBCDD consumption was used for the manufacturing of polymer dispersions (PD) as FR for textile coating (see Table 44). Also, at EU level, even before 2007, the use of HBCDD has significantly declined in previous years (see Figure 26). Relevant information regarding the European situation is presented in BiPRO 2011. Further details are available in [BiPRO 2015].

The occurrence of FRs within textiles differs between European countries as fire safety requirements for furniture vary across Europe. Most stringent standards can be found in the UK and Ireland.

HBCDD-containing textiles have a typical lifetime of 10 years (+/- 3 years) (see Table 45). HBCDD in waste streams stemming of textiles is still relevant at low levels but is already significantly decreasing [BiPRO 2011; BiPRO 2015].

4.7.10.5 Waste management

Specific quantitative data on the utilisation and disposal of HBCDD flame retarded textiles is not available. The available data on textile wear suggests that the majority of textiles are bulked together including household textiles, workwear and clothing. Within the EU around 25% of discarded textiles are recycled. The remaining 75% are incinerated or landfilled [FOEE 2013].

Fire-retarded textiles from the automotive sector are usually removed and recycled during ELV treatment. These fabrics usually end up in the shredder light fraction. ELV management and of the shredder light fraction is described in section 4.1.7.5.
4.7.10.6 Substance and mass flow

HBCDD is no longer used in the production of textiles since 2007. Already in the years before 2007 the use levels have declined (see Figure 26). Nevertheless, HBCDD from previous uses may continue to appear at low levels in textile waste streams and recycled materials due to the assumed lifetime of textiles (10 +/- 3 years) and from imports.

Due to the longer lifetime of cars (15 to 20 years), HBCDD will continue to occur in ELV but will level out during the 2020s.

Since the relevance of HBCDD in current waste streams is considered comparatively low, specific waste streams are not established.
5. RISK EVALUATION

5.1.1 Scenarios for SCCP

The average SCCP concentration in rubber conveyor belts for the use in underground mining amounts to 3.3 %, and to ca. 100,000 mg/kg in the corresponding rubber fraction of the conveyor belts. In the absence of detailed information concerning the processing and disposal of used rubber conveyor belts it is assumed that the rubber fraction of the conveyor belts is treated/disposed of together with other rubber wastes. Under this assumption, the theoretical SCCP concentration of the total waste stream decreases to about 420 mg/kg.

The average SCCP concentration in sealants and adhesives for the construction sector averages to 200,000 mg/kg. It is assumed that due to their properties, a major proportion of the applied sealants and adhesives adheres to the surface of other construction materials (e.g. on concrete, tiles, bricks and ceramics) and is therefore also treated together with this waste. In the absence of more precise information, it is assumed that about 2/3 (~1,373 t) of the SCCP-containing waste stream is treated along with the other construction and demolition waste and that ca. 1/3 (~687 t) of the relevant waste stream is disposed of separately in hazardous waste incineration plants. The concentration of SCCP in the mixed construction and demolition waste decreases to ca. 1 mg/kg.

At a low POP concentration limit <420 mg/kg, theoretically the entire rubber waste in the EU would be affected (~2,630 kt) whereas at a low POP concentration limit <1 mg/kg theoretically the entire construction and demolition waste (~276,910 kt) as well as the entire rubber waste in the EU (~2,630 kt) would be concerned (see Figure 37).

![Figure 35: Scenarios for low POP concentration limits (SCCP)](image)

A possible POP concentration limit for SCCPs <1,000 mg/kg would have no impact on the current waste management of the above described waste streams. At a low POP concentration limit of <420 mg/kg, theoretically about 1,430 kt SCCP-containing rubber waste (currently recycled) would have to be treated alternatively and at a low POP concentration limit <1 mg/kg a considerably larger waste amount of 276,762 kt would be concerned (see the following table).
Table 47: Possible LPCLs and their effects on the present treatment operation (SCCP)

<table>
<thead>
<tr>
<th>Possible low POP concentration limits</th>
<th>0.1 mg/kg</th>
<th>420 mg/kg</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Concerned waste streams for different low POP concentration limits</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rubber waste [kt/a]</td>
<td>2,630</td>
<td>2,630</td>
</tr>
<tr>
<td>Mineral waste from C&amp;D [kt/a]</td>
<td>276,910</td>
<td></td>
</tr>
<tr>
<td><strong>Total [kt/a]</strong></td>
<td>279,540</td>
<td>2,630</td>
</tr>
<tr>
<td><strong>Concerned waste streams which theoretically would have to be treated alternatively for different low POP concentration limits</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rubber waste (incl. SCCP-containing rubber waste) [kt/a]</td>
<td>1,430</td>
<td>1,430</td>
</tr>
<tr>
<td>Mineral waste from C&amp;D (incl. SCCP-containing sealants and adhesives) [kt/a]</td>
<td>275,332</td>
<td></td>
</tr>
<tr>
<td><strong>Total [kt/a]</strong></td>
<td>276,762</td>
<td>1,430</td>
</tr>
</tbody>
</table>

At this point it needs to be mentioned, that a possible POP concentration limit for SCCPs <1,000 mg/kg would still impact several waste streams in the EU (except the ones discussed above). In particular, imports of new products and/or recycled products contaminated with or containing SCCPs (also in concentrations above the set limit values in Regulation EU 2015/2030) are to be expected and are regularly detected during enforcement activities. It is expected that most of those imported articles will be landfilled or incinerated together with municipal solid waste streams; however, a possible recycling of respective articles cannot be generally excluded. However, up-to-date information on recycling activities in EU MS and quantitative information on SCCP concentrations and amounts in waste and recyclates is not available. No information on SCCP concentrations in waste and recyclates could be obtained during the stakeholder consultation for this project (April 2018). In addition, considering the large diversity of the relevant imported new articles potentially containing SCCPs and the fact that imports cannot be traced it is not possible to further investigate these waste streams.

5.2 Overview of relevant sectors

Based on the substance and material flows as well as additional information concerning the occurrence in products and waste, Table 48 demonstrates in which sectors the substances/substance groups typically occur and may cause relevant risks.

Table 48: Overview of POPs, relevant sectors and corresponding amounts of the POPs in waste; highlighted sectors (in bold) are considered in the risk evaluation (see Table 49)

<table>
<thead>
<tr>
<th>Potential sector of origin</th>
<th>Substance</th>
<th>Estimated relevance in the project context</th>
<th>Waste (t/y)</th>
</tr>
</thead>
<tbody>
<tr>
<td>WEEE plastics</td>
<td>DecaBDE</td>
<td>High relevance in products and in the future in waste and recyclate due to long lifetimes</td>
<td>630</td>
</tr>
<tr>
<td>Plastic &amp; textiles in ELVs</td>
<td>DecaBDE</td>
<td>High relevance in products and in the future in waste and recyclate due to long lifetimes</td>
<td>97.9</td>
</tr>
<tr>
<td>Construction material</td>
<td>DecaBDE</td>
<td>Potential relevance in waste streams due to long lifetime. Data gaps however exist</td>
<td>n.a.</td>
</tr>
<tr>
<td>Category</td>
<td>Substance</td>
<td>Description</td>
<td>Value</td>
</tr>
<tr>
<td>----------------------------------</td>
<td>-----------</td>
<td>-------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
<td>-------</td>
</tr>
<tr>
<td>Textiles</td>
<td>DecaBDE</td>
<td>Potential relevance in waste streams, however partially already contained in ELV treatment paths. Moreover, decaBDE in textiles occur in low quantities and contaminated waste streams are already decreasing. Furthermore, most of textiles are incinerated or landfilled within the EU.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Sewage sludge</td>
<td>DecaBDE</td>
<td>Not relevant due to minor share of decaBDE and very low concentration found</td>
<td>0.92</td>
</tr>
<tr>
<td><strong>Rubber from conveyor belts</strong></td>
<td>SCCPs</td>
<td>Relevant in waste and recyclate</td>
<td>1,100</td>
</tr>
<tr>
<td><strong>Sealants and adhesives</strong></td>
<td>SCCPs</td>
<td>Relevant in waste and recyclate</td>
<td>274.68</td>
</tr>
<tr>
<td>Unintentional production – Incineration processes waste</td>
<td>HCBD</td>
<td>Not relevant, analytical results indicate HCBD does not occur in relevant amounts in incineration residues</td>
<td>0.004</td>
</tr>
<tr>
<td>Sewage sludge</td>
<td>HCBD</td>
<td>Not relevant, analytical results indicate that HCBD does only occur in negligible amounts</td>
<td>0.031</td>
</tr>
<tr>
<td>Stockpiles</td>
<td>Dicofol</td>
<td>Not relevant</td>
<td>n.a.</td>
</tr>
<tr>
<td>C&amp;D waste</td>
<td>Dicofol</td>
<td>Not relevant</td>
<td>n.a.</td>
</tr>
<tr>
<td><strong>Fire fighting foam</strong></td>
<td>PFOA, its salts and related compounds</td>
<td>The size of in-use stockpiles may be significant (but no quantitative information available). AFFFs are typically incinerated in hazardous waste incinerators after their use stage. Recycling and/or recovery is not a common waste management operation applied in the EU and it is therefore not expected that PFOA accumulates in recycled products</td>
<td>n.a.</td>
</tr>
<tr>
<td>Textiles</td>
<td>PFOA, its salts and related compounds</td>
<td>Still relevant in certain textiles and will remain relevant in specific waste streams within the EU. Most textiles will be incinerated or landfilled (together with municipal solid waste). Currently, textile recycling is not a common waste management operation in the EU and it is therefore not expected that PFOA accumulates in recycled products.</td>
<td>n.a.</td>
</tr>
<tr>
<td><strong>Semiconductors</strong></td>
<td>PFOA, its salts and related compounds</td>
<td>Information regarding wastes and recyclates is missing. However, the use of semiconductors in various products may result in various waste streams (e.g. in WEEE, ELV, municipal solid waste) which will need to be managed (also in the EU). As some of the waste streams may be recycled, it cannot be generally excluded that PFOA accumulate in recyclates / new products.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Industry</td>
<td>Pesticides</td>
<td>Environmental Impact</td>
<td>Quantitative Data</td>
</tr>
<tr>
<td>-----------------------------</td>
<td>-------------------------------------</td>
<td>--------------------------------------------------------------------------------------</td>
<td>-------------------</td>
</tr>
<tr>
<td>Medical devices</td>
<td>PFOA, its salts and related compounds</td>
<td>Not considered relevant due to the low amounts expected to be used.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Printing inks</td>
<td>PFOA, its salts and related compounds</td>
<td>The use only continues in printers that are no longer manufactured, and therefore a phase-out is underway. A clear decreasing trend in the amounts used has been reported. Current consumption volumes are not known, but it is expected that the consumption is decreasing. No information is available on wastes and recyclates.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Photo-imaging</td>
<td>PFOA, its salts and related compounds</td>
<td>Due to the comparably low and decreasing use amounts in very specific applications and the fact that these specific applications will become obsolete, this use is not considered relevant in the EU. No significant waste amounts are expected to arise in the EU.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Automotive applications</td>
<td>PFOA, its salts and related compounds</td>
<td>The amounts of PFOA in use as well as the respective waste amounts might be relevant. However, in the absence of further information (e.g. relevant application areas) and quantitative data on the amounts used, it is not possible to further investigate this use.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Production of pharmaceutical products</td>
<td>PFOA, its salts and related compounds</td>
<td>Not relevant as no significant waste amounts are expected.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Fire fighting foam</td>
<td>PFHxS, its salts and related compounds</td>
<td>It can be assumed that there may still be relevant stockpiles of AFFFs containing PFHxS also in the EU (but no quantitative information available). AFFFs are typically incinerated in hazardous waste incinerators after their use stage. Recycling and/or recovery is not a common waste management operation applied in the EU and it is therefore not expected that PFOA accumulates in recycled products.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Metal plating</td>
<td>PFHxS, its salts and related compounds</td>
<td>Not relevant, as waste streams are assumed to be collected by specialised companies and are subject to controlled waste management. It is not clear to what extent such waste streams are recycled and/or recovered in the EU.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Carpets and Textiles</td>
<td>PFHxS, its salts and related compounds</td>
<td>PFHxS use was reported in stain protection products for carpets and upholstery. However, no relevant information on a deliberate use of PFHxS in products for carpets and upholstery could be identified in the EU. Besides, information related to wastes and recyclates is not available. Currently, textile recycling is not a common waste management operation in the EU and it is therefore not expected that PFHxS accumulates in recycled products.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Application</td>
<td>Substance</td>
<td>Relevance in Products and Waste</td>
<td>Remarks</td>
</tr>
<tr>
<td>-------------</td>
<td>-----------</td>
<td>--------------------------------</td>
<td>---------</td>
</tr>
<tr>
<td>Leather</td>
<td>PFHxS, its salts and related compounds</td>
<td>The same applies as for carpets and textiles</td>
<td>n.a.</td>
</tr>
<tr>
<td>Lubricants</td>
<td>PFHxS, its salts and related compounds</td>
<td>The use of PFHxS in lubricants has been indicated during the recent stakeholder consultation for this project. However, no additional information on the use in lubricants has been identified. Considering that the major share of spent oils is recovered/recycled in the EU, it cannot be excluded that PFHxS-containing waste oils enter recycling streams and consequently end up in recycled oils. However, no qualitative or quantitative information on waste and recyclates containing PFHxS is available.</td>
<td>n.a.</td>
</tr>
<tr>
<td>Semiconductors</td>
<td>PFHxS, its salts and related compounds</td>
<td>Not clear if PFHxS has been used in the EU in the production of semiconductors. However, considering that semiconductors are traded on a global level, it cannot be excluded that PFHxS-containing semiconductors or finished products containing those semiconductors enter the EU. Information on imported volumes is not available. Information regarding wastes and recyclates is also missing. However, the use of semiconductors in various products may result in various waste streams (e.g. in WEEE, ELV, municipal solid waste) which will need to be managed. As some of the waste streams may be recycled it cannot be excluded that PFHxS accumulate in recyclates / new products.</td>
<td>n.a.</td>
</tr>
</tbody>
</table>

### Relevance Charts

<table>
<thead>
<tr>
<th>Application</th>
<th>Substance</th>
<th>Relevance</th>
</tr>
</thead>
<tbody>
<tr>
<td>EPS for construction</td>
<td>HBCDD</td>
<td>High relevance due to long lifetime and increasing trend in use</td>
</tr>
<tr>
<td>XPS for construction</td>
<td>HBCDD</td>
<td>High relevance due to long lifetime and increasing trend in use</td>
</tr>
<tr>
<td>EPS in the packaging sector</td>
<td>HBCDD</td>
<td>At present no longer relevant in products; also, in waste no longer relevant due to limited lifetime of products (used mostly in packaging) with continuously decreasing relevance</td>
</tr>
<tr>
<td>High Impact Polystyrene (HIPS) for electrical and electronic devices</td>
<td>HBCDD</td>
<td>At present no longer relevant in products with continuously decreasing relevance. Also, only minor relevance in waste as waste streams are already managed under WEEE treatment; all measures applied to BFR in WEEE will apply to HBCDD in HIPS</td>
</tr>
<tr>
<td>Polymer dispersions for textiles</td>
<td>HBCDD</td>
<td>No longer relevant in products, also in waste only minor relevance is expected as relevant shares are already managed under ELV treatment</td>
</tr>
</tbody>
</table>
5.3 Risk evaluation to derive recommendations for environmentally sound management (ESM)

In the risk assessment, especially the following questions are considered:

- Are disposal and recovery operations (D9, D10, R1, R4), permitted according to Annex V, part 1, appropriate to destroy or irreversibly transform the relevant substances/substance groups?
- Can the corresponding processes lead to an unintended generation of new POPs?
- Can disposal or recovery cause a risk to human health or the environment (i.e. a relevant exposure of humans or the environment)? The emission of significant amounts of POPs into the environment is generally considered a risk

Further information on the methodology and environmental and health impacts is provided in section 6.2.2.2
<table>
<thead>
<tr>
<th>POPs</th>
<th>Sector</th>
<th>Amount of substance (in t/a or qualitatively)</th>
<th>Treatment operation</th>
<th>Possible risks regarding the POP content and strategy for ESM of wastes</th>
</tr>
</thead>
<tbody>
<tr>
<td>DecaBDE</td>
<td>WEEE plastics</td>
<td>88.76</td>
<td>Recycling</td>
<td>Recycling with prior treatment of WEEE plastics includes the manual disassembly or mechanical treatment in shredders. Plastic containing brominated flame retardants exceeding 2,000 mg/kg are separated and treated by combustion. However, plastic containing brominated flame retardants below this threshold is recycled. The decaBDE content is not destroyed and therefore it occurs in recyclates. This practice entails a further transfer of the HBCDD content into various plastic products, thus resulting in releases during further product cycles (see [BiPRO 2017]) and end of life treatment (see [BiPRO 2017]) and consequently in uncontrolled global distribution and possible releases of significant quantities of decaBDE. The emission of significant amounts of POPs into the environment is generally considered a risk. The risk can be minimised through a higher yield of separation of WEEE plastics containing decaBDE as far as possible or through stopping recycling of WEEE plastics. Mechanical treatment in shredders can result in increased releases of PBDEs (including decaBDE; see BiPRO 2017). A specific health or environmental risk due to exceeding critical environmental concentrations or exposure limits is not expected.</td>
</tr>
<tr>
<td>519 + 15.35</td>
<td>Incineration/Energy recovery</td>
<td>Controlled incineration in advanced solid waste incineration, hazardous waste incineration, cement kiln co-incineration or in thermal and metallurgical production of metals of WEEE plastics containing decaBDE at high temperatures, along with compliance with BAT/BEP guidelines is an effective way of destruction and of avoiding generation of other POPs [UNEP/CHW.13/6/Add.1/Rev.1 2017]. A specific health or environmental risk due to exceeding critical environmental concentrations or exposure limits due to appropriate incineration is not expected.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>6.18</td>
<td>Landfill</td>
<td>A share of 5.6% of plastics from WEEE (low bromine fraction) is landfilled. DecaBDE is not destroyed. The generation of other POPs does not occur. There is the potential risk of ‘leaking’ to the surrounding environment. Health or environmental risks cannot be excluded and are represented in the long run especially due to the risk of uncontrolled global distribution and the corresponding risks for human health and the environment. The landfilled share of plastic pertains to the low brominated fraction of WEEE plastics. The risk is therefore considered low. The risk can be managed through appropriate landfills, monitoring, and control.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POPs</td>
<td>Sector</td>
<td>Amount of substance (in t/a or qualitatively)</td>
<td>Treatment operation</td>
<td>Possible risks regarding the POP content and strategy for ESM of wastes</td>
</tr>
<tr>
<td>------</td>
<td>--------</td>
<td>---------------------------------------------</td>
<td>---------------------</td>
<td>--------------------------------------------------------------------------------</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Further minimised by avoiding the landfilling of plastics</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>A specific health or environmental risk due to exceeding critical environmental concentrations or exposure limits is not expected.</td>
</tr>
<tr>
<td>Plastic &amp; textiles in ELVs</td>
<td>20.5+28.7</td>
<td>Recycling</td>
<td>Treatment of plastics and textiles in ELVs includes the manual disassembly and mechanical treatment in shredders. The decaBDE content is not destroyed and therefore it occurs in recyclates. This practice entails a further transfer of the decaBDE content into various plastic products, thus resulting in releases during further product cycles and end of life treatment and consequently in uncontrolled global distribution and possible releases of significant quantities of decaBDE. The emission of significant amounts of POPs into the environment is generally considered a risk. The risk can be minimised through a higher yield of separation of ELV plastics containing decaBDE. This can be achieved by increasing the share of advanced PST of the shredder light fractions. Advanced post shredder technologies enable to accumulate decaBDE rich fractions in the shredder residues with a higher density. High density fractions could be incinerated to destroy the decaBDE content. Mechanical treatment in shredders can result in increased releases of PBDEs. A specific health or environmental risk due to exceeding critical environmental concentrations or exposure limits is not expected.</td>
<td></td>
</tr>
<tr>
<td>23.2</td>
<td>Incineration</td>
<td>Controlled incineration in advanced solid waste incineration, hazardous waste incineration, cement kiln co-incineration or in thermal and metallurgical production of metals of the ELV SLF containing decaBDE at high temperatures, along with compliance with BAT/BEP guidelines is an effective way of destruction and of avoiding generation of other POPs [UNEP/CHW.13/6/Add.1/Rev.1 2017]. A specific health or environmental risk due to exceeding of critical environmental concentrations or exposure limits due to appropriate incineration is not expected.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>25.8</td>
<td>Landfill</td>
<td>A share of 37% of the ELV SLF is landfilled. DecaBDE is not destroyed. The generation of other POPs does not occur. There is the potential risk of 'leaking' to the surrounding environment. Health or environmental risks cannot be excluded and are represented in the long run especially due to the risk of uncontrolled global distribution and the corresponding risks for human health and the environment. The risk is considered low and can be further minimised by avoiding the landfilling of plastics SLF fractions. A specific health or environmental risk due to exceeding critical environmental concentrations or exposure limits is not expected.</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
### POPs

<table>
<thead>
<tr>
<th>Sector</th>
<th>Amount of substance (in t/a or qualitatively)</th>
<th>Treatment operation</th>
<th>Possible risks regarding the POP content and strategy for ESM of wastes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Construction material</td>
<td>n.a.</td>
<td>Recycling, Incineration, Landfilling</td>
<td>In general, the same considerations as for other decaBDE-containing plastic wastes apply (e.g. from WEEE or ELVs apply (see above). Relevant quantities of decaBDE-containing plastics from the construction sector are expected in the future (e.g. an estimated quantity of 1,500 t/a from 2040 to 2060). If they will not be identified and the decaBDE content will not be destroyed there will be the risk of uncontrolled global distribution and the corresponding risks for human health and the environment from recycling and landfilling. The risk can be minimised by identification and separation of decaBDE-containing plastics from other C&amp;D plastics as far as reasonably possible and by destroying the decaBDE content. Appropriate destruction technologies are available. A specific health or environmental risk due to exceeding critical environmental concentrations or exposure limits is not expected.</td>
</tr>
</tbody>
</table>

### SCCPs

<table>
<thead>
<tr>
<th>Sector</th>
<th>Amount of substance</th>
<th>Treatment operation</th>
<th>Possible risks regarding the POP content and strategy for ESM of wastes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rubber from conveyor belts</td>
<td>589.71</td>
<td>Recycling</td>
<td>In the absence of information on the disposal and processing of used rubber conveyor belts from underground mining, it is assumed that the relevant rubber waste is managed together with other rubber waste in the EU. A large share of the rubber waste in the EU is directed to material recovery. In this context, it cannot generally be excluded that SCCP enter recyclate streams and accumulate in recyclates and new products. During the recycling processes, POPs are usually neither destroyed nor generated. Therefore, there is a specific risk that SCCP might be released into recyclates (e.g. floorings for halls, playgrounds etc.), resulting in uncontrolled distribution and corresponding risks for human health and the environment. The risk can be minimised through separation of SCCP-containing conveyor belts from underground mining as far as possible and through the choice of appropriate treatment operations (e.g. incineration). During handling of SCCP-containing conveyor belt and especially during pre-treatment operation before recycling, dust generation should be avoided as far as possible. Inhalation of contaminated dust can be avoided by the application of breathing masks.</td>
</tr>
</tbody>
</table>

SCCP are thermally decomposed at about 200 °C [BiPRO 2011]. Due to high incineration temperatures (>800°C), it can be expected that the SCCP content in conveyor belts is destroyed during (hazardous) waste incineration. A potential health or environmental risk may occur in case SCCP-containing rubber need to be broken down mechanically prior to incineration (dust generation). To minimise environmental and health risks, dust generation should be avoided as far as possible. Inhalation of
<table>
<thead>
<tr>
<th>POPs</th>
<th>Sector</th>
<th>Amount of substance (in t/a or qualitatively)</th>
<th>Treatment operation</th>
<th>Possible risks regarding the POP content and strategy for ESM of wastes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>contaminated dust can be avoided by the application of breathing masks.</td>
</tr>
<tr>
<td>8.36</td>
<td>Landfill</td>
<td></td>
<td></td>
<td>In general, landfilling involves the risk that POPs may be released into the environment. Due to the comparably low SCCP amounts from this application, no relevant environmental or health risks are expected. Separation of waste and separate treatment (e.g. incineration) to the largest extent possible can minimise the effect of transfer and distribution of POPs into the environment.</td>
</tr>
<tr>
<td>Sealants and adhesives</td>
<td>240.95</td>
<td>Recycling</td>
<td></td>
<td>In the absence of detailed information regarding the waste management of sealants and adhesives waste in the EU, and considering their material properties, it is assumed that a certain share of sealants and adhesives used in the construction sector adheres to the surface of other building materials (e.g. on concrete, tiles, bricks). Therefore, in practice, a certain amount of sealants and adhesive waste will be managed together with other construction and demolition wastes (including recycling operations). Consequently, it cannot be excluded that SCCPs enter recycle streams or are released to the environment. While the theoretically concerned waste amount would be significant, the SCCP concentration within this mixed waste stream is expected to be low. To minimize this risk, SCCP-containing sealants and adhesives should be separated from other construction and demolition wastes as far as possible and separately treated (e.g. hazardous waste incineration).</td>
</tr>
<tr>
<td>1.57+137.32</td>
<td>Incineration</td>
<td></td>
<td></td>
<td>In case the sealants and adhesives can be successfully separated, it is assumed that this waste stream is subsequently incinerated (including incineration as hazardous waste (e.g. in case of suspected PCB-containing joint sealants)). SCCP are thermally decomposed at about 200 °C [BiPRO 2011]. Due to high incineration temperatures (&gt;800°C), it can be expected that the SCCP content in sealants and adhesives is destroyed during waste incineration.</td>
</tr>
<tr>
<td>32.17</td>
<td>Landfill</td>
<td></td>
<td></td>
<td>In general, landfilling involves the risk that POPs may be released into the environment. Separation of waste and separate treatment (e.g. incineration) to the largest extent possible can minimise the effect of transfer and distribution of POPs into the environment.</td>
</tr>
<tr>
<td>PFOA, its salts and related compounds</td>
<td>Fire fighting foam</td>
<td>n.a.</td>
<td>Recycling</td>
<td>Recycling and/or recovery of AFFFs is not a common waste management operation applied in the EU and it is therefore not expected that PFOA accumulates in recycled products.</td>
</tr>
<tr>
<td>n.a.</td>
<td>Incineration</td>
<td></td>
<td></td>
<td>It is expected that AFFFs waste is typically managed in hazardous waste incinerators. Controlled incineration with high temperatures (e.g., at 1,000°C) is effective to destroy PFOA [UNEP/POPS/POPRC.14/3 2018]. It is however, currently unclear to what extent formation of PFOA may occur in municipal waste incinerators where (1) flue gases may reach temperatures of 850°C or greater and may result in different degradation products; (2) other substances coexist and may interfere with the</td>
</tr>
<tr>
<td>POPs</td>
<td>Sector</td>
<td>Amount of substance (in t/a or qualitatively)</td>
<td>Treatment operation</td>
<td>Possible risks regarding the POP content and strategy for ESM of wastes</td>
</tr>
<tr>
<td>------</td>
<td>--------</td>
<td>---------------------------------------------</td>
<td>-------------------</td>
<td>--------------------------------------------------------------------------------</td>
</tr>
<tr>
<td></td>
<td></td>
<td>thermolysis of fluoropolymers (e.g., thermolysis of PTFE is inhibited by a hydrogen or chlorine atmosphere in contrast to steam, oxygen or sulphur dioxide, which accelerate decomposition; and (3) technologies such as activated carbon injection (ACI) coupled with baghouse filtration (BF) may be installed to remove dioxin or mercury and may also trap PFCAs (for further details see [UNEP/POPS/POPRC.14/3 2018]). In the absence of conclusive information regarding formation and destruction efficiencies of PFOA in municipal solid waste incinerators, it is recommended to rather manage respective waste streams in hazardous waste incinerators.</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>n.a. Landfill</td>
<td>Textiles</td>
<td>It is assumed that landfilling is not a commonly applied waste management option for end of life firefighting foams in the EU.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>n.a. Recycling</td>
<td>Textiles</td>
<td>Textile recycling is currently not a common waste management operation applied in the EU (might change in the future) and it is therefore not expected that PFOA accumulates in recycled products.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>n.a. Incineration</td>
<td>Textiles</td>
<td>Even though textiles might be collected, reused (exported) and/or recycled, it is expected that most textiles in the EU are incinerated and landfilled (usually together with municipal solid waste). Due to high incineration temperatures it can be assumed that PFOA, its salts and related compounds are largely destroyed (see additional information provided above on uncertainties regarding formation and destruction efficiencies of PFOA in municipal waste incinerators).</td>
</tr>
<tr>
<td></td>
<td></td>
<td>n.a. Landfill</td>
<td>Textiles</td>
<td>In general, landfilling involves the risk that POPs could be released from the landfill site into the environment. To minimise the risk, textiles containing PFOA should be separated from other textiles and treated separately, as far as possible. The practical challenge to separate &quot;POP-rich&quot; from &quot;POP-free&quot; textiles still remains.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>n.a. Recycling</td>
<td>Semiconductors</td>
<td>Information regarding wastes and recyclates is missing. However, the use of semiconductors in various products will result in various waste streams (e.g. in WEEE, ELV, municipal solid waste) which will need to be managed. Besides, as some of the waste streams may be recycled it cannot be generally excluded that PFOA its salts and PFOSA-related compounds accumulate in recyclates and subsequently in new products. Manual disassembly or mechanical treatment in shredders can result in dust generation and therefore presents a potential environmental and health risk. In order to minimise the risk, dust generation should be avoided as far as possible. Inhalation of contaminated dust can be avoided by the application of breathing masks.</td>
</tr>
</tbody>
</table>
### POPs

<table>
<thead>
<tr>
<th>Sector</th>
<th>Amount of substance (in t/a or qualitatively)</th>
<th>Treatment operation</th>
<th>Possible risks regarding the POP content and strategy for ESM of wastes</th>
</tr>
</thead>
<tbody>
<tr>
<td>POPs</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Automotive applications</td>
<td>n.a.</td>
<td>Incineration</td>
<td>See above</td>
</tr>
<tr>
<td></td>
<td>n.a.</td>
<td>Landfill</td>
<td>See above</td>
</tr>
<tr>
<td>Fire fighting foam</td>
<td>n.a.</td>
<td>Recycling</td>
<td>The amounts of PFOA and PFOA-related compounds in use as well as the respective waste amounts in the EU might be relevant. However, in the absence of information on the relevant application areas / automotive parts and quantitative data on the amounts used in the sector, it was not possible to investigate this use and related waste streams in detail. In general, automotive service and replacement parts and vehicles containing PFOA, its salts and related compounds may be recycled. Therefore, it cannot be excluded that PFOA, its salts and related compounds potentially accumulate in recyclates / new products. The risks can be minimised by separating automotive service and replacement parts and vehicles containing PFOA, its salts and related compounds to the largest extent possible with a subsequent environmentally sound waste treatment. However, as indicated, it is essential to know the main application areas of PFOA, its salts and PFOA-related compounds within the automotive sector to be able to separate relevant wastes.</td>
</tr>
<tr>
<td>Lubricants</td>
<td>n.a.</td>
<td>Incineration</td>
<td>A certain share of automotive service and replacement parts will be incinerated. Due to high incineration temperatures it can be assumed that PFOA, its salts and related compounds are largely destroyed (see additional information provided in previous sections on uncertainties regarding formation and destruction efficiencies of PFOA in municipal waste incinerators).</td>
</tr>
<tr>
<td></td>
<td>n.a.</td>
<td>Landfill</td>
<td>In general, landfilling involves the risk that POPs could be released from the landfill site into the environment. Potential release of PFOA, its salts and PFOA-related compounds from landfill leachate to the environment cannot be generally excluded. The risk can be minimised through decreasing the landfill by directing contaminated products to appropriate treatment options.</td>
</tr>
<tr>
<td>PFHxS, its salts and related compounds</td>
<td>n.a.</td>
<td>Recycling</td>
<td>Similar risks and recommendations for ESM apply as in the case of PFOA, its salts and related compounds (see previous sections).</td>
</tr>
<tr>
<td></td>
<td>n.a.</td>
<td>Incineration</td>
<td>Similar risks and recommendations for ESM apply as in the case of PFOA, its salts and related compounds (see previous sections).</td>
</tr>
<tr>
<td></td>
<td>n.a.</td>
<td>Landfill</td>
<td>Similar risks and recommendations for ESM apply as in the case of PFOA, its salts and related compounds (see previous sections).</td>
</tr>
<tr>
<td></td>
<td>n.a.</td>
<td>Recycling</td>
<td>The use of PFHxS in lubricants has been indicated during the recent stakeholder consultation for this project. However, no additional qualitative or any quantitative information on the use of PFHxS in lubricants has been identified.</td>
</tr>
<tr>
<td>Sector</td>
<td>Amount of substance (in t/a or qualitatively)</td>
<td>Treatment operation</td>
<td>Possible risks regarding the POP content and strategy for ESM of wastes</td>
</tr>
<tr>
<td>-----------------</td>
<td>---------------------------------------------</td>
<td>---------------------</td>
<td>-----------------------------------------------------------------------</td>
</tr>
<tr>
<td>n.a.</td>
<td>Incineration</td>
<td></td>
<td>According to the newest data from Eurostat, a significant share of used oils in the EU is recovered/recycled (about 86% in 2014). Considering that a major share of spent oils is recovered/recycled in the EU, it cannot be excluded that PFHxS-containing or contaminated waste oils enter recycling streams and consequently may accumulate in new recovered/recycled oils. Therefore, lubricants containing PFHxS should be separated from other spent oils as far as possible and separately managed (preferably incinerated).</td>
</tr>
<tr>
<td>n.a.</td>
<td>Landfill</td>
<td></td>
<td>Around 13% of waste oils is incinerated (with and without energy recovery) according to data from Eurostat. It can be assumed, that controlled incineration with high temperatures (e.g., at 1,000°C) is effective to destroy PFHxS. However, the destruction efficiencies of PFHxS in waste incinerators are not known.</td>
</tr>
<tr>
<td>Semiconductors</td>
<td>n.a.</td>
<td>Recycling</td>
<td>Landfilling of lubricants and oils is not a commonly applied waste management operation in the EU.</td>
</tr>
<tr>
<td>n.a.</td>
<td>Incineration</td>
<td></td>
<td>It is not clear if PFHxS, its salts and related compounds have also been used in the EU in the production of semiconductors. However, considering that semiconductors are manufactured and traded on a global level, it cannot be excluded that PFHxS-containing semiconductors or finished products containing respective semiconductors enter the EU market. Information regarding wastes and recyclates is missing. However, the use of semiconductors in various products will result in various waste streams (e.g. in WEEE, ELV, municipal solid waste) which will need to be managed. Besides, as some of the waste streams may be recycled it cannot be generally excluded that PFHxS, its salts and PFHxS-related compounds accumulate in recyclates and subsequently in new products. Manual disassembly or mechanical treatment in shredders can result in dust generation and therefore presents a potential environmental and health risk. In order to minimise the risk, dust generation should be avoided as far as possible. Inhalation of contaminated dust can be avoided by the application of breathing masks.</td>
</tr>
<tr>
<td>n.a.</td>
<td>Landfill</td>
<td></td>
<td>It can be assumed, that controlled incineration with high temperatures (e.g., at 1,000°C) is effective to destroy PFHxS. However, the destruction efficiencies of PFHxS in waste incinerators are not known.</td>
</tr>
<tr>
<td>n.a.</td>
<td>Landfill</td>
<td></td>
<td>In general, landfilling involves the risk that POPs could be released from the landfill site into the environment. Potential release of PFHxS, its salts and PFHxS-related compounds from landfill leachate to the environment cannot be generally excluded. The risk can be minimised through decreasing the landfill by directing products containing PFHxS-containing semiconductors to other.</td>
</tr>
<tr>
<td>POPs</td>
<td>Sector</td>
<td>Amount of substance (in t/a or qualitatively)</td>
<td>Treatment operation</td>
</tr>
<tr>
<td>------</td>
<td>--------</td>
<td>---------------------------------------------</td>
<td>---------------------</td>
</tr>
<tr>
<td>HBCDD</td>
<td>EPS/XPS for construction</td>
<td>9.1+8.3</td>
<td>Recycling</td>
</tr>
<tr>
<td>480.2+443.3</td>
<td>Incineration</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POPs</td>
<td>Sector</td>
<td>Amount of substance (in t/a or qualitatively)</td>
<td>Treatment operation</td>
</tr>
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<td>------</td>
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<td>---------------------------------------------</td>
<td>---------------------</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>200.9+185.3</td>
<td>Landfill</td>
<td>A relevant share (almost 30%) of EPS/XPS panels containing HBCDD is landfilled. HBCDD is not destroyed. The generation of other POPs does not occur. There is the potential risk of 'leaking' to the surrounding environment. Health or environmental risks cannot be excluded and are represented in the long run especially due to the risk of uncontrolled global distribution and the corresponding risks for human health and the environment. A specific health or environmental risk due to exceeding critical environmental concentrations or exposure limits is not expected. The risk can be minimised through decreasing the landfill of EPS/XPS panels containing HBCDD by directing them to other appropriate treatment options.</td>
<td></td>
</tr>
<tr>
<td>EPS in the packaging sector</td>
<td>1.26</td>
<td>Recycling</td>
<td>In practice, cutting losses of HBCDD-containing EPS insulation materials are unintentionally mixed and recycled with packaging waste. Thus, waste which is not contaminated or contains only minor amounts of HBCDD (EPS packaging waste), is mixed with HBCDD-containing waste (EPS for C&amp;D) and contaminated recycled PS pellets are generated in the consequence. This practice entails a further transfer of the HBCDD content into various plastic products, thus resulting in releases during further product cycles and end of life treatment and consequently in uncontrolled global distribution and possible releases of significant quantities of HBCDD. The emission of significant amounts of POPs into the environment is generally considered a risk. A specific health or environmental risk due to the exceeding of critical environmental concentrations or exposure limits is not expected.</td>
</tr>
<tr>
<td>POPs</td>
<td>Sector</td>
<td>Amount of substance (in t/a or qualitatively)</td>
<td>Treatment operation</td>
</tr>
<tr>
<td>------</td>
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<td>---------------------</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.35</td>
<td>Incineration</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.27</td>
<td>Landfill</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

n.r.: Not relevant
n.a.: Not available
6. SCENARIOS AND EVALUATION OF THE LIMITATION CRITERIA

In section 6 typical substance/substance group concentrations of the relevant waste fractions and product categories as well as the waste amounts which arose within the individual waste streams are used to derive different scenarios of the concerned waste amounts for possible LPCLs.

Further, the evaluation of the limitation criteria as described in 2.3 is applied for the proposal of LPCLs.

6.1 Scenarios for realistic limit values and their effects on treatment operations

6.1.1 SCENARIOS FOR DECBDE

Using typical decaBDE concentrations of the relevant waste fractions as well as the waste amounts which arose for the individual fractions, different scenarios of the concerned waste amounts for possible LPCLs can be derived.

It is assumed that the WEEE fraction containing Br > 2,000 mg/kg is separated in accordance with the voluntary CENELEC standard (see [CENELEC TS 50625-3-1 2016]). Assuming that the separated share is incinerated, the quantity of the waste fraction containing decaBDE subject to incineration is 96,000 t in the high Br fraction and 153,456 t in the low Br fraction. This leads to two possible derived limit values. Firstly, the average assumed concentration of the high Br fraction with 5,410 mg/kg. Secondly, a limit value of less than 100 mg/kg where only WEEE plastics (1,200 kt/a) below 2,000 mg/kg Br would be affected. Since Br > 2,000 mg/kg is disposed of, establishing a possible limit value at 5,410 mg/kg is considered irrelevant. However, a limit value of 5,410 mg/kg is considered as a scenario.

Under the presumption that the ELV plastic fraction disposed of has a typical concentration of 0.109% of decaBDE a possible limit value of 109 mg/kg can be derived. Any possible LPCL below this limit value would result in the waste fraction of ELV plastics to be concerned as POP waste (900.88 kt/a) in addition to the aforementioned 1,200 kt/a resulting from WEEE plastics POP waste. In total, a theoretically concerned waste amount of 2,100.88 kt/a can be derived.

At a possible low POP concentration limit of 0.3 mg/kg theoretically all waste streams would be concerned including sludges from waste water treatment (3,060 kt/a) which would result theoretically in a concerned waste amount of 5,159.78 kt/a.

Figure 36 graphically illustrates the theoretically concerned waste amounts within each of the different scenarios.
Scenarios for low POP concentration limits (decaBDE)

Possible LPCLs would also lead to a change in the waste streams. Thus, waste exceeding possible LPCLs would have to be directed to an alternative treatment operation.

Table 50 elaborates the concerned waste streams for the different possible LPCLs explained in the previous section. Further, decaBDE-containing waste streams which would theoretically have to be directed to alternative treatment operations are listed.

Table 50: Possible LPCLs and their effects on the present treatment operation (decaBDE)

<table>
<thead>
<tr>
<th>Possible low POP concentration limits</th>
<th>0.3 mg/kg</th>
<th>100 mg/kg</th>
<th>109 mg/kg</th>
<th>5,410 mg/kg</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Concerned waste streams for different low POP concentration limits</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ELV plastics</td>
<td>900.9</td>
<td>900.9</td>
<td>900.9</td>
<td>-</td>
</tr>
<tr>
<td>WEEE plastics high Br fraction</td>
<td>96.0</td>
<td>96.0</td>
<td>96.0</td>
<td>96.0</td>
</tr>
<tr>
<td>WEEE plastics low Br fraction</td>
<td>1,104.0</td>
<td>1,104.0</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Sludges</td>
<td>3,060.0</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Total [kt/a]</strong></td>
<td>5,160.9</td>
<td>2,100.9</td>
<td>996.9</td>
<td>96.0</td>
</tr>
<tr>
<td><strong>Concerned waste streams which theoretically would have to be treated alternatively for different LPCLs</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ELV plastics</td>
<td>664.6</td>
<td>664.6</td>
<td>664.6</td>
<td>-</td>
</tr>
<tr>
<td>WEEE plastics high Br fraction</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>WEEE plastics low Br fraction</td>
<td>950.5</td>
<td>950.5</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Sludges</td>
<td>2,550.0</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Total [kt/a]</strong></td>
<td>4,165.1</td>
<td>1,615.1</td>
<td>664.6</td>
<td>0.0</td>
</tr>
</tbody>
</table>

The limit value of 5,410 mg/kg, as shown in the table above, would not lead to waste capacities which would have to be treated alternatively, as the high Br fraction of WEEE plastics is currently incinerated.

A possible LPCL of 109 mg/kg would lead to a total of 664.9 kt/a which would have to be treated in an alternative treatment operation. This amount results from the ELV plastic waste which would
have to be treated to a greater share with incineration and the share of recycling and landfilling of ELV plastics would be reduced.

A LPCL of 100 mg/kg would theoretically lead to about 1,615.1 kt/a of decaBDE-containing waste, which would have to be treated alternatively.

Lastly, with a LPCL of 0.3 mg/kg, a considerably larger waste amount of 4,165.1 kt/a would have to be treated alternatively.

6.1.2 Scenarios for SCCP

The average SCCP concentration in rubber conveyor belts for the use in underground mining amounts to 3.3 %, and to ca. 100,000 mg/kg in the corresponding rubber fraction of the conveyor belts. In the absence of detailed information concerning the processing and disposal of used rubber conveyor belts it is assumed that the rubber fraction of the conveyor belts is treated/disposed of together with other rubber wastes. Under this assumption, the theoretical SCCP concentration of the total waste stream decreases to about 420 mg/kg.

The average SCCP concentration in sealants and adhesives for the construction sector averages to 200,000 mg/kg. It is assumed that due to their properties, a major proportion of the applied sealants and adhesives adheres to the surface of other construction materials (e.g. on concrete, tiles, bricks and ceramics) and is therefore also treated together with this waste. In the absence of more precise information, it is assumed that about 2/3 (~1,373 t) of the SCCP-containing waste stream is treated along with the other construction and demolition waste and that ca. 1/3 (~687 t) of the relevant waste stream is disposed of separately in hazardous waste incineration plants. The concentration of SCCP in the mixed construction and demolition waste decreases to ca. 1 mg/kg.

At a low POP concentration limit <420 mg/kg, theoretically the entire rubber waste in the EU would be affected (~2,630 kt) whereas at a low POP concentration limit <1 mg/kg theoretically the entire construction and demolition waste (~276,910 kt) as well as the entire rubber waste in the EU (~2,630 kt) would be concerned (see Figure 37).

![Figure 37: Scenarios for realistic low POP concentration limits (SCCP)](image)

A possible POP concentration limit for SCCPs <1,000 mg/kg would have no impact on the current waste management of the above described waste streams. At a low POP concentration limit of
<420 mg/kg, theoretically about 1,430 kt SCCP-containing rubber waste (currently recycled) would have to be treated alternatively and at a low POP concentration limit <1 mg/kg a considerably larger waste amount of 276,762 kt would be concerned (see the following table).

Table 51: Possible LPCLs and their effects on the present treatment operation (SCCP)

<table>
<thead>
<tr>
<th>Possible low POP concentration limits</th>
<th>0.1 mg/kg</th>
<th>420 mg/kg</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Concerned waste streams for different low POP concentration limits</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rubber waste [kt/a]</td>
<td>2,630</td>
<td>2,630</td>
</tr>
<tr>
<td>Mineral waste from C&amp;D [kt/a]</td>
<td>276,910</td>
<td></td>
</tr>
<tr>
<td><strong>Total [kt/a]</strong></td>
<td>279,540</td>
<td>2,630</td>
</tr>
<tr>
<td><strong>Concerned waste streams which theoretically would have to be treated alternatively for different low POP concentration limits</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rubber waste (incl. SCCP-containing rubber waste) [kt/a]</td>
<td>1,430</td>
<td>1,430</td>
</tr>
<tr>
<td>Mineral waste from C&amp;D (incl. SCCP-containing sealants and adhesives) [kt/a]</td>
<td>275,332</td>
<td></td>
</tr>
<tr>
<td><strong>Total [kt/a]</strong></td>
<td>276,762</td>
<td>1,430</td>
</tr>
</tbody>
</table>

At this point it needs to be mentioned, that a possible POP concentration limit for SCCPs <1,000 mg/kg would still impact several waste streams in the EU (except the ones discussed above). In particular, imports of new products and/or recycled products contaminated with or containing SCCPs (also in concentrations above the set limit values in Regulation EU 2015/2030) cannot be excluded and are regularly detected during enforcement activities. It is expected that most of those imported articles will be landfilled or incinerated together with municipal solid waste streams; however, a possible recycling of respective articles cannot be generally excluded. However, up-to-date information on recycling activities in EU MS and quantitative information on SCCP concentrations and amounts in waste and recyclates is not available. No information on SCCP concentrations in waste and recyclates could be obtained during the stakeholder consultation for this project (April 2018). In addition, considering the large diversity of the relevant imported new articles potentially containing SCCPs and the fact that imports cannot be traced it is not possible to further investigate these waste streams.

### 6.1.3 SCENARIOS FOR HCBD

The assumed concentration of HCBD in ashes and residues from the incineration of municipal hazardous waste is 0.0008 mg/kg. Since 43 % of all ashes and residues from the incineration of municipal hazardous waste are recycled and 56 % are landfilled at present, setting the low POP concentration limit at 0.0008 mg/kg would result in a necessary change of waste treatment for almost 99 %, or 5,230 kt, of ashes and residues. However, a re-combustion of ashes and residues from the incineration of municipal hazardous waste is excluded as an alternative treatment method [BiPRO 2011]. Thus, only the underground disposal of 5,230 t would be a relevant option. Since the concentration with 0.0008 mg/kg is quite low and the alternative treatment option would be very unrealistic it is omitted to select this as a scenario.

In section 4.3.5 mass flows were established for sewage sludge from waste water treatment plants for HCBD. However, HCBD concentration estimated is negligible. On that account, the report refrains from elaborating scenarios of the concerned waste amounts for possible low POP concentrations.
concentration limits. In all probability, an LPCL would not concern any relevant waste fractions or recyclates.

### 6.1.4 SCENARIOS FOR DICOFOL

There is no specific information indicating relevant dicofol concentrations in certain waste fractions or products in the EU. On that account, scenarios of the concerned waste amounts for possible low POP concentration limits cannot be elaborated. Probably, an LPCL would not concern any relevant waste fractions or recyclates.

### 6.1.5 SCENARIOS FOR PFOA

PFOA, its salts and related compounds have been widely used in many applications including consumer products across various sectors. However, quantitative information on used amounts in different application areas is limited. Besides, information on wastes and recyclates is missing.

Due to the limited quantitative information on PFOA production and use as well as missing information related to waste streams and recyclates containing or contaminated with those substances, concerned waste amounts for a possible LPCL cannot be derived. Nevertheless, the current relevance of PFOA, its salts and related compounds in products, wastes and recyclates in the EU is at least qualitatively evaluated / discussed in this report and recommendations for possible limit values and environmentally sound waste management are included in section 8 report.

### 6.1.6 SCENARIOS FOR PFHXS

PFHxS, its salts and related compounds have been widely used in many applications including consumer products across various sectors. However, quantitative information on used amounts in different application and information on wastes and recyclates is missing. Therefore, concerned waste amounts for a possible LPCL cannot be derived. Nevertheless, the current relevance of PFHxS, its salts and related compounds in products, wastes and recyclates in the EU is at least qualitatively evaluated / discussed in this report and recommendations for possible limit values and environmentally sound waste management are included in section 8 of the report.

### 6.1.7 SCENARIOS FOR HBCDD

Using typical HBCDD concentrations of the relevant waste fractions as well as the waste amounts which arose within the individual fractions (cf. material flows in section 4.7.6.6), different scenarios of the concerned waste amounts for possible low POP concentration limits can be derived.

Under the presumption that the flame retarded C&D waste is disposed of has a typical concentration of 1.5% of HBCDD a possible limit value of 15,000 mg/kg can be derived. Since the limit value of HBCDD is defined at the EU POP Regulation with 1,000 mg/kg, the discussion around a possible limit value of 15,000 mg/kg may be irrelevant. However, according to the calculated mass flows, a certain share of XPS from C&D waste is still being landfilled, which according to the POP Regulation is not allowed. For this reason, the limit value of 15,000 mg/kg is still considered as a scenario in order to see the effect if the POP Regulation would actually be followed. For a limit value of less than 15,000 mg/kg only one fraction of the concerned waste streams, namely XPS (32.9+ kt/a), would be affected.

Considering the mass flow of EPS from C&D waste in the EU, a further possible limit value of 7,000 can be derived. Since the limit value of HBCDD is defined at the EU POP Regulation with 1,000 mg/kg the discussion of a possible limit value of 7,000 mg/kg would also be irrelevant. Nevertheless, as with XPS, a certain share of EPS from C&D waste is still being recycled and
landfilled, which according to the POP Regulation is not allowed. For this reason, the limit value of 7,000 mg/kg is still considered as a scenario in order to see the effect, if the POP Regulation would actually be followed. Below the theoretical limit value of 7,000 mg/kg, consequently the EPS C&D waste would be concerned (98.6 kt/a) in addition to the aforementioned 32.9 kt/a from XPS C&D waste, resulting in a theoretically concerned waste amount of 131.5 kt/a in total.

As illustrated in section 4.7, it is assumed that at present HBCDD is used predominantly in the construction sector and that therefore no relevant waste amounts arise in other sectors. However, the analysis results indicate that the average HBCDD contamination of packaging waste might amount to about 10 mg/kg. Hence, for a possible limit value of 10 mg/kg, an additional 388.3 kt/a of waste would be concerned in addition to the aforementioned 32.9 kt/a and the 98.6 kt/a (total: 519.8 kt/a).

Figure 38 graphically illustrates the theoretically concerned waste amounts within each of the different scenarios.

Figure 38: Scenarios for low POP concentration limits (HBCDD)

Possible low POP concentration limits would also lead to modification in the waste streams. Therefore, waste exceeding a possible LPCLs has to be directed to an alternative treatment e.g. certain waste no longer being allowed for recycling or landfilling.

Table 52 details the concerned waste streams for the different possible LPCLs explained previously. Furthermore, HBCDD-containing waste streams which would theoretically have to be directed to alternative treatment operations are listed.

Table 52: Possible LPCLs and their effects on the present treatment operation (HBCDD)

<table>
<thead>
<tr>
<th>Concerned waste streams for different low POP concentration limits</th>
<th>10 mg/kg</th>
<th>7,000 mg/kg</th>
<th>15,000 mg/kg</th>
</tr>
</thead>
<tbody>
<tr>
<td>EPS - C&amp;D waste [kt/a]</td>
<td>98.6</td>
<td>98.6</td>
<td>-</td>
</tr>
<tr>
<td>XPS - C&amp;D waste [kt/a]</td>
<td>32.9</td>
<td>32.9</td>
<td>32.9</td>
</tr>
<tr>
<td>EPS - packaging [kt/a]</td>
<td>388.3</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>
A possible LPCL of 15,000 mg/kg would lead to a total of 9.7 kt/a which would have to be treated alternatively. This amount results from the XPS C&D waste which would have to be incinerated instead of landfilled.

A LPCL of 7,000 mg/kg would theoretically lead to about 39.7 kt/a of HBCDD-containing waste, which would have to be treated alternatively. Lastly, at a LPCL of 10 mg/kg, a considerably larger waste amount of 292.9 kt/a would be concerned.
6.2 Evaluation of lower and upper limitation criteria

6.2.1 EVALUATION OF THE LOWER LIMITATION CRITERIA OF THE LPCL

6.2.1.1 (A) Analytical potential

Analysis costs and detection/quantification limits depend on the individual substance, the examined matrix, as well as on the laboratory costs of the analysis. Further, it has to be considered whether an analytical method is appropriate to be used under relevant conditions e.g. in industrial recycling processes. For the evaluation, not the minimum detection and or quantification limit of an analytical method is used, but, as far as available, the detection and or quantification limit in the usual waste matrices under usual effort. The LOD is hereby the lowest analyte concentration likely to be reliable at which the detection is feasible. The limit of quantification (LOQ) is the lowest concentration at which the analyte can not only be reliably detected but also at which set parameters for bias and imprecisions are met [Armbruster and Pry 2008]. The LOQ can thereby be equal to the LOD or it could be a higher concentration. As a rule of thumb if the LOQ is not given and is not equal to the LOD, the LOD is multiplied times 3 to extrapolate the LOQ. This is in accordance with regulation (EU) 333/2007 [EU 2007] laying down the methods of sampling and analysis for the official control of the levels of several contaminants in foodstuffs, specifies the LOD as 3 times the standard deviation of the mean of blank determinations and LOQ as six or 10 times the standard deviation of the mean of blank determinations. Ten times the standard deviation of the blank corresponds, under the condition of constant standard deviations at low levels, to a relative standard deviation of 10 %. Depending on the definition of LOQ, ratios between LOQ and LOD of 2 or about 3.3 are obtained. This document specifies the relation between LOD and LOQ as

\[ x_{\text{LOQ}} = 3 \times x_{\text{LOD}} \]

Analysis costs include costs for typical sample preparation. Costs up to €500 per individual measurement are regarded as economically available. The actual costs partly range significantly below this value and differ distinctly depending on the substance/substance group.

DecaBDE

DecaBDE belongs to the group of BFRs. The analytical quantitative determination in plastic and textiles is performed using either: (1) state-of-the-art laboratory techniques accompanied by matrix extraction and further clean-up methods or (2) XRF spectrometry.

Methods

Individual BFRs, like decaBDE, can be quantified by state-of-the-art laboratory techniques those include gas chromatography-mass spectrometry (GC-MS) as well as informative techniques such as ion attachment mass spectrometry (IA-MS) coupled with direct injection probe (DIP) and high-pressure liquid chromatography technique coupled with an UV-Detector (HPLC-UV) [BSEF Sub. 2018; International Electrotechnical Commission (IEC) 62321-6:2015]. GC-MS and the two informative techniques are validated in IEC 62321-6:2015 for the determination of PBDEs in electrotechnical products [IEC 62321-6:2015]. According to several stakeholder submissions, GC-MS is the up-to-date measurement method for identifying the presence and levels of decaBDE in waste [ARN Sub. 2018; IPEN & Arnika Sub. 2018]. The advantage of GC-MS is the simultaneous analysis with other BFRs such as polybrominated diphenyl ethers (PBDEs) and thus GC-MS as a measurement can be used to verify the quality on a regular basis (e.g. monthly) and to check the efficiency of the sorting process [Plastic Recyc. Europe Sub. 2018]. As the process of measuring decaBDE via GC-MS is a relatively expensive method, it is more often used for scientific analyses than in daily practice.
Generally, GC-MS and the other mentioned techniques are considered to be relatively time-consuming, since matrix extraction and further sample preparation have to take place. Therefore, in practice, also a second determination method is used: XRF spectrometry [EuRIC Sub. 2018b]. XRF spectrometry is further validated for measuring a concentration of total Br in electrotechnical products [IEC 62321-3-1:2013]. This method has several advantages, as it is non-destructive and allows in-situ (on site) analysis, which makes time-consuming sample treatment obsolete. In addition, it is possible to buy handheld and/or portable XRF spectrometers. The big disadvantage of this technique is, that the total bromine amount is determined. This means, that for example also other BFRs like, HBCDD and other potentially unknown brominated compounds are identified and/or quantified. In practice, the total bromine is regarded as a first indication for BFR-contamination of waste. Resulting in a differentiation between Br-positive and -negative samples however as [Furl et al. 2012] found only a small fraction of the total Br present were PBDEs. It was nevertheless concluded that there was no case where PBDEs were detected in the absence of XRF-measured Br, indicating that XRF is useful as a screening tool to identify samples that could contain PBDEs.

**LOD/LOQ**

The theoretical LOD for GC-MS is in the sub- µg/kg range, however in practice a limit of detection/quantification of around 0.01-0.1 mg/kg in some matrices is more reasonable. The [IEC 62321-6:2015] standard is evaluated for individuals PBDEs between 20 mg/kg and 2,000 mg/kg in PC/ABS. Thus, a LOQ of 20 mg/kg can reasonable be assumed for decaBDE. The LOQ of the ED (energy based)-XRF method in EEE is 100 mg/kg [Beccagutti 2016]. The stakeholder submission of [AT Ministry of Sustain. and Tourism Sub. 2018b] reported an even higher range LOD of 100-150 mg/kg based on the acceptable relative error of the device and the accuracy of the results. It was pointed out that only new XRF equipment would be able to detect below 10 mg/kg. Within the validated standard (IEC 62321-3-1:2013]) concentrations starting from 25 mg/kg in HIPS and ABS and 96 mg/kg in PE for total Br were measured. It was indicated within the stakeholder responses that the standard is validated for 1,000 mg/kg for total Br [BSEF Sub. 2018].

Considering the literature findings and the reported measured concentrations within the standard a LOQ of below 100 mg/kg for Br is considered reasonably achievable. Making analogue consideration as for HBCDD, if the method results in a bromine content of 100 mg/kg it is ensured that the decaBDE content is below 200 mg/kg decaBDE, as decaBDE consists of 83.3% Br (w/w). Therefore, a Br content of 100 mg/kg corresponds to a decaBDE content of 120 mg/kg, under the assumption that decaBDE is the only brominated substance within the plastic. Accordingly, an LOQ for decaBDE of 120 mg/kg is considered achievable.

**Costs**

The individual measurements cost for GC-MS are around €200. [Plastic Recyc. Europe Sub. 2018] and [ARN Sub. 2018] however, pointed out that high costs of laboratory analysis can occur and have to be considered, as several analyses have to be done to provide reliable measurements on a batch of material. Thus, several analyses and calculating the average measurement or changing the sampling protocol are proposed which potentially increase the total costs. According to [ARN Sub. 2018], internal XRF measurements is a known inexpensive screening method for identifying decaBDE in waste. This screening methods shows comparable results as Gas Chromatography – Mass Spectrometry (GC/MS) (DIN EN ISO 22032) measurement method, with slight differences. Some values measured with XRF are higher and some lower [ARN Sub. 2018]. In 2018, the acquisition cost of a XRF-handheld device (XRF gun) is between €16,000 - €45,000 based on different suppliers, applications, calibrations and hardware. There are no additional costs per single measurement. [EuRIC Sub. 2018] however, pointed out that XRF can be
qualified as expensive as some portable detectors can cost up to 10,000 € and that the process is not adapted to industrial processes.

**Summary**

In summary it can be concluded that screening methods to check whether plastic or textile waste contains decaBDE above levels of 200\(^{55}\) mg/kg are available. There are standards for XRF and GC-MS available which are validated for 1,000 mg/kg [IEC 62321-3-1:2013; IEC 62321-6:2015]. Considering the plan to implement a lower LPCL for decaBDE and that existing standards are evaluated regularly, it is assumed that an adjustment of standards to measure lower concentration is under development and will be available in a few years.

**SCCPs**

**Methods**

The analytical quantitative determination of SCCPs in various matrices is performed using: (1) *state-of-the-art laboratory techniques* accompanied by matrix extraction and further clean-up methods (*GC-ECNI-MS, GCxGC-ECD, carbon skeleton GC-FID, TOF-MS*).

In general, extraction and clean-up procedures for the analysis of SCCPs in biological and environmental samples are comparable to those developed for the analysis of other halogenated compounds such as PCBs and chlorinated pesticides. In general, the procedures are based on batch or Soxhlet extraction with organic solvents (toluene, n-hexane), clean-up of the extracts by adsorption and gel permeation chromatography and determination by gas chromatography electron capture mass spectrometric detection. However, in contrast to PCB and chlorinated pesticides, the analysis of SCCPs is particularly challenging, as they are complex mixtures of compounds and isomers thus difficult to separate chromatographically.

Instrumental techniques used for all matrices are mostly determined by gas chromatography in combination with *electron capture negative ion mass spectrometry (GC-ECNI-MS)*. A challenge of this method is that some of the quantifier signals of SCCPs can be overloaded when high MCCPs (medium chain length) are present in the sample. Additional disadvantages pointed out by [Mourik et al. 2015], in their review of recent developments for analysing CPs, are that GC-ECNI-HRMS is relatively expensive and GC-ECNI-LRMS can lead to incorrect concentration self-interference between CPs and interference from compounds such as chlordanes, toxaphenes and PCBs. In the case of SCCPs (C\(_{10}\)-C\(_{13}\)) with a chlorination degree between 49 and 67 % by mass there are up to 8,000 homologues, diastereomers and enantiomers possible. Furthermore, although the hydrocarbons used to prepare SCCPs are primarily linear n-alkanes, they also contain branched alkanes and probably other hydrocarbons which would also add to the complexity of the mixtures. Even if only a small percentage of the theoretically possible number of chloroalkanes are readily formed, it can be assumed that commercial SCCP formulations contain many thousand compounds. There is also a lack of suitable standards, their low response in various detection systems and their high complex nature which leads to a difficulty in chromatographic separation between CPs and between CPs and other organohalogen compounds as described by [Mourik et al. 2015].

Three International Standards Organization (ISO) methods have been published that enhance the standardized analyses of SCCPs in water, sediment, sewage sludge, suspended matter and leather. Method ISO 12010:2012 is applicable to the determination of the sum of SCCPs in unfiltered

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\(^{55}\) 120 mg/kg would be possible; in order to increase the margin of safety a concentration of 200 mg/kg is suggested.
surface water, ground water, drinking water and wastewater using gas chromatography-mass spectrometry with electron capture negative ionization (GC-ECNI-MS). Method 18635:2016 is applicable for the quantitative determination of SCCPs in sediment and suspended (particulate) matter, sewage sludge, and soil using GC-ECNI-MS. Method ISO 18219:2015 specifies a chromatographic method to determine the amount of SCCPs in processed and un-processed leathers [UNEP/POPS/POPRC.12/11/Add.3 2016]. A new standard for analysis of SCCPs in textiles is under development (i.e. ISO/NP 22818), [UNEP/CHW/OEWG.11/INF/10 2018]. Further it can be extracted from dust, waste and pine needles as well of polymer samples like for example coated textiles, printed textiles and coated leather samples. The extraction involves toluene as a solvent. In the case of uncoated leather samples, the extraction is done by n-Hexane. Both extraction techniques are described in DIN EN ISO 18219. For SCCPs determination in water, no further extraction is needed, according to DIN EN ISO 18219:2016-02.

According to Mourik et al (2015), comprehensive two-dimensional gas chromatography is another promising tool for the analysis but is currently only employed in few laboratories due to the complicated optimisation, the cost factor and the amount of time that is required for quantification [Mourik et al 2015]. As stated by [Euro Chlor Sub. 2018] during the recent stakeholder consultation for this project, 2-dimensional gas chromatography combined with electron capture detection (GCxGC-ECD) is state-of-the-art CA detection method. The GCxGC separation method can qualitatively identify groups of CA isomers by carbon chain length and chlorination level, although this is very difficult due to the complex nature of chlorinated alkanes. Advantages of this technique include the detection of lower chlorinated congeners, the high separation power of congeners with different chlorination levels and the ability to detect groups of congeners with equal chlorine levels.

Another approach is carbon skeleton analysis by gas chromatography with flame ionisation detection (carbon skeleton FID-GC) after simultaneous dechlorination and hydrogenation. Moreover, novel and promising use of high resolution time of flight Mass Spectrometry (TOF-MS) has also been reported. In addition, improved clean-up procedures have been found to remove interfering compounds, and new instrumental techniques, which distinguish between MCCPs and SCCPs, have been developed. The study also states that new CP quantification methods have emerged, including the use of mathematical algorithms, multiple linear regression and principal component analysis which however are not yet widely used. Further, [Euro Chlor Sub. 2018] states that many state laboratories use standards for comparison which are based on chlorinated end groups which may add to the detection problem. The laboratories detect matches for the C10-13 standard (which do not reflect historic, commercially produced SCCPs) and conclude that SCCP has been detected. Up to now, this drawback remains, therefore, the preparation of certified reference materials (standard solutions, sediment and biota samples) is needed. [Euro Chlor Sub. 2018] further points out that, whilst GC-ECNI-MS is a popular method, this method has difficulty in accurately separating different congeners with the same chlorine number, and the detection of congeners containing low numbers of chlorine atoms (≤ CI5).

LOD/LOQ

The LODs of GC-ECNI-MS are relatively high for most analysis and as methods develop and LODs decrease blank contamination may become a greater issue [Mourik et al 2015]. Reported LOD values for SCCPs in water were around 50 to 100 ng/l. For SCCPs in sediment and soil typical reported LOD are<0.144 ng/g but are highly dependent on the clean-up efficiency of Sulphur [Mourik et al. 2015]. A practical LOQ of GC-ECNI-MS indicated by the Danish EPA is 0.1 mg/kg [Danish EPA 2014a].

Costs

The individual cost for the analysis is in a range of €190-380, depending on the matrix examined.
Summary

For the evaluation of the lower limitation criteria, 0.1 mg/kg is considered applicable for criterion A.

HCBD

The analytical quantitative determination of HCBD in various matrices is performed using state-of-the-art laboratory techniques accompanied by matrix extraction and further clean-up methods (GC-ECD, GC-MS, GC-HECD, GC-PID).

Methods

Hexachlorobutadiene in environmental samples (soil, water, air) is measured using GC coupled with electron capture detector (GC-ECD), mass spectrometry (GC-MS), or a halogen electrolytic conductivity detector (GC-HECD), or a photoionization detector (GC-PID) [BiPRO 2011]. US EPA Method 8260D is used to determine volatile organic compounds in a variety of solid waste matrices. This method is applicable to nearly all types of samples, regardless of water content, including various air sampling trapping media, ground and surface water, aqueous sludges, caustic liquors, acid liquors, waste solvents, oily wastes, mousse, tars, fibrous wastes, polymeric emulsions, filter cakes, spent carbons, spent catalysts, soils, and sediments [US EPA 2018].

LOQ/LOD

The limit of quantification (LOQ) for HCBD in water is 0.01 µg/L [RIVM 2015]. The US EPA method 8260D (volatile organic compounds by GC-MS) describes a method detection limit (MDL) of 0.1 µg/L.

Costs

The costs lay in the range of 200-400 € [US EPA 2018].

Summary

In summary it can be concluded that screening methods are available to check whether waste contains HCBD above levels of 0.001 mg/L by means of GC-MS.

Dicofol

The analytical identification and quantification of dicofol is done using GC or LC-MS/MS from various matrices (DIN CEN/TR 16699 and DIN EN 12393-2).

Methods

Dicofol is mostly analysed simultaneously with up to 700 other pesticides. This can be done in all environmental samples (water, soil, sediment) and other matrices (food stuff, leather, clothing).

There are various methods for extraction and clean-up, for example acetone and liquid-liquid partition with dichloromethane/light petroleum. Dicofol is well known to readily degrade to p,p’-dichlorobenzophenone (DCBP) at various steps of the analytical procedures (e.g. during milling, extraction, clean-up, extract storage and GC-injection). DCBP is not part of the residue definition of dicofol. It is thus important to detect dicofol as such and to conduct analysis in a way that minimizes dicofol losses. Therefore, it is recommended to use an internal (deutered) standard [EURL-SRM 2013].

LOQ/LOD

In the case of pesticides in water, a LOQ of around 0.01 µg/mL is reported. For soil samples a LOD of 0.01 mg/kg is realistic. Starting from the LOD of the method (0.01 mg/kg) a LOQ of 0.03 mg/kg (3 times LOD) can be derived.
Costs

These comprehensive pesticides screenings cost around 170 € for water analysis and around 220 € for soil analyses [my-lab 2017].

Summary

In summary, any limit value above 0.03 mg/kg can be controlled with acceptable certainty and the described methodology.

PFOA

Detection methods for PFOA are primarily based on liquid chromatography (LC) or high-performance liquid chromatography (HPLC) coupled with tandem mass spectrometry (MS/MS).

Methods

In the case that water quality is to be checked, solid phase extraction is to be performed as shown in ISO 25101 and EPA method 537. According to US EPA, the linear isomer represents the predominant form of PFOA, but samples may also have some degree of branched-chain isomers. To account for linear and branched isomers of PFOA, EPA recommends that integration and quantitation of water samples include peaks that represent both linear and branched isomers [US EPA 2016]. In addition, PFOA is the main product of environmental biodegradation processes of several other perfluoroalkyl substances as shown exhaustively by [Butt et al. 2014]. Thus, when PFOA is measured also the precursor substances need to be considered.

LOQ/LOD

In LC-MS/MS or HPLC-MS/MS, the respective detection limits are in the ng/L range (0.000001 mg/kg). According to the recently published toxicological profile for perfluoroalkyls published by the US Agency for Toxic Substances and Disease Registry (ATSDR) and the US EPA typically achieved limits of detection range between 0.11–0.75 µg/kg for soil and 0.21–1.2 µg/kg in sediment [ATSDR 2018]. However, most laboratories report a LOQ of 1 mg/kg.

Costs

Costs of around 200 € for laboratory methods with a LOQ of 1 mg/kg can be expected [my-lab 2017].

Summary

In summary, any limit value above 1 mg/kg can be detected with the described methodology.

PFHxS

The analytical methods for the identification and quantification of PFHxS are the same as for PFOA. Furthermore, also the cost and LOQ are the same, leading to a result of 1 mg/kg for criterion A.

HBCDD

To detect and quantify HBCDD in plastics, different relevant test methods exists: (1) simple XRF spectroscopy, (2) XRF spectroscopy including extraction and (3) state-of-the-art laboratory techniques (GC-FID, GC-MS, LC-MS, UHPLC- ESI-MS/MS).

Methods

**XRF spectroscopy (without extraction)** is limited to the detection solely of bromine in polymers, without any capability to identify the type of brominated FR compound and needs a specific screening approach to differentiate between HBCDD and brominated PS polymer. In
addition, the test is only validated for the concentration limit of 1,000 mg/kg of HBCDD. If HBCDD with a lower content than 1,000 mg/kg has to be identified, this method is not applicable, according to [Giraf Results 2018]. IEC 62321-3-1 describes the screening analysis of five substances including total bromine (Br) in uniform materials found in electrotechnical products, using the analytical technique of X-ray fluorescence (XRF) spectrometry. Within the standards concentrations starting from 25 mg/kg in HIPS for total Br are tested. XRF is considered a reliable screening tool for the compliance with LPCLs. According to [Sharkey et al., 2018], XRF mistakenly identifies only 6% of samples as exceeding LPCLs.

**XRF spectroscopy including extraction** is based on the fact that PolyFR is not extractable whereas HBCDD (a low molecular substance) is. Following a rapid extraction (with acetone) of HBCDD the BFR is identified and quantified via bromine analysis using a handheld XRF instrument. The screening approach for the rapid identification of polystyrene foam wastes containing HBCDD or its alternative polymeric brominated flame retardant (Poly-FR) by X-ray fluorescence spectroscopy is described by [Schlummer et al. 2015].

Gas chromatography mass spectrometry (GC-MS) and liquid chromatography mass spectrometry (LC-MS) are both used for the analysis of HBCDD. An important advantage of LC-MS is the separation of individual diastereomers. This allows the use of mass labelled internal standards, which improves the accuracy of the results. The advantage of GC-MS is the simultaneous analysis with other BFRs such as polybrominated diphenyl ethers (PBDEs).

[Straková et al. 2018a] published very recently a tandem mass spectrometry technique with electrospray ionization in negative mode (UHPLC-ESI-MS/MS).

According to [PS PlasticsSupport Sub. 2018], in 2015 an industry group (HBCDD ANA WG) released an analytical method for PS foams. The method is based on detection via GC and Flame Ionization Detection (GC-FID). The method has been shared with relevant UNEP and EU Commission bodies as well as with engaged laboratories and standardisation bodies including the IEC TC111 WG3 [European HBCDD IG 2018].

**LOD/LOQ**

The LOD of XRF is 10 mg/kg for total bromine (which leads to a derived LOQ\(^{56}\) 30 mg/kg for total bromine [PS PlasticsSupport 2018a].

The LOD of the XRF with extraction method can be calculated as 35 ± 7 mg/kg bromine, which corresponds to 47 ± 9 mg/kg HBCDD [Schlummer et al. 2015]. According to [PS PlasticsSupport Sub. 2018] this screening method enables to identify polymers which contain HBCDD concentrations below 100 mg/kg. If the method results in a bromine content below 60 mg/kg it is ensured that the HBCDD content is below 100 mg/kg HBCDD.

The internationally standardised GC-MS and LC-MS can be applied to detect the HBCDD component with a LOD of 10-100 mg/kg, but the test results take a few days.

[Straková et al. 2018b] suggest a LOQ for HBCDD of 10 µg/kg (=0.01 mg/kg), achieved by LC interfaced with tandem mass spectrometry with electrospray ionization in negative mode (UHPLC-ESI-MS/MS).

The detection limit of the GC-FID technique is 35 mg/kg of HBCDD in the sample. The method enables to ensure that a limit value of 1,000 mg/kg can be ascertained. The method is considered economically available at global level. A description is available via the Basel Convention website\(^{57}\)

\(^{56}\) Applying standard rule LOD x 3

\(^{57}\)
In 2017 the group has initiated the investigation of analytical methods for determining HBCDD at a 100 mg/kg level in solid PS matrices, building upon the experience gained with the 1000 mg/kg method. Also here an essential consideration has been to validate a method that would be specific for the substance, in this case the HBCDD, and specific for the matrix, here solid PS pellets and granulates [European HBCDD IG 2018]. A limit value of 100 mg/kg is established as UTC for HBCDD in Annex I of the EU POP regulation. The group has identified several hurdles for the analysis of HBCDD at a 100 mg/kg level in PS matrices [European HBCDD IG 2018]. According to [PS PlasticsSupport Sub. 2018] the use of GC-FID is not appropriate to ensure that control of the limit values of 100 mg/kg or 500 mg/kg can be ensured. Considering the LOD of 35 mg/kg and a statistical approach this would be related to an LOQ of 120 mg/kg, below which specific figures are reported as “<LOQ”. Considering an LPCL of 500 mg/kg, the LOQ of 120 mg/kg would be at 24% of the limit value. [PS PlasticsSupport Sub. 2018] does not consider this sufficiently accurate to ensure a judicially secured control of a limit value of 500 mg/kg.

However, it is questionable whether such a broad margin of safety is required to control the implementation of the LPCL. Taking the LOD or LOQ into account and an uncertainty of three times the standard deviation could be considered an appropriate approach for taking uncertainty into account. The description of the GC-FID method does not indicate a standard deviation. Assuming a standard deviation of 10 %, the following consideration can be made: Starting from the LOD of the method (35 mg/kg) and an LOQ of 105 mg/kg (3 times LOD) and considering 3 times the assumed standard deviation (3 x 10.5) results in a value of 136.5 mg/kg HBCDD which can be quantified with an acceptable certainty. Based on these considerations, any limit value above 136.5 mg/kg can be controlled with acceptable certainty with the described GC-FID methodology. In conclusion, with even more certainty any limit value at or above 200 mg/kg HBCDD could therefore be considered implementable based on the GC-FID method.

Several stakeholders provided their view on a possible LPCL which can be found in the Annex in section 10.11.

Costs

The internationally standardised GC-MS and LC-MS take a few days for the results and costs range between 170-340 € [Giraf Results 2018]. According to another source, costs about 150 € per sample are reported [my-lab 2017].

Summary

Since 2015 the International Electrotechnical Commission (IEC) is working on an IEC standard for HBCDD under the standards family IEC 62321 on the determination of certain substances in electrotechnical products at IEC TC111 WG3). In the development LC-MS, GC-MS and to a certain extent also GC-FID are considered. The objective in 2014 was to develop a validated standard for 1,000 mg/kg, however in the meantime the objective also includes developing the standard for lower levels of HBCDD in polymers58 (down to 50 mg/kg). In 2015 an international interlaboratory study was initiated which is not yet concluded. According to the current time schedule it is foreseen to finalise a standard (planned standard IEC 62321-9 HBCDD in polymers by HPLC-MS) by the end of 2019. It can be expected that the planned standard will also enable validated

58 Note: A validated method is e.g. also required in view of the concentration limit for HBCDD in Annex I of the POP Regulation which is established at 100 mg/kg.
quantification of the HBCDD in polymers at levels well below 1,000 mg/kg (summarised based on [PS PlasticsSupport Sub. 2018]).

In summary it can be concluded that screening methods are available to check whether plastic waste contains HBCDD above or below levels of 100 mg/kg (LOD 47 mg/kg HBCDD; see [Schlummer et al. 2015]). An XRF screening standard is available which is validated for 1,000 mg/kg (IEC 62321-3-1). A validated standard for HBCDD determination is under development and may be available by the end of 2019 validated for 1,000 mg/kg and possibly well below (possibly down to 50 mg/kg). Moreover, a GC-FID method which is economically available at global level with a detection limit of 35 mg/kg enables to ensure that a limit value of 200 mg/kg can be kept.

Results for criterion A

The detection limits of analytical methods available at reasonable economic conditions are summarised in the table below.

Table 53: Practical Detection limits of analytical methods

<table>
<thead>
<tr>
<th>Substance/substance group</th>
<th>Method</th>
<th>Quantification limits [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>DecaBDE</td>
<td>XRF</td>
<td>200</td>
</tr>
<tr>
<td>SCCP</td>
<td>GC-ECNI-MS</td>
<td>0.1</td>
</tr>
<tr>
<td>HCBD</td>
<td>GC/MS</td>
<td>0.001 mg/L</td>
</tr>
<tr>
<td>Dicofol</td>
<td>GC/LC-MS/MS</td>
<td>0.03</td>
</tr>
<tr>
<td>PFOA</td>
<td>(HP)LC-MS/MS</td>
<td>2</td>
</tr>
<tr>
<td>PFHxS</td>
<td>(HP)LC-MS/MS</td>
<td>1</td>
</tr>
<tr>
<td>HBCDD</td>
<td>XRF spectroscopy including extraction</td>
<td>100</td>
</tr>
</tbody>
</table>
6.2.1.2 (B) Background contamination

The evaluation of environmental levels should take into account the fact that the lower boundary for the development of low POP concentration limits for POPs might be set by elevated POP concentrations in various environmental media [BiPRO 2011].

Any limit value should be significantly higher than the average or background level observed in the environment. A LPCL below the environmental background concentrations would cause severe economic problems and problems of acceptability if e.g. soil with usual background concentration would have to be treated as POP waste as soon as it is excavated.

Thus, background contamination levels have to be taken into account. For this purpose, highest environmental concentration levels for soil, and in absence of values for soil, sediments for Europe, if available, have been compiled. In order to express the uncertainty related to the availability of representative data an uncertainty factor of 10 is applies to derive the value of this lower limitation criterium.

DecaBDE

DecaBDE due to its intrinsic properties and given its low water solubility is expected to adsorb strongly to organic matter in suspended particles, sewage sludge, sediment and soil. Monitoring data outlined in [ECHA 2013a] showed that decaBDE is widely dispersed in the environment [UNEP/POPS/POPRC.10/10/Add.2 2014].

Levels of decaBDE in soil are primarily reported for China with the highest ranging values from 296 up to 5,850 ng/g dw on an e-waste recycling site. Ranges for Europe are typically lower with the highest recorded range going from 26 ng/g up to 440 ng/g in Sweden.

Wirth regards to sediment concentration, highest values recorded in Europe up to 1,650 ng/g dw were reported. Higher values of 98,125 μg/kg dw and 12,495dw μg/kg dw were reported in literature for Europe but stems from a disposal area or were taken in proximity of an industrial park and are therefore not further considered. Moreover, high values were found in Sweden but no indication on sampling location was given. Worldwide, the highest range for sediment was recorded in China with 21.3-7,340 ng/g in the Dongjiang River area not resulting from a potentially contaminated disposal site or e-waste recycling site [UNEP/POPS/POPRC.10/INF/5 2014]. Further elaboration can be found in [UNEP/POPS/POPRC.10/INF/5 2014 Table 5.1]

For the assumption of the background contamination of decaBDE for this report, the highest European value of 1,650 ng/g measured in Sweden was chosen as typical background contamination.

<table>
<thead>
<tr>
<th>Table 54: Calculation of decaBDE background contamination</th>
</tr>
</thead>
<tbody>
<tr>
<td>Highest value found in soil in the EU [ng/g]</td>
</tr>
<tr>
<td>1,650 ng/g</td>
</tr>
</tbody>
</table>

For the evaluation of the limitation criteria, an initial rounded value of 2 mg/kg as found in sediment is assumed for the background contamination of decaBDE and an uncertainty factor of 10 is applied. This results in the value of 20 mg/kg.

SCCPs

As outlined in [BiPRO 2015] and [BiPRO 2011], the existing SCCP background contamination in sediments can range from the detection limit to a maximum reported concentration level of
0.484 mg/kg (Japan). In Germany, concentrations reaching 0.083 mg/kg were measured (see BiPRO 2011, Tab. 6-47). The concentration values for Germany are in line with other values reported for the rest of Europe. Higher concentration within Europe were reported in soil and sediment e.g. in sediment found in Spain (with 2,090 ng/g dw) but were all located near a contaminated or industrial site and are therefore not further considered [UNEP/POPS/POPRC.6/INF/15 2010].

For the evaluation of the lower limitation criteria, an initial value of 0.1 mg/kg is assumed for the background contamination of SCCP and an uncertainty factor of 10 is applied. This results in the value of 1.0 mg/kg for criterion B.

**HCBD**

According to [BiPRO 2011] and [BiPRO 2015], most of the data on existing background contamination in sediments and soils exist for Canada and the United States, but also for Europe. The HCBD contamination in sediments amounts to about 0.001 mg/kg. The highest background contamination reported is indicated as 0.55 mg/kg, detected in the Great Lakes area in Canada. Different studies focusing on Germany identified concentrations of up to 0.005 mg/kg.

For the evaluation of the lower limitation criteria, an initial value below 0.01 mg/kg is assumed for the background contamination of HCBD and an uncertainty factor of 10 is applied. This results in the value of 0.1 mg/kg.

**Dicofol**

As summarised in [UNEP/POPS/POPRC.12/11/Add.1 2016] and cited from [Vulliet et al. 2014], no dicofol was detected in any of the analysed sediments samples in France. This is in accordance with other publications focusing on France. The highest recorded values were found in the USA in California’s Central Valles with up to 250 ng/g dw, whereas in an earlier publication concentration mean of 36 ng/g dw were reported. Other sediments samples e.g. for the lower regions of Yangtze River detected dicofol levels with a mean of 0.46 ng/g dw in sediments and in China highest concentrations of 18 ng/g dw in surface sediment were reported.

For the evaluation of the limitation criteria, an initial value of 0.3 mg/kg is assumed for the background contamination and an uncertainty factor of 10 is applied resulting in a value of 3 mg/kg.

**PFOA**

According to [ECHA, 2018a], most of the data on existing background contamination in sediments and soils exist for China, but values for sediment for Europe are available. The PFOA contamination in sediments in Europe amounts to 0.079 – 0.157 μg/kg dw and 0.061 – 0.684 μg/kg dw. The highest background contamination is therefore assumed with 0.684 μg/kg dw, detected in the Western Baltic Sea area. A study focusing on Canada identified concentrations ranges of up to 0.0075 μg/kg dw. For background contamination in soils the highest values are reported for China with an average of 0.0501 μg/kg dw (for further details see [ECHA 2018a] Appendix Table A.B.4-8).

For the evaluation of the limitation criteria, an initial value of 0.001 mg/kg is assumed for the background contamination of PFOA and an uncertainty factor of 10 is applied resulting in a value of 0.01 mg/kg.

**PFHxS**

In [UNEP/POPS/POPRC.14/INF/4 2018] measured levels of PFHxS in biota were reported. Highest values in Europe in sediment were recorded in the Czech Republic with 3.8 μg/kg dw however in
an industrial area (not used for estimations). Lower limits were detected in France with an upper range of 0.63 ng/g dw. The highest value worldwide reported for sediment was found in Vietnam between 2013 and 2015 with 18.3 ng/g. For soil, values below the detection limit and up to 99.7 pg/g dw are reported for Europe.

For the evaluation of the limitation criteria, an initial value 0.001 mg/kg is assumed for the background contamination of PFHxS and an uncertainty factor of 10 is applied resulting in a value of 0.01 mg/kg.

**HBCDD**

According to several publications (as cited in [UNEP/POPS/POPRC.6/13/Add.2 2010]) HBCDD levels within the environment are generally increasing in all matrices. According to several authors, sediment core analyses show higher levels of HBCDD in the top layer and lower concentration in the deeper layers in Asia and Europe. First sediment detections in Europe reach back to the mid-1980 in Switzerland according to Kohler et al. (as cited in [UNEP/POPS/POPRC5WG/ASSES-HBCDD/draftRP 2010])

[UNEP/POPS/POPRC.6/INF/25 2010] provides an overview of concentration measures in the ambient environment. For sediments in Europe, highest concentrations were reported from [Morris et al. 2004] in England with a total range of HBCDD from not detected to 1,680 μg/kg dw (as cited in [UNEP/POPS/POPRC.6/INF/25 2010]). The highest reported concentration, however, was detected in the vicinity of a BFR manufacturing site and is therefore not further considered. In unspecified sediments from the river Clyde (Scotland) HBCDD concentrations of up to 187 μg/kg were measured. Reported concentrations of total HBCDD outside of Europe range from below 0.075 to 3.7 ng/g dw in North America.

For the evaluation of the limitation criteria, an initial value below 0.2 mg/kg is assumed for the background contamination of total HBCDD and an uncertainty factor of 10 is applied. This results in the value of 2 mg/kg.

**Results for criterion B**

Table 55 present an overview of the results elaborated for the criterion “background contamination”:

<table>
<thead>
<tr>
<th>Substance</th>
<th>Highest reported background concentration in Europe (if available) [mg/kg]</th>
<th>Background contamination [mg/kg]</th>
<th>Uncertainty factor</th>
<th>Result [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>DecaBDE</td>
<td>1.6</td>
<td>&lt;2</td>
<td>10</td>
<td>20</td>
</tr>
<tr>
<td>SCCP</td>
<td>0.083&lt;sup&gt;59&lt;/sup&gt;</td>
<td>&lt;0.1</td>
<td>10</td>
<td>1.0</td>
</tr>
<tr>
<td>HCBD</td>
<td>0.005&lt;sup&gt;60&lt;/sup&gt;</td>
<td>&lt;0.01</td>
<td>10</td>
<td>0.1</td>
</tr>
</tbody>
</table>

<sup>59</sup> Highest international background contamination reported: 0.484 mg/kg (Japan)

<sup>60</sup> Highest international background contamination reported: 0.55 mg/kg (Canada)
<table>
<thead>
<tr>
<th>Substance</th>
<th>Highest reported background concentration in Europe (if available) [mg/kg]</th>
<th>Background contamination [mg/kg]</th>
<th>Uncertainty factor</th>
<th>Result [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dicofol</td>
<td>0.25&lt;sup&gt;61&lt;/sup&gt;</td>
<td>&lt;0.3</td>
<td>10</td>
<td>3</td>
</tr>
<tr>
<td>PFOA</td>
<td>0.000684&lt;sup&gt;62&lt;/sup&gt;</td>
<td>&lt;0.001</td>
<td>10</td>
<td>0.01</td>
</tr>
<tr>
<td>PFHxS</td>
<td>0.00063</td>
<td>&lt;0.001</td>
<td>10</td>
<td>0.01</td>
</tr>
<tr>
<td>HBCDD</td>
<td>0.187</td>
<td>&lt;0.2</td>
<td>10</td>
<td>2</td>
</tr>
</tbody>
</table>

<sup>61</sup> Highest international background contamination reported: 250 ng/g (USA)

<sup>62</sup> Highest international background contamination reported: 0.0501 μg/kg in soil (China)
6.2.1.3 (DR) Disposal and recovery capacities

Limit values should be established in a way that the (new) required capacities for recovery and disposal are realistically available. The criterion „disposal and recovery capacities“ assesses whether (new) required capacities for recovery and disposal are realistically available for different possible low POP concentration limits. Possible low POP concentration limits might implicate considerable changes in waste treatment, as some wastes will not be permissible for recycling or landfilling anymore but will inevitably have to be directed to energy recovery/thermal disposal facilities. This proceeding guarantees that the POP content is destroyed or irreversibly transformed without undue delay. Based on those scenarios and the elaborated material flows it is subsequently assessed whether sufficient treatment capacities are available for changed waste treatment operations.

According to [Wilts et al. 2017] the mixed municipal waste incineration capacity in the EU was 76,75 Mio tonnes in 2014. In the same year 64,22 Mio tonnes of mixed municipal waste have been incinerated or energetically recovered in the EU [Eurostat 2018], meaning a remaining capacity of 11,98 Mio tonnes (16%) (see also Table 56). Thus, for the derivation of a proposed limit value, an additional increase of up to 5% of the currently required capacities of the relevant thermal waste treatment sites is considered justifiable.

The required capacities in relation to every substance/substance group are calculated via a comparison of the changes in the required thermal treatment capacities resulting from different possible low POP concentration limits with the total treatment capacities available in the EU.

Table 56: Overview of the capacities available in the EU for the thermal treatment of mixed residual waste in 2014 [Wilts et al. 2017; Eurostat 2018]

<table>
<thead>
<tr>
<th>Type of facility</th>
<th>Total incineration capacity [million t/a]</th>
<th>Total nominal load [million t/a]</th>
<th>Remaining capacity [million t/a]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Waste incineration plants for MSW</td>
<td>76,20</td>
<td>64,22</td>
<td>11,98</td>
</tr>
</tbody>
</table>

**DecaBDE**

In order to assess the possible consequences of the different LPCLs on possible additional capacities, the current used treatment for the relevant waste streams are taken into consideration. The following amounts for recovery and disposal and incineration of the considered decaBDE-containing waste can be derived.

- **Incineration (R10)**: 992.79 kt/a (~189 %),
- **Landfill**: 2,057.00 (~36 %),
- **Recycling (including reuse)**: 2,298.50 (~45 %).

It can be assumed, that the decaBDE content is almost completely destroyed during energy recovery and incineration processes. During material recovery processes it cannot be excluded, that decaBDE might be released into recyclates.

In Table 57 the current state of waste treatment is used to compare with the possible state at specific LPCLs. The resulting changes are calculated in kt/a and in percentage.
<table>
<thead>
<tr>
<th>DecaBDE</th>
<th>Current state [kt/a]</th>
<th>State at specific POP concentration limit [kt/a]</th>
<th>Change in [kt/a]</th>
<th>Change in [%]</th>
</tr>
</thead>
<tbody>
<tr>
<td>5,410 mg/kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>995.8</td>
<td>995.8</td>
<td>0.0</td>
<td>+ 0.0</td>
</tr>
<tr>
<td>Landfilling</td>
<td>1,865.3</td>
<td>1,865.3</td>
<td>0.0</td>
<td>+ 0.0</td>
</tr>
<tr>
<td>Recycling</td>
<td>2,298.7</td>
<td>2,298.7</td>
<td>0.0</td>
<td>+ 0.0</td>
</tr>
<tr>
<td>109 mg/kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>995.8</td>
<td>1,660.3</td>
<td>664.6</td>
<td>+ 66.7</td>
</tr>
<tr>
<td>Landfilling</td>
<td>1,865.3</td>
<td>1,601.8</td>
<td>-263.5</td>
<td>-14.1</td>
</tr>
<tr>
<td>Recycling</td>
<td>2,298.7</td>
<td>1,897.6</td>
<td>-401.1</td>
<td>-17.4</td>
</tr>
<tr>
<td>100 mg/kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>995.8</td>
<td>2,609.8</td>
<td>1,614.0</td>
<td>+ 162.1</td>
</tr>
<tr>
<td>Landfilling</td>
<td>1,865.3</td>
<td>1,540.0</td>
<td>-325.3</td>
<td>-17.4</td>
</tr>
<tr>
<td>Recycling</td>
<td>2,298.7</td>
<td>1,010.0</td>
<td>-1,288.7</td>
<td>-56.1</td>
</tr>
<tr>
<td>0.3 mg/kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>995.8</td>
<td>5,159.8</td>
<td>4,164.0</td>
<td>+ 418.2</td>
</tr>
<tr>
<td>Landfilling</td>
<td>1,865.3</td>
<td>0.0</td>
<td>-1,865.3</td>
<td>-100.0</td>
</tr>
<tr>
<td>Recycling</td>
<td>2,298.7</td>
<td>0.0</td>
<td>-2,298.7</td>
<td>-100.0</td>
</tr>
</tbody>
</table>

Currently, waste above a concentration of above 2,000 mg/kg Br is incinerated. The scenarios with 5,410 mg/kg decaBDE is much higher and thus already affected under the current limit. The need for thermal capacity would thus not change.

At a LPCL of 109 mg/kg, changed treatment paths as compared to the current state would have consequences for WEEE plastic and ELV plastic waste. A LPCL of 109 mg/kg would lead to an added need of incineration capacity of 664.6.70 kt/a, which corresponds to 66.7%.

At a LPCL of 100 mg/kg, theoretically 1,614 kt/a of waste, which is recovered or landfilled at present, would have to be incinerated additionally and/or would have to be directed to energy recovery. Those 1,614 kt/a correspond to a 162.1% of the nominal load (waste incineration and hazardous waste incineration plants) available in the EU.

The most significant changes, compared to the current situation, result from a LPCL of less than 0.3 mg/kg. At this limit value additional incineration capacities of 4,164 kt/a would be needed. All concerned waste stream in this scenario would exclusively be directed to incineration and both for landfilling and recycling no capacities whatsoever would be needed.

Table 58 presents the results and shows the percentage changes resulting from the discussed LPCLs.
Table 58: Change as compared with the total capacity of the relevant thermal waste treatment plants in Europe, depending on possible LPCL for decaBDE

<table>
<thead>
<tr>
<th>Possible LPCL [mg/kg]</th>
<th>Additionally required incineration capacities [kt]</th>
<th>Change as compared with existing total capacity of relevant incineration plants in Europe [%]</th>
<th>Remaining capacity at different low POP concentration limits [kt]</th>
</tr>
</thead>
<tbody>
<tr>
<td>5,410</td>
<td>+ 0.0</td>
<td>+ 0.00</td>
<td>11,982</td>
</tr>
<tr>
<td>109</td>
<td>+ 664.6</td>
<td>+ 0.87</td>
<td>11,317</td>
</tr>
<tr>
<td>100</td>
<td>+ 1,614.0</td>
<td>+ 2.12</td>
<td>10,368</td>
</tr>
<tr>
<td>0.3</td>
<td>+ 4,164.0</td>
<td>+ 5.46</td>
<td>7,818</td>
</tr>
</tbody>
</table>

For the derivation of a proposed limit value, an additional increase of up to 5% of the currently required capacities of the relevant thermal waste treatment sites is considered justifiable. For decaBDE this means, that all possible LPCLs, with the exception of 0.3 mg/kg would result in an additional increase of less than 5% and therefore all LPCL above 0.3 mg/kg are theoretically possible. Thus, in regard to criterion DR for decaBDE a possible LCPL of above 0.3 mg/kg is derived because of the precautionary principle (it is the lowest LPCL).

**SCCPs**

The waste streams relevant in the EU are SCCP-containing rubber wastes from conveyor belts for the use in underground mining and SCCP-containing sealants and adhesives from construction and demolition wastes. Besides, several other waste streams (mostly resulting from imported articles) are of relevance and are regularly detected during enforcement activities but cannot be quantified.

Table 59: Changes in treatment operations at specific POP limit values (SCCP)

<table>
<thead>
<tr>
<th>SCCP</th>
<th>Current state [kt/a]</th>
<th>State at specific POP concentration limit [kt/a]</th>
<th>Change in [kt]</th>
<th>Change in [%]</th>
</tr>
</thead>
<tbody>
<tr>
<td>420 mg/kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>2,778</td>
<td>4,208</td>
<td>+ 1,429.9</td>
<td>+ 52</td>
</tr>
<tr>
<td>Landfilling</td>
<td>32,446</td>
<td>32,426</td>
<td>- 20.0</td>
<td>- 0.1</td>
</tr>
<tr>
<td>Recycling</td>
<td>244,315</td>
<td>242,905</td>
<td>- 1,409.9</td>
<td>- 0.6</td>
</tr>
<tr>
<td>1 mg/kg</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>2,778</td>
<td>279,540</td>
<td>+ 276,761.5</td>
<td>+ 9.961</td>
</tr>
<tr>
<td>Landfilling</td>
<td>32,446</td>
<td>0</td>
<td>- 32,446.1</td>
<td>- 100</td>
</tr>
<tr>
<td>Recycling</td>
<td>244,315</td>
<td>0</td>
<td>- 244,315.4</td>
<td>- 100</td>
</tr>
</tbody>
</table>

Assuming that the SCCP-containing rubber waste is disposed of/recovered together with other rubber waste, the SCCP concentration of the total waste stream decreases to about 420 mg/kg. Approximately 54% (1,410 kt/a) of this waste is currently directed to recycling (production of granulates for further use), about 1,200 kt/a (~46%) to incineration plants and a comparably low waste amount is currently landfilled (~20 kt/a; <1%).

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At a low POP concentration limit of 420 mg/kg, theoretically ~1,410 kt/a of rubber waste, which is recycled, would have to be incinerated additionally (see Table 59). The same applies for 20 kt/a, which go to landfill at present and would have to be incinerated instead. Those 1,430 kt/a correspond to an additional necessary capacity of relevant incineration plants in Europe of about 1.9 % and should not pose any capacity problems.

At present 0.57 % (1,578 kt/a) of construction and demolition waste (including 66.65 % of SCCP-containing sealants and adhesives) are incinerated, 11.71 % (32,426 kt/a) are landfilled and 87.72 % (242,905 kt) are recycled. The theoretically concerned waste amount, which would have to be treated alternatively at a possible concentration limit of 1 mg/kg, would be considerable >276,761 kt/a (including waste from the C&D waste fraction and rubber fraction) (see Table 59). Thus, the criterion DR for SCCP should be set above 1 mg/kg.

**HCBD**

Due to the information presented in chapter 4.3, it can be assumed that HCBD does not occur in relevant amounts in products, waste and recyclates in the EU. Thus, possible LPCL will have no relevant effects on disposal and recovery capacities. Therefore, the LPCL can be set arbitrarily low regarding these capacities.

**Dicofol**

Dicofol does not occur in products, waste and recyclates in the EU. Thus, possible LPCL will have no relevant effects on disposal and recovery capacities.

**PFOA**

Due to the limited quantitative information on PFOA production and use as well as missing information related to waste streams and recyclates containing or contaminated with those substances, concerned waste amounts for a possible LPCL cannot be derived. Consequently, it is also not possible to estimate required changes in treatment operations and derive criterion DR.

**PFHxS**

Due to the missing quantitative information on PFHxS production and use amounts in different application as well as missing information on wastes and recyclates containing or contaminated with those substances, concerned waste amounts for a possible LPCL cannot be derived. Consequently, it is also not possible to estimate required changes in treatment operations and derive criterion DR.

**HBCDD**

To assess the potential consequences of different low POP concentration limits on existing and possibly required additional capacities, the currently used treatment operations for the identified application fields are taken into consideration. The following amounts for recovery, disposal and incineration of the considered HBCDD-containing waste can be derived.

- **Incineration (D10 or R1)**: 226.9 kt/a (~44%),
- **Landfill**: 165.2 kt/a (~32%),
- **Recycling (including mechanical and feedstock recycling)**: 127.7 kt/a (~25%).

In Table 60 the current state of the waste treatment is used as a starting point and compared with the state at specific LPCLs, with the resulting changes being calculated (in kt and as a percentage, referring to the current total amounts for thermal treatment, landfiling and recycling).
Currently, the concentration limit for HBCDD lies at 1,000 mg/kg. The scenarios with 7,000 mg/kg and 15,000 mg/kg are much higher and thus already affected under the current limit. Nevertheless, the mass flows show that the waste of XPS and EPS is not treated in accordance with the regulation. Treating XPS and EPS waste from C&D in accordance with the POP Regulation, additional 39.7 kt of EPS and XPS waste would have to be incinerated. The need for thermal capacity would increase by a total of 17.5 % and recycling and landfilling would decrease accordingly.

The most significant changes, compared to the current situation, would result from a possible low POP concentration limit of less than 10 mg/kg. At this limit value, waste of all relevant application areas would have to be exclusively directed to thermal treatment. Therefore, the required thermal capacity would rise to 292.9 kt/a. Both, for landfilling and recycling no capacities whatsoever would be needed at this limit value.

Table 61 presents the results and shows the percentage changes resulting for each possible low POP concentration limit, based on the total capacities.

### Table 60: Overview of quantitative changes in recovery and treatment operations, depending on possible LPCLs

<table>
<thead>
<tr>
<th>HBCDD</th>
<th>Current state [kt/a]</th>
<th>State at specific POP concentration limit [kt/a]</th>
<th>Change in [kt/a]</th>
<th>Change in [%]</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>15,000 mg/kg</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>226.9</td>
<td>236.6</td>
<td>+ 9.7</td>
<td>+ 4.3</td>
</tr>
<tr>
<td>Landfilling</td>
<td>165.2</td>
<td>155.5</td>
<td>- 9.7</td>
<td>- 5.9</td>
</tr>
<tr>
<td>Recycling</td>
<td>127.7</td>
<td>127.7</td>
<td>+ 0.0</td>
<td>+ 0.0</td>
</tr>
<tr>
<td><strong>7,000 mg/kg</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>226.9</td>
<td>266.6</td>
<td>+ 39.7</td>
<td>+ 17.5</td>
</tr>
<tr>
<td>Landfilling</td>
<td>165.2</td>
<td>126.8</td>
<td>- 38.4</td>
<td>- 23.2</td>
</tr>
<tr>
<td>Recycling</td>
<td>127.7</td>
<td>126.4</td>
<td>- 1.3</td>
<td>- 1.0</td>
</tr>
<tr>
<td><strong>10 mg/kg</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>226.9</td>
<td>519.8</td>
<td>+ 292.9</td>
<td>+ 129.1</td>
</tr>
<tr>
<td>Landfilling</td>
<td>165.2</td>
<td>0.0</td>
<td>- 165.2</td>
<td>- 100.0</td>
</tr>
<tr>
<td>Recycling</td>
<td>127.7</td>
<td>0.0</td>
<td>- 127.7</td>
<td>- 100.0</td>
</tr>
</tbody>
</table>

Currently, the concentration limit for HBCDD lies at 1,000 mg/kg. The scenarios with 7,000 mg/kg and 15,000 mg/kg are much higher and thus already affected under the current limit. Nevertheless, the mass flows show that the waste of XPS and EPS is not treated in accordance with the regulation. Treating XPS and EPS waste from C&D in accordance with the POP Regulation, additional 39.7 kt of EPS and XPS waste would have to be incinerated. The need for thermal capacity would increase by a total of 17.5 % and recycling and landfilling would decrease accordingly.

The most significant changes, compared to the current situation, would result from a possible low POP concentration limit of less than 10 mg/kg. At this limit value, waste of all relevant application areas would have to be exclusively directed to thermal treatment. Therefore, the required thermal capacity would rise to 292.9 kt/a. Both, for landfilling and recycling no capacities whatsoever would be needed at this limit value.

Table 61 presents the results and shows the percentage changes resulting for each possible low POP concentration limit, based on the total capacities.

### Table 61: Change as compared with the total capacity of the relevant thermal waste treatment plants in Europe, depending on possible LPCL for HBCDD

<table>
<thead>
<tr>
<th>Possible low POP concentration limit [mg/kg]</th>
<th>Additionally required incineration capacities [kt]</th>
<th>Change as compared with existing total capacity of relevant incineration plants in Europe [%]</th>
<th>Remaining capacity at different low POP concentration limits [kt]</th>
</tr>
</thead>
<tbody>
<tr>
<td>15,000</td>
<td>+ 9.7</td>
<td>+ 0.01</td>
<td>11,972.3</td>
</tr>
<tr>
<td>7,000</td>
<td>+ 39.7</td>
<td>+ 0.05</td>
<td>11,942.3</td>
</tr>
</tbody>
</table>
As an additional increase of up to 5% of the currently required thermal treatment capacities is considered feasible, no scenario would cause a shortage. In regard to criterion (DR), the LPCL could be set at or above 10 mg/kg.

Results for criterion DR

Table 62 gives an overview on the evaluation of the disposal and recovery operations.

Table 62: Overview of the result for the limitation criterion DR

<table>
<thead>
<tr>
<th>Substance/ Substance group</th>
<th>Result criterion DR [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
<td>0.3</td>
</tr>
<tr>
<td>SCCP</td>
<td>1</td>
</tr>
<tr>
<td>HCBD</td>
<td>-</td>
</tr>
<tr>
<td>Dicofol</td>
<td>-</td>
</tr>
<tr>
<td>PFOA</td>
<td>-</td>
</tr>
<tr>
<td>PFHxS</td>
<td>-</td>
</tr>
<tr>
<td>HBCDD</td>
<td>10</td>
</tr>
</tbody>
</table>


6.2.1.4 **(E) Economic feasibility**

Criterion E assesses the economic impacts of changed treatment costs which accompany possible low POP concentration limits. The analysis of possible disposal operations can indicate how costs for the future disposal and recovery operations will change (e.g., changed costs when a distinct type of waste can no longer be landfilled but has to be incinerated in the future), it does however not take into account for potential loss of employment or of capital in recycling activities.

In case the POP content of waste exceeds the LPCL, specific disposal and recovery operations are permitted (D9, D10, R1, R4, permanent underground disposal).

Waste with a POP content exceeding the proposed limit value causes the following possible changes in disposal and recovery operations:

- at present: deposition in future: incineration (D10 or R1)
- at present: recycling/re-use in future: incineration (D10 or R1)
- at present: deposition in future: underground disposal (e.g. waste that cannot be incinerated as ashes)
- at present: recycling/re-use in future: underground disposal

Theoretically POP-containing waste can also be deposited after stabilisation\(^{63}\). However, stabilisation is not relevant for the waste streams analysed in this research project and will not be taken into account in the economic impact assessment.

The resulting changes in costs for possible modifications of disposal and recovery operations have to be taken into account as well. An assessment of those economic consequences is carried out calculating the average costs for modified disposal and recovery operations.

The costs for waste incineration (taxes + gate fees) in the EU range between € 46 in the CZ and € 174 in DE per tonne of waste [Bio 2012]. For the economic impact assessment, an average of € 102 is calculated. At this point a differentiation between the incineration in hazardous waste incineration plants and municipal waste incineration plants or between hazardous and non-hazardous waste is not necessary in the scope of this project. According to the Directive 2000/76/EC on the incineration of waste or the Industrial Emissions Directive (IED) 2010/75/EU on industrial emissions, there is no legal obligation to incinerate hazardous waste exclusively in hazardous waste incineration plants. The plants rather have to hold specific permits for the waste categories incinerated and comply with certain provisions for hazardous waste.

The total costs for landfilling one tonne of waste in the EU ranges between 3.7 € in RO and 150 € in LU [EEA 2013]. For the economic assessment average costs of € 44 are used for the calculations.

According to information from German operators of underground disposal sites, costs for underground disposal lie between € 260 and € 900 per tonne, irrespective of the deposited waste category [EC 2010]. The upper end of the price refers to waste which for example requires specific storage (e.g. separate storage or storage in an isolated area; see [EC 2010]). For the economic

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\(^{63}\) Stabilisation is a chemical/physical treatment (D9) and is therefore permissible for POP-containing waste in accordance with Annex V, Part 1 of the POP Regulation. Decision 2000/532/EC states: „Stabilisation processes change the dangerousness of the constituents in the waste and thus transform hazardous waste into non-hazardous waste.”
assessment average costs of € 260 per tonne are used for the calculations, since no European data is available.

General recycling costs for the EU are not available. [BiPRO 2011] includes data of recycling/re-use costs in Germany and UK and concludes that the option recycling/re-use costs an average € 40 per tonne of waste. Although recycling/re-use costs vary widely across the different EU MS and must have changed over time since 2011, this data will be used for further calculations, due to poor data availability.

Table 63 shows both the calculated costs for the different disposal and recovery operations as well as the cost changes resulting in case the possible LPCL was exceeded.

**Table 63: Changes in costs for relevant disposal and recovery operations**

<table>
<thead>
<tr>
<th>Current treatment</th>
<th>Future treatment in case LPCL is exceeded</th>
<th>Cost changes per tonne of waste</th>
</tr>
</thead>
<tbody>
<tr>
<td>Deposit (€ 44)</td>
<td>Incineration: D10 or R1 (€ 102)</td>
<td>+ € 58</td>
</tr>
<tr>
<td>Recycling/Re-use (€ 40)</td>
<td>Incineration: D10 or R1 (€ 102)</td>
<td>+ € 62</td>
</tr>
<tr>
<td>Deposit (€ 44)</td>
<td>Underground disposal (€ 260)</td>
<td>+ € 216</td>
</tr>
<tr>
<td>Recycling/Re-use (€ 40)</td>
<td>Underground disposal (€ 260)</td>
<td>+ € 220</td>
</tr>
</tbody>
</table>

The cost changes per tonne of waste shown in Table 63 serve as a basis for the assessment of the economic impacts of possible LPCL. Only those LPCL values are considered, which are below and the additional costs of 50 million € per year.

**DecaBDE**

Currently, fraction which contain Br above 2,000 mg/kg are incinerated in the high Br fraction. Consequently, the economic assessment of the limits 5,410 mg/kg show the economic effects resulting from compliance with the present legislation.

Considering the concentration limit of 5,410 mg/kg, no additional treatment costs would arise as not waste would have to be treated alternatively.

A possible limit value above 109 mg/kg, theoretically would result in an increase of approximately 40 million Euros total per year. This would be ascribable to changed treatment operations for ELV plastic waste and the high Br fraction of WEEE plastic. In total, the recycling of around 401.08 kt per year would have to be renounced.

At a possible LPCL of 100 mg/kg, theoretically additional treatment costs of 98.45 million Euros per year would arise in total. The increase in costs of a further 58.43 million Euros per year as compared with a possible LPCL of 109 mg/kg results from the additional thermal treatment of low Br WEEE plastic waste.

A possible LPCL of 0.3 mg/kg would theoretically result in an increase of 249.88 million Euros total per year as all previous waste categories would have to be directed to incineration.

The results which are based on the current treatment costs, also considered total costs at possible low POP concentration limits and additional costs arising (changes as compared with the current state) were calculated and are shown in Table 64 below:
Table 64: Economic impacts at possible LPCLs (decaBDE)

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>5,410 mg/kg</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>101.37</td>
<td>101.37</td>
<td>+ 0.00</td>
<td>+ 0.00</td>
</tr>
<tr>
<td>Landfilling</td>
<td>82.07</td>
<td>82.07</td>
<td>+ 0.00</td>
<td>+ 0.00</td>
</tr>
<tr>
<td>Recycling</td>
<td>91.95</td>
<td>91.95</td>
<td>+ 0.00</td>
<td>+ 0.00</td>
</tr>
<tr>
<td></td>
<td>109 mg/kg</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>101.37</td>
<td>169.02</td>
<td>+ 67.65</td>
<td>+ 40.02</td>
</tr>
<tr>
<td>Landfilling</td>
<td>82.07</td>
<td>70.48</td>
<td>- 11.59</td>
<td></td>
</tr>
<tr>
<td>Recycling</td>
<td>91.95</td>
<td>75.90</td>
<td>- 16.04</td>
<td></td>
</tr>
<tr>
<td></td>
<td>100 mg/kg</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>101.37</td>
<td>265.68</td>
<td>+ 164.31</td>
<td>+ 98.45</td>
</tr>
<tr>
<td>Landfilling</td>
<td>82.07</td>
<td>67.76</td>
<td>- 14.31</td>
<td></td>
</tr>
<tr>
<td>Recycling</td>
<td>91.95</td>
<td>40.40</td>
<td>- 51.55</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.3 mg/kg</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Incineration (D10 or R1)</td>
<td>101.37</td>
<td>525.27</td>
<td>+ 423.90</td>
<td>+ 249.88</td>
</tr>
<tr>
<td>Landfilling</td>
<td>82.07</td>
<td>0.00</td>
<td>- 82.07</td>
<td></td>
</tr>
<tr>
<td>Recycling</td>
<td>91.95</td>
<td>0.00</td>
<td>- 91.95</td>
<td></td>
</tr>
</tbody>
</table>

For the assessment of criterion E, only those LPLC values are considered, which are below the additional costs of 50 million € per year. For decaBDE this means, that only a LPCL above 100 mg/kg can be considered. Considering a margin of safety an LPCL at or above 200 is derived for the criterion E.

**SCCPs**

At a limit values above 420 mg/kg, theoretically no waste would have to be treated alternatively, which would avoid an increase in costs for waste management. At a low POP concentration limit of 420 mg/kg, theoretically ~1,410 kt/a of rubber waste, which is recycled at present, would have to be incinerated additionally. The same applies for 20 kt/a, which go to landfill at present and would have to be incinerated instead. Those additional 1,430 kt/a would result in an increase in costs amounting to more than 80 million Euros per year. Besides the additional costs the limit value would also entail the fact that ~1,410 kt/a of the rubber waste could not be recycled. At a possible low concentration limit of 1 mg/kg, the theoretically concerned waste amount, which would have to be treated alternatively (incineration is not an option for this kind of waste), would be significant >276,761 kt/a (additional waste from the C&D waste fraction and rubber fraction) and would in any case require an additional cost far above 50 million € per year.

According to the methodology applied, neither an LPCL at 420 mg/kg nor an LPCL at 1 mg/kg would be economically feasible. Consequently, the limit value should be set above 420 mg/kg (e.g. 500 mg/kg or 1,000 mg/kg).
**HCBD**

The data basis indicates that HCBD-containing waste is not relevant in the EU. Therefore, LPCLs have no relevant economic impacts and can be set arbitrarily low.

**Dicofol**

Dicofol does not occur in products, waste and recyclates in the EU. Thus, possible LPCL will have no relevant effects on economic feasibility criteria E.

**PFOA**

Due to the limited quantitative information on PFOA production and use as well as missing information related to waste streams and recyclates containing or contaminated with those substances, concerned waste amounts for a possible LPCL cannot be derived. Consequently, it is also not possible to estimate the economic feasibility and derive criterion E.

**PFHxS**

Due to the missing quantitative information on PFHxS production and use amounts in different application as well as missing information on wastes and recyclates containing or contaminated with those substances, concerned waste amounts for a possible LPCL cannot be derived. Consequently, it is also not possible to estimate the economic feasibility and derive criterion E.

**HBCDD**

In order to assess the changes in treatment costs for HBCDD-containing waste, the current waste treatment costs are shown in Table 65.

The current LPCL for HBCDD lies at 1,000 mg/kg. Consequently, the economic assessment of the limits 15,000 mg/kg and 7,000 mg/kg show the economic effects resulting from compliance with the present LPCL under the POP Regulation.

Considering the concentration limit of 15,000 mg/kg, additional treatment costs of about 560,000 euros per year would arise in total. Setting off additional thermal treatment costs of 990,000 million Euros per year against the declining landfilling costs of 430,000 Euros per year.

Considering the concentration limit of 7,000 mg/kg, theoretically additional treatment costs of 2.30 million Euros per year would arise in total. The increase in costs by another 1.7 million Euros per year as compared with a possible LPCL of 15,000 mg/kg results from the additional thermal treatment of EPS in C&D waste.

A possible LPCL of 10 mg/kg would theoretically result in an increase of 17.4 million Euros total per year. This would be ascribable to changed treatment operations for EPS packaging waste as well as XPS and EPS C&D waste. In total, the recycling of 127.7 kt of EPS and XPS would have to be renounced.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Incineration (D10 or R1)</td>
<td>23.10</td>
<td>24.08</td>
<td>+ 0.99</td>
<td>+ 0.56</td>
</tr>
<tr>
<td>Landfilling</td>
<td>7.27</td>
<td>6.84</td>
<td>- 0.43</td>
<td></td>
</tr>
<tr>
<td>-------------------------</td>
<td>-----------------------------</td>
<td>-------------------------------------</td>
<td>-----------------------</td>
<td>---------------------------</td>
</tr>
<tr>
<td>Recycling</td>
<td>5.11</td>
<td>5.11</td>
<td>+ 0.00</td>
<td></td>
</tr>
</tbody>
</table>

7,000 mg/kg

| Incineration (D10 or R1) | 23.10                      | 27.14                                | + 4.04                | + 2.3                     |
| Landfilling             | 7.27                        | 5.58                                 | - 1.69                |                           |
| Recycling               | 5.11                        | 5.06                                 | - 0.05                |                           |

10 mg/kg

| Incineration (D10 or R1) | 23.10                      | 52.91                                | + 29.82               | + 17.4                    |
| Landfilling             | 7.27                        | 0.00                                 | - 7.27                |                           |
| Recycling               | 5.11                        | 0.00                                 | - 5.11                |                           |

Since additional costs of 50 million € per year would still be acceptable, a LPCL of 10 mg/kg could be taken into account under pure economic terms. However, this would mean that EPS packaging waste (126 kt) will no longer be recycled (the 1.3 kt of EPS waste from C&D should actually not be recycled today anymore). For this reason, the LPCL should be set at or above 10 mg/kg and the already specified 1,000 mg/kg.

**Results for criterion E**

Table 66 sums up the evaluation of the economic impacts associated with different possible LPCLs.

**Table 66: Overview of the results for criterion E**

<table>
<thead>
<tr>
<th>Substance/ Substance group</th>
<th>Result criterion E [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
<td>200</td>
</tr>
<tr>
<td>SCCP</td>
<td>&gt;420</td>
</tr>
<tr>
<td>HCBD</td>
<td>-</td>
</tr>
<tr>
<td>Dicofol</td>
<td>-</td>
</tr>
<tr>
<td>PFOA</td>
<td>-</td>
</tr>
<tr>
<td>PFHxS</td>
<td>-</td>
</tr>
<tr>
<td>HBCDD</td>
<td>≥10</td>
</tr>
</tbody>
</table>
6.2.2 EVALUATION OF THE UPPER LIMITATION CRITERIA OF THE LPCL

6.2.2.1 (LV) Limit values

Existing limit values on a national and international scale were taken into account as upper limitation criterion, in order to avoid possible conflicts between existing limit values and proposed LPCL. The evaluation was especially focused on the question whether and how existing limit values delimit possible LPCLs.

DecaBDE

Under Directive 2011/65/EU (also known as the RoHS Directive), decaBDE is considered in the PBDEs with a maximum concentration value tolerated of 0.1% by weight in homogenous materials (for the PBDE as a substance group).

POP-PBDEs in articles produced from recycled materials in Annex I of the POPs Regulation, i.e. the level of 1,000 mg/kg (0.1% by weight) of POP-PBDEs allowed in articles produced partially or fully from recycled materials now [EC POP CA 2007].

Conclusion

DecaBDE LPCL should not be above those values that are in place today for the recycled materials i.e. current levels of RoHS with 1,000 mg/kg in homogeneous material.

SCCPs

According to Annex I of the POP Regulation, the following specific exemptions on intermediate use or other specifications are defined for SCCPs:

1. "By way of derogation, the production, placing on the market and use of substances or preparations containing SCCPs in concentrations lower than 1 % by weight or articles containing SCCPs in concentrations lower than 0.15 % by weight shall be allowed.

2. Use shall be allowed in respect of:
   a. conveyor belts in the mining industry and dam sealants containing SCCPs already in use before or on 4 December 2015; and
   b. articles containing SCCPs other than those referred to in (a) already in use before or on 10 July 2012.

3. Article 4(2) third and fourth subparagraphs shall apply to the articles referred to in point 2 above.

It is further stipulated that articles containing SCCP in concentrations lower than 0.15 % by weight are allowed to be placed on the market and used, as this is the amount of SCCP that may be present as an impurity in an article produced with MCCPs.

Under the REACH Regulation (EC) No 1907/2006, SCCP have been included to the candidate list of substances of very high concern in 2008. Producers and importers must notify ECHA about substances if they are present in a concentration of 0.1% or the substance is present in those articles in quantities totalling over a tonne per producer or importer per year.

Further, chlorinated paraffins C10-13 are classified, among others, as aquatic chronic 1 (H410) under the CLP Regulation, which means that any waste containing ≥ 2,500 mg/kg would be classified as hazardous. This would imply additional requirements regarding safe management of...
hazardous waste and thus require a stricter control regime (including additional labelling and identification documents, record keeping, monitoring and control obligations, etc.).

Conclusion:
The LPCL for SCCPs should not be above 10,000 mg/kg (1 %) as specified in Anne I of the POP Regulation.

**HCBD**
HCBD is subject to various national and international regulations (see section 4.3.3).

Under the Stockholm Convention HCBD is listed in Annex A (elimination) with no exemptions and C (Unintentional production) with a concentration limit of 1,000 mg/kg. Thus, concentration limits for Annex III of the EU POP Regulation need to be proposed. Additionally, HCBD is listed in Annex IV of the EU POP Regulation with a LPCL of 100 mg/kg and a MPCL of 1,000 mg/kg.

Conclusion
The LPCL should not exceed 100 mg/kg, unless there are relevant reasons for that.

**Dicofol**
Dicofol is subject to various national and international regulations (see section 4.4.3). Product or waste related limit values for dicofol are not defined.

**PFOA**
PFOA is subject to regulatory measures under REACH and CLP. It was included in the REACH candidate list and will be restricted from 2020 due to an entry in REACH Annex XVII.

Annex XVII specified the following:
1) PFOA 'shall not be manufactured, or placed on the market as substances on their own from 4 July 2020’
2) PFOA 'shall not, from 4 July 2020, be used in the production of, or placed on the market in:
   a) another substance, as a constituent;
   b) a mixture;
   c) an article, in a concentration equal to or above 0.025 mg/kg of PFOA including its salts or 1 mg/kg of one or a combination of PFOA-related substances.

Section 4.5.3 further stipulated important exemptions.
At its thirteenth meeting in October 2017, the POPRC adopted the RME on PFOA, its salts and PFOA-related compounds and recommended to the COP that it consider listing the chemicals in Annex A or B to the Convention with specific exemptions specified in decision POPRC-13/2. However, the Committee was unable to reach conclusions on whether exemptions may be needed for specific uses [UNEP/POPS/POPRC.13/7/Add.2 2017; UNEP/POPS/POPRC.13/7/Add.2 2017].

Conclusion
PFOA, its salts and related compounds are subject to various national and international regulations (see section 4.5.3). Currently, there is no limit value which enables to derive an upper limitation criteria LV.
**PFHxS**

PFHxS was added to the REACH Candidate List and is further currently reviewed under the Stockholm Convention. Relevant limit values for PFHxS are not defined.

**Conclusion**

Currently, there is no limit value which enables to derive an upper limitation criteria LV.

**HBCDD**

HBCDD is listed in Annex A of the Stockholm Convention. HBCDD was added to Annex IV of the POP Regulation by Regulation (EU) 2016/460 with a concentration limit of 1,000 mg/kg, subject to review by the Commission by 20.4.2019. Therefore, waste containing HBCDD with a concentration above 1,000 mg/kg shall be disposed of in accordance with Article 7 par. 2 of the POP regulation.

**Conclusion**

The LPCL should not exceed the existing limit value in accordance with Annex IV by Regulation (EU) 2016/460 set at 1,000 mg/kg.

**Results for criterion LV**

Table 67 presents a summary of the results elaborated for the criterion LV.

**Table 67: Overview of the results for the limitation criterion LV**

<table>
<thead>
<tr>
<th>Substance/Substance group</th>
<th>Existing limit value</th>
<th>Result criterion LV [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
<td>RoHS-Directive 2011/65 / EU: 1,000 mg/kg CENELEC: 2,000 mg/kg</td>
<td>1,000 mg/kg</td>
</tr>
<tr>
<td>SCCP</td>
<td>Annex I to the EU POP Regulation specifies a concentration threshold of 1% by weight.</td>
<td>10,000 mg/kg</td>
</tr>
<tr>
<td>HCBD</td>
<td>Annex IV of EU POP Regulation specifies a LPCL of 100 mg/kg</td>
<td>100 mg/kg</td>
</tr>
<tr>
<td>Dicofol</td>
<td>No existing limit value</td>
<td>-</td>
</tr>
<tr>
<td>PFOA</td>
<td>Limit values in place but no limit value to derive upper limitation criteria</td>
<td>-</td>
</tr>
<tr>
<td>PFHxS</td>
<td>No existing limit value</td>
<td></td>
</tr>
<tr>
<td>HBCDD</td>
<td>Annex IV of EU POP Regulation specifies LPCL of 1,000 mg/kg</td>
<td>1,000 mg/kg</td>
</tr>
</tbody>
</table>

6.2.2.2 (R) Possible adverse effects on human health and the environment

Limit values should be established in a way that adverse effects on human health and the environment are avoided and human health and the environment are protected from persistent organic pollutants as far as possible. In order to assess possible effects, (Predicted) no effect concentration ((P)NEC) values, environmental quality standards (EQS) were investigated. Table 68 gives an overview of the available (P)NEC values. As a starting point the ECHA REACH registration dossiers for the individual substances were searched. As highlighted above also EQS limit values are considered. According to article 16 of the Water Framework Directive (2000/60/EC) (WFD), these limits should establish "Strategies against pollution of water", outlining the steps to be
taken. This framework regulation was amended in 2008 by directive 2008/105, that laid down specific EQS in accordance with the provisions and objectives of Directive 2000/60/EC. In 2013, Directive 2013/39/EU of the European Parliament was published, that is amending Directives 2000/60/EC and 2008/105/EC in terms of listing new priority substances in the field of water policy. There are 45 substances or group of substances listed. It is important to note, that for most of the (group of) substances, there are different EQS: annual average value (AA-EQS), maximum allowable concentration (MAC-EQS). Furthermore, the EQS are divided in values for inland surface water and other surface water, whereby inland surface water encompasses rivers and lakes and related artificial or heavily modified water bodies. The biota EQS relates to fish and is considered in Table 68 as food chain.

The available (P)NEC values can serve as information for the derivation of a quantitative limitation criterion. For the derivation of the limitation criterion it has to be taken into consideration, whether a certain waste can be released directly into the environment (sewage sludge for instance) or if and how it can indirectly cause an exposure of humans or the environment to POPs. In case the POP-containing waste can be released directly into the environment, the (P)NEC can be used directly as upper limitation for the relevant compartment.

In other cases, the risk is reduced in accordance with the specific frame conditions. In order to analyse the frame conditions, the pre-treatment, treatment and finally the disposal/recovery of the waste are discussed. In the analysis especially, the following questions were taken into account:

- Are disposal and recovery operations (D9, D10, R1, R4), permitted according to Annex V, part 1, appropriate to destroy or irreversibly transform the relevant substances/substance groups?
- Can the corresponding processes lead to an unintended generation of new POPs?
- Can disposal or recovery cause a risk to human health or the environment (i.e. a relevant exposure of humans or the environment)? The emission of significant amounts of POPs into the environment is generally considered a risk.
<table>
<thead>
<tr>
<th>Substance/ Substance group</th>
<th>Freshwater</th>
<th>Seawater</th>
<th>Sediment Freshwater</th>
<th>Sediment Seawater</th>
<th>Soil</th>
<th>Food chain</th>
<th>Microbiol. activity</th>
</tr>
</thead>
<tbody>
<tr>
<td>decabioE [ECHA 2018d] (Registration dossier)</td>
<td>no data: aquatic toxicity unlikely</td>
<td>no data: aquatic toxicity unlikely</td>
<td>≥384 mg/kg dry sediment</td>
<td>10 mg/kg sediment dw</td>
<td>≥98 mg/kg dry soil</td>
<td>222 mg/kg</td>
<td>1.5 mg/L (PNEC STP)</td>
</tr>
<tr>
<td>SCP [ECHA 2018e] (Registration dossier)</td>
<td>0.5 µg/L</td>
<td>0.15 µg/L</td>
<td>6.5 mg/kg sediment dw</td>
<td>1.3 mg/kg sediment dw</td>
<td>5.95 mg/kg (wet)</td>
<td>5.5 mg/kg food</td>
<td>6 mg/L</td>
</tr>
<tr>
<td>HCBD [Euro Chlor 2002] and [BMU 2013]</td>
<td>0.6 µg/L (MACEQS)</td>
<td>0.13 µg/L</td>
<td>No data</td>
<td>24.4 µg/kg (dry)</td>
<td>No data</td>
<td>55 µg/kg (biota EQS)</td>
<td>No data</td>
</tr>
<tr>
<td>Dicofol [Dicofol EQS dossier 2011]</td>
<td>0.01 µg/L [International Office for Water and INERIS 2009] 0.000032 µg/L</td>
<td>0.0237 µg/l</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
<td>33 µg/kg (biota EQS)</td>
<td>No data</td>
</tr>
<tr>
<td>PFOA [RIVM EQS 2017]</td>
<td>0.1 µg/L (AA-EQS)</td>
<td>0.02 µg/L (AA-EQS)</td>
<td>Insufficient data</td>
<td>Insufficient data</td>
<td>No Data</td>
<td>1.53 µg/kg (biota EQS)</td>
<td>No data</td>
</tr>
<tr>
<td>PFHxS [Valsecchi 2015; LAWA 2017]</td>
<td>0.1 µg/L (GFS) [LAWA 2017]</td>
<td>No data</td>
<td>According to [Valsecchi 2015]: For compounds with carbon chain &lt;C6 it is not possible to derive a QS for sediments because do not accumulate in sediment and data</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
</tr>
<tr>
<td>Substance/Substance group</td>
<td>Freshwater</td>
<td>Seawater</td>
<td>Sediment Freshwater</td>
<td>Sediment Seawater</td>
<td>Soil</td>
<td>Food chain</td>
<td>Microbiol. activity</td>
</tr>
<tr>
<td>---------------------------</td>
<td>-----------</td>
<td>----------</td>
<td>---------------------</td>
<td>-------------------</td>
<td>------</td>
<td>------------</td>
<td>-------------------</td>
</tr>
<tr>
<td>HBCDD [ECHA 2018c] (Registration dossier)</td>
<td>0.31 µg/L</td>
<td>0.062 µg/L</td>
<td>10 mg/kg sediment dw</td>
<td>1 mg/kg sediment dw</td>
<td>5.9 mg kg⁻¹ [Environment Agency 2009]</td>
<td>222.22 mg/kg food</td>
<td>0.15 mg/L</td>
</tr>
<tr>
<td></td>
<td>data are lacking</td>
<td>data are lacking</td>
<td>167 µg/kg ww (biota EQS)</td>
<td>1 mg/kg sediment dw</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
(P)NEC values are most relevant for sediments or soils in order to evaluate the risks posed by different wastes. Wastes containing the relevant substances/substance groups above this concentration should not enter the environment (e.g. be applied on agricultural soil as sewage sludge).

Table 69 contains a comparison of (P)NEC values for soils and sediments for the relevant substances/substance groups and typical contents in sewage sludge.

Table 69: Comparison of (P)NEC values for soils and sediments for the relevant substances/substance groups and typical contents in sewage sludge

<table>
<thead>
<tr>
<th>Substance/Substance groups</th>
<th>(P)NEC values for soils and sediments [mg/kg]</th>
<th>Typical contents in sewage sludge [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
<td>Soil: ≥98 mg/kg</td>
<td>248 ug/kg dw</td>
</tr>
<tr>
<td></td>
<td>Sediment: ≥384 mg/kg dry</td>
<td>UNEP/POPS/POPRC9 SUBM BDE</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Denmark 1 2014 as cited</td>
</tr>
<tr>
<td></td>
<td></td>
<td>[Christensen et al. 2003]</td>
</tr>
<tr>
<td>SCCP [Bogdal et al. 2015]</td>
<td>Soil: 5.95 mg/kg (wet)</td>
<td>0.135 and 0.584 mg/kg dry weight (dw)</td>
</tr>
<tr>
<td>HCBD [BIPRO 2015]</td>
<td>Sediment: 24.4 µg/kg</td>
<td>~ 6 µg/kg</td>
</tr>
<tr>
<td></td>
<td></td>
<td>39-597 µg/kg [JRC 2012]</td>
</tr>
<tr>
<td>Dicofol</td>
<td>Sediment: 0.0237 µg/l</td>
<td>Insufficient data</td>
</tr>
<tr>
<td>PFOA</td>
<td>Insufficient data</td>
<td>0.011-0.018 [JRC 2012]</td>
</tr>
<tr>
<td>PFHxS</td>
<td>Insufficient data</td>
<td>Insufficient data</td>
</tr>
<tr>
<td>HBCDD [BIPRO 2015]</td>
<td>0.17 - 5.9 mg/kg</td>
<td>~ 60 µg/kg</td>
</tr>
</tbody>
</table>

The comparison given in Table 69 indicates that typical sewage sludges pose no immediate risk for human health and the environment regarding their content of the relevant substance/substance group.

In case the waste management operations of other relevant types of waste containing the relevant substances/substance groups lead to the exceeding of (P)NEC values or occupational exposure limits, possible adverse effects on human health or the environment must be expected.

If the evaluation is based on the lowest (P)NEC value for each substance, it can be assumed that the corresponding environmental concentrations cause no risk. Provided that the risk of causing environmental contamination is reduced by a factor of 10,000 through appropriate recovery and disposal operations (i.e. due to the operations, the concentration in waste will occur in the environment in not more than a tenthothousandth of this concentration), it can be concluded that the waste will cause no risk when treated appropriately.

On the basis of these premises, (P)NEC values can be multiplied by 10,000 and serve as a quantitative indicator for the upper limitation criterion (R). Table 70 shows the results of this procedure:

Table 70: Derivation of proposals for the limitation criterion R

<table>
<thead>
<tr>
<th>Substance/Substance group</th>
<th>Lowest (P)NEC for soils/sediments</th>
<th>Factor</th>
<th>Result criterion R [mg/kg]</th>
<th>Proposal criterion R</th>
</tr>
</thead>
</table>

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Further relevant information for the evaluation of possible risks for human health and the environment regarding to the relevant substances/substance groups is presented in the following. Table 71 shows an overview of the essential exposition pathways for the relevant substances/substance groups.

Table 71: Overview of essential exposure pathways for the relevant substances/substance groups

<table>
<thead>
<tr>
<th>Substance/ Substance group</th>
<th>Essential exposure pathway</th>
<th>Comment</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
<td>Inhalation of dust</td>
<td>The key routes of human exposure are thought to be from use in household consumer products, and their presence in house dust, and not from dietary routes [US EPA 2010].</td>
</tr>
<tr>
<td>SCCP [BIPRO 2015]</td>
<td>Oral exposure</td>
<td>The general public is not expected to handle SCCP materials and mixtures except for a few exceptions such as sealants and paints, commercial tents, etc. (see [RIVM 2010]). Food is considered to be the main route of intake of SCCP for humans, however no health risk was established in a study on Japan in 2003. [POPRC.5/2 /Rev. 1]</td>
</tr>
<tr>
<td>HCBD [BIPRO 2015]</td>
<td>Oral exposure</td>
<td>Exposure of the general public to HCBD mainly occurs indirectly via drinking-water and food with high lipid content [IPCS 1993].</td>
</tr>
<tr>
<td>Dicofol</td>
<td>Ingestion of food</td>
<td>Monitoring data indicate that the general population may be exposed to dicofol via ingestion of food containing dicofol</td>
</tr>
</tbody>
</table>

65 Considering the calculation Criterion (R) is not applicable for decaBDE
<table>
<thead>
<tr>
<th>Substance/ Substance group</th>
<th>Occupational exposure</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>PFOA [ECHA 2013b]</td>
<td>Oral exposure (food and drinking water). Ingestion of contaminated food, water or soil. Breathing of indoor air: PFOA-contaminated dust from carpets, upholstery, clothing, etc.</td>
<td>Exposure of the general public to HCBD mainly occurs indirectly via drinking-water. Mothers excrete PFOA via breast milk and transfer PFOA to infants.</td>
</tr>
<tr>
<td>PFHxS [ECHA 2017b]</td>
<td>Oral (food and drinking water) and inhalation (dust) exposure</td>
<td>Exposure of the public to HCBD mainly occurs indirectly via drinking-water. Gestational and lactational transfer.</td>
</tr>
<tr>
<td>HBCDD [BIPRO 2015]</td>
<td>Inhalation, oral and dermal exposure</td>
<td>Humans may be exposed to HBCDD through inhalation of fumes or dust, food intake or skin contact. In addition, a risk exists for babies during pregnancy and breastfeeding. (see [ECB 2008])</td>
</tr>
</tbody>
</table>

Table 72 gives an overview of relevant information on occupational exposure.

**Table 72: Information on occupational exposure**

<table>
<thead>
<tr>
<th>Substance/ Substance group</th>
<th>Occupational exposure</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
<td>Occupational exposure is possible from dust ingestion/dermal contact, inhalation, and diet in the use of the different decaBDE applications (e.g. as a FR in different electrical and textile applications).</td>
<td>[US EPA 2010]</td>
</tr>
<tr>
<td>SCCP [BIPRO 2015]</td>
<td>Occupational exposure is possible via the dermal route but also through inhalation in the preparation, formulation and use of the different SCCP applications. (see BiPRO 2011, Table 9-38)</td>
<td>[ERA 2000; ECHA 2009]</td>
</tr>
<tr>
<td>HCBDD [BIPRO 2015]</td>
<td>Occupational exposure possible in the manufacturing and use of HCBD-containing substances or products. (see BiPRO 2011, Table 9-38)</td>
<td>[IARC 1999]</td>
</tr>
<tr>
<td>Dicofol</td>
<td>Occupational exposure to dicofol may occur through inhalation and dermal contact with this compound at workplaces where dicofol is produced or used.</td>
<td>TOXNET: Dicofol</td>
</tr>
<tr>
<td>PFOA</td>
<td>Workers at fluorochemical production: Breathing air that contains PFOA. Also professional ski waxers exhibit high level of PFOA in their blood.</td>
<td>[SRHD 2017]</td>
</tr>
<tr>
<td>PFHxS</td>
<td>Occupational exposure is possible</td>
<td>[ECHA 2017b]</td>
</tr>
</tbody>
</table>
via inhalation, oral and dermal exposure. Inhalation as the most important way of exposure. Workers at PFOA manufacturing facilities and ski wax technicians are characterized in the literature.

<table>
<thead>
<tr>
<th>Substance/substance group</th>
<th>Proposal for criterion R [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
<td>Not applicable</td>
</tr>
<tr>
<td>SCCP [BIPRO 2015]</td>
<td>18,000</td>
</tr>
<tr>
<td>HCBDD [BIPRO 2015]</td>
<td>200</td>
</tr>
<tr>
<td>Dicofol</td>
<td>200</td>
</tr>
<tr>
<td>PFOA</td>
<td>-</td>
</tr>
<tr>
<td>PFHxS</td>
<td>-</td>
</tr>
<tr>
<td>HBCDD [BIPRO 2015]</td>
<td>1,000</td>
</tr>
</tbody>
</table>

Results for criterion R

Table 73 lists the results for the upper limitation criterion R.

Table 73: Overview of the results for the limitation criterion R
6.2.3 EVALUATION OF CRITERIA OF THE MPCL

6.2.3.1 Worst case scenario for human health risks

The compliance with agreed limit values cannot exclude all risks related to the POP substances in question. However, it can be concluded that acute risks do not emanate from wastes with lower contamination levels. The risk that substances may pose to humans and the environment is not only a function of its toxicological properties but also of the corresponding exposure.

Performing a complete risk assessment for the different exposure settings is not in the scope of the project, even less as crucial parameters such as leakage from different applications, environmental transfer rates, and effects on food contamination have not yet been quantified and can be controversially discussed.

Above which POP concentration risks can occur in a worst-case scenario to humans or the environment that do not allow application of an environmental preferable solution other than destruction or irreversible transformation is therefore estimated based in the methodology developed by [BiPRO 2011] and [BiPRO 2005].

The two approaches which are outlined in detail in [BiPRO 2011] are applied. Both approaches rely on leaching rates for waste which were determined for the evaluated POPs in [BiPRO 2011]. The approach developed in 2011, considering the use of the assumed leaching potentially, however might not be perfectly applicable to the newly listed POPs and POP candidates.

The results of the two approaches, as outlined in more detail in [BiPRO 2011] are presented in Table 74. Values for the calculations are based on the EQS concentrations reported and highlighted in Table 68 and the lowest background values as detailed in section 10.5 of the Annex.

### Table 74: Results for approach 1 and 2

<table>
<thead>
<tr>
<th>Substance</th>
<th>Approach 1 [mg/kg]</th>
<th>Approach 2 [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>DecaBDE</td>
<td>N/A</td>
<td>27 – 1,100</td>
</tr>
<tr>
<td>SCCPs</td>
<td>10,000</td>
<td>5,000– 83,000</td>
</tr>
<tr>
<td>HCBD</td>
<td>12,000</td>
<td>200 – 3,000</td>
</tr>
<tr>
<td>Dicofol</td>
<td>26 or 200 (depending on initial value)</td>
<td>36,000</td>
</tr>
<tr>
<td>PFOA</td>
<td>2,000</td>
<td>61 – 684</td>
</tr>
<tr>
<td></td>
<td></td>
<td>50 – 1,800</td>
</tr>
<tr>
<td>PFHxS</td>
<td>200</td>
<td>200 – 630</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2,950 – 14,630</td>
</tr>
<tr>
<td>HBCDD</td>
<td>6,200</td>
<td>50 or 100 – 730 or</td>
</tr>
<tr>
<td></td>
<td></td>
<td>140,000 (depending on reported concentration)</td>
</tr>
</tbody>
</table>
The results compiled in Table 74 are related to the above mentioned limitations and can only give a rough indication on the dimension of MPCLs based on risks related to potential leaching. Further studies or modelling work on leaching behaviour (from waste into leachate water) would be needed for the individual substances in order to enable more comprehensive estimates. Considering the presented results and the limitations of the methodology, no quantitative results which allow to arrive at justifiable MPCLs are derived. Therefore, MPCLs are derived taking into consideration the existing MPCLs and other relevant available information (see chapter 7).
6.2.4 RESULTS OF THE EVALUATION OF THE LPCL AND MPCL

Table 75 shows the results of the evaluation of the upper and lower limitation criteria for the substances/substances groups. The results present an essential discussion basis for recommending LPCLs and appropriate waste management operations, particularly in combination with the elaborated substance and mass flows and the risk assessment.
Table 75: Results of the preliminary evaluation of the upper and lower limitation criteria for the selected substances/substances groups [mg/kg]

<table>
<thead>
<tr>
<th></th>
<th>decaBDE</th>
<th>SCCPs</th>
<th>HCBD</th>
<th>Dicofol</th>
<th>PFOA</th>
<th>PFHxS</th>
<th>HBCDD</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Lower limitation criteria of the LPCL</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(A) Analytical potential</td>
<td>200</td>
<td>0.1</td>
<td>0.001</td>
<td>0.03</td>
<td>1</td>
<td>1</td>
<td>100</td>
</tr>
<tr>
<td>(B) Background contamination</td>
<td>≥20</td>
<td>1.0</td>
<td>0.1</td>
<td>3</td>
<td>0.01</td>
<td>0.01</td>
<td>2</td>
</tr>
<tr>
<td>(DR) Disposal and recovery capacities</td>
<td>0.3</td>
<td>1</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>10</td>
</tr>
<tr>
<td>(E) Economic feasibility</td>
<td>200</td>
<td>420</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>10</td>
</tr>
<tr>
<td><strong>Upper limitation criteria of the LPCL</strong></td>
<td></td>
<td></td>
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<td></td>
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<td></td>
</tr>
<tr>
<td>(LV)</td>
<td>1,000</td>
<td>10,000</td>
<td>100</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
<tr>
<td>(R)</td>
<td>n.a.</td>
<td>18,000</td>
<td>200</td>
<td>200</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
<tr>
<td><strong>Current value</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Current LPCL:</td>
<td>-</td>
<td>10,000</td>
<td>100</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,000</td>
</tr>
<tr>
<td>Current MPCL:</td>
<td>-</td>
<td>10,000</td>
<td>1,000</td>
<td>-</td>
<td>-</td>
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<td>1,000</td>
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<tr>
<td>n.a.: Not applicable</td>
<td></td>
<td></td>
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7. REASSESSMENT OF LIMIT VALUES FOR PCB AND PCDD/F

7.1 Introduction

In order to assess whether notable developments have occurred, a screening whether there is information available which indicates a possible need to adjust already established LPCLs or MPCLs has been performed.

A main conclusions was that there was a particular need to review the LPCL for PCDD/Fs and to possibly particularly consider the dl-PCBs on a TEQ basis.

In the context of this project, for PCDD/Fs consideration related to risks and selected economic feasibility considerations are made, since there is no indication that other relevant criteria have been subject to change. The risk criterion includes possible adverse effects on human health and the environment. The LPCL should be established in a way that adverse effects on human health and the environment are avoided.

With regard to PCDD/Fs, information from studies prepared by the Swedish Environmental Protection Agency (EPA) in 2011 as well as information provided by IPEN indicates the need for reviewing and possibly adjusting the currently established limit values for PCDD/Fs. Further, possible impact / contribution of dl-PCBs in waste pointed out by the Swedish EPA and IPEN and therewith associated implications on PCDD/F limit values. According to the Swedish EPA, as long as dl-PCBs are not included in the LPCL for PCDD/Fs, the estimated risk associated to food chain transfer of pollutants in waste may be underestimated [SE EPA 2011]. It could be considered to include the dl-PCBs in the TEQ-based POP-contents levels of PCDD/Fs and/or to lower the LPCL for PCDD/Fs.

The chemical and physical properties, as well as the toxicity and environmental fate of PCB is affected by the degree of chlorination. There is a positive correlation between increasing chlorination, fat solubility, environmental persistency resulting in a bioaccumulative potential of highly chlorinated PCB. In addition, the group of PCB include 12 congeners that have been assigned to have dioxin like properties (also called dioxin-like PCB or short dl-PCB). These twelve congeners have, due to steric hindrance, a coplanar structure where the two rings are on the same level thus not twisted. This structural motive is similar to those of dioxin, and relaunches the same toxic mode of action [Safe 1985]. These congeners were classified as carcinogenic to humans (Group 1) by the International Agency for Research on Cancer in 2014 [Lauby-Secretan 2016].

In June 2005, a World Health Organization (WHO)-International Programme on Chemical Safety expert meeting was held in Geneva during which the toxic equivalency factors (TEFs) for dioxin-like compounds, including some polychlorinated biphenyls (PCBs), were reevaluated. At the meeting the so called “WHO 2005 TEF” values were decided. These include TEFs for PCDDs, PCDFs and the dl-PCBs.

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66 See chapter 4.2 of the interim report from 14.3.2018
68 See http://www.who.int/ipcs/assessment/tef_values.pdf
### 7.2 Definition of PCB in EU legislation and the Stockholm Convention

Within EU legislation PCBs are defined in different places in different way. Therefore, in the frame of this project also the differences in the definition of PCB between EU Legislation (Council Directive 96/59/EC of 16 September 1996) and the Stockholm Convention are addressed.

**Council Directive 96/59/EC:** "On the disposal of polychlorinated biphenyls and polychlorinated terphenyls (PCB/PCT)"

In article 2 of the directive, PCB are defined as:

- polychlorinated biphenyls
- polychlorinated terphenyls,
- Monomethyl-tetrachlorodiphenyl methane, Monomethyl-dichloro-diphenyl methane, Monomethylidibromo-diphenyl methane
- any mixture containing any of the abovementioned substances in a total of more than 0.005 % by weight;

Thus, next to the 209 congeners of PCB also other classes of chemicals are covered by the definition. The group of polychlorinated terphenyls (also called polychlorinated triphenyls) possess one additional phenolic ring compared to biphenyls.

Also halogenated monomethyl diphenyl methane\(^69\) are defined as PCB. This includes, the (theoretically) 69 isomers of monomethyl-tetrachloro diphenyl methane (CAS: 76253-60-6, EC: 278-404-3, trade name Ugilec 141) and the isomers of monomethyl- dichloro- diphenyl methane (no CAS, EC: 400-140-6, trade name Ugilec 121 or Ugilec C 21). These compounds have been marketed since the early 1980s as less hazardous substitutes for PCBs. However, the share similar physicochemical properties to PCBs and were applied for similar purposes like dielectric fluids in capacitors and transformers or as hydraulic fluids in coal mine equipment. Moreover, they were considered less harmful in case of fire due to a much lower potential to form chlorinated dioxins/furans [Herrmann, 2001].

Next to chlorinated also dibrominated monomethyl diphenyl methane are defined as PCB (CAS: 99688-47-8, EC: 402-210-1).

As observable by the title, this regulation only addresses the disposal of PCB. However, as every other chemical also PCB need to be transported. In this context, the committee of Experts on transport of dangerous goods, on GHS-classification and labelling of chemicals included halogenated monomethyl biphenyls as PCBs not only for the purpose of disposal, but also for the purpose of transport [UNECE 2015].

**EU Regulation No 850/2004 (POP Regulation)**

The EU POP Regulation cites a CAS number for the chemical definition of PCB (CAS: 1336-36-3). This CAS describes the group of all 209 congeners. However, the CAS number is accompanied by the phrase "and others".

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\(^{69}\) Please note, that in this context: monomethyl biphenyls = monomethyldiphenylmethanes = benzyltoluenes
In the context of the concentration limit, the regulation cites European standards EN 12766-1\textsuperscript{70} and EN 12766-2\textsuperscript{71}, that shall apply when they need to be calculated. In the context of this norm, PCB is defined the same way as in directive 96/59/EC of 1996-09-16 on the disposal of polychlorinated biphenyls and polychlorinated terphenyls. As described already above, the definition includes all PCB congeners, PCTs and PCBTs (polychlorinated benzyltoluenes).

However, EN 12766-2 covers two calculation methods specifically related to PCB congeners:

- method A uses the sum of the contribution of all congeners to produce a measurement for PCB content, whereas
- method B uses six congeners (28, 52, 101, 153, 138, 180) for the calculation of an intermediate sum, which is then multiplied by a multiplication factor to yield the PCB content. The multiplication factor five constitutes an averaged factor, representing an averaged content of the six selected congeners in a multitude of technical waste materials, experienced in many European countries during the last 15 years (EN 12766-2).

In conclusion, the term “PCB” in the POP Regulation can be interpreted to include PCB (all congeners), PCTs and PCBTs, however, waste related concentration limits are specifically related to the PCB congeners. Also calculation methods in relevant standards are related to the PCB congeners.

**Stockholm convention**

In Annex C, Part IV 1(a) of the Stockholm Convention PCB are defined for the purposes of that Annex, as “aromatic compounds formed in such a manner that the hydrogen atoms on the biphenyl molecule (two benzene rings bonded together by a single carbon-carbon bond) may be replaced by up to ten chlorine atoms”. By this the 209 PCB-congeners are covered, exclusively. However, no exact definition is given at any place other that this annex. On the SC internet site the following quote can be found: “Of the 209 different types of PCBs, 13 exhibit a dioxin-like toxicity.”. In other words, it can be considered that the SC defines PCB as the 209 congeners of PCB and that polychlorinated terphenyls and halogenated monomethyl biphenyls are not defined as PCB.

### 7.3 Considerations related to adverse effects of PCDD/F and the established LPCL

PCDD/Fs are included in the list of substances subject to waste management provisions set out in Article 7 of the POP Regulation. The concentration limit referred to in Article 7(4)(a) is 15 µg/kg (0.015 mg/kg). The limit is calculated as PCDD and PCDF according to the TEFs indicated in Annex IV to the POP Regulation (which are in line with the WHO TEQ from 2005). PCDD/Fs are also included in the list of substances subject to release reduction provisions of the POP Regulation (Annex III).

The report “Low POP Content Limit OF PCDD/F in Waste - Evaluation of human health risks” by the Swedish environmental agency investigated whether the LPCL of 15 µg/kg (15 000 ng TEQ kg\textsuperscript{-1}) for PCDD/Fs in waste is low enough to protect humans from health risks [SE EPA 2011].

\textsuperscript{70} EN 12766-1: Petroleum products and used oils - Determination of PCBs and related products - Part 1: Separation and determination of selected PCB congeners by gas chromatography (GC) using an electron capture detector (ECD)

\textsuperscript{71} EN 12766-2: Petroleum products and used oils - Determination of PCBs and related products – Part 2: Calculation of polychlorinated biphenyl (PCB) content.
The report itself is based on two field studies that were conducted at sites where wastes and ashes are managed and stored open, one in Thailand and one in Peru. The aim of these studies was to investigate the degree of local environmental impact of open waste/ash management and storage. Based on the data of the fieldwork in Peru, human exposure levels were assessed using a fate and exposure model (CalTOX). According to the bio transfer factors (BTF) calculated from the literature, 3 pg WHO-TEQ g^{-1} fat in eggs can be reached already at soil concentrations of 1-70 ng TEQ kg^{-1} dw (0.001-0.07 µg/kg). Therefore, the authors point out that there is a risk that the conclusions made in the BiPRO report (2005) significantly underestimates the risk for transfer of PCDD/F into eggs at soil concentrations of 1-000 ng TEQ kg^{-1} (1 µg/kg). The calculations determine also a critical soil level of 7-25 ng WHO-TEQ kg^{-1} (0.007-0.025 µg/kg) for PCDD/Fs for children in rural and self-supporting settings. The criterion was that the total exposure summed for multiple exposure routes should not exceed the tolerable daily intake (TDI). If additional exposure from dl-PCBs is assumed to account for 50% of TDI, this limit should be adjusted downwards to 3-13 ng WHO-TEQ kg^{-1} (0.003-0.013 µg/kg).

In the report also, the health risks for adults are evaluated. For adults, ingestion of soil or ash can become an important exposure route during e.g. occupational exposure. Critical solid matrices concentrations for adults can probably be found in the range of 200-1,000 ng WHO-TEQ kg^{-1} (0.2-1 µg/kg) depending on the assumed internal bio accessibility of solid matrices, the soil ingestion rate and what fraction of TDI that is considered as an acceptable exposure dose. This range is also far below the suggested Low POP Content Limit of 15 µg/kg.

In context to dl-PCB, the author found that only limited data exist for concentrations in waste. According to the Swedish report, dl-PCBs account only for a minor fraction of the total TEQ in waste. But because of the higher accessibility of this compound group may lead to a significant contribution to total TEQ if the waste is introduced into the environment and its pollution enter food chains. So far, the impact of dl-PCBs in waste has not been fully investigated. The authors conclude, that the investigation of human health risks associated to PCDD/Fs is complex and is not easily handled, since it involves environmental and toxicological sciences as well as political, technical and economical perspectives. Thus, the Swedish report does not suggest specific new LPCLs for PCDD/Fs (or dl-PCBs) in waste.

[CEWEP Sub. 2018b] criticises that the report is based on data from Peru and Thailand. According to CEWEP, the practices outlined in the report are no current practice in Europe (fly ash as feed or bedding, fertiliser etc.). They advocate that a lower LPCL would not have any impacts on environment or human health in Europe.

A further PCDD/F-risk assessment is available from IPEN/Arnika. The corresponding report was released in April 2017 [Petrlik and Bell 2017a] and a paper presented at the Dioxin 2017 Conference [Petrlik and Bell 2017b].

The report is based on a review entitled "Toxic ash poisons our food chain", that investigated more than 300 scientific reports and articles, that cover waste incineration management practices in general and data on dioxins in particular. A key finding of the report is, that the amount of dioxins released (contained) in waste incineration fly ash is highly underestimated (its content is 3 – 10-times as much as previously estimated). In exact numbers, the author estimates that for EU countries the amount could be between approx. 7 kg I-TEQ of dioxins released into waste incineration residues annually (at a minimum), although the final figure is more likely closer to or over 10 kg TEQ/year of PCDD/Fs. The levels of PCDD/Fs in fly ash were also addressed in this report. For Europe levels of 140 ng TEQ g^{-1} are reported for Czech Republic [ALS 2012]. Considerably lower levels were reported by [Vehlow et al. 2006] that collected data about measurements from 2001 – 2004 for 47 waste incineration plants and they observed a range of PCDD/Fs in fly ash from 0.1 – 9.4 ng TEQ g^{-1} [Vehlow et al. 2006].
In general, fly ash is reused for a broad range of open and closed uses. The authors name the following uses, that are amongst the most critical and difficult to control: feed and bedding for poultry, fertilizer or as an amendment to soil for agricultural use, use for roads and as pathway pavement in areas with locally grown food. Waste incineration fly ash is also the subject of waste traffic, thus the minimization transboundary movement makes control of dioxin releases even harder [Petrlik and Bell 2017a]. Against this background, [IPEN & Arnika Sub. 2018b] suggest a soil specific LPCL for application to soil.

[CEWEP Sub. 2018b] however, pointed out that fly ash use practices such as the mentioned feed or bedding for poultry, fertilisers for agriculture, or other application without a previous stabilization are not practice in Europe.

This statement of [CEWEP Sub. 2018b] is considered to be correct, since there is no specific information on current relevant use of PCDD/Fs-containing fly ash as feed and/or bedding for poultry, fertilizer or as an amendment to soil for agricultural in Europe. Such uses are therefore not considered current practice within the EU and therefore, a possible soil-specific LPCL, as suggested by IPEN/Arnika, for this group of chemicals is not considered necessary.

Based on calculations, the authors proof that the open application of wastes to soil or on terrain surface (without stabilization) with a level of PCDD/F and dl-PCB above 0.05 µg/kg can lead to food chain contamination. This is particular true for free-range poultry eggs, which have been observed to have increased levels of dioxins. These levels are above the currently used safety limits (2.5 pg WHO-TEQ g-1 fat), in cases revealing a 10-fold exceedance. Therefore, locally produced food is of great importance in developing countries and rural locations in developed countries in particular [Petrlik and Bell 2017a].

In conclusion, the IPEN/Arnika report proposes to adopt the LPCL for PCDD/F of 1 µg/kg to avoid the risk of transboundary movement of incinerator residues and loss of control of the fate and transport of large volumes of POPs contaminated residues. Dl-PCB should be integrated in the LPCL of PCDD/PCDF, so the level of 1 µg/kg will be applicable for both PCDD/Fs and dl-PCB as a total expressed in WHO-TEQ. An additional suggestion is to ban the use/application of wastes to soil or on terrain surface (without stabilization) with a level of PCDD/Fs and DL PCBs above 0.05 and/or at least 0.1 µg/kg.

The authors also highlight, that the lowering of the LPCL is in accordance with the national POP-NIPs (national implementation plans) of Japan and South Korea, as those countries already elaborated more strict levels for PCDD/Fs control in waste and/or contaminated soil, with respective values of 1 µg/kg or 3 µg/kg [NIP Japan 2005] and [NIP South Korea 2009].

The revision of the PCDD/F’s LPCL is also the topic of another publication of IPEN and Arnika published in 2018 in the context of the Dioxin 2018 conference [Petrlik and Bell 2018a]. In this publication, the authors present recent results of PCDD/F measurements around sites (located in UK, Czech Republic, Thailand) near waste incinerators that burn waste containing POP substances. Based on the results, the authors highlight that the potential exposure routes of wastes contaminated by POPs (PCDD/Fs, dl-PCB) - dust/soil/sediment - biota (poultry, birds, fish) is clearly demonstrated in the cases described. Therefore, according to IPEN and Arnika, waste containing PCDD /Fs below the current LPCL of 15 µg/kg (15,000 pg TEQ/ g) can lead to significant contamination around sites where the waste is not handled as required by the Stockholm Convention. Furthermore, also waste above - 0.02 / 0.05 µg/kg can contaminate soil if used on surface without any treatment. Based on the findings of this and other studies, the authors recommend a new LPCL for PCDD/Fs in wastes at 1 µg/kg, and to limit use of wastes containing PCDD /Fs + dl-PCBs above 0.05 µg/kg on surface soils without pre-treatment. This is in accordance to the claims made in the 2017 publication by the same authors.
The report of IPEN/Arnika is also recognized by several stakeholders. Thus, two stakeholders (ToxicoWatch Foundation (NE) and the society for earth) also suggest lowering the limit values as follows: PCDD/F – 1 ng/g (1 µg/kg) and 0.05 ng/g limit for waste application on terrain surface (including dioxin-like PCBs). However, one stakeholder also raises concerns about lowering the limit value for PCDD/Fs based on the findings on fly ashes. The stakeholder points out that fly ashes are partially recycled in closed applications like concrete and asphalt, since this is usage is believed to be without any risk for human health or environment. As a starting point for resolving this issue, the stakeholder suggest making an inventory and overview of the present concentrations in fly-ashes of waste incinerators and other thermal processes. The final destination of those ashes (landfill, incineration) should be part of that inventory.

In summary, the available information from studies prepared by the Swedish EPA as well as a report provided by IPEN indicates the need for reviewing and possibly adjusting the currently established limit values for PCDD/Fs. For PCDD/PCDF the evidence indicates, that the current limit value (15 µg/kg) could be adjusted. The possible contribution of dl-PCB in waste as pointed out by the Swedish EPA and IPEN indicates the need to include dl-PCBs in the LPCL for PCDD/Fs. This should be done based on the TEQ-based POP-content levels of PCDD/Fs and dl-PCBs. The advice is to change to values of 1 µg/kg as LPCL and 0.05° µg/kg for waste application on terrain surface.

7.4 Considerations related to the economic feasibility of lowering the LPCL for PCDD/F

From MSWI treatment, solid wastes remain. Two types of ashes are usually present: bottom ash, typically collected at the bottom of the combustion chamber and fly ash.

Fly ashes are usually collected with flue-gas treatment equipment that also capture other pollutants from the flue-gas. This corresponds to flue-gas treatment (FGT) residues, also sometimes called air pollution control (APC) residues. Strictly speaking, APC residues covers the solid residues arising from FGT equipment, including the reacted and unreacted reagents but excluding the fly-ash portion. However, from the waste management point of view, all these solid residues are usually handled in combination, although a separate collection and management is also possible [BREF WT FD 2017].

Waste incineration concentrates the environmentally harmful substances present in the waste in the FGT residues which amount to 3-5 % of the mass of the waste entering the incineration plant [CEWEP Sub. 2018b]. Considering an input of 90 million tonnes of municipal solid waste to MSWI plants per year in EU28, approximately 3 million tonnes of FGT residues are generated each year [CEWEP Sub. 2018a; CEWEP Sub. 2018b].

[Dias-Ferreira et al. 2016] in their study evaluated the influence of electrodialytic remediation on dioxin (PCDD/F) levels in fly ash and APC. It was found that fly ash shows higher concentration than other residues e.g. APC or bottom ash. The concentration reported for fly ash ranged from 0.2 to 2 µg I-TEQ/kg, whereby the higher concentration was found in fly ash from small waste incinerators working at lower temperatures. The following graph highlights the latest values reported for PCDD/PCDF concentrations highlighting values exceeding 1 and / or 3 ng I-TEQ/g. After the year 2000 the concentrations decreased 10-fold, mostly falling in the range 0.01-10 ng I-TEQ/g. The values obtained in the current work are in the lower range of the chart: relatively low values and low variability were found in the APC residues (between 0.18 and 0.35 I-TEQ/g dry matter), while for fly ash the value was one order of magnitude higher. Class 2-incinerators were found to have emission factors approximately 30 times higher than Class 4-incinerators.
In a Waste-to-Energy plant the bottom ash is collected at the end of the furnace’s grate. It consists of non-combustible materials and is the residual part from the incineration of waste. Both the remaining ferrous metals and non-ferrous metals can be taken out of the bottom ash and recycled. After the metals are taken out, the rest of the ashes, which are like gravel or sand, are stored for a period before being used in road construction or as a covering layer on landfill sites. Further, FGT residues are collected after the filtration process and are carefully stored to ensure no escape of the material into the local environment. The material is then transported in sealed containers to hazardous landfill sites, to treatment plants or salt mines [CEWEP 2018].

According to the [BREF WI 2006] in the Netherlands, around a third of the total fly ash from MSWI plants was applied as filling material for road construction materials (asphalt). The PCDD/F content measured in Dutch fly ashes used as raw material ranged from 0.17 to 3.8 µg/kg as evaluated in 2005, the maximum content of the total scope of Dutch fly ash was four times higher [Lamers et al. 2005]. Moreover, it was reported that projects in Belgium used fly ash as a composite filler in asphalt for road construction [Royal Haskoning 2004]. As stated by [CEWEP Sub. 2018b] fly ash is sometimes used as secondary raw material, however this is only done if permitted by national legislation, as is the case in the Netherlands, and with stringent monitoring of possible pollutants.

Combustion ash and FGT residues are one of the main waste streams treated by stabilization and solidification processes, directly within the combustion plant or at dedicated waste management facilities. Other methods are for instance vitrification, purification and recycling of some components (e.g. salts). The residues may also substitute raw materials in waste treatment or may undergo further treatment to enable them to be used e.g. in construction applications [BREF WT FD 2017].

Earlier attempts at managing residues mainly focused on pre-treatment or stabilization to reduce the leaching of potentially toxic substances prior to landfilling. In a recent study by [Bio 2015], the impact of different classification approaches for hazard property “HP 14” was assessed for fly ash from incinerators (19 01 13*/19 01 14), and it emphasized the limited recovery options of MSWI fly ash in Europe [Quina et al. 2018]. According to [Bio 2015], recovery of MSWI fly ash as a
second-hand raw material, for example in cement or in other building materials, is not widespread in Europe. The available information mainly focuses on the management of hazardous MSWI fly ash and deals with treatment before landfilling or potential recovery. One of the most common stabilization process prior to landfilling has been the solidification of fly ash using a hydraulic binder, mostly Portland cement (e.g. in France and Italy). Indeed, a large part of available literature describes research on the technical and environment feasibility of fly ash reuse: stabilization for reuse in construction (buildings and roads), investigation of leaching from the product using pre-treated (or “washed”) fly ash, mechanical properties of the materials built from fly ash, etc. However, no evidence of large-scale industrial projects using reused fly ash was found [Bio 2015].

The main barrier to reusing or recycling fly ash is finding a suitable way to manage risks linked to the hazardous properties of this waste. Although effective washing processes have been developed to treat fly ash prior to reuse, it seems that the high costs of such treatments, along with a negative perception of ashes, have prevented an uptake of fly ash recycling in Europe. Another factor hindering reuse of fly ash is the highly variable composition of this waste, which makes a burden of the need for characterization and regular analysis for recovery [Bio 2015].

[CEWEP Sub. 2018b] provided input indicating that studies conducted in countries which use fly ash as raw material for composite asphalt fillers have found no risk for human health and the environment. In the Netherlands the methodology of the overall quality supply structure is laid out in National Assessment Guideline 9041 and covers requirements related to environmental and work hygiene. As a result, the total PCDD/F contents found in asphalt concrete with composite asphalt fillers are negligible according to [CEWEP Sub. 2018b]. [BiPRO 2005] considers the use of fly ash with concentrations of 15 ng/g as filler in asphalt as safe.

Seeing the amount of research regarding safe ways to reuse fly ash and avoid landfilling, it seems that fly ash management is going towards the applications processes allowing fly ashes to be reused or at least landfilled as inert waste at a low cost [Bio 2015]. Besides, since high costs may be involved for landfilling (150 – 500 €/t), the recycling/recovery options may become interesting alternatives in Europe [Quina et al. 2018]. This is also reflected in the current draft BREF document on waste treatment (Final Draft 2017), describing possible recycling and recovery options for fly ash in addition to the currently commonly applied processes and techniques such as solidification. The applied processes and techniques include among others the following [BREF WT FD 2017]:

- **Immobilization of solid and/or pasty waste:** Immobilization aims at minimizing the rate of contaminant migration to the environment and/or reducing the level of toxicity of contaminants, to alter or improve the characteristics of the waste so that it can be disposed off. Immobilization relies on the properties of the reagent to produce an immobilized output, even when this output does not have a solid form. The pre-treatment allows the treatment of fly ashes and salts arising from the dichlorination of fumes in household waste incineration. The outputs consist of a filter cake with a reduced toxicity and solubility, and salty water. It helps reduce the leachability of the output and therefore the risk of contamination by the leaching out of soluble compounds. The process is more sophisticated than solidification.

- **Solidification:** Solidification changes the physical properties of the waste input by using additives. Cement solidification for example, based on mixing waste with cement, is a chemical process that aims at developing bonds between the binder and the waste. Another technique, involves the curing of fly ash waste, for example with aqueous neutral solution, to give a granular output prior to landfill. It is probably the most commonly used method for the treatment of FGT residues and is widely used in Europe and Japan.
• **Physico-chemical treatment:** Physico-chemical treatment of waste prior to backfilling consists of adapting the structural and physical characteristics of the waste input (mainly fly ashes) in accordance with local conditions for long-term storage in the mine. If wastes cannot be used directly as backfilling material, they are treated in dedicated physico-chemical treatment plants. The waste input is mainly flu-gas treatment residues from incineration or combustion plants (fly ashes), and a small amount of fine-grained pulverized waste. The output is a material with adequate structural and physical characteristics for backfilling. Plants for physico-chemical treatment of wastes as a pre-treatment before backfilling in salt and potash mines are found in Germany.

• **Recycling of Residual Sodium Chemicals from solid FGT residues:** The technology realizes the separation of soluble and insoluble parts of the FGT waste, solidifying the insoluble part and purifying and reusing the soluble part (constituted of inorganic salts) in some industries. Two plants in Europe using this process or a slight variation of it are in Italy and France. Both supply purified brine as raw material to a soda ash plant.

• **Recovery of salts from liquid FGT residues by solution/evaporation:** The main objective of the operation is to avoid the discharge of saline waste water to the sewerage system. This is achieved by evaporation of the scrubber liquid from the wet FG system.

• **Washing of FGT residues and their use as a raw material for production of construction products:** The washing process treats waste powders such as FGT residues to make them suitable for use in the manufacture of construction products. The waste input powders consist of useful mineral phases but also contain undesirable soluble fractions. High concentrations of soluble material are not desirable in the production of construction products and have previously restricted the recycling of FGT residues. Two plants in the United Kingdom have reported using this process.

**Implications on waste management due to proposed adjustment of POP limit values for PCDD/PCDF**

Available data indicate that PCDD/F TEQ concentration levels in fly ash from currently operating MSWI plants in Europe are in a range of 0.2 to 2 µg TEQ/kg (see Figure 39) and that dl-PCBs contribute less than 10% to the total TEQ (including PCDD/F and dl-PCB) [SE EPA 2011]. However, for Europe there are only a few measurement data from recent years available (particularly from a literature review in [Dias-Ferreira et al. 2016]).

[CEWEP Sub. 2018b] provided information on concentrations of PCDD/F in fly ash with the conclusion that out of the provided 35 observations the average concentration was around 2.5 ng I-TEQ/g. 20 out of the 35 observations exceeded a concentration of 1 µg/kg TEQ, 7 values exceeded a concentration of 3 µg/kg TEQ, 4 values exceeded a concentration of 5 µg/kg TEQ and 1 value exceeded a concentration of 10 ng µg/kg TEQ. The provided data is given in Annex section 10.12.72

[CEWEP Sub. 2018b] voiced their concern that setting a LPCL close to the operating levels of the plants will increase the costs of monitoring and the no additional environmental benefit would be achieved. They stated that a LPCL close to the operating levels would hamper with the safe recycling of fly ashes and thus resource efficiency. Further it was noted, that including dl-PCB can potentially lead to additional costs of analysis with no benefits for the environment and human health as reported by [CEWEP Sub. 2018b].

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72 Considering similar values for I-TEQ and WHO-TEQ
Based on available information, it is expected that currently recycling in the construction sector is not widespread in Europe. The current extent of recycling fly ash as filler in asphalt as previously reported for the Netherlands is not known (also not the PCDD/F TEQ concentration levels of fly ash from currently operating MSWI plants in the Netherlands). [Dutch Waste Management Association Sub. 2018], however, pointed out that fly ash below the current LPCL of below 15 µg/kg TEQ are used within the Netherlands. According to personal communication with [CEWEP Sub. 2018a], typical treatment operations for fly ash at EU level are underground disposal and hazardous waste landfilling. These waste treatment options are also possible for fly ashes exceeding the LPCL. Hence, relevant economic impacts are not expected at EU level, however, such impacts cannot be excluded for the Netherlands and Belgium and possibly in the UK. If the LPCL would be below typical fly ash contamination levels, the management option of using fly ash as filler in asphalt would be hampered.

According to the recent data provided by CEWEP 11% and 3% of the fly ash would exceed an LPCL of 5 µg/kg TEQ and 10 µg/kg TEQ respectively.

Against this background it can be expected that there will be no unacceptable economic impacts resulting at a LPCL in range between 5 and 10 µg/kg TEQ (total TEQ including PCDD/F and excluding dl-PCB). It could therefore be recommended to lower the LPCL for PCDD/F to 5 µg/kg TEQ.

It should be noted that this conclusion is based on the considerations made above related to fly ash from municipal solid waste incineration which do not represent a full assessment of the economic feasibility.

7.5 Considerations related to the analytical potential

Methods

When PCDD/PCDFs are to be measured in soil, sediment or by products from waste incineration processes traditionally high resolution magnetic sector GC-MS (GC-HRMS) is the gold standard technique. However, also GC-MS/MS delivers comparable results [Silcock et al. 2012]. For adequate quantification, the isotope dilution method is used whereby 17 labelled PCDD/F and 12 labelled PCB internal standards are determined. The extracts for the GC/MS measurements contain one or two recovery standards. The determined concentrations then need to be proportionately calculated in TEFs scheme and a corresponding TEQ ("Toxicity Equivalent") in accordance to the specification by the World Health Organisation (WHO).

LOD/LOQ

Reported LOQ for waste residues, ash, and slag is 0.001 µg WHO-TEQ/kg [Eurofins 2013].

One stakeholder (CEWEP, The Confederation of European Waste-to-Energy Plants) points that the uncertainty linked to the determination of PCDD/Fs and PCB-like dioxins at very low (also realistic) concentrations is very high, due uncertainty of the instrumental analytical procedure itself and sampling, transport, storage, sample pre-treatment, etc. According to CEWEP, this is of particular importance in the case of solid samples, and even more of residues, this uncertainty is usually higher than with liquid or gaseous samples, due to heterogeneity and the increased sample pre-treatment need to transform the solid sample into a sample capable to be instrumentally processed (either liquid or gaseous). Therefore, to avoid compliance issues without a real benefit, a margin between the values reported by the plants and the LCPL is needed.

Costs

For testing dioxins in solid matter (soil, sediments, compost) a cost of around 410 € is reported [my-lab 2017].
8. RECOMMENDATIONS ON LIMIT VALUES AND ENVIRONMENTALLY SOUND MANAGEMENT OF POP WASTE

On the basis of the results of this project, proposals for limit values and recommendations for the Environmentally Sound Management (ESM) of POP waste treatment operations can be derived. In the following, recommendations on limit values and environmentally sound management of relevant POP wastes are discussed substance by substance in the light of the project results.

The proposals for limit values concern the LPCL and MPCL which are laid down in the EU POP Regulation in its Annexes IV and V respectively. On the one hand they are intended to ensure pollutant removal to the maximum extent possible and on the other hand to enable environmentally sound recycling processes.

It should be noted that in carrying out the tasks under the present project, in several occasions relevant difficulties were encountered with respect to the availability of reliable data to derive well justified limit values. Information which was missing in specific cases was particularly related to (a) concentration levels of substances in waste, (b) consumption of substances for specific products and related quantities of products and wastes, and (c) reliable data on the practical applicability of test methods which enable to detect and/or quantify the substances and information on related LOD/LOQs. Corresponding data are required to apply the established methodology to derive justified concentration limits. As a consequence, justified assumptions had to be made at several steps. Where such assumptions are made this is indicated in order to provide a transparent information basis on how the recommendations for limit values were derived. Table 76 outlines the limitations for the individual substances.

Table 76: Limitations encountered in deriving a LPCL for the individual substances

<table>
<thead>
<tr>
<th>Substance</th>
<th>Limitations</th>
<th>Relevant Chapter</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
<td>Lack of data availability on decaBDE in C&amp;D; Unclear situation regarding the management of decaBDE-containing textile waste in Europe; Uncertainties are related to reliable data on the practical applicability of test methods. Moreover, cost estimated are lacking for health benefits of a lower LPCL or compliance costs under a certain LPCL.</td>
<td>4.1.8, 4.1.9 6.2 &amp; 8.1</td>
</tr>
<tr>
<td>SCCP</td>
<td>Information and data reported in recent literature sources is often outdated (e.g. reported concentrations of SCCPs detected quite often represent applications where SCCPs were typically used in the past). Inspection and enforcement activities carried out in EU MSs, where SCCPs are banned have found the continued presence of SCCPs in articles (including consumer products). These findings demonstrate that new products continue to be a source of SCCPs. Considering that currently SCCPs are expected not to be deliberately used in the EU, the relevant products and/or recycled products contaminated with or containing SCCPs are probably associated with imported articles and materials into the EU. However, considering the large diversity of imported articles reported to contain SCCPs and the fact that imports cannot be reliably traced it was not possible to further investigate associated waste streams and develop detailed substance/mass flows.</td>
<td>4.2 &amp; 8.2</td>
</tr>
</tbody>
</table>
In addition, a possible contribution from other CPs (e.g. MCCPs) to SCCP concentrations detected in various products cannot be excluded. Stakeholders’ opinion on this topic differ largely. This is mainly due to the acknowledged complexity of the mixtures, making the chemical analysis challenging. Application of suitable and further development of new analytical method would allow to quantify possible contribution of MCCPs to SCCP concentrations in wastes and recyclates and is needed to obtain reliable additional data on EU level.

<table>
<thead>
<tr>
<th>Substance</th>
<th>Description</th>
<th>Section</th>
</tr>
</thead>
<tbody>
<tr>
<td>HCBD</td>
<td>Missing observations of concentrations of HCBD in sludge in various countries in Europe. Estimates are based on the available data</td>
<td>4.3</td>
</tr>
<tr>
<td>Dicofol</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>PFOA, its salts and related compounds</td>
<td>Information on current production volumes and past and present uses of PFOA, its salts and PFOA related compounds in different applications is limited. Further, quantitative information on imported products containing PFOA and related compounds as well as information related to waste streams and recyclates containing or contaminated with those substances is missing. Based on the available information and data, it was not possible to investigate relevant waste streams in detail as it has been done for other substances in the report. Further, it was not possible to derive any upper limiting criteria which, is a clear limitation to proposing a range for a possible POP waste limit value.</td>
<td>4.5 &amp; 8.5</td>
</tr>
<tr>
<td>PFHxS, its salts and related compounds</td>
<td>Information on current production volumes and past and present uses in different applications is mostly missing and rather qualitative then quantitative. Available information is partly considered not to be reliable (e.g. information from market research reports). Further, quantitative information on imported products containing PFHxS and related compounds as well as information related to waste streams and recyclates containing or contaminated with those substances is missing. Based on the available information and data, it was not possible to investigate relevant waste streams in detail as it has been done for other substances in the report. Further, it was not possible to derive any upper limiting criteria which, is a clear limitation to proposing a range for a possible POP waste limit value.</td>
<td>4.6 &amp; 8.6</td>
</tr>
<tr>
<td>HBCDD</td>
<td>Lack of data availability on waste management of XPS in construction and is largely based on information submitted during the stakeholder consultation. Uncertainties are associated with the concentration on packaging were higher values are reported for the Netherlands but are based on waste contaminated with EPS from construction. Further quantitative data on concentrations in packaging from other EU countries would be valuable.</td>
<td>4.7, 6.2 and 8.7</td>
</tr>
</tbody>
</table>
Uncertainties are attached to reliable data on the practical applicability of test methods. Moreover, cost estimated are lacking for health benefits of a lower LPCL or compliance costs under a certain LPCL.

<table>
<thead>
<tr>
<th>PCB</th>
<th>TEFs for dl-PCBs may be overestimated</th>
</tr>
</thead>
<tbody>
<tr>
<td>PCDD/F</td>
<td>Lack of data regarding the availability of concentrations of fly ash, which is currently based on literature findings and data submitted during the stakeholder consultation. Moreover, quantitative data on economic implications is lacking.</td>
</tr>
</tbody>
</table>

For the improvement of ESM for POP waste three key elements are of particular relevance:

1. Improvement of relevant knowledge and technologies;
2. Better/more efficient identification and separation of POP-containing waste;
3. Higher degree of destruction or irreversible transformation of the POP content.

All three elements can be applied to all the relevant substances and will be discussed below, substance by substance. ESM of POP waste should aim at minimising possible risks related to the occurrence of POPs in wastes and their management. Corresponding risks are identified in chapter 5 and are used as a systematic basis for the discussion of ESM of the relevant waste streams.

Identification and separation of POP-containing waste - can be separately described for the identified relevant waste streams. In general, it can be stated that the dismantling, disassembling and mechanic separation can be used to reduce the volume of POP-BDE waste [UNEP/CHW.12/5/Add.6/Rev.1 2015] and that separation should take place as early as possible in the waste phase.

It should be noted that guidance for the ESM of POP wastes is available in the general and specific technical guidelines on the environmentally sound management of wastes consisting of, containing or contaminated with persistent organic pollutants.73

### 8.1 DecaBDE

#### 8.1.1 LOW POP CONCENTRATION LIMIT

The current POP Regulation does not yet contain a limit value for decaBDE. Based on the lower limit and upper limitation criteria, considering that analytical potential is a limiting factor for the LPCL, as detection at or below 100 mg/kg can be difficult, and limit values is another limitation factor based on existing legislation, a range of 200 mg/kg to 1,000 mg/kg for decaBDE or for the sum of POP-BDEs is proposed as LPCL. The relevant standards (IEC 62321-3-1:2013 and IEC 62321-6:2015) should be validated for concentration levels in line with the LPCL which will be selected. However, implementation can already be based on existing analytical methods.

The rationale behind this recommendation is the following:

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Background

C-decaBDE production and consumption is relevant in the period from 1970 to 2020 with the most relevant annual consumption over a period of 20 years (from 1990 to 2010) with 7,000 to 9,000 tonnes per year. It was and is used in functional concentrations in a typical range between 10 and 15% by weight.

Most relevant waste streams due to historic (and to a certain extent ongoing) use are (1) WEEE plastics, (2) plastics and textiles from ELVs and (3) plastics from C&D waste.

Due to past and current recycling activities decaBDE occurs in recycled plastics and products made thereof (including toys) in concentrations ranging from a few mg/kg up to several thousand mg/kg.

Waste management, relevance of waste flows and occurrence of decaBDE in the waste flows

WEEE

The usual treatment of WEEE plastics includes the manual disassembly or the mechanical treatment in shredders. Manual disassembly allows the manual separation of the individual plastic parts. Shredder processes are often associated with an automatic sorting process.

According to the WEEE Directive, plastics containing brominated flame retardants have to be removed from any separately collected WEEE. This obligation includes, among other things, plastics containing decaBDE as brominated flame retardants. Since in practice not all brominated plastics are separated, decaBDE occurs in recyclates.

Separation is typically carried out according to CENELEC TS 50625 which requires the separation of plastic of all WEEE categories other than large appliances and refrigerators and freezers, that contain a total bromine concentration exceeding 2,000 mg/kg from the plastic recycling stream (the so called high-Br-fraction). Currently 1.2 million tonnes of WEEE plastics result annually from separately collected WEEE in the EU. They are separated into a high-Br-fraction (8% of the total WEEE plastics) and a low-Br-fraction (92% of the total WEEE plastics). The high-Br-fraction is typically incinerated as waste. The low-Br-fraction is typically used in recycling processes.

DecaBDE consists of 83.3% Br (w/w). Therefore, a Br content of 2,000 mg/kg corresponds to a decaBDE content of 2,401 mg/kg. As several different BFRs are statistically found in WEEE plastics (TBBPA (36%), decaBDE (22%), DBDPE (20%), BTBPE (9%) and octaBDE (7%)), it can be concluded that usually the content of decaBDE and of POP-BDEs is well below 1,000 mg/kg (average content < 530 mg/kg\(^{74}\) for decaBDE and < 700 mg/kg\(^{75}\) for decaBDE and octaBDE) for a total Br content below 2,000 mg/kg.

The available data suggest that according to existing waste management practice, the major share of decaBDE contained in WEEE plastics is destroyed (\(~85\% \text{ or } 534 \text{ t/y}\)). The bulk of the decaBDE which is destroyed, is contained in the high Br fraction which is separated during WEEE recycling (\(~82.5\% \text{ or } 519 \text{ t}\)). However, a relevant share is not destroyed. Approximately \(~15\% \text{ or } 95 \text{ t} \text{ of decaBDE is not destroyed since it is recycled (\(~14\% \text{ or } 89 \text{ t} \text{) or landfilled (1%) and will continue to occur in recyclates and products made thereof.}

DecaBDE occurs in waste streams in concentrations ranging from zero to functional concentrations. According to calculations based on available data from literature median concentrations for the

\(^{74}\) Calculation: 2401 mg/kg x 22% corresponds to \(~530 \text{ mg/kg decaBDE}

\(^{75}\) Calculation: 2401 mg/kg x (22%+7%) corresponds to \(~700 \text{ mg/kg POP-BDEs}
The majority of the concentrations mentioned in literature and from the stakeholder consultation are in the range from 100 to 1,000 mg/kg. Preliminary results from a research with a sample consisting of 289 pieces and an estimated Br concentration of 632 mg/kg provided by the French Ministry for an ecological and solary transition state that in 91% of the pieces within the sample, the Br concentration was smaller than 100 mg/kg. In 9 pieces, representing 3.1% of the sample, the Br concentration laid above 2,000 mg/kg. The authors of that research estimate that those 9 pieces are responsible for almost 90% of the total Br within the sample.

**ELV**

The selective dismantling of decaBDE-containing components from ELVs is not considered feasible for two reasons because the dismantlers cannot be provided with detailed information on parts which contain decaBDE and it would be a too complex and time-consuming process.

The materials that cannot be readily dismantled and reused enter the shredding process. After shredding, the ASR is separated into three different fractions: (1) Ferrous fraction, (2) Non-ferrous metallic fraction (shredder heavy fraction, SHF) and (3) Shredder light fraction (SLF). The SHF as well as the SLF contain plastic components. The plastic particles within both fractions are assumed to contain the brominated compounds. In this respect, the SLF is most relevant (>700,000 t/y). DecaBDE levels in ELV materials range from 0 to 27,000 mg/kg. Based on literature data an average content of 109 mg/kg decaBDE in the SLF is estimated.

State-of-the-art post shredder technologies based on density separation allow to accumulate bromine rich fractions in the shredder residues with higher density (e.g. above 1.1 g/cm³). These high-density fractions contain high bromine levels (median values 2,220 and 2,600 mg/kg) and account for around 19% of the SLF. They can e.g. be incinerated at appropriate conditions to destroy the PBDE content (including the decaBDE content). The low-density fractions contain low levels of decaBDE, typically well below 100 mg/kg.

Estimations based on EUROSTAT data for 2014/15 demonstrate that only one third of the SLF including its corresponding decaBDE freight (annually 77.7 t) is destroyed whereas significant shares are landfilled (37.0%) or recycled (29.9%) and there is a risk that they will be released to the environment and they will continue to occur in recyclates and products made thereof.

According to [Öko-Institut 2018] more than one third of the SLF is currently not subject to onsite or off-site post shredder treatment. Increasing the rate of PST enables to increase the rate of destruction of decaBDE in SLF. To this end, by applying state of the art PST, high density fractions (18.7% of the SLF which contain high bromine levels) can be directed to feedstock recycling, incineration or chemical recycling processes.

**Construction and demolition waste**

Typical decaBDE concentrations in electrical insulation range from 10-30% and in epoxy adhesives it is below 30%. Specific information on decaBDE levels in construction materials is very scarce.

Though specify data for the EU are not available data from other countries suggest that 20% of the total decaBDE use in the EU was consumed within the construction sector. The plastic quantity from C&D waste in the EU amounts to 930,000 tonnes per year.

Considering a typical lifetime of a building of 50 +/-25 years and scenario C2 of [Earnshaw et al. 2013], it can be deduced that decaBDE in construction waste related to its use around 1970 starts to occur around now (2020) in C&D waste and will increase constantly over the following 20 years. As about 20% of the consumed decaBDE was used in the construction sector, we can expect a plateau of approximately 1,500 tonnes decaBDE per year to become waste via the C&D waste stream for the period from 2040 to 2060. Considering a quantity of 1,500 t of decaBDE in C&D
waste per year with the tonnage of plastic within C&D waste in EU per year results in an expected hypothetical decaBDE concentration within plastics from C&D waste of approximately 1,600 mg/kg.

Although the recycling of waste containing decaBDE above the LPCL (to be defined) will not be permitted, in management practice for C&D waste, no distinction is currently made e.g. between streams of brominated and non-brominated material.

Evaluation of limitation criteria

Figure 40 illustrates the results of the evaluation of the limitation criteria for decaBDE.

![Figure 40: Concentration range for an LPCL for decaBDE according to the evaluation of the limitation criteria](image)

According to the evaluation of the limitation criteria, an appropriate LPCL can be set in a range between 200 and 1,000 mg/kg.

The criteria which are possibly limiting the LPCL are (A) analytical potential, the (E) economic feasibility and the (LV) existing limit values:

(A) Analytical Potential

DecaBDE belongs to the group of BFRs. The analytical quantitative determination in plastic and textiles is performed using either (1) state-of-the-art laboratory techniques accompanied by matrix extraction and further clean-up methods or (2) XRF spectrometry.

State-of-the-art laboratory techniques (e.g. GC-MS, IA-MS). GC-MS and the two informative techniques are validated for the determination of PBDEs in electrotechnical products [IEC 62321-6:2015]. The advantage of GC-MS is the simultaneous analysis with other BFRs such as polybrominated diphenyl ethers (PBDEs) and thus GC-MS as a measurement can be used to verify the quality on a regular basis (e.g. monthly) and to check the efficiency of the sorting process [Plastic Recyc. Europe Sub. 2018]. As the process of measuring decaBDE via GC-MS is a relatively expensive and time-consuming method, it is more often used for scientific analyses than in daily practice.

XRF spectrometry is validated for a concentration for total Br in electrotechnical products [IEC 62321-3-1:2013]. This method has several advantages, as it is non-destructive and allows in-situ (on site) analasys, which makes time-consuming sample treatment obsolete. In addition, it is possible to buy handheld and/or portable XRF spectrometers. The disadvantage of this technique is, that the total bromine amount is determined. This means, that for example also other BFRs like, hexabromocyclododecane (HBCDD) and other potentially unknown brominated compounds are identified and/or quantified. In practice, the total bromine is regarded as a first indication for BFR-contamination of waste. Resulting in a differentiation between Br-positive and -negative samples.

The IEC 62321-6:2015 standard is evaluated for individual PBDEs between 20 mg/kg and 2,000 mg/kg in PC/ABS. An LOQ of 20 mg/kg can reasonable be assumed for decaBDE. Considering literature findings and reported measured concentrations within the standard IEC
62321-3-1:2013, an LOQ at or below 100 mg/kg for bromine is considered reasonably achievable. Making analogue considerations as for HBCDD, if the method results in a bromine content of 100 mg/kg it is ensured that the decaBDE content is below 120 mg/kg decaBDE as decaBDE consists of 83.3% Br (w/w). Accordingly, an LOQ for decaBDE of 120 mg/kg is considered achievable.

In summary it can be concluded that screening methods to check whether plastic or textile waste containing decaBDE above levels of 120 mg/kg are available. There are both standards for XRF and GC-MS available which are validated for 1,000 mg/kg [IEC 62321-3-1:2013; IEC 62321-6:2015]. For an effective implementation of an LPCL below a concentration of 1,000 mg/kg for decaBDE, the existing standards should be validated for concentration levels in line with the LPCL which will be selected. Considering that existing standards are evaluated regularly it is assumed that an adjustment of standards to measure lower concentration is under development and will be available in a few years.

Based on the information given in the results of criterion “analytical potential” for decaBDE given in section 6.2.1.1, several options for establishing an LPCL for decaBDE are possible:

- 1,000 mg/kg decaBDE related to an existing validated standard screening methods for XRF and GC-MS
- 500 mg/kg decaBDE related to existing reliable methods (GC-MS and XRF)
- 200 mg/kg\textsuperscript{76} decaBDE related to existing reliable methods (GC-MS and XRF)

Selecting 1,000 mg/kg as an LPCL for decaBDE corresponds to the current situation and would have no relevant impacts regarding possible (measurements to ensure) implementation based on reliable measurements. Implementation can be based on available screening and state of the art chemical analysis.

Selecting 500 mg/kg is considered an option which can currently be implemented without relevant impacts regarding implementation. Implementation can be based on the XRF screening method or the laboratory technique GC-MS.

Selecting 200 mg/kg is considered an option which can currently be implemented without relevant impacts regarding implementation. Implementation can be based on the XRF screening method or the laboratory technique GC-MS.

A relevant question is, whether a separate limit value should be established for decaBDE alone or a limit value should be established for the sum of all POP-BDEs (i.e. for tetrabromodiphenyl ether, pentabromodiphenyl ether, hexabromodiphenyl ether, heptabromodiphenyl and decabromodiphenyl ether).

The presented possible range for decaBDE could also be considered for a sum of all PBDEs as tetrabromodiphenyl ether, pentabromodiphenyl ether, hexabromodiphenyl ether, heptabromodiphenyl ether, heptabromodiphenyl are expected to occur in relevant waste streams together with decaBDE. A safety factor should be applied to the proposed decaBDE range for the inclusion of the already regulated POP-BDEs. This safety factors can be based on the average share of c-octaBDE and c-pentaBDE in the relevant waste streams. According to EERA, decaBDE, octaBDE and pentaBDE accounted for 22%, 7% and 0% respectively of the BFRs in WEEE plastics in 2011. Based on data derived from [Ineris 2017] and [IVM 2013] shares of c-octaBDE from 8 to 12.5 % still can occur in

\textsuperscript{76} 120 mg/kg would be possible; in order to increase the margin of safety a concentration of 200 mg/kg is suggested
WEEE. Lower values of c-pentaBDE are estimated for ELVs. Considering this information, a safety factor of 25% can be derived to take account of the other POP-BDEs contained in the relevant waste streams.

Considering this factor via XRF an LOQ of 150 mg/kg for the sum of the POP-BDEs is considered reasonably achievable. Hence, considering the analytical potential an LPCL range from 200 to 1,000 mg/kg can be recommended for decaBDE and/or the sum of the POP-BDEs including tetrabromodiphenyl ether, pentabromodiphenyl ether, hexabromodiphenyl ether, heptabromodiphenyl and decabromodiphenyl ether.

(E) Economic feasibility

The assessment of this criterion is not a complete assessment of socio-economic consequences. The assessment considers the economic impacts of changed treatment costs which accompany possible low POP concentration limits. The analysis of possible disposal operations can indicate how costs for the future disposal and recovery operations will change (e.g. changed costs when a distinct type of waste can no longer be landfilled but has to be incinerated in the future). Changed treatment costs exceeding 50 million € per year are considered critical. The assessment does neither take into account potential losses of employment or of capital in recycling activities or other negative or positive (e.g. reduced environmental of health costs) economic impacts.

Relevant changes in treatment costs occur at a hypothetical LPCLs of 100 mg/kg and 109 mg/kg (in total 98.45 million € per year) since these concentration levels are the estimated average decaBDE concentration levels in relevant WEEE related (low-Br-fraction) and ELV related (total SLF77) waste streams.

It is therefore not recommended to establish the LPCL below these limits. Considering a margin of safety an LPCL at or above 200 mg/kg for decaBDE or the sum of the POP-BDEs including decaBDE is recommended.

(LV) Limit Values

Under the RoHS Directive, the maximum concentration value tolerated for PBDEs is 0.1% (= 1,000 mg/kg) by weight in homogenous materials. The LPCL should not be established above that concentration limit.

Discussion of the recommendation

Against this background we conclude that a range of 200 mg/kg to 1,000 mg/kg for decaBDE or for the sum of POP-BDEs can be recommended as LPCL.

To decide where within this possible range the LPCL should be established, the following considerations should be taken into account:

The objective of the EU POP Regulation in its Article 1 is directly related to the precautionary principle. Applying the precautionary principle to the evaluation of the limitation criteria would require to reduce the limit value to the lowest limit value within the feasible range of options (here: 200 mg/kg).

On the other hand, the objective of the EU POP Regulation is related to feasibility. Therefore, considerations regarding e.g. economic, technical and other practical feasibility criteria and efficiency should be taken into account. In this respect an import question is related to efficiency, i.e. the question whether a lower LPCL provides significant health and environmental benefits in

Note that the decaBDE levels in the light fractions of the SLF are well below 100 mg/kg
terms of amounts of POPs removed and a reduction of POPs in end products and/or released to the environment.

Finally, the selected LPCL should consider other important policy objectives such as resource efficiency and circular economy.

Feasibility considerations

Compliance monitoring

Practical methods to measure and control the implementation of the LPCL in the recommended range are considered available at economically reasonable conditions (see section 6.2.1.1).

Regarding feasibility, another important question is related to where specifically the LPCL does apply. From a legal perspective it applies to waste. The most relevant wastes related to decaBDE are plastic and textile wastes. In a first instance the LPCL applies to large waste items or their components (e.g. cars, WEEE). These should be separated as far as reasonably possible if they contain decaBDE above the LPCL. Since, for the major share of relevant waste streams, this is not considered technically feasible, the large waste items are typically shredded.

After shredding, the question is: Does the LPCL apply in a second instance to an average concentration of the waste stream as a whole or to the concentrations in the individual particles within that waste stream (which varies from zero to functional concentrations)?

Existing separation methods in the WEEE and ELV SLF treatment can decrease the average concentration of specific waste streams (i.e. in the corresponding low-density fractions), the variation of the concentrations in the waste streams will, however, remain high. This is because the technically available automated identification and separation methods cannot sharply separate particles e.g. containing bromine at levels above or below 2,000 mg/kg bromine. There are always "mistakes" in separation and as a consequence some particles with high concentrations will always remain even in the low density waste fractions. As a consequence, the variation within the waste stream remains high and, if the LPCL applies to the individual particles within the waste stream, the whole waste stream will not comply with the LPCL.

Based on these considerations it might be reasonable to apply the LPCL to an average concentration of the whole waste streams. If the LPCL will apply to the average concentration of the waste stream, it can be selected at a lower level. If it will apply to the concentrations of the individual particles, it should be selected at a higher level.

Regarding the economic feasibility, costs for measuring based on GC-MS and XRF methods are indicated in section 6.2.1.1. Specific costs will, however, particularly depend on how the practice of monitoring and control to ensure compliance with the LPCL will be required by the competent MS authorities. Crucial issues in this respect are (1) frequency of measurements, (2) required method for measurements (i.e. GC-MS or XRF-screening) and (3) the sampling procedure for measurements.

In industrial recycling, in a first instance, large items which are known or suspected to contain decaBDE should be removed as far as reasonably possible. This should be based on existing knowledge or XRF-screening and take into account economic feasibility. For shredded materials and the subsequent processes, a pragmatic approach could be to take mixed samples at or close to the output streams of WEEE or ELV recycling processes which are intended for recycling (e.g. flakes or pellets resulting from the processing of the low-Br fraction from WEEE processing or the low density PST fraction from ELV processing) in a simple sampling procedure at regular intervals in order to control the bromine content of the samples. Samples at or close to the output of the process are appropriate to provide a good picture of an average concentration of decaBDE. If the bromine content of such samples indicates that the decaBDE content is well below the established LPCL, the material can be used as a recyclate and regular monitoring can continue, possibly at lower frequencies. If the bromine content indicates that the decaBDE content is close to or above
the established LPCL or shows high variations, the monitoring frequency should be increased, and it should be ensured, that batches with decabDE (or other POP-BDEs) above established LPCL should be removed and should be treated accordingly. GC-MS measurements should be initiated to identify the specific concentration of decabDE and, if relevant, action should be taken to decrease the input of decabDE-containing plastics to the process.

The output from recycling streams must comply with specific quality requirements (among other with the future UTC level which is to be established in Annex I of the POP Regulation). Hence, it is in the interest and duty of recyclers to ensure a high quality of recyclates which enables their use for the production, placing on the market and use in another life-cycle. Therefore, costs for sampling and measuring the content of substances of concern, including decaBDE, already arise in the frame of required quality management measures. Consequently, additional costs for monitoring and controlling the LPCL for decabDE are not expected or not considered relevant. Competent MS authorities should establish pragmatic and proportionate monitoring and control requirements.

**Disposal and recovery capacities**

Already in current or planned waste treatment of decabDE-containing waste, specific fractions containing higher loads of decabDE from WEEE and ELV are separated and the decabDE content is usually or can be destroyed. It is not expected that an LPCL within the recommended range will cause relevant quantities of waste which will have to be treated differently and would thus possibly lead to relevant economic effects. The assessment of disposal and recovery capacities (see 6.2.1.3) demonstrates that the existing treatment capacities within Europe are much more than sufficient to treat the waste quantities.

**Impacts on waste treatment costs**

Criterion E assesses the economic impacts of changed treatment costs which accompany possible low POP concentration limits (see section 6.2.1.4). Relevant cost impacts would be expected if the LPCL would be set at low concentration levels at which WEEE plastics and the ELV SLF would have to be treated as POP waste because they would exceed the possible LPCL. The hypothetical additional costs would a be around 40 million €/a and 98 million €/a respectively. If the LPCL will be selected within the recommended range, significant cost impacts due to changed waste treatment are not expected.

**Impacts on environmental and health costs**

It is generally considered that a significant decrease of releases of POPs to the environment contributes to reducing environmental or health costs which can be significant (see e.g. [Trasande et al. 2016; Grandjean and Bellanger 2017; Lam et al. 2017]), however, it is difficult to quantify the cost impacts. In the cost benefit assessment of the EU REACH restriction dossier on decabDE, it was not possible to estimate the benefits related to a specific emission reduction though the quantity of the emission reduction was known (see [ECHA 2016]).

**Other cost impacts**

Specific information on other cost impacts such as enforcement costs or compliance control costs are not available and are thus not considered within the report. This lack of information on potential costs associated with an adjustment of limit values was also stated within the RAC and SEAC Opinion for decabDE [ECHA 2015b].

**Efficiency considerations**

Usually in relevant shredded plastic waste the distribution of the concentrations is that only a low percentage of the particles contains bromine above 2,000 mg/kg which are responsible for the
most relevant quantity of bromine within the waste stream. Most particles contain bromine in a range from zero to 100 mg/kg (see example above provided by the French Ministry for an ecological and solidary transition). Accordingly, it can be expected that lowering the LPCL within the range of 200 to 1,000 mg/kg will not have a relevant impact on sorting. If for example the separation would not be made according to CENELEC at threshold of 2,000 mg/kg but of 1,000 mg/kg, only a few additional individual particles containing a concentration between 1,000 and 2,000 mg/kg would be separated compared to the CENELEC threshold.

As a consequence, significant environmental and health benefits would not be expected from lowering the LPCL within that range.

However, significant benefits could be achieved by a comparatively low LPCL if it would stimulate (1) better separation of relevant large waste items or components prior to shredding, and (2) improvements in the efficiency of separation techniques.

Consideration of other policy objectives

The declared political objective of European and national waste policy is on the one hand to promote a circular economy in order to save natural resources. On the other hand, waste policy also aims to guarantee the protection of human health and the environment. The provisions concerning the permissibility of waste treatment operations according to the waste hierarchy demonstrate the generally intended balance between the two goals: In those cases where recycling leads to a cycle of pollutants, which is undesirable from an environmental and health perspective, the hierarchical priority of recycling is no longer valid. In these cases, waste or the contained pollutants can and should be removed from the economic cycle in the sense of the protection of environment and health.

The balance between the two fundamental goals of waste policy has to be found especially concerning the POPs content in waste and mixtures and products as an output of recycling processes (recyclates), in particular through the determination of the pollutant-specific LPCLs. Whether the limit value for decaBDE will be chosen at the lower end of the suggested concentration range or in the upper end depends on the weighing of the mentioned policy objectives.

If absolute priority is given to the protection of human health and the environment, the LPCL could even be chosen below the suggested range at very low levels, in order to stop recycling of plastics from the relevant waste streams. If absolute priority is given to circular economy, the LPCL could be chosen above the suggested range, in order to facilitate corresponding recycling.

Policy objectives related to circular economy are e.g. specifically addressed in recycling quota such e.g. for WEEE and ELVs (see WEEE directive, Article 11; ELV Directive, Article 7). Under current waste management practice, the quota is typically achieved.

Regarding WEEE, we expect that the suggested range for the LPCL will not have a relevant impact on the quantity of recycling of plastics from WEEE (note: plastics content in WEEE typically ranges between 5 and 33%). Hence, we do not expect an impact on the achievability of the existing recycling quota. If the LPCL will be set at a level at which recycling of WEEE plastics will stop, it needs to be specifically assessed whether recycling quota can still be achieved for the different WEEE categories or whether they need to be adjusted.

Regarding ELVs, we expect that the recycling quota can still be achieved (note: plastics content in cars typically 9-12%) based on the following considerations: The recycling quota are currently a minimum of 95% reuse and recovery and 85% re-use and recycling. [Öko-Institut 2018] state that "PST is considered as a necessary operation to fulfil the recycling quotas defined by the ELV Directive." Öko-Institut 2018 recommend (in their section 4.6) for the relevant high-density fractions from PST use in blast furnaces for feedstock recycling and chemical treatment processes
or waste incineration. Against this background, it can be concluded that, using PST, existing recycling quota according to the ELV Directive can be achieved. Accordingly, [Öko-Institut 2018] concludes that the listing of decaBDE in the POP Regulation would not have an effect on the recovery rate, as the separated materials containing decaBDE can be treated in waste incinerators with energy recovery or in cement kilns as an energy carrier, or can be used as a reducing agent (feedstock recycling).

8.1.2 MAXIMUM POP CONCENTRATION LIMIT

DecaBDE itself is not considered to meet the toxicity criterion. It should be noted that the usual Predicted environmental concentration (PEC)/PNEC comparison methods cannot apply to this situation. Accordingly, it is not possible to derive direct adverse effects to human health or the environment from decaBDE.

However, debromination of BDE-209 in environmental matrices and biota to more persistent, toxic and bioaccumulative PBDEs including those already listed under the EU-POP Regulation ("old POP-BDEs") is considered to be of concern.

Due to debromination of c-decaBDE and past releases of commercial penta- and octabromodiphenyl ether organisms are often exposed to a multitude of PBDEs. On the basis of common modes of action (not fully established) and common adverse outcomes, there is concern that BDE-209 and other PBDEs may act in combination, in an additive or synergistic manner and induce developmental neurotoxicity in both humans and wildlife at environmentally relevant concentrations [UNEP/POPS/POPRC.10/10/Add.2 2014].

Based on previous evaluations, the MPCL for the "old" POP-BDEs was established at 10,000 mg/kg for the sum of the concentrations of tetraBDE, pentaBDE, hexaBDE and heptaBDE (see Annex V of the EU POP Regulation).

Based on the consideration that decaBDE contributes via debromination to the toxicity of the sum of all POP-BDEs, we suggest that decaBDE should be integrated in the MPCL at a level of 10,000 mg/kg for the sum of the concentrations of tetraBDE, pentaBDE, hexaBDE, heptaBDE and decaBDE.

8.1.3 RECOMMENDATIONS FOR ENVIRONMENTALLY SOUND MANAGEMENT

WEEE plastics

Recycling of WEEE

Recycling with prior treatment of WEEE plastics includes the manual disassembly or mechanical treatment in shredders. Plastic containing brominated flame retardants exceeding 2,000 mg/kg are separated and treated by combustion. However, plastic containing brominated flame retardants below this threshold is recycled. The decaBDE content is not destroyed and therefore it occurs in recyclates. This practice entails a further transfer of the decaBDE content into various plastic products, thus resulting in releases during further product cycles and end of life treatment and consequently in uncontrolled global distribution and possible releases of significant quantities of decaBDE. The risk can be minimised through a higher yield of separation of WEEE plastics containing decaBDE as far as possible or through stopping recycling of WEEE plastics.

If stopping recycling of WEEE plastics is not considered a desirable option, a higher yield of separation of WEEE plastics containing decaBDE should be envisaged. To this end two options are possible:
1. Increase the share of identification and separation of decaBDE-containing WEEE or WEEE components prior to the shredding process based on increased knowledge (information on relevant WEEE and WEEE components) and efforts (e.g. invest more time and effort in screening prior to shredding).

2. Increase the yield of separation in industrial separation processes based on improved identification and separation technology (e.g. invest in better separation technology or consider to adjust the CENELEC standard to a lower threshold e.g. 1,000 mg/kg bormine).

Considering the voluntary CENELEC treatment standard, separation and incineration as high Br fraction of plastics containing Br is done above a Br concentration of 2,000 mg/kg. The current status quo is appropriate to achieve an LPCL of 1,000 in a sum of the concentrations of tetraBDE, pentaBDE, hexaBDE, heptaBDE and decaBDE. There are however, as stated, significant quantities of POP-BDEs which are not destroyed. It could be evaluated if an adjustment of the CENELEC standard to separate a high-Br-fraction based on a concentration of 1,000 mg/kg Br. However, this option has to be evaluated for its technical and economic feasibility and efficiency (see above).

Pre-treatment techniques such as the CreaSolv® can be used to separate contaminated fractions as this process offers solutions for the removal of brominated flame-retardants from WEEE [Fraunhofer IVV 2018; Defra Sub. 2018]. Also, tailored ionic liquid solvent extraction of BFRs can be considered [Defra Sub. 2018].

Mechanical treatment in shredders can result in increased releases of PBDEs to the environment. A specific health or environmental risk due to exceeding critical environmental concentrations or exposure limits is not expected. It should be considered whether such releases are relevant and how corresponding releases could be reduced e.g. by appropriate technical measures and by separating brominated plastics from WEEE prior to shredding to the largest extent possible.

**Landfilling of WEEE plastics**

A share of 5.6% of plastics from WEEE (low bromine fraction) is landfilled. DecaBDE is not destroyed. As a consequence, there is a potential risk of ‘leaking’ to the surrounding environment. Health or environmental risks cannot be excluded and are represented in the long run especially due to the risk of uncontrolled global distribution and the corresponding risks for human health and the environment.

The landfilled share of plastic pertains to the low brominated fraction of WEEE plastics. The risk is therefore considered low. The risk should be minimised by avoiding the landfilling of plastics from WEEE treatment.

**Export of used EEE**

Used EEE is frequently exported from the EU. Adequate disposal or recovery leading to the destruction of the POP content is not necessarily ensured in the importing countries. Thus, the disposal of WEEE in the recipient countries and its contribution to global distribution may cause risks to human health and the environment [UBA 2010]. This risk can be minimised by a restriction of corresponding exports to countries, where appropriate recovery and disposal of WEEE are ensured.

**Plastics and textiles from ELVs**

**Recycling**

Due to recycling of SLF, a relevant share of the decaBDE content is not destroyed and therefore occurs in recyclates. This practice entails a further transfer of the decaBDE content into various plastic products, thus resulting in releases during further product cycles and end of life treatment and consequently in uncontrolled global distribution and possible releases of significant quantities of decaBDE. The risk can be minimised through a higher yield of separation of ELV textiles and
plastics containing decaBDE. This can be achieved by increasing the share of advanced PST of the shredder light fractions. Advanced PST enables to accumulate decaBDE rich fractions in the shredder residues with a higher density. High density fractions should be incinerated to destroy the decaBDE content. Moreover, tailored ionic liquid solvent extraction of BFRs, CreaSolv® or supercritical fluid technology can be used in recycling processes [Defra Sub. 2018; IPEN & Arnika Sub. 2018]. Supercritical waste oxidation and/or supercritical fluid technology allows the recycling of crosslinked polymers by selective decrosslinking reactions and could be a potential option for the destruction of decaBDE and other POPs [IPEN & Arnika Sub. 2018a].

Mechanical treatment in shredders can result in increased releases of PBDEs and should be minimised (see above).

Landfilling of ELV SLF

A significant share of the ELV SLF is landfilled. DecaBDE is not destroyed. There is the potential risk of ‘leaking’ to the surrounding environment. Health or environmental risks cannot be excluded and are represented in the long run especially due to the risk of uncontrolled global distribution and the corresponding risks for human health and the environment. The risk is considered low and can be further minimised by avoiding the landfilling of plastics SLF fractions.

Export of used cars

The considerations related to export of used EEE apply in a similar way the export of used cars.

C&D Waste

ESM of the plastic fraction of C&D waste is crucial to avoid future decaBDE releases via C&D waste since currently there is no identification and separation from C&D waste in place.

Relevant quantities of decaBDE-containing plastics from the construction sector are expected in the future (e.g. an estimated quantity of 1,500 t/a from 2040 to 2060). If they will not be identified, separated and the decaBDE content will not be destroyed there will be the risk of uncontrolled global distribution and the corresponding risks for human health and the environment from recycling and landfilling.

The risk can be minimised by identification and separation of decaBDE-containing plastics from other C&D plastics as far as reasonably possible and by destroying the decaBDE content. Appropriate destruction technologies are available.

It is therefore vital that for C&D waste, options to identify and separate decaBDE and other POP-BDE-containing plastics have to be further explored to make relevant progress in the ESM of C&D waste.

In general, the same considerations as for other decaBDE-containing plastic wastes apply (e.g. from WEEE or ELVs apply (see above).

WEEE, ELVs and C&D Waste

Key element three, the destruction or irreversible transformation of the PBDE content in waste is an important aspect of ESM of POP waste. Appropriate operations for the environmentally sound destruction and irreversible transformation of the PBDE content in waste are commercially available and should be applied. The revised technical guidelines on the environmentally sound management of waste under the Basel Convention [UNEP/CHW.12/5/Add.6/Rev.1 2015] lists several available destruction or irreversible transformation technologies including advanced solid waste incineration, cement kiln co-incineration and hazardous waste incineration, to name but a few. It has however to be noted that Polybrominated dibenzo-dioxins (PBDDs)/ Polybrominated dibenzofurans (PDBFs) can be generated from the combustion and incineration of POP-BDE waste. For further information on the general technical guidelines on the environmentally sound
management of wastes consisting of, containing or contaminated with persistent organic pollutants of waste please refer to the technical guidelines under the Basel Convention [UNEP/CHW.13/6/Add.1/Rev.1 2017].

8.1.4 CONTINUED NEED FOR THE DEROGATION PROVIDED FOR POP-PBDES

Annex I of the POPs Regulation provides a derogation for the recycling of the “old” POP-BDEs by means of a concentration level of 1,000mg/kg (0.1% by weight) for each individual POP-PBDE, allowed in articles and preparations when produced partially or fully from recycled materials or materials from waste prepared for re-use.

Since the listed POP-BDEs still occur in the relevant waste streams at significant levels, the exemption can still be justified. DecaBDE and octaBDE accounted for 22% and 7% respectively of the BFRs in WEEE plastics in 2011. Based on data derived from [Ineris 2017] and [IVM 2013] shares of c-octaBDE from 8 to 12.5 % still can occur in WEEE. Lower values of c-pentaBDE are estimated for ELVs.

Considering current typical levels, it could be considered to lower the corresponding concentration levels for the exemption from currently 1,000 mg/kg to 100 mg/kg per POP-BDE, i.e. for tetraBDE, pentaBDE, hexaBDE and heptaBDE [Hennebert and Filella 2017; Ineris 2017].

Based on the stakeholder input (see section 10.4 in the Annex) and considering our assessment we recommend that there is a continued need for the exemption but to consider to lower the concentration limit to 100 mg/kg with the intention, that recycling of relevant plastics should not be significantly hampered.

In its resolution of 13 September 2018 on implementation of the circular economy package: options to address the interface between chemical, product and waste legislation, the European Parliament states that “in accordance with the waste hierarchy, prevention takes priority over recycling and that, accordingly, recycling should not justify the perpetuation of the use of hazardous legacy substances” and that “the possibility to recycle materials containing substances of concern should only be envisaged when there are no substitute materials without substances of concern” and “considers that any such recycling should take place in closed or controlled loops without endangering human health, including workers’ health, or the environment” [European Parliament 2018]. Taking the view from the European Parliament, a derogation for recycling PBDEs is not appropriate anymore and could be abolished with possible relevant impacts on the recycling of relevant plastics such as from WEEE. In a strict interpretation, the Parliament’s position is an argumentation to establish the LPCL at the same level as the UTC in order to minimise recycling of POPs to the largest extent possible.
8.2 SCCPs

8.2.1 LOW CONCENTRATION LIMIT

The LPCL for SCCPs is currently set at 10,000 mg/kg. Based on the assessment of lower and upper limitation criteria and considering that the economic feasibility and the existing limit values are the main limitation factors, a range of >420 mg/kg to 10,000 mg/kg is proposed as LPCL. The rationale behind the recommendation is explained below:

Background and relevance in products and wastes

Based on the available information, it can be summarized that SCCPs are currently neither produced nor used in the EU (at least not above the limit values defined in Annex I of Regulation EU 2015/30). As the production in the EU ceased some years ago, remaining stocks are not likely to occur. Nevertheless, some of the historic uses, especially in products with typically long lifetimes, may still be relevant from the waste management perspective (including conveyor belts managed together with other rubber waste streams and sealants and adhesives managed together with construction and demolition waste). In addition, possible imports of new products and/or recycled products contaminated with or containing SCCPs cannot be generally excluded as such products are regularly detected during enforcement activities in EU member states. Further, also in connection to the previous statement, a possible contribution from other chlorinated paraffins to SCCPs concentrations has been controversially discussed together with the complex chemical analysis of SCCPs.

Evaluation of limitation criteria

Based on the detailed evaluation of the lower and upper limitation criteria in Chapter 6, a potential LPCL for SCCPs can be set in a range between >420 to 10,000 mg/kg (see Figure 41).

Figure 41: Concentration range for an LPCL for SCCPs according to the evaluation of the limitation criteria

The main criteria limiting the LPCL are on the one hand the economic feasibility (E) and on the other hand the existing limit values (LV). These will be briefly discussed in the following, whereas further detailed information is available in Chapter 6.

(E) economic feasibility

Selecting a limit value above 420 mg/kg (e.g. 500 mg/kg or 1,000 mg/kg), which is the estimated theoretical average SCCPs concentration in rubber waste in the EU, would imply comparably low changes in costs for waste management compared to the current situation. A limit value above 420 mg/kg would also not result in any disposal and recovery capacity bottlenecks in the EU.

At a low POP concentration limit below 420 mg/kg, theoretically about 1,430 kt/a of rubber waste, which is currently recycled and landfilled, would have to be managed alternatively (i.e. incinerated). This would correspond to an additional necessary capacity of incineration plants in the EU of about
1.9 % and should also not pose any capacity problems. However, the cost increase of more than 80 million € per year would, according to the methodology applied, not be economically feasible.

At this point it should also be mentioned, that a possible POP concentration limit for SCCPs above 420 mg/kg would still have an impact on several other waste streams in the EU. In particular, as imports of new products and/or recycled products contaminated with or containing SCCPs cannot be excluded (also in concentrations above the limit values defined in Annex I of Regulation EU 2015/30). Such products are detected during enforcement activities in EU member states and the measured concentrations range up to several thousand mg/kg. It is expected that most of those (typically low-cost) imported articles will be landfilled or incinerated anyways together with municipal solid waste, but a possible recycling of respective wastes cannot be generally excluded. Considering the large diversity of the imported articles potentially containing SCCPs and the fact that imports cannot be reliably traced, it was not possible to further investigate these waste streams in the present study. Consequently, it was also not possible to estimate associated additional costs for waste management.

(LV) existing limit values

According to Annex I of the POP Regulation, the following specific exemptions on intermediate use or other specifications are defined for SCCPs:

1. "By way of derogation, the production, placing on the market and use of substances or preparations containing SCCPs in concentrations lower than 1 % by weight or articles containing SCCPs in concentrations lower than 0.15 % by weight shall be allowed.

2. Use shall be allowed in respect of:
   a. conveyor belts in the mining industry and dam sealants containing SCCPs already in use before or on 4 December 2015; and
   b. articles containing SCCPs other than those referred to in (a) already in use before or on 10 July 2012.

3. Article 4(2) third and fourth subparagraphs shall apply to the articles referred to in point 2 above.

It is further stipulated that articles containing SCCP in concentrations lower than 0.15 % (1,500 mg/kg) by weight are allowed to be placed on the market and used, as this is the amount of SCCP that may be present as an impurity in an article produced with MCCPs.

Therefore, considering the existing limit values, it is not recommended to set the LPCL for SCCPs above 10,000 mg/kg (1 %) as specified in Annex I of the POP Regulation.

It also needs to be considered that waste streams containing ≥2,500 mg/kg SCCPs would in addition to their POP classification be also considered hazardous (according to Annex III of the EU WFD) and thus require a stricter control regime (e.g. additional labelling and identification documents, record keeping, monitoring and control obligation, etc.) and stricter requirements for waste treatment. This may also result in higher costs for waste management.

Discussion of the recommendation

Lowering the current LPCL for SCCPs from 10,000 mg/kg, to values higher 1,500 mg/kg, would theoretically have no impact on the historically relevant uses and related waste streams. Nevertheless, according to legislation, placing on the market of articles in concentrations higher than 0.15 % is not allowed, however, higher SCCP concentration (several thousand mg/kg) have been reported during enforcement activities in EU member states. Therefore, a possible limit value below 10,000 mg/kg could still have an impact on the produced and/or recycled products contaminated with or containing SCCPs entering the EU market. The potential impact cannot be quantified based on the currently available information but is likely linked to comparably low
changes in costs for waste management compared to the current situation and is not expected to result in bottlenecks regarding disposal and recovery capacities in the EU.

Lowering the LPCL further below 1,500 mg/kg (e.g., 1,000 mg/kg or 500 mg/kg) would restrict the placing on the market and use of articles containing SCCPs in concentrations lower than 0.15 % by weight and probably also the alternative MCCPs as this is the amount of SCCPs that may be present as an impurity in an article produced with MCCPs (according to the EU POP Regulation). This has also been stipulated by the Netherlands during the consultation phase for this project. As already mentioned, considering the large diversity of the imported articles potentially containing SCCPs and the fact that imports cannot be reliably traced, it was not possible to investigate associated waste streams in detail and estimate the potentially required costs for waste management. However, a limit value below 1,500 mg/kg would theoretically still have no impact on the management of waste streams resulting from historic applications (e.g., rubber from conveyor belts and sealants and adhesives).

A POP concentration limit below 420 mg/kg (e.g., 100 mg/kg as suggested by IPEN and Arnika) would result in an additional cost increase which is considered not economically feasible. All other implications described in previous section also apply here (e.g., related to the placing on the market and use of articles containing SCCPs in concentrations lower than 0.15 %). The waste disposal and recovery capacities are still estimated to be sufficient even at potential limit values below 420 mg/kg but are estimated not sufficient at values below 1 mg/kg.

8.2.2 MAXIMUM CONCENTRATION LIMIT

For SCCPs, considering that no indication was found that the current MPCL is inadequate, it is proposed to maintain the current value of 10,000 mg/kg, as established within the Regulation. This is also in accordance with the stakeholder consultation.

8.2.3 RECOMMENDATIONS FOR ENVIRONMENTALLY SOUND MANAGEMENT

SCCP are no longer deliberately produced or used in Europe. Continued presence of SCCPs in articles in new products are probably associated with imported articles and materials into the European market.

On the basis of former consumption quantities of SCCP, the relevant waste amounts were estimated (as further discusses in chapter 4.2). The most relevant waste streams are rubber waste from used rubber conveyor belts, sealants and adhesive from construction and demolition waste.

Rubber waste

Recycling of rubber waste

In Europe, rubber from used rubber conveyor belts is partially directed to material recovery. In this context, it cannot be excluded that SCCP might also be released into recyclates.

To minimise the risk, SCCP-containing rubber conveyor belts from underground mining should be separated as far as possible and directed to appropriate treatment operations (energy recover or incineration). Moreover, further environmental and health risks before the actual material recovery cannot be excluded.

As larger metal components would damage the shredders and mills, metal fraction of the conveyor belt should be separated from the rubber fraction as far as possible before the actual material recovery. This can result in dust generation and therefore presents a potential environmental and health risk. In order to minimise the risk, dust generation should be avoided as far as possible. The inhalation of contaminated dust can be avoided by the application of breathing masks.

Incineration of rubber waste
SCCP are thermally decomposed at about 200 °C [BiPRO 2011]. Due to high incineration temperatures (>800°C), it can be expected that the SCCP content in conveyor belts is destroyed during (hazardous) waste incineration.

A potential health or environmental risk may occur in case SCCP-containing rubber needs to be broken down mechanically prior to incineration (dust generation). Thus, dust generation and inhalation should be avoided as far as possible (application of breathing masks).

**Landfilling of rubber waste**

Only a small fraction is landfilled, SCCPs are thereby not destroyed and thus involves the risk that it may be released to the environment. Due to the comparably low SCCP amounts from this application however, no relevant environmental or health risks are expected. Separation of waste and separate treatment (e.g. incineration) to the largest extent possible can minimise the effect of transfer and distribution of SCCPs into the environment.

**Sealants and Adhesives**

**Recycling of C&D waste**

In Europe sealants and adhesives might partially end up in the material recovery fraction. In this context, it cannot be excluded that SCCP might also be released into recyclates and the environment as it is assumed to be treated with other mineral waste from construction and demolition.

The risk of releases can be minimised through separation of SCCP-containing sealants and adhesives as far as possible. However, currently, sorting and separation techniques specific to SCCPs are not available.

When removing sealants from buildings, fast running machines should not be used in order to avoid heat generation. In order to minimise the risk, dust generation should be avoided as far as possible. Inhalation of contaminated dust can be prevented by the use of breathing masks.

**Incineration of sealants and adhesives**

In case the sealants and adhesives can be successfully separated and is subsequently incinerated (including incineration as hazardous waste (e.g. in case of suspected PCB-containing joint sealants)), SCCPs are thermally decomposed at about 200 °C [BiPRO 2011]. Due to the high incineration temperatures (>800°C), it can be expected that the SCCP content in sealants and adhesives is destroyed during waste incineration.

**Landfilling of rubber waste**

Similarly, to rubber containing SCCPs, only a small fraction is of contaminated sealants and adhesives are landfilled. The same principle applies (see Landfilling of rubber waste).

**Other waste streams**

Other waste streams (e.g. resulting from consumer products) might be difficult to control as they mostly stem from imported products. It is expected that most of the relevant waste is managed together with MSW (largely incinerated and landfilled in the EU). However, it cannot be generally excluded that certain waste streams are also recycled (possibly leading to accumulation of SCCPs in new recycled products). A measure to control the concentration of import products and related waste however is only feasible if reliable analytical methods/standards are used to distinguish between MCCPs and SCCPs.

In any case, the measures on environmentally sound disposal proposed in the Draft technical guidelines on the environmentally sound management of wastes consisting of, containing or
contaminated with short-chain chlorinated paraffins should be considered [UNEP/CHW/OEWG.11/INF/10 2018].
8.3 HCBD

8.3.1 LOW CONCENTRATION LIMIT

HCBD is listed in the EU POP Regulation with an LPCL of 100 mg/kg.

Based on the assessments in the present report, a concentration range of 0.1 to 100 mg/kg for the LPCL is possible. There is no indication that there is the need to adjust the established concentration limit of 100 mg/kg HCBD and that there would be beneficial impact on environment or health from lowering the LPCL.

Based on these considerations 100 mg/kg HCBD is proposed as LPCL.

The rationale behind this recommendation is the following:

Background

In Europe the commercial production of HCBD ended in the late 1970s. As of 2012 all applications of HCBD in the European Union have ceased. HCBD is not known to be currently intentionally produced or used. The intentional production of HCBD in Europe ended in the late 1970s and is therefore not considered of relevance as a potential waste stream.

The remaining generation of HCBD is unintentional during the production of chlorinated solvents, plastics and magnesium as well as residues (ashes and slag) from incineration of municipal, clinical and hazardous waste and sludge from municipal and industrial sewage treatment. The relevance of such unintentional production for potential waste stream is discussed in the report. Wastes consisting of, containing or contaminated with HCBD may potentially be found in:

- Unintentional HCBD chemical production during
  - the manufacture of chlorinated chemicals,
  - the manufacturing of plastics, and
  - electrolysis in magnesium manufacturing;
- Residues (ashes and slags) from incineration of unintentional produced HCBD;
- Electrical transformers, heat exchangers, electrical hydraulic fluids, cooling and absorbent liquids;
- Other industrial electrical equipment, including removed lining (ebonite) and graphite electrodes from chlorine electrolysis cells;
- Rubber compounds;
- Agricultural insecticides, fungicides and biocides;
- Residues (ashes and slag) from incineration of municipal, clinical and hazardous waste
- Sludge from municipal and industrial sewage treatment

At EU level, no waste streams containing HCBD in relevant quantities or concentration levels were identified and thus no economic or adverse environment and health consequences are expected by maintaining the currently established LPCL.

Evaluation of limitation criteria

Figure 42 illustrates the results of the evaluation of the limitation criteria for HCBD.
According to the evaluation of the limitation criteria, an appropriate LPCL can be set in a range between 0.1 and 100 mg/kg. The criteria which are most relevant are the (B) background contamination and the (LV) existing limit values.

8.3.2 MAXIMUM CONCENTRATION LIMIT

HCBD is listed in the EU POP Regulation with an MPCL of 1,000 mg/kg. There is no indication that there is a need to adjust the established concentration limit and that there would be beneficial impact on environment or health from lowering the MPCL. Based on these considerations **1,000 mg/kg HCBD is proposed as MPCL**.

8.3.3 RECOMMENDATIONS FOR ENVIRONMENTALLY SOUND MANAGEMENT

Relevant waste streams were not identified.
8.4 Dicofol

8.4.1 LOW CONCENTRATION LIMIT

For dicofol no limit values are currently established under the EU POP Regulation. Based on the assessment of lower and upper limitation criteria a range of 3 mg/kg to 200 mg/kg is proposed as LPCL for dicofol. A possible LPCL of 50 mg/kg, which is well within the proposed range, and would align with the LPCLs currently established for other POP pesticides could be considered. The rationale behind this recommendation is the following:

Background and relevance in products and wastes

Dicofol is an organochlorine pesticide which is chemically related to dichlorodiphenyltrichloroethane. It is used as a miticidal pesticide and acaricide in many countries worldwide on a variety of fruits, vegetables, ornamental and field crops. However, the use has been largely restricted or prohibited in many countries and the use as a pesticide has markedly decreased.

Dicofol is no longer produced in EU member states since 2006. The withdrawal of dicofol of the authorisations for plant protection products containing that substance was agreed in 2008. National registration of dicofol was no longer possible after March 2009 and any transitional period granted by EU MS expired by 30 March 2010. It can therefore be expected that all the remaining stockpiles have been either exported and/or already been subject to controlled waste management. The absence of remaining stockpiles in the EU has been additionally confirmed by several member states, who submitted information on dicofol in response to a questionnaire conducted during this project. Further, no EU imports and exports of dicofol and dicofol-containing products have been indicated within the member states’ responses. Besides, there is no known occurrence of dicofol in the EU as well as a decreasing relevance and occurrence on the global scale, with bans being in place in various countries around the globe.

Evaluation of limitation criteria

Based on the detailed evaluation of the lower and upper limitation criteria in Chapter 6, a potential LPCL for dicofol can be set in a range between 3 mg/kg to 200 mg/kg (see Figure 43).

![Figure 43: Concentration range for an LPCL for dicofol according to the evaluation of the limitation criteria](image)

Limit values above 200 mg/kg are not recommended to avoid possible adverse effects on human health and the environment. A possible LPCL of 50 mg/kg, which is well within the proposed range, and would align with the LPCLs currently established for other POP pesticides could be considered. Theoretically, a lower limit value could be selected as well but not below the derived background contamination of 3 mg/kg.

8.4.2 MAXIMUM CONCENTRATION LIMIT

As in the case of the LPCL, a maximum concentration of 5,000 mg/kg, in alignment with other POP pesticides, could be also established for dicofol.
8.4.3 RECOMMENDATIONS FOR ENVIRONMENTALLY SOUND MANAGEMENT

As indicated in the previous section, dicofol is currently neither produced nor used in the EU. All the remaining stockpiles have been either exported and/or already been subject to controlled waste management. General recommendations for recovery and disposal operations are for instance available in the Basel Convention guidelines on POP pesticides [UNEP/CHW.13/6/Add.5/Rev.1 2017].
8.5 PFOA

8.5.1 LOW CONCENTRATION LIMIT

For PFOA, its salts and PFOA-related compounds no limit values are currently established under the EU POP Regulation. Based on the assessment of the lower limitation criteria it is not recommended to establish a potential LPCL below 1 mg/kg. Due to the limited data availability, it was not possible to derive any of the upper limiting criteria for PFOA, its salts and PFOA-related compounds. In the absence of upper limiting criteria, the same limit value as currently established for PFOS of 50 mg/kg could be considered. The rationale behind this recommendation is the following:

Background and relevance in products and wastes

PFOA, its salts and PFOA-related compounds belong to the family of perfluoroalkyl and polyfluoroalkyl substances (PFASs). PFASs, which can be degraded to PFOA are referred to as PFOA-related substances. PFOA, its salts and PFOA-related compounds have been used in many different applications and consumer products across sectors due to their physico-chemical properties. In recent years they have become subject to a number of regulations with bans, restrictions and phase-out dates. PFOA and its salts are mostly used as processing aids for the production of fluoroelastomers and fluoropolymers. PFOA-related compounds are widely used as surfactants and surface treatment agents e.g. in textiles, paper and paints and fire-fighting foams, and for the manufacturing of side-chain fluorininated polymers.

Literature findings indicate that manufactures of PFOAs and its salts primarily takes place in companies in China and South America. Production of PFOA, and its salts within the EU as well as in the U.S. and Japan has ceased. The EU will further restrict the manufacturing, placing on the market and use, which includes imports, of PFOA, its salts and certain related compounds as well as articles containing these substances with an exemption for certain uses under REACH. [ECHA 2015a] estimates that production outside of PFOA and its salts has however increased, with China tripling its production between 2004 and 2012. Up-to-date information on production volumes of PFOA, its salts and PFOA-related compounds is not available.

Due to their numerous technical properties PFOA, its salts and related compounds have been widely used in many applications including consumer products across various sectors in the EU. Further details related to different areas of application are provided in Chapter 4 of this report. However, quantitative information on used amounts in different application areas is limited or missing. Therefore, as far as possible, the relevance of PFOA, its salts and related compounds in products, wastes and recyclates in the EU was been at least qualitatively evaluated in Chapter 6 of this report. Waste streams associated with specific textiles (e.g. membranes and textiles for the protection of workers from risks to their health and safety), end of life vehicles and possibly also semiconductors deserve particular attention to avoid PFOA accumulation in products and recyclates.

Evaluation of limitation criteria

For PFOA, its salts and PFOA-related compounds no limit values are currently established under the EU POP Regulation. Based on the detailed evaluation of the lower and upper limitation criteria in Chapter 6, it is not recommended to establish a potential LPCL for PFOA, its salts and PFOA-related compounds below 1 mg/kg to ensure that the proposed limit value can be reliably measured applying currently available analytical methods (see Figure 44). Considering the limited quantitative information available for PFOA, its salts and PFOA-related compounds, no upper limitation criteria could be derived.
In the absence of any upper limiting criteria, the same limit value as for PFOS could be possibly considered, in particular 50 mg/kg for PFOA and 2,000 mg/kg for PFOA-related substances
(applying a factor 40 as proposed by the Committee for Risk Assessment (RAC) and similarly to the limit values established in the REACH regulation for unintentional production, i.e. 25 ppb for PFOA and its salts and 1,000 ppb for PFOA related substances).

![Figure 44: Concentration range for an LPCL for PFOA according to the evaluation of the limitation criteria](image)

The higher LPCL for PFOA-related substances can be justified by the fact that PFOA-related compounds only degrade to a certain and unknown extent to PFOA and because it is a concentration limit for the sum of PFOA-related compounds. However, it is important to point out that there is no clear indication whether the factor 40 is appropriate in this specific case and that the LPCL for PFOS might need to be adjusted/updated, considering that it has been established already some years ago (i.e. listed in the Stockholm Convention in 2009 and its Risk Profile was completed in 2006).

In this connection, the project team believes, that the approaches of the Zürich Statement (see Chapter 4 for further details) are important for future global action on PFASs and relevant for POP waste management in general. Especially the grouping approach is important for the formulation of possible future limit values for the entire PFAS families (potentially including PFOS, PFOA and PFHxS as well as their related substances). Further, the need for additional information is strongly supported by the project team, especially information and data on production and use of PFASs in different application areas is still very limited or in certain cases entirely missing, but essential for a well justified derivation of POP waste limit values.

It can be expected that EU member states will continue with or will initiate projects related to PFAS in articles, wastes and recyclates which will help to improve the current situation regarding the data availability for the EU.

### 8.5.2 MAXIMUM CONCENTRATION LIMIT

As in the case of the lower concentration limit, for PFOA and its salts a MPCL of 50 mg/kg and for the PFOA-related substances a MPCL for the sum with 2,000 mg/kg could be considered (the same as currently established for PFOS).

### 8.5.3 RECOMMENDATIONS FOR ENVIRONMENTALLY SOUND MANAGEMENT

Based on the currently available information and data, it was not possible to investigate waste streams in detail as it has been done for other substances in this report (and develop substance and mass flows). Nevertheless, typical waste management options have been discussed quantitatively in Chapter 4 and environmentally sound management options have been proposed in this report as far as possible. The main considerations are summarized in the following:

Most of the relevant waste streams are/will be typically incinerated. Controlled incineration with high temperatures (e.g., at 1,000°C) is effective to destroy PFOA and to prevent the formation of PFOA from the thermolysis of highly fluorinated polymers, e.g.:

- Firefighting foams waste
It can be expected that currently most AFFFs are typically incinerated in hazardous waste incinerators after their use stage. Recycling and/or recovery of AFFFs is not a common waste management operation applied in the EU.

- **Textile waste**
  Even though textiles might be collected, reused (exported) and/or recycled, it can be expected that most textiles are currently incinerated and landfilled (usually together with municipal solid waste). Textile recycling is currently not a common waste management operation in the EU (but might change in the future).

Other waste streams will be rather difficult to trace, for instance semiconductors (containing wastes):

- Various waste streams expected (e.g. in WEEE, ELV, MSW). As some of the waste streams may be recycled, it cannot be excluded that PFOA its salts and related compounds potentially accumulate in recyclates and new products.

Besides, limited information is available to assess several other relevant uses and wastes in the EU, e.g. automotive applications.

For PFOS, pre-treatment is mentioned as an important aspect of environmental sound disposal and methods should be selected based on the nature and type of the PFOS waste. This can also apply to PFOA waste. Incineration of hazardous waste, according to the general technical guidelines, is at least one of the destruction and irreversible transformation methods applicable for the environmentally sound disposal of wastes with a content of PFOS, its salts or PFOSF at or above 50 mg/kg and could also be applied for PFOA. For further information on the general technical guidelines on the environmentally sound management of wastes consisting of, containing or contaminated with persistent organic pollutants of waste please refer to the technical guidelines under the Basel Convention [UNEP/CHW.13/6/Add.1/Rev.1 2017].
8.6 PFHxS

8.6.1 LOW CONCENTRATION LIMIT

For PFHxS, its salts and PFHxS-related compounds no limit values are currently established under the EU POP Regulation. Based on the assessment of the lower limitation criteria it is not recommended to establish a potential LPCL below 1 mg/kg. Due to the limited data availability, it was not possible to derive any of the upper limiting criteria for PFHxS, its salts and PFHxS-related compounds. In the absence of upper limiting criteria, the same limit value as currently established for PFOS of 50 mg/kg could be considered. The rationale is explained following:

Background and relevance in products and wastes

Perfluorohexane sulfonic acid (PFHxS), its salts and PFHxS-related compounds belong to the per- and poly-fluoroalkyl substances (PFAS) group. The group of PFHxS, its salts and related substances comprises a large number of chemicals. A study published by the Norwegian Env. Agency identified about 80 commercially available compounds. PFHxS is one of the most frequently detected PFAS in human blood in the general population and has a very long half-life in humans (range 5.3–35 years). High concentrations of PFHxS have been detected in soil, ground and drinking water near airports or fire-fighting training sites, sludge and wastewater from waste water treatment plants, near PFAS/PFHxS production/usage plants and in leachate from landfills.

Due to their technical properties (thermal and chemical stability as well as the hydro- and oleophobicity), PFHxS, its salts and related compounds are widely used in many applications including consumer products. PFHxS has been used in similar application areas as other PFAS including PFOS and PFOA. Quantitative information on production and use is mostly missing or considered not reliable.

Evaluation of limitation criteria

For PFHxS, its salts and PFHxS-related compounds no limit values are currently established under the EU POP Regulation. Based on the detailed evaluation of the lower and upper limitation criteria in Chapter 6, it is not recommended to establish a potential LPCL for PFHxS, its salts and PFHxS-related compounds below 1 mg/kg to ensure that the proposed limit value can be reliably measured applying currently available analytical methods (see Figure 45). Considering the limited quantitative information available, no upper limitation criteria could be derived.

In the absence of any upper limiting criteria, the same limit value as currently established for PFOS could be considered, i.e. 50 mg/kg for PFHxS and 2,000 mg/kg for PFHxS-related substances.

\[
\begin{array}{c}
\text{PFHxS} \\
\text{-related Subs.} \\
0.01 \\
1 \\
50 \ (2,000)
\end{array}
\]

\[C_{\text{POP}} \ (\text{mg/kg})\]

Figure 45: Concentration range for an LPCL for PFHxS according to the evaluation of the limitation criteria

The higher LPCL for PFHxS-related substances can be justified by the fact that PFHxS-related compounds only degrade to a certain and unknown extent to PFHxS and because it is a concentration limit for the sum of PFOA-related compounds. As already indicated in case of PFOA...
also in this case it needs to be mentioned that there is no clear indication whether the applied factor of 40 is appropriate in this specific case and that the LPCL for PFOS might be outdated.

In general, additional information, especially on production and use of PFASs is still very limited or in the case of PFHxS mostly missing, but essential for a well justified derivation of POP waste limit values. It can be expected that EU member states will continue with or will initiate projects related to PFAS in articles, wastes and recyclates which will help to improve the current situation regarding the data availability.

8.6.2 MAXIMUM CONCENTRATION LIMIT

As in the case of the lower concentration limit, for PFHxS and its salts a MPCL of 50 mg/kg and for the PFHxS-related substances a MPCL for the sum with 2,000 mg/kg could be considered (the same as currently established for PFOS).

8.6.3 RECOMMENDATIONS FOR ENVIRONMENTALLY SOUND MANAGEMENT

Information on production and use of PFHxS, its salts and related compounds in the EU is very scarce and mostly qualitative. Based on the currently available information and data, it was not possible to investigate waste streams in detail as it has been done for other substances in this report (and develop substance and mass flows). Nevertheless, typical waste management options have been discussed quantitatively in Chapter 4 and environmentally sound management options have been proposed in this report as far as possible. The main considerations are summarized in the following:

Most of the relevant waste streams identified are or will in the future be typically incinerated, e.g.

- **Firefighting foams waste**
  It can be expected that most AFFFs are typically incinerated in hazardous waste incinerators after their use stage. Recycling and/or recovery of AFFFs is not a common waste management operation applied in the EU.

- **Carpets and Textile waste**
  Even though textiles might be collected, reused (exported) and/or recycled, it can be expected that most textiles are currently incinerated and landfilled (usually together with municipal solid waste). Textile recycling is currently not a common waste management operation in the EU (might change in the future).

For other relevant waste streams, it can be assumed that these will be collected by specialized companies and are subject to controlled waste management in the EU. It is not clear to what extent such waste streams are recycled or recovered in the EU, e.g.

- **Metal plating** (incl. exchangers in the chromium(VI) cycle, chromium hydroxide sludge from WWT, ion exchange resins...)

Other waste streams, for instance semiconductors in waste, will be rather difficult to control. Various waste streams are expected (e.g. in WEEE, ELV, MSW) to contain PFHxS, its salts and related compounds. As some of the waste streams may be recycled, it cannot be excluded that PFHxS its salts and related compounds potentially accumulate in recyclates and new products or find their way into the environment, especially as only limited information is available to assess the relevant uses and wastes in the EU (e.g. the use in automotive applications).
8.7 HBCDD

8.7.1 LOW CONCENTRATION LIMIT

HBCDD is listed in the EU POP Regulation with an LPCL of 1,000 mg/kg, subject to review by the Commission by 20 April 2019.

Based on the results of the present study a range of 100 to 1,000 mg/kg HBCDD is proposed as LPCL. The relevant standard (IEC 62321-3-1:2013) should be validated for concentration levels in line with the LPCL which will be selected. However, implementation can already be based on existing analytical methods.

The rationale behind this recommendation is the following:

Background

HBCDD entered the world market in the late 1960s and since then was used in insulation boards in the construction sector. The main use of HBCDD in the EU (90 %) was in EPS and XPS for the building sector. Approximately 6 % of HBCDD was used particularly for EPS packaging (but also for EPS and XPS insulation panels used for transportation vehicles and other applications). About 2 % of the total consumption of HBCDD was in HIPS for electronic products and articles. Around 2 % was used in textile coating. The consumption/demand of HBCDD in EPS/XPS outside the construction industry has stopped in 2014. HBCDD was used in the EU in EPS and XPS in the construction and building industry until 2016 in typical functional concentrations of 0.7 and 1.5% respectively.

HBCDD in waste streams stemming of HIPS from the electronic sector is still relevant at low levels but will decrease from around the beginning of the 2020s. The relevance of HBCDD in textile waste is already decreasing. Most relevant waste streams due to historic use, imports or cross-contamination are (1) EPS in construction, (2) XPS in construction and (3) EPS in packaging.

Due to past and current recycling activities HBCDD occurs in recycled plastics and products made thereof (including toys) in concentrations ranging from a few mg/kg up to more than one thousand mg/kg.

Waste management, relevance of waste flows and occurrence of HBCDD in the waste flows

EPS from construction and demolition waste

EPS is a coarse-pored foam or expanded polystyrene (EPS) which is mainly used in the construction sector as an insulating material. EPS for construction is related to 45% of the historic HBCDD consumption in the EU.

EPS and XPS are here not the primary target for the mechanical recycling of C&D waste. They are separated as far as possible from the mineral content and separately recovered or disposed. However, a clean separation in practice is very difficult due to the attachment to e.g. mineral components (concrete, brick, plaster, etc.). This leaves impurities in mineral fractions which enter the corresponding wastes and recyclates and an unquantifiable proportion of EPS waste is therefore recycled or disposed together with mineral construction and demolition waste.

Only a small proportion of the C&D plastic waste is related to EPS. EPS waste is mainly caused by the dismantling of buildings (insulation). In 2017, post-consumer EPS construction waste from C&D in the EU amounted to 98,600 tonnes. The different fractions are either separated during the dismantling process or in a subsequent process.

In the future, an increasing amount of HBCDD-containing products will leave the use stage and enter the waste management sector. As a consequence, the amounts of HBCDD contained in C&D
waste to be managed will increase. The bulk of this waste will be disposed of in the future with an expected peak around the 2050s or later.

EPS as insulation material can be recovered in mechanical (material) recycling, energetic recovery or feedstock recycling (e.g. in the CreaSolv® process). The latter process will be applied in a planned demonstration plant in Terneuzen, Netherlands which is planned to start operating in 2019.

In 2017, at EU level, of the quantity of 98,600 tonnes of the EPS C&D waste (containing 690 tonnes of HBCDD), 1.32% was recycled, 69.57% was incinerated with energy recovery and 29.11% was landfilled. These data demonstrate that the major share of HBCDD is destroyed in incineration (69.6%) but a small share of HBCDD is recycled (1.3%) and a significant share is landfilled (29.1%). Consequently, the corresponding HBCDD content (9.1 t + 200.9 t) is not destroyed and there is a risk that it will be partly released to the environment and HBCDD will continue to occur in recyclates and products made thereof.

XPS from construction and demolition waste

XPS is a fine-pored foam or extruded polystyrene (XPS) which is mainly used in the construction sector as an insulating material. In general, similar considerations as for EPS apply (see above). XPS waste from construction is related to 41.5% of the historic HBCDD consumption in the EU.

The collected waste quantity in 2017 amounted to approximately 33,000 t (containing around 637 t HBCDD). At EU level 70.5 % of XPS waste from construction is incinerated and 29.5 % are landfilled. No recycling takes places. Accordingly, the major share of HBCDD is destroyed in incineration (70.5%), however, a significant share is landfilled (29.5%). Consequently, the corresponding HBCDD content (145 t) is not destroyed and there is a risk that it will be partly released to the environment.

EPS packaging waste

At European level, about 5.6 % of the HBCDD consumption was used particularly for EPS packaging (at low relevance also for EPS and XPS insulation panels used for transportation vehicles, and other applications).

The last use of HBCDD in the EU was in 2014. Packaging is generally disposed of immediately after the use or within one year and should not enter the waste streams anymore in relevant quantities via domestic sources. However, it is reported that HBCDD-containing EPS packaging enters the EU and the EPS packaging waste streams via imports and also that EPS from construction cross-contaminate the packaging waste streams and thus recyclates.

Available data indicate, that median concentration levels of HBCDD in packaging waste are comparatively low (around 10 mg/kg). However, in packaging waste streams average concentration levels around 200 to 300 mg/kg are reported. It is assumed that the comparatively high HBCDD concentrations in the packaging waste flows are particularly a result of cross-contamination stemming from EPS from C&D waste. This is the case because EPS from C&D waste cannot always be differentiated from Packaging EPS and enters during collection or waste treatment the packaging waste flow. Hence, concentrations will vary between MS depending on the waste collection operations in place.

HBCDD-free EPS packaging waste can be easily mechanically recycled as EPS or PS, if collected separately. Usually the collection of EPS packaging waste is covered by the household and/or similar commercial waste collection. Although HBCDD-containing packaging could be sorted out before the actual waste treatment (e.g. with the XRF screening), this screening process is rarely undertaken.

Compared to the quantity of HBCDD EPS and XPS from construction, the quantitative relevance of HBDC in packaging is low: About 388.3 kt of EPS packaging waste containing an estimated
quantity of 3.9 tonnes of HBCDD was collected in 2017 in the EU. Thereof 33 % were mechanically recycled, 35 % were energetically recovered and 32 % were landfilled.

**Evaluation of limitation criteria**

Figure 46 illustrates the results of the evaluation of the limitation criteria for HBCDD.

![Figure 46: Concentration range for an LPCL for HBCDD according to the evaluation of the limitation criteria](image)

According to the evaluation of the limitation criteria, an appropriate LPCL can be set in a range between 100 and 1,000 mg/kg.

The criteria which are most relevant are the (A) analytical potential and the (LV) existing limit values:

**A) Analytical Potential**

To detect and quantify HBCDD in plastics, different relevant test methods exist: (1) simple XRF spectroscopy, (2) XRF spectroscopy including extraction and (3) state-of-the-art laboratory techniques (GC-FID, GC-MS, LC-MS, UHPLC- ESI-MS/MS).

According to industry stakeholders practically feasible test methods would allow reliable quantification of HBCDD at 1,000 mg/kg but not at 500 or 100 mg/kg.

However, the LOD of the XRF with extraction method can be calculated as $35 \pm 7$ mg/kg bromine, which corresponds to $47 \pm 9$ mg/kg HBCDD. This screening method enables to identify polymers which contain HBCDD concentrations below 100 mg/kg. If the method results in a bromine content below 60 mg/kg it is ensured that the HBCDD content is below 100 mg/kg HBCDD.

The internationally standardised GC-MS and LC-MS can be applied to detect the HBCDD component with a LOD of 10-100 mg/kg, but the test results take a few days.

Based on available information on test methods it can be stated that screening methods are available to check whether plastic waste contains HBCDD above or below levels of 100 mg/kg. An XRF screening standard is available which is validated for 1,000 mg/kg (IEC 62321-3-1). For an effective implementation of an LPCL below a concentration of 1,000 mg/kg for decaBDE, standards should be validated for concentration levels in line with the LPCL which will be selected.

A validated standard for HBCDD determination is under development and may be available by the end of 2019 validated for 1,000 mg/kg and possibly well below (possibly down to 50 mg/kg).

A GC-FID method which is economically available at global level with a detection limit of 35 mg/kg enables to ensure that a limit value of 200 mg/kg can be kept.

Based on this information the result for the criterion "analytical potential" for HBCDD provides for several options:

- 1,000 mg/kg HBCDD related to an existing validated standard screening method
- 200 mg/kg or above HBCDD related to an existing reliable method (GC-FID)
• 100 mg/kg or above HBCDD related to an existing screening method (rapid extraction and XRF)
• Possibly down to 50 mg/kg or above related to a planned validated standard (HPLC-MS) which may be available by the end of 2019 (planned IEC 62321-9)

Selecting 1,000 mg/kg as an LPCL for HBCDD corresponds to the current situation and would have no relevant impacts regarding possible implementation based on reliable measurements. Implementation can be based on available screening and state of the art chemical analysis. A validated standard (1,000 mg/kg) is expected by 2019.

Selecting 200 mg/kg or above is considered an option which can currently be implemented without relevant impacts. Implementation can be based on two screening methods (XRF, rapid extraction and XRF) and a state of the art chemical analysis. A validated standard (possibly appropriate for 200 mg/kg) is expected by 2019.

Selecting 100 mg/kg or above is considered an option with possible barriers in implementation based on the current status. An appropriate screening method is available (rapid extraction and XRF). Regarding chemical analysis, the available GC-FID method seems not appropriate. However, a validated standard (possibly appropriate for 100 mg/kg) is expected by 2019.

Selecting 50 mg/kg or above is an option where possible barriers in implementation are considered not acceptable at the current status because an appropriate screening method is not available. Elaborate methods for chemical analysis are available, however, at considerable costs. A validated standard (possibly appropriate for 50 mg/kg) may be available by 2019.

(LV) Limit Values

The LPCL for HBCDD should not exceed the already established LPCL at 1,000 mg/kg. It should be set a 1,000 mg/kg HBCDD or below. The relevant standard (IEC 62321-3-1:2013) should be validated for concentration levels in line with the LPCL which will be selected. However, implementation can already be based on existing analytical methods.

Discussion of the recommendation

Against this background we conclude that a range from 100 mg/kg to 1,000 mg/kg for HBCDD can be recommended as LPCL.

To decide where within this possible range the LPCL should be established, the following considerations should be taken into account:

Applying the precautionary principle to the evaluation of the limitation criteria would require to reduce the limit value to the lowest limit value within the feasible range of options (here: 100 mg/kg).

However, also considerations regarding e.g. economic, technical and other practical feasibility criteria and efficiency and other important policy objectives should be taken into account.

Feasibility considerations

Compliance monitoring

Practical methods to measure and control the implementation of the LPCL in the recommended range are considered available at economically reasonable conditions (see section 6.2.1.1).

The cost of measuring based on GC-MS and XRF, giving an indication about the economic feasibility, methods are indicated in section 6.2.1.1. Specific costs, however, will depend on how the practice of monitoring and control to ensure the compliance of the LPCL will be required by the competent MS authorities. Crucial issues in this respect are (1) frequency of measurements, (2)
required method for measurements (i.e. GC-MS or XRF-screening) and (3) the sampling procedure for measurements.

An important question with respect to the most relevant HBCDD-containing waste streams, is to what extent cross contamination from EPS C&D waste into packaging waste can be avoided. Current collection and waste treatment practices can lead to a contamination of packaging waste streams up to several 100 mg/kg. Hence, if the LPCL would be set at 100 mg/kg, it would have to be ensured that EPS waste from construction is better kept separate from EPS packaging waste. Otherwise, the desired recycling of HBCDD-free packaging would be hampered. If appropriate measures are taken (see recommendations for ESM below) it is considered that average concentrations in packaging waste streams well below 100 mg/kg can be achieved.

The same question as related to decaBDE (see feasibility considerations related to decaBDE) is also here relevant: Does the LPCL apply in a second instance to an average concentration of the waste stream as a whole or to the concentrations in the individual particles within that waste stream (which varies from zero to functional concentrations)? If the LPCL will apply to the average concentration of the waste stream, it can be selected at a lower level. If it will apply to the concentrations of the individual particles, it should be selected at a higher level.

Sampling procedures should be established. If the bromine content of such samples indicates that the content is well below the established LPCL, the material can be used as a recyclate and regular monitoring can continue, possibly at lower frequencies. If the bromine content indicates that the HBCDD content is close to or above the established LPCL or shows high variations, the monitoring frequency should be increased, and it should be ensured, that batches with HBCDD above established LPCL should be removed and should be treated accordingly. Action should be taken to decrease the input of HBCDD into the process. Since the output of recycling streams must comply with certain quality requirements, amongst them the UTC, it should be of interest to recyclers to ensure the quality of their recyclate to use within another life-cycle. Thus, cost for sampling and measuring to adherence to the LPCL should not be substantial. Similarly to decaBDE, competent MS authorities should establish pragmatic and proportionate monitoring and control requirements.

Disposal and recovery capacities

In the current and future waste treatment of HBCDD-containing waste, certain fractions with higher HBCDD loads from e.g. construction EPS and XPS are separated and the HBCDD content is destroyed. It is not expected that an LPCL within the recommended range will cause relevant quantities of waste which will have to be treated differently and would thus possibly lead to relevant economic effects. The assessment of disposal and recovery capacities (see 6.2.1.3) demonstrates that the existing treatment capacities within Europe are much more than sufficient to treat the waste quantities.

Impacts on waste treatment costs

Criterion E, which assessed the economic impacts of changed treatment costs which accompany possible low POP concentration limits (see section 6.2.1.4), established that no relevant cost impacts would be expected if the LPCL would be set at low concentration levels. The hypothetical additional costs to include EPS packaging with a median concentration of 10 mg/kg would be around 17.4 million €/a. As the proposed range lies above, a LPCL within the range will not lead to significant cost impacts due to changed waste treatment.

Impacts on environmental and health costs

It is generally considered that a significant decrease of releases of POPs to the environment contributes to reducing environmental or health costs which can be significant (see e.g. [Trasande et al. 2016; Grandjean and Bellanger 2017; Lam et al. 2017]). The difficulties, however, lie in the quantification of these cost impacts. Within the RAC and SEAC Opinion, the RAC also pointed out,
that considerable uncertainties were attached to the socio-economic analysis. As a result, RAC determined that it was not possible to conclude on the remaining risk reduced if the authorisation for the manufacturing of flame retarded EPS for building applications was not granted [ECHA 2015c].

Efficiency considerations

Considering that in collecting practice, HBCDD-containing EPS also from other applications, particularly from construction, enters the EPS packaging waste flow, a higher HBCDD concentration could be applied to calculate the mass flow. For example, recent results from the Netherlands indicate an average HBCDD concentration of 245 mg/kg in collected packaging waste. The comparatively high concentration is presumably due to the fact that EPS form construction enters the waste stream. Assuming a concentration of 250 mg/kg in packaging waste streams would mean that via packaging waste approximately 97 tonnes of HBCDD would be treated at EU level with significant shares being landfilled or recycled. This illustrates the importance to keep EPS from construction and from packaging separated from each other.

Lowering the LPCL e.g. to 500 mg/kg would not cause any impacts on EPS packaging waste streams (considering average concentrations).

Lowering the LPCL for HBCDD e.g. to 100 mg/kg would entail the obligation to better keep EPS and XPS wastes separate in order to ensure achievement of concentration levels in the EPS packaging waste stream which allow to recycle (i.e. not to destroy or irreversibly transform its POP content) the EPS packaging waste (i.e. below 100 mg/kg).

Usually EPS from construction contains HBCDD in a functional concentration of 7,000 mg/kg. Practical methods to identify EPS items containing HBCDD above 100 mg/kg are considered available (e.g. XRF screening with extraction).

A better separation of EPS construction and EPS packaging waste could significantly reduce the carryover of HBCDD via recycling into new products.

The intention by recyclers to achieve an average concentration in EPS packaging waste is already existing since the output from recycling processes should preferably be below 100 mg/kg, so that the recycled material can be directly used for the production, placing on the market and use in accordance with the unintentional trace contamination (100 mg/kg) laid down in Annex I of the EU POP Regulation.

Therefore, it is not expected that lowering the LPCL to 100 mg/kg will significantly change the current situation. It is however recommended to harmonise the LPCL with the UTC level and to foster a better separation of EPS construction and EPS packaging waste.

Consideration of other policy objectives

The considerations related to other policies explained in relation with decaBDE also apply for HBCDD. Since packaging HBCDD should principally be free of HBCDD, it is definitely desirable to recycle it. An LPCL which stimulates a better separation of EPS construction and EPS packaging waste in order to avoid cross-contamination is therefore recommendable.

8.7.2 MAXIMUM CONCENTRATION LIMIT

The MPCL in Annex V of the EU POP regulation is established at 1,000 mg/kg. There is no indication that the current MPCL is not appropriate. It is proposed to maintain the established concentration limit.
8.7.3  RECOMMENDATIONS FOR ENVIRONMENTALLY SOUND MANAGEMENT

EPS and XPS from construction and demolition waste

Recycling
In general, insulation panels from construction waste are not recycled. A comparatively small amount of HBCDD is unintentionally recycled in EPS/XPS from C&D waste, which is not separated from the rest of C&D waste, together with other polymeric materials.

POPs are usually neither destroyed nor generated in the recycling processes. Presumably small HBCDD quantities are preserved in the recyclates for long periods and involve the risk of a further transfer of the HBCDD content into various plastic products, thus resulting in uncontrolled global distribution and the corresponding risks for human health and the environment. A specific health or environmental risk due to the exceeding of critical environmental concentrations or exposure limits is not expected.

The risk can be minimised by separating EPS/XPS insulation materials to the largest extent possible in construction and destruction and by treating them in an appropriate way. During construction and destruction works, where EPS/XPS insulation panels are handled, dust generation should be avoided. It might be reasonable to apply personal protection measures, in order to avoid potential inhalation of HBCDD-containing dust.

An innovative process for the recycling of EPS containing HBCDD is CreaSolv®. The process consists of the CreaSolv® Technology and a Bromine Recovery Unit (BRU). In the process HBCDD is destroyed to a large extent while PS and bromine can be recycled and used for further applications. A planned demonstration plant will be located in Terneuzen, Netherlands and start operation in 2019.

Incineration
EPS and XPS insulation panels can be incinerated appropriately in state-of-the-art municipal solid waste incineration plants up to a share of 2 % of the total waste. Crucial for a professional incineration is the expert mixture in the bunker of the plant. For a share of 1 percent by weight (corresponding to about 15 percent by volume) the insulation panels can be handled without any problems in the incineration process.

During the product lifetime, nearly 100% of the HBCDD content remains in the insulation panels. During construction and demolition works, where EPS/XPS insulation panels are handled as well as during the handling of the boards for recovery and disposal, dust could be generated and cause an exposure via inhalation. In order to minimise environmental and especially health risks, dust generation should be avoided, and it might be reasonable to apply personal protection measures (breathing masks), in order to avoid potential inhalation of HBCDD-containing dust.

In the view of circular economy, it is recommendable to deviate EPS and XPS from incineration to appropriate chemical recycling processes.

Landfilling
A relevant share (almost 30%) of EPS/XPS panels containing HBCDD is landfilled. HBCDD is not destroyed. Long term risks for human health and the environment cannot be excluded.

The risk can be minimised through decreasing the landfill of EPS/XPS panels containing HBCDD by directing them to other appropriate treatment options.

EPS Packaging waste
A share of 5.6% of plastics from WEEE (low bromine fraction) is landfilled. DecaBDE is not destroyed. As a consequence, there is a potential risk of ‘leaking’ to the surrounding environment.
Health or environmental risks cannot be excluded and are represented in the long run especially due to the risk of uncontrolled global distribution and the corresponding risks for human health and the environment.

**Recycling**

About one third of EPS packaging waste is recycled. HBCDD is not used in EPS packaging anymore. However, in practice, cutting losses of HBCDD-containing EPS insulation materials are unintentionally mixed and recycled with packaging waste. Thus, waste which is not contaminated or contains only minor amounts of HBCDD (EPS packaging waste), is mixed with HBCDD-containing waste (EPS from C&D) and in consequence, contaminated recycled PS pellets are produced.

This practice entails a further transfer of the HBCDD content into various plastic products, thus resulting in releases during further product cycles and end of life treatment.

The risk can be minimised through a better separation of EPS waste from C&D and packaging EPS waste. To this end specific measures should be implemented to (1) raise the awareness about the necessity to separate EPS from construction at waste producers (consumers and professional users), recyclers and waste collection facilities and to (2) increase the use of screening methods to identify HBCDD-containing EPS at collection facilities and recycling facilities in order to achieve a better separation of EPS from construction.
8.8 PCBs and PCDD/Fs

8.8.1 LOW POP CONCENTRATION LIMIT

The review of the LPCL for PCBs showed, that there is a need to consider this group of substances in bilateral way. This means, that non-dioxin-like (ndl-) and dioxin-like (dl-) PCBs should be considered independently from each other. Only when this is done, it can be guaranteed that the POP-regulations addresses the toxicological characteristics in a scientific adequate manner and potential adverse outcomes on the environment and humans are prevented. These toxicological characteristics are based on the fact, that dl-PCBs exhibit toxicological features that are the same as of PCDD/Fs also this group is involved in the LPCL-considerations of dl-PCBs. In addition, the results of this report indicate, that the LPCL for PCDD/F of this group might be too high.

Therefore, the following recommendations are made

- For ndl-PCBs, there is no indication, that the LPCL (50 mg/kg) should be adjusted.
- DL-PCBs need to be addressed separately from ndl-PCBs, for this there are two options:
  - Integration of dl-PCBs in the TEQ based LPCL of PCDD/Fs (This value might be also lowered).
  - Establishment of an individual TEQ based LPCL for dl-PCBs, this LPCL might lay in the range of 10-times higher than the LPCL of PCDD/Fs, as the highest TEF of all dl-PCB is 0.1 for PCB 126.
- For PCDD/Fs there is indication that the LCPL of 15 µg TEQ/kg is too high. Based on the results of this study a lower value in the range for example of 5-10 µg TEQ/kg should be considered. In the case, that dl-PCBs are included in the LPCL of PCDD/Fs it might be reasonable to consider a value at the upper limit of this range.

The rationale behind this recommendation is the following:

**Background**

For PCBs and PCDD/PCDFs the LPCL review process was performed considering scientific (meaning new findings on adverse effects) and technical developments solely. Thus, the methodology to derive LPCLs of these substances was not applied to the full extent.

The group of dl-PCBs need to be treated with considerable caution based on the diverse toxicological features of a special set of congeners. These 12 congeners exhibit a coplanar structure where the two rings lay on the same level and are thus not twisted. This structural motive is similar to those of dioxin and relaunches the same toxic mode of action [Safe 1985]. These congeners were classified as carcinogenic to humans (Group 1) by the International Agency for Research on Cancer in 2014 [Lauby-Secretan 2016].

For PCDD/PCDFs there are indications, based on scientific input, that the current LPCL is too high to prevent potential adverse effect on environment. Information from studies prepared by the Swedish EPA in 2011 as well as information provided by IPEN indicates the need for reviewing and possibly adjusting the currently established limit values for PCDD/Fs. Further, possible impact / contribution of dl-PCBs in waste pointed out by the Swedish EPA and IPEN and therewith associated implications on PCDD/F limit values. According to the Swedish EPA, as long as dl-PCBs are not included in the LPCL for PCDD/Fs, the estimated risk associated to food chain transfer of pollutants in waste may be underestimated [SE EPA 2011].

According to a recently published assessment, EFSA’s expert Panel on Contaminants in the Food Chain (CONTAM) has completed the Authority’s first comprehensive review of the risks to human and animal health from these substances in food and feed. The European Commission asked EFSA for this risk assessment following its 2015 review of differences between tolerable intake levels set by various scientific advisory bodies. The Panel has set a new tolerable weekly intake (TWI) for dioxins and dioxin-like PCBs in food of 2 picograms*/ per kilogram of body weight. The new TWI is seven-times lower than the previous EU tolerable intake set by the European Commission’s former Scientific Committee on Food in 2001 [EFSA 2018].

The toxicity of the most harmful dioxin-like PCB may be overestimated. The expert panel would support a review of the TEFs for both dioxins and dioxin-like PCBs in light of new scientific data. If confirmed to be less toxic, this would reduce the concern for consumers [EFSA 2018].

This information is another indication based on new scientific evidence, that the risks related to PCDD/Fs where underestimated and that also dl-PCBs should be considered in the LPCLs in the POP regulation.

* Waste management, relevance of waste flows and occurrence of PCBs, dl-PCBs, and PCDD/Fs in the waste flows

Fly ashes from municipal solid waste incineration are considered an important mass flow. Approximately 3 million tonnes of fly ashes are generated each year in EU28.

Recent data on concentrations of PCDD/F in fly ash from European MSWI plants (35 observations) demonstrate that the average concentration was around 2.5 ng I-TEQ/g. 20 out of the 35 observations exceeded a concentration of 1 µg TEQ/kg, 7 values exceeded a concentration of 3 µg TEQ/kg, 4 values exceeded a concentration of 5 µg TEQ/kg and 1 value exceeded a concentration of 10 ng µg TEQ/kg. Hence, 11% and 3% of the fly ash would exceed an LPCL of 5 µg TEQ/kg and 10 µg TEQ/kg respectively.

In some European (e.g. NL) countries fly ashes are used as a second raw material for composite asphalt fillers. This application is considered to be safe and is in accordance with the European circular economy strategy. However, stakeholders claim when the LPCL of PCDD/Fs is lowered, the recycling would be hampered and that a proper data collection of PCDD/F content in fly ashes should be conducted before suggesting a new LCPL value, with a proper assessment of the impact on environment and human health when current practices are followed. However, if the LPCL would be lowered to of 5 µg TEQ/kg and 10 µg TEQ/kg respectively, the majority of the fly ash can still be used for this purpose. Assuming that the provided concentrations are representative for Europe with 2.5 ng I-TEQ/g only minor economic impacts for waste treatment operations are expected.

Reported LOQ for waste residues, ash, and slag is around 0.001 µg WHO-TEQ/kg (equals 0.001 µg/kg) [Eurofins 2013]. Stakeholders refer to the fact that the uncertainty linked to the determination of PCDD/Fs and PCB-like dioxins at those very concentrations is very high, due uncertainty of the instrumental analytical procedure itself and sampling, transport, storage, sample pre-treatment, etc. However, based on the fact that there is a factor in the range of thousands between the LPCL and the LOQ this uncertainty is not considered relevant.
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[UBA 2018b]  

[UBA Sub. 2018]  

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[UNECE 2015]  

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[UNEP/CHW/OEWG.11/INF/10 2018]

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[UNEP/POPS/COP.6/SC-6/13 2013]
United Nations Environment Program (UNEP), Conference of the Parties to the Stockholm

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[UNEP/POPS/COP.7/SC-7/1 2015]

[UNEP/POPS/COP.8/INF/18 2017]

[UNEP/POPS/COP.8/SC-8/10 2017]

[UNEP/POPS/COP.8/SC-8/11 2017]

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[UNEP/POPS/POPRC.12/11/Add.2 2016]

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10. ANNEX

10.1 Questionnaire for the informal stakeholder consultation

February 2018

QUESTIONNAIRE

"Study to Support the Review of Waste-related issues in Annexes IV and V of Regulation (EC) 850/2004"\(^1\)

1) Aim of the questionnaire

The following questionnaire has been prepared by BiPRO GmbH (part of Ramboll)\(^2\) in close coordination with the European Commission. The questionnaire aims at gathering up-to-date information and quantitative data on Persistent Organic Pollutants (POPs) and more specifically on waste related issues of certain “new POPs”, “candidate POPs” and “already listed POPs”. The information will provide the EU Commission with the necessary scientific basis to propose amendments to the POP Regulation (EC) 850/2004 (hereafter called “POP Regulation”), due to the listing of new substances and to the review of concentration limits for substances already listed. In addition, the study shall provide guidance on how wastes containing the new POPs may be managed.

2) Background information

POPs are a group of organic compounds that possess toxic properties, persist in the environment, bioaccumulate through the food web and pose a risk to human health and the environment. POPs are transported across international boundaries far from their sources through air, water and migratory species.

The “Protocol to the regional UNECE Convention on Long-Range Transboundary Air Pollution” (CLRTAP) and the Global “Stockholm Convention” on POPs are international, legally binding instruments aiming to reduce and eliminate the production, use and releases of POPs in the territories of all participating parties. Both contain provisions on the environmentally sound management of wastes consisting of, containing or contaminated by POPs (hereafter called “POP waste”).

Although substantial progress has been achieved in limiting the use and application of POPs and reduce their emission into the environment, there are ongoing releases into the environment as well as a constant cycling of substances released in the past. For an optimised approach to elimination, all sectors in the life cycle of a product and of anthropogenic emission sources need to be considered. In this framework, proper waste management can contribute substantially to the reduction of POP releases into the environment, and a comprehensive legislation on POP waste is a necessary pre-requisite.

The Stockholm Convention was implemented into EU Community law in 2004 by the POP

\(^1\) Information related to this project on behalf of the European Commission is provided on a dedicated project website at [http://popswaste.bipro.de](http://popswaste.bipro.de).

\(^2\) BiPRO GmbH (part of Ramboll), Munich, Germany (www.bipro.de), has been contracted by the European Commission to carry out the “Study to support the review of waste-related issues in Annexes IV and V of Regulation (EC) 850/2004”
Regulation. It foresees an obligation to generally destroy or irreversibly transform the POP content of waste above certain concentration limits (the 'low POP content'). In addition, in exceptional cases, waste above the limits may be otherwise managed with defined operations for specified waste types if destruction or irreversible transformation do not represent the environmentally preferable option and the concentration in such wastes are below another threshold (the 'high POP content').

In 2017, the Conference of the Parties of the Stockholm Convention (SC) decided to add three new substances to the relevant Annexes. Every time a substance is listed as a POP by the SC, the parties have to reflect the listing in domestic legislation. The EU as a party to the Convention is requested to amend the POP Regulation by May 2018 to include these 'new POPs'.

In addition, three substances are currently under review procedures and are likely to be added to the list of POPs under the SC in the next years (the so-called 'candidate POPs'). For the new POPs and the candidate POPs, there is a need to improve the knowledge basis regarding quantities that were used in the past, their concentrations and sources, as well as regarding aspects related to waste management in terms of disposal and recycling paths. This information is needed to assess possible disposal options and to establish concentration limits for waste. Consequently, further analysis is needed for the following substances:

- "new POPs": Decabromodiphenylether (decaBDE), short-chain chlorinated paraffins (SCCPs) and Hexachlorobutadiene (HCBD)
- "candidate POPs": Dicofol, Pentadecafluoroocanoic acid (PFOA, perfluoroocanoic acid) and its salts and PFOA-related compounds, Perfluorohexanoic acid (PFHxS) and its salts and PFHxS-related compounds

Furthermore, new scientific information on three substances already listed in the annexes of the SC has raised the necessity of reviewing already established concentration limits. Therefore, the following substances require renewed analysis and, potentially adjustment of the concentration limits:

- "already listed POPs": Hexabromocyclododecane (HBCD), Polychlorinated Biphenyls (PCB), Polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/PCDF)

4) Instructions on using the questionnaire

Please note that some of the relevant questions might have already been addressed in requests for information under the SC or the Basel Convention (BC). Submissions from EU member states related to these requests for information will be reviewed and considered as appropriate for the purpose of the actual study.

---

3 Article 7 of Regulation EC (No) 850/2004
4 Note that for SCCPs and HCBD concentration limits are already established in the Annexes IV and V of the POP Regulation. Only for decaBDE new concentration limits need to be established.
February 2018

The present questionnaire is provided as an MS-Word-file and it would be much appreciated to receive your responses using this electronic version. All questions are numbered and highlighted in grey. All fields where input is desired are blue.

The questionnaire is structured in three main sections.

- **Section I** – “new POPs”: decaBDE, SCCPs and HCBD
- **Section II** – “candidate POPs”: dicofol, PFOA and PFHxS
- **Section III** – “already listed POPs”: HBCD, PCB and PCDD/PCDF

Please only fill in the sections that you consider relevant and where you can provide specific information. Please provide specific references wherever possible and do not hesitate to also send us additional background information in form of documents, reports, data sets or as links to websites.

We recommend to quickly screen all questions of a section before starting to fill in the information. If a question is unclear or if you desire to discuss a certain aspect, please do not hesitate to contact one of the following contact persons of the project team:

<table>
<thead>
<tr>
<th>Contact person</th>
<th>E-mail address</th>
<th>Telephone</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mr Alexander Potrykus</td>
<td><a href="mailto:apot@ramboll.com">apot@ramboll.com</a></td>
<td>+49 89 978970-100</td>
</tr>
<tr>
<td>Mr Milos Mihunov</td>
<td><a href="mailto:mmo@ramboll.com">mmo@ramboll.com</a></td>
<td></td>
</tr>
</tbody>
</table>

All information provided will be used in a transparent and traceable way for the present study. Provided that your agreement is expressed below, submitted non-confidential information will be published on the dedicated project related website. If you wish to submit information on a confidential basis, please indicate this in your response. Any confidential information will only be used in anonymous or aggregated form.

Please indicate in the following table whether you agree with the publication of your answers to the Questionnaires:

<table>
<thead>
<tr>
<th>Section</th>
<th>Y/N</th>
</tr>
</thead>
<tbody>
<tr>
<td>I.I Occurrence of “new POPs” decaBDE, SCCPs and HCBD</td>
<td></td>
</tr>
<tr>
<td>I.II Recycling Operations/Waste Management Options for “new POPs”</td>
<td></td>
</tr>
<tr>
<td>I.III Concentration Limits for decaBDE, SCCPs and HCBD</td>
<td></td>
</tr>
<tr>
<td>II.I Occurrence of “candidate POPs” Dicofol, PFOA and PFHxS</td>
<td></td>
</tr>
<tr>
<td>II.II Recycling Operations/Waste Management Options for “candidate POPs”</td>
<td></td>
</tr>
</tbody>
</table>

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II.III. Concentration Limits for dicofol, PFOA and PFHxS

III. Already listed POPs

Please return the completed questionnaire and any related documents to popwaste2018@ramboll.com before 29 March 2018.

In case you are rather interested to discuss with us by telephone, please let us know when we could reach you.

Name of Institution:
Country:
City/ ZIP Code:
Street:
Contact person:
E-mail:
Phone:
Remark:
Section I – “new POPs”
Decabromodiphenylether (decaBDE); short-chain chlorinated paraffins (SCCPs); Hexachlorobutadiene (HCBD)

I. Occurrence of “new POPs” decaBDE, SCCPs and HCBD (articles in use, waste, recycled articles as well as (production) processes and unintentional releases)

1. a) Please indicate whether your country / company has stockpiles of decaBDE, SCCPs or HCBD.

   b) If yes, please provide information on types, quantity, concentrations, etc.

   a) 
   b) 

2. Please indicate any known occurrence and concentration of the “new POPs” decaBDE, SCCPs and HCBD in different articles in use, waste categories and recycled articles.

<table>
<thead>
<tr>
<th>decaBDE</th>
<th>Specification of waste/article</th>
<th>Concentration [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Articles in use</td>
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<tr>
<td>Wastes</td>
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<tr>
<td>Recycled articles</td>
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</tbody>
</table>

<table>
<thead>
<tr>
<th>SCCPs</th>
<th>Specification of waste/article</th>
<th>Concentration [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Articles in use</td>
<td></td>
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<tr>
<td>Wastes</td>
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<td>Recycled articles</td>
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</tbody>
</table>

<table>
<thead>
<tr>
<th>HCBD</th>
<th>Specification of waste/article</th>
<th>Concentration [mg/kg]</th>
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</thead>
<tbody>
<tr>
<td>Articles in use</td>
<td></td>
<td></td>
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<tr>
<td>Wastes</td>
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<tr>
<td>Recycled articles</td>
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</tbody>
</table>
3. Please provide information on quantities of waste containing “new POPs” that are currently generated, then disposed of or recycled

<table>
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<tr>
<th></th>
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<tbody>
<tr>
<td>decaBDE</td>
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<td></td>
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<tr>
<td>SCCPs</td>
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<tr>
<td>HCBP</td>
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</table>

4. Please provide information on wastes containing “new POPs” that are currently recycled (now or in the near future) and on the extent of recycling. If possible, please specify the types of new articles produced from the recycled material.

<table>
<thead>
<tr>
<th>&quot;new POPs&quot;</th>
<th>Types of waste recycled</th>
<th>Recycling rate [%]</th>
<th>New articles produced from recycled material</th>
</tr>
</thead>
<tbody>
<tr>
<td>decaBDE</td>
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<td></td>
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<tr>
<td>SCCPs</td>
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<tr>
<td>HCBP</td>
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</tbody>
</table>

Remarks:

5. Please indicate up-to-date (reference) measurement (analytical) methods for identifying the presence and levels of the listed “new POPs” in waste.
6. Please indicate known inexpensive **screening methods** for identifying wastes containing “new POPs”

7. Please indicate any known (production) processes using decaBDE, SCCPs and HCBD as well as options for the environmental management of their operation and potential related **unintentional releases of these POPs** into the environment.

II. Waste Management Options/Recycling Operations for “new POPs”

8. Waste separation for decaBDE, SCCPs and/or HCBD containing wastes:
   a) How can be **distinguished** between contaminated and non-contaminated waste?
   b) Which **separation operations** should preferably be used in practice to separate contaminated from non-contaminated waste (please provide further details if available)?
   c) What should be the preferred **waste management options for the contaminated waste fraction** (please provide justification and further details if available e.g. related costs)?

<table>
<thead>
<tr>
<th>Preferable separation operations</th>
<th>Relevant waste / new POP</th>
<th>Possible health risks for workers during separation of waste</th>
<th>Separation costs / ton of waste</th>
<th>Explanation / further information</th>
</tr>
</thead>
<tbody>
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<td>1.</td>
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<td>2.</td>
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</tbody>
</table>

Remarks:
### February 2018

<table>
<thead>
<tr>
<th>Preferable waste management operations</th>
<th>Relevant waste / new POP</th>
<th>Possible health risks for workers during waste management (e.g., recycling)</th>
<th>Management costs / ton of waste</th>
<th>Explanation / further information</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.</td>
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<td>3.</td>
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</tbody>
</table>

Remarks:

### III. Concentration Limits for decaBDE, SCCPs and HCBD

9. Are you aware of **any existing concentration limits for decaBDE in waste**?

---

10. Which **concentration limits for decaBDE in waste** according to the POP Regulation would you recommend? Please justify.

---

11. At which lower concentration limits for decaBDE in waste would you **expect relevant impacts** (e.g., on recycling industry)? Please justify.

---

12. Is there a **continued need for the derogation provided for POP-PBDEs in articles produced from recycled materials in the POPs Regulation** (i.e., level of 1,000 mg/kg or 0.1% by weight) of POP-PBDEs allowed in articles produced partially or fully from recycled materials? Please justify.

---

See Annex I Regulation (EC) No 850/2004
13. Is an adjustment of existing POP limit values for SCCPs\(^7\) and HCBD\(^8\), as specified in Annex IV and V of the EU POP Regulation, and/or additional measures required (e.g. due to any notable developments)? Please justify.

14. Please indicate if, beyond the EU POP Regulation, there are any adjustments to EU legislation needed, resulting from the listing of the "new POPs" decaBDE, SCCPs and HCBD under the Stockholm Convention.

15. Can you provide any other information or information sources relevant to Section I of this questionnaire on the "new POPs"?

---

\(^7\) Regulation (EC) 850/2004, Annex IV. Concentration limit referred to in Article 7(4)(a): 10 000 mg/kg; Maximum concentration limits of substance listed in Annex IV: 10 000 mg/kg

\(^8\) Regulation (EC) 850/2004, Annex IV. Concentration limit referred to in Article 7(4)(a): 100 mg/kg; Maximum concentration limits of substance listed in Annex IV: 1000 mg/kg
Section II – “candidate POPs”

Dioctyl, Pentadecylfluorooctanoic acid (PFOA, perfluorooctanoic acid), its salts and PFOA-related compounds, Perfluorohexanoic acid (PFHxS), its salts and PFHxS-related compounds

I. Occurrence of “candidate POPs” Dioctyl, PFOA and PFHxS (articles in use, waste, recycled articles as well as production processes and unintentional releases)

16. a) Please indicate whether your country/company has stockpiles of “candidate POPs” listed.

b) If yes, please provide information on types, quantity, concentrations, etc.

   a) 
   b) 

17. Please indicate any known occurrence and concentration of the “candidate POPs” in different articles in use, waste categories and recycled articles.

<table>
<thead>
<tr>
<th>a) Dioctyl</th>
<th>Specification of waste/article</th>
<th>Concentration [mg/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Articles in use</td>
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<tr>
<td>Wastes</td>
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<tr>
<td>Recycled articles</td>
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</tr>
<tr>
<td>b) PFOA, its salts and PFOA-related compounds</td>
<td>Specification of waste/article</td>
<td>Concentration [mg/kg]</td>
</tr>
<tr>
<td>Articles in use</td>
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<tr>
<td>Wastes</td>
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<tr>
<td>Recycled articles</td>
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<tr>
<td>c) PFHxS, its salts and PFHxS-related compounds</td>
<td>Specification of waste/article</td>
<td>Concentration [mg/kg]</td>
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<tr>
<td>Articles in use</td>
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<tr>
<td>Wastes</td>
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<tr>
<td>Recycled articles</td>
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</table>
18. Please provide information on quantities of waste containing “candidate POPs” that are currently generated, then disposed of or recycled.

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<tbody>
<tr>
<td>Dicofol</td>
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<tr>
<td>PFOA, its salts and PFOA-related compounds</td>
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<td></td>
</tr>
<tr>
<td>PFHxS, its salts and PFHxS-related compounds</td>
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</tbody>
</table>

19. Please provide information on wastes containing “candidate POPs” that are currently recycled (or possibly in the future) and the extent of recycling. If possible, please specify the types of new articles produced from the recycled material.

<table>
<thead>
<tr>
<th>“candidate POPs”</th>
<th>Types of waste recycled currently (or in the future)</th>
<th>Recycling rate [%]</th>
<th>New articles produced from recycled material</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dicofol</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PFOA, its salts and PFOA-related compounds</td>
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<td></td>
</tr>
<tr>
<td>PFHxS, its salts and PFHxS-related compounds</td>
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</tbody>
</table>

Remarks:

20. Please indicate up-to-date (reference) measurement (analytical) methods for identifying the presence and levels of the listed “candidate POPs” in waste.
21. Please indicate known inexpensive screening methods for identifying waste to be classified as POPs wastes due to their content of the listed “candidate POPs”.

22. Please indicate any known (production) processes using dicofol, PFOA and PFHxS as well as options for the environmental management of their operation and potential related unintentional releases of these POPs into the environment.

II. Waste Management Options/Recycling Operations/for “candidate POPs”

23. Waste separation for dicofol, PFOA and PFHxS containing waste:
   a) How can be distinguished between contaminated and non-contaminated waste?
   b) Which separation operations should preferably be used in practice to separate contaminated from non-contaminated waste (please provide further details if available)?
   c) What should be the preferred waste management option for the contaminated waste fraction (please provide justification and further details if available e.g. related costs)?

<table>
<thead>
<tr>
<th>Preferable separation operation</th>
<th>Relevant waste/candidate POP</th>
<th>Possible health risks for workers during separation of waste</th>
<th>Separation costs /ton of waste</th>
<th>Explanation / further information</th>
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**Remarks:**

**III. Concentration Limits for dicofol, PFOA and PFHxS**

**24. Are you aware of any existing concentration limits for dicofol, PFOA and PFHxS in waste?** (please list limits individually)

**25. Which concentration limits for dicofol, PFOA and PFHxS in waste according to the POP Regulation would you recommend? Please justify.**

**26. At which lower concentration limits for dicofol, PFOA and PFHxS in waste would you expect relevant impacts (e.g. on recycling industry)? Please justify.**

**27. Please indicate if, beyond the EU POP Regulation, there are any adjustments to EU legislation needed, resulting from the listing of the candidate POPs under the Stockholm Convention.**
28. Can you provide any other information or information sources relevant to Section II of this questionnaire on the “candidate POPs”? 
Section III – “already listed POPs”
Hexabromocyclododecane (HBCD), Polychlorinated Biphenyls (PCB), Polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/PCDF)

29. Are the existing concentration limits in Annex IV and V of the EU POP Regulation for HBCD, PCB and PCDD/F appropriate to ensure a sufficient level of environmental and health protection or is it necessary to adjust them (e.g. due to any notable developments such as new scientific data and technical progress, etc.)? Please justify.

30. If the existing limit values need to be adjusted, which concentration limits for HBCD, PCB and PCDD/F in waste would you recommend and why?

31. What would be the major impacts from a possible adjustment of existing limit values of Annex IV or V of the EU POP Regulation? Please justify.
10.2 Stakeholder response documentation

Table 77: Documentation of stakeholder responses

<table>
<thead>
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Submission under the questionnaire to the informal stakeholder consultation
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Submission on background information – extraordinary meeting of the Expert Group on the 26 October 2018

<p>|      | Ministry of Infrastructure and Environment, Netherlands | 5                         |       |      |       |         |      |      |       |     |        |
| 2    | EERA- Submission of EERA, EuRIC, PRE and BSEF          | 2                         |       | x    |      |         |      |      |       |     |        |
| 3    | European HBCDD Industry Group and EUMEPS              | 14                        |       |      |       |         |      |      |       |     |        |
| 4    | Ministry of Infrastructure and Wastemanagement, Netherlands | 2                         |       | x    | x    | x       | x    | x    | x     |     |        |
| 5    | Confederation of European Waste-to-Energy Plants (CEWEP) | 3                         |       |      |      |         |      |      |       |     |        |</p>
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<td>2</td>
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**Sum of submitted Documents: 112**
10.3 Stakeholder input related to other adjustments to EU legislation

According to the UK’s Department for Environment, Food and Rural Affairs (DEFRA), certain industry members argue that the REACH regulations need to be adjusted to reflect the new evidence on the new POPs. They believe keeping the 1,000 mg/kg threshold for total Bromine content does provide a simple easy monitoring system. However, the decaBDE congener needs to reflect the 100 mg/kg limit for the presence in articles (and substance and mixtures – i.e. recycled plastic compounds) [DEFRA Sub. 2018].

According to EERA, there is no need for further adjustments to EU legislations beyond the EU POP Regulation, because there is enough legislation and related confusion related to legislative texts around BFR as it is [EERA Sub. 2018b].

However, in a later submission of [EERA Sub. 2018c], who submitted for BSEF, EERA, EuRIC and Plastic Recyclers Europe, they did advocate that if there is a risk for children from exposure to toys, the toys directive allows to set specific consent of migration limits to prevent such an unacceptable exposure. A refinement to the REACH restriction to include childcare articles to reduce the concentration of decaBDE in such articles could be a viable option without eliminating the recycling industry, whilst protecting human health.

Romania states, that it is currently unknown whether there is a need for adjustments to EU legislation beyond the EU POP Regulation, resulting from the listing of decaBDE under the Stockholm Convention [RO MoE Subm. 2018].

According to EuRIC, there should be legal obligations related to eco-design. Recyclers have to deal with end-of-life product containing decaBDE, so manufacturers benefiting from exemptions under the Stockholm Convention who will continue to include decaBDE in articles should be obliged to inform recyclers of the decaBDE content and location of the parts containing decaBDE in the products. And further, bear the added cost of treatment at end-of-life linked to decaBDE [EuRIC Sub. 2018b].

The German Environment Agency (UBA) states that no adjustments beyond the EU POP Regulation result from the listing of decaBDE under the Stockholm Convention [UBA Sub. 2018].

According to the Finish Environment Institute, all EC waste legislation that sets specific recycling targets (ELV-directive 2000/53/EC, waste framework directive 98/2008/EU, WEEE-directive 2012/19/EU, etc.) should have a general derogation stating that all recycling targets are calculated from the material left after the removal of materials that are required to be disposed of in accordance with the POPs regulation. Further, this amendment should apply to all POP substances and not just the new POPs [FI Env. Institute Sub. 2018].

The Danish EPA states, that the classification of POP-waste as hazardous waste should be amended, so that POP waste is classified as hazardous waste when the concentration of POPs in waste is greater or the same as the Annex IV limit value and not at concentrations greater than the Annex IV limit value, as is the case today (cf. Commission Decision of 18 December 2014 amending Decision 2000/532/EC on the list of waste79). The reason being, that it will align the classification as hazardous waste with the requirement for destruction [Danish EPA Sub. 2018].

The Netherlands Ministry of Infrastructure and Watermanagement stated that no new adjustments are needed due to the decreasing presence [NL MoIW Sub. 2018].

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Spolchemie’s response indicated that there is the need to maintain the possibility to remove waste e.g. by combustion under defined conditions for chlorinated hydrocarbons [Spolchemie Sub. 2018].

10.4 Stakeholder input related to the need for the derogation provided for POP-PBDEs

According to Norway, there is no continued need for the derogation provided for POP-PBDEs in articles produced from recycled materials in the POPs Regulation. This is due to available data from a Norwegian report from [Swerea 2018]. According to the report, up-to-date practices for sorting out Br-containing plastics by use of X-ray fluorescence (XRF in combination with density separation or sensor based sorting result in low levels of POP-BDEs in plastics destined for recycling. Therefore, no derogation is needed for the level of POP-PBDEs in articles produced from recycled materials [NO EA Sub. 2018b].

Luxembourg also sees no continued need for the derogation provided for POP-PBDEs in the POPs Regulation. According to the submission, the derogation should be stopped as soon as possible to allow high quality upcycling opportunities instead of low quality downcycling of materials [LU Ministry of Sust. Dev. and Infrastructure Sub. 2018].

According to the EERA, there is a continued need for the derogation provided for POP-PBDE because the EU wants to stimulate a circular economy including the recycling of plastics and further wants to reduce CO2 emissions [EERA Sub. 2018b].

According to Romania, no information is available whether there is a continued need for the derogation. The Romanian POP-PBDEs inventory will be updated during their National Implementation Plan update [RO MoE Subm. 2018].

According to IPEN & Arnika, the derogation provided for POP-PBDEs in articles produced from recycled materials in the POPs Regulation was never needed. It was suggested by many studies and requested by some Parties to Stockholm Convention at last COP to end the derogation [IPEN & Arnika Subm. 2018]. The POPRC examination of the recycling exemption for the COP recommended in 2011 to “eliminate brominated diphenyl ethers from the recycling streams as swiftly as possible” noting that, “Failure to do so will inevitably result in wider human and environmental contamination and the dispersal of brominated diphenyl ethers into matrices from which recovery is not technically or economically feasible and in the loss of the long-term credibility of recycling.” [IPEN & Arnika Subm. 2018].

Furthermore IPEN & Arnika state, that updated information presented at COP 880 indicates that “developing countries also receive articles that may contain POP-BDEs in the form of second-hand/used goods or as wastes, originating mainly from developed countries. (...) It is estimated that at least 50 % of WEEE is collected outside of the official take-back systems in the EU, part of which is then exported to developing countries as used equipment or illegally. Illegal shipments originate mainly from Europe, North America, Japan and Australia the USA with common destinations in Asia (including China, Hong Kong, India, Pakistan and Vietnam) and Africa (including Ghana, Nigeria, and Benin). In addition to WEEE, plastics from WEEE are also reported to be exported to developing countries in Asia.” [IPEN & Arnika Subm. 2018].

80 UNEP/POPS/COP.8/7
According to the Netherlands, no incidents or non-compliance has been reported as long as the derogation provided for POP-PBDEs in articles is in place. As the POP-BDEs were prohibited 15 years ago, one could expect that the presence of articles containing POP-BDEs will decrease, depending on the lifecycle of those products [NL Sub. 2018].

According to Plastic Recyclers Europe (PRE) it would not be possible to recycle without the current derogation for POP-PBDEs. Any stricter limits would increase the burden on the recycling industry. Data available in some studies show that the concentration of PBDEs is decreasing. Therefore, in due time, limits could be lowered [Plastic Recyc. Europe Sub. 2018].

According to EuRIC, there is a continued need for the derogation provided for POP-PBDEs in articles, especially since this is also the level set in the REACH restriction currently in force for decaBDE in articles. It is neither technically nor economically viable to envisage a lower limit than 1,000 mg/kg, given the state of the art of sorting techniques currently existing, and the concentrations of brominated compounds in plastics in particular from electronic waste. It might be that this derogation might not be needed anymore once POP-PBDEs will have been phased out entirely, but as of today, and taking into account the existing exemptions granted under the Stockholm Convention for decaBDE (for automotive producers for example, running until 2030 – which means that decaBDE will be found in waste well after that date), such a derogation is fully justified. In addition, EuRIC opposes the continuation of derogation for decaBDE under the Stockholm Convention since such exemptions delay the complete phase out of regulated substances and create in legislation itself legacy issues [EuRIC Sub. 2018b].

According to the United Kingdom's DEFRA, the continuation of 1,000 mg/kg (0.1% by weight) of POP-PBDEs allowed in articles produced partially or fully from recycled materials does conform with the industry standard REACH for total Bromine testing using XRF methods. However – for the banned PBDE’s (tetra- octa penta-) industry evidence shows that a much lower level could be tolerated – perhaps down to 50 ppm per individual congener for five more years. For decaBDE certain industry members argue that a period with a 100 ppm allowed level in articles made from recycled plastic, will allow for all the trace levels of decaBDE in recycled extruded pellet compound to gradually reduce. A period of ten years is proposed to enable this to happen in the existing waste processing system for WEEE and ELV plastics (Polymers Recycling Industry in [DEFRA Sub. 2018]).

According to the Finish Environment Institute, it is not legally possible to have a general derogation in the EU POP Regulation for decaBDE in articles manufactured from recycled material because when decaBDE was listed in the Stockholm Convention81 no recycling exception was included for decaBDE. This principle was already confirmed when HBCDD was included in the EU POP Regulation82 and it did not have a recycling derogation in the Stockholm Convention. Finland does not see a need for a continued general derogation for tetra-, penta-, hexa- and heptaBDE in articles produced from recycled material because decaBDE is the most commonly detected BDE in plastic wastes and its concentration in recycled plastics is typically 10-100 times higher than the other POP-BDEs [FI Env. Institute Sub. 2018].

According to the Danish EPA, the level of 1,000 mg/kg (0.1% by weight) of POP-PBDEs allowed in articles produced partially or fully from recycled materials should be re-evaluated at fixed intervals to ensure that it reflects the latest scientific knowledge and newest technologies [Danish EPA Sub. 2018].

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81 Decision SC-8/10
82 Decision SC-6/13
According to the Austrian Ministry of Sustainability and Tourism, a further derogation is needed for decaBDE in articles produced from recycled materials (e.g. WEEE plastics, ELV plastics) in the POPs Regulation for an intermediate time period which should be reviewed. In the Stockholm Convention no exemptions are foreseen [AT Ministry of Sustain. and Tourism Sub. 2018].
### 10.5 Lowest observed background values

**Table 78: Lowest observed background values**

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<td>Schweden</td>
<td>[UNEP/POPS/POPRC.10/INF/5 2014]</td>
</tr>
<tr>
<td>SCCP</td>
<td>&lt;5-83</td>
<td>Sediment</td>
<td>Germany</td>
<td>[UNEP/POPS/POPRC.6/INF/15 2010]</td>
</tr>
<tr>
<td>HCBD</td>
<td>&lt;0.2 and 3</td>
<td>Sediment</td>
<td>Europe</td>
<td>[BiPRO 2011]</td>
</tr>
<tr>
<td>Dicofol</td>
<td>36</td>
<td>Mean sediment value</td>
<td>USA</td>
<td>[UNEP/POPS/POPRC.12/11/Add.1 2016]</td>
</tr>
<tr>
<td>PFOA</td>
<td>&lt;0.05 – 1.82</td>
<td>Soil</td>
<td>China</td>
<td>[ECHA 2018a]</td>
</tr>
<tr>
<td>PFOA</td>
<td>0.061 – 0.684</td>
<td>Sediment</td>
<td>Europe</td>
<td>[ECHA 2018a]</td>
</tr>
<tr>
<td>PFHxS</td>
<td>2.95–14.63</td>
<td>Soil</td>
<td>Africa</td>
<td>[UNEP/POPS/POPRC.14/INF/4 2018]</td>
</tr>
<tr>
<td>PFHxS</td>
<td>&lt;0.2-0.63 ng/g</td>
<td>Sediment</td>
<td>France</td>
<td>[UNEP/POPS/POPRC.14/INF/4 2018]</td>
</tr>
<tr>
<td>HBCDD</td>
<td>0.1 – 0.73</td>
<td>Sediment</td>
<td>Switzerland</td>
<td>[UNEP/POPS/POPRC.6/INF/25 2010]</td>
</tr>
<tr>
<td>HBCDD</td>
<td>nd – 0.05</td>
<td>Sediment</td>
<td>Norway</td>
<td>[UNEP/POPS/POPRC.6/INF/25 2010]</td>
</tr>
<tr>
<td>HBCDD</td>
<td>140</td>
<td>Mean soil value</td>
<td>Europe</td>
<td>[UNEP/POPS/POPRC.6/INF/25 2010]</td>
</tr>
</tbody>
</table>
### 10.6 Occurrence and concentrations of decaBDE in articles made from recyclate

#### Table 79: Occurrence and concentrations of decaBDE in articles made from recyclate

<table>
<thead>
<tr>
<th>Country/region</th>
<th>Product/product category</th>
<th>Product category</th>
<th>Concentration [mg/kg]</th>
<th>Evidence of other POP-BDEs</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Austria</td>
<td>Toy gun</td>
<td>Toy</td>
<td>147 - 359</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Austria</td>
<td>Fidget spinner</td>
<td>Toy</td>
<td>307</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Austria</td>
<td>Hair clip</td>
<td>Hair accessorie</td>
<td>132 - 458</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Austria</td>
<td>Hair brush</td>
<td>Hair accessorie</td>
<td>101</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Belgium</td>
<td>Telescope</td>
<td>Toy</td>
<td>660</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Belgium</td>
<td>Hair brush</td>
<td>Hair accessorie</td>
<td>57 - 86</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Belgium</td>
<td>Hair clip</td>
<td>Hair accessorie</td>
<td>26</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>United Kingdom</td>
<td>Black plastic kitchen utensils</td>
<td>Kitchen utensil</td>
<td>&lt; 0.0026 - 140</td>
<td></td>
<td>[Kuang et al. 2018]</td>
</tr>
<tr>
<td>Hungary, Canada, USA, Kyrgyzstan, Nepal and Thailand</td>
<td>Carpet paddings</td>
<td></td>
<td>1 - 166</td>
<td></td>
<td>[DiGangi et al. 2011]</td>
</tr>
<tr>
<td>Czech Republic, Slovakia</td>
<td>Combs and headdress</td>
<td>Hair accessorie</td>
<td>0 - 195</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Italy, Czech Republic, Germany</td>
<td>Different toys</td>
<td>Toy</td>
<td>210 - 9,225</td>
<td></td>
<td>[Guzzonato et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Gun</td>
<td>Toy</td>
<td>511</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Kitchen set</td>
<td>Kitchen utensil</td>
<td>24</td>
<td>no</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Pastry brush set</td>
<td>Kitchen utensil</td>
<td>24</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Knife</td>
<td>Kitchen utensil</td>
<td>9 - 195</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Fork</td>
<td>Kitchen utensil</td>
<td>144</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Concentration [mg/kg]</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
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</tr>
<tr>
<td>Czech Republic</td>
<td>Scissors (handle, rivet)</td>
<td>Kitchen utensil</td>
<td>&lt;LOQ - 4</td>
<td>no</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Comb</td>
<td>Hair accessory</td>
<td>26</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Eye-glasses</td>
<td>Other</td>
<td>1</td>
<td>no</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Coat hanger</td>
<td>Other</td>
<td>302</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Stapler</td>
<td>Other</td>
<td>652</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Hair accessory</td>
<td>Hair accessory</td>
<td>6.43 – 1,402</td>
<td>yes</td>
<td>[Straková et al. 2017a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Transformer</td>
<td>Toy</td>
<td>7.03 – 42.33</td>
<td>yes</td>
<td>[Straková et al. 2018b]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Car</td>
<td>Toy</td>
<td>52.34</td>
<td>yes</td>
<td>[Straková et al. 2018b]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Shoe toy</td>
<td>Toy</td>
<td>269.98</td>
<td>yes</td>
<td>[Straková et al. 2018b]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Cube (mole)</td>
<td>Toy</td>
<td>2,234.12</td>
<td>yes</td>
<td>[Straková et al. 2018b]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>344.24 – 1,194.71</td>
<td>yes</td>
<td>[Straková et al. 2018b]</td>
</tr>
<tr>
<td>Czech Republic, Netherlands</td>
<td>Other plastic toys</td>
<td>Toy</td>
<td>1 - 145</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Plastic polymers in WEEE/ELV</td>
<td>EEE/articles</td>
<td></td>
<td>0.2 - 590</td>
<td></td>
<td>[UBA Sub. 2018]</td>
</tr>
<tr>
<td>produced in Czech Republic</td>
<td>Plastic roofing (recycled PVC)</td>
<td></td>
<td>21.06</td>
<td></td>
<td>[IPEN &amp; Arnika Sub. 2018a].</td>
</tr>
<tr>
<td>Denmark</td>
<td>Magic cube star</td>
<td>Toy</td>
<td>4</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Denmark</td>
<td>Fidget cube</td>
<td>Toy</td>
<td>16</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Denmark</td>
<td>Toy gun</td>
<td>Toy</td>
<td>3 – 33</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Denmark</td>
<td>Diadem</td>
<td>Hair accessory</td>
<td>2</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Concentration [mg/kg]</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
</tr>
<tr>
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<tr>
<td>Denmark</td>
<td>Hairclip</td>
<td>Hair accessorie</td>
<td>71</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>France</td>
<td>Toy gun</td>
<td>Toy</td>
<td>1,043</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>France</td>
<td>Musical mobile phone</td>
<td>Toy</td>
<td>21</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>France</td>
<td>Water gun</td>
<td>Toy</td>
<td>322</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>France</td>
<td>Diadem</td>
<td>Hair accessorie</td>
<td>2 - 293</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Hairclip</td>
<td>Hair accessorie</td>
<td>718</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Formula 1</td>
<td>Toy</td>
<td>33</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Magic cube</td>
<td>Toy</td>
<td>25 - 442</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Toy gun</td>
<td>Toy</td>
<td>247</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Hair brush</td>
<td>Hair accessorie</td>
<td>30 - 93</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Comb</td>
<td>Hair accessorie</td>
<td>17</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Mini hair claws</td>
<td>Hair accessorie</td>
<td>&lt;LOQ</td>
<td></td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Massage roller</td>
<td>Other</td>
<td>178</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Germany</td>
<td>Waste bin</td>
<td>Other</td>
<td>165</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Water gun</td>
<td>Toy</td>
<td>&lt;LOQ</td>
<td></td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Diadem</td>
<td>Hair accessorie</td>
<td>20</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Hair clip</td>
<td>Hair accessorie</td>
<td>569</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Poland</td>
<td>Car</td>
<td>Toy</td>
<td>118</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Concentration [mg/kg]</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
</tr>
<tr>
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<tr>
<td>Poland</td>
<td>IQ cube</td>
<td>Toy</td>
<td>624</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Poland</td>
<td>Diadem</td>
<td>Hair accessorie</td>
<td>6 - 84</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Poland</td>
<td>Hair clip</td>
<td>Hair accessorie</td>
<td>97</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Poland</td>
<td>Hair brush</td>
<td>Hair accessorie</td>
<td>25</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Portugal</td>
<td>Chess tray</td>
<td>Toy</td>
<td>1,494</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Portugal</td>
<td>Small guitar</td>
<td>Toy</td>
<td>3,310</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Portugal</td>
<td>Comb</td>
<td>Hair accessorie</td>
<td>21 - 37</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Portugal</td>
<td>Diadem</td>
<td>Hair accessorie</td>
<td>2,491</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Spain</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>152</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Spain</td>
<td>Toy gun</td>
<td>Toy</td>
<td>598</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Spain</td>
<td>Water game</td>
<td>Toy</td>
<td>571</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Spain</td>
<td>Hair clip</td>
<td>Hair accessorie</td>
<td>764 - 853</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Spain</td>
<td>Diadem</td>
<td>Hair accessorie</td>
<td>898</td>
<td>yes</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Sweden</td>
<td>Toy camera</td>
<td>Toy</td>
<td>&lt;LOQ</td>
<td></td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Sweden</td>
<td>Toy</td>
<td>Toy</td>
<td>&lt;LOQ</td>
<td></td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Sweden</td>
<td>Diadem</td>
<td>Hair accessorie</td>
<td>&lt;LOQ - 8</td>
<td>no</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Sweden</td>
<td>Comb</td>
<td>Hair accessorie</td>
<td>&lt;LOQ - 2</td>
<td>no</td>
<td>[Straková et al. 2018a]</td>
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<tr>
<td></td>
<td>Recycled articles</td>
<td></td>
<td>&lt;1,000</td>
<td></td>
<td>[Plastic Recyc. Europe Sub. 2018]</td>
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<td></td>
<td>Recycled polymer products</td>
<td></td>
<td>&lt; 5</td>
<td></td>
<td>[DEFRA Sub. 2018]</td>
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<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Concentration [mg/kg]</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
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<tr>
<td></td>
<td>Axpoly ABS</td>
<td></td>
<td></td>
<td></td>
<td>[DEFRA Sub. 2018]</td>
</tr>
<tr>
<td>Recycled polymer products Axpoly ABS</td>
<td>Non-Detect</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>26 countries worldwide</td>
<td>Rubik’s and Rubik’s like cubes</td>
<td>Toy</td>
<td>0 - 672</td>
<td></td>
<td>[DiGangi and Straková 2016; DiGangi et al. 2017]</td>
</tr>
<tr>
<td>(7 EU countries)</td>
<td>Rubik’s and Rubik’s like cubes</td>
<td>Toy</td>
<td>0 - 400</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Thermal cup</td>
<td>Kitchen utensil</td>
<td>6</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>EU</td>
<td>Thermal cups</td>
<td>Kitchen utensil</td>
<td>775 - 778</td>
<td></td>
<td>[Guzzonato et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Toys</td>
<td>Toy</td>
<td>7.03 – 2,234</td>
<td></td>
<td>[Straková et al. 2017a; Straková et al. 2017b]</td>
</tr>
<tr>
<td>International references 2009-2011</td>
<td>Various recycled articles</td>
<td></td>
<td>100 - 780 (0.01-0.78%)</td>
<td></td>
<td>[SE EPA Sub. Add. Info 2018]</td>
</tr>
<tr>
<td></td>
<td>Vehicle components made of recycled materials</td>
<td>Vehicle component</td>
<td>No DecaBDE contained in articles produced after mid-2018 and made of recycled materials.</td>
<td></td>
<td>[DEFRA Sub. 2018; ACEA Sub. 2018a]</td>
</tr>
<tr>
<td>Argentina</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0 – 359</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Bangladesh</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>33 – 96</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Belarus</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>134 – 153</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Brazil</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>1 – 6</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Canada</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>20 - 297</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>China</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>2 – 36</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>2 – 96</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Germany</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>3 – 4</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Hungar</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0 – 58</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Concentration [mg/kg]</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
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<tr>
<td>India</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0 – 516</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0 – 63</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Japan</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>1 -17</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Kenya</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>18 – 171</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Mexico</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>17 – 152</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Nepal</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>19 – 234</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Philippines</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>5 – 293</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Poland</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0 - 79</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Russia</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0 – 217</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Serbia</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>36 – 47</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Slovakia</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>98</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>South Africa</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>98 – 281</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Sri Lanka</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>44 – 131</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Sweden</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Thailand</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>21 – 23</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>UK</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>10 - 400</td>
<td></td>
<td>[DiGangi et al. 2017]</td>
</tr>
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</table>
### 10.7 Occurrence and concentrations of decaBDE in material, articles and waste for/of EEE

**Table 80:** Occurrence and concentration of decaBDE in material, articles and waste for/of EEE

<table>
<thead>
<tr>
<th>Country/ region</th>
<th>Product/product category</th>
<th>Product category</th>
<th>Waste category</th>
<th>Production year and country of origin</th>
<th>decaBDE content (mean*/range) [mg/kg]</th>
<th>Detection frequency</th>
<th>Evidence of other POP-BDEs</th>
<th>Reference</th>
<th>Comment/further info</th>
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<tbody>
<tr>
<td><strong>Material</strong></td>
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<td></td>
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</tr>
<tr>
<td>Switzerland</td>
<td>FR2 laminates</td>
<td>EEE/material</td>
<td>WEEE</td>
<td></td>
<td>36 g/m²</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2003]</td>
<td>Assumptions based on literature; more data available on EEE and household appliances, but only assumptions and extrapolations</td>
</tr>
<tr>
<td>Switzerland</td>
<td>PE insulating foam</td>
<td>EEE/material</td>
<td>WEEE</td>
<td></td>
<td>20,000</td>
<td>no</td>
<td></td>
<td>[Morf et al. 2003]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>PE plastic sheeting</td>
<td>EEE/material</td>
<td>WEEE</td>
<td></td>
<td>100,000</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2003]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>PP plastic sheeting</td>
<td>EEE/material</td>
<td>WEEE</td>
<td></td>
<td>100,000</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2003]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>PVC plastic sheeting</td>
<td>EEE/material</td>
<td>WEEE</td>
<td></td>
<td>50,000</td>
<td>no</td>
<td></td>
<td>[Morf et al. 2003]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>plastic casings for EE products</td>
<td>EEE/material</td>
<td>WEEE</td>
<td>1998</td>
<td>115,000</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2003]</td>
<td></td>
</tr>
<tr>
<td><strong>China/Guangzhou City/Pearl Delta River</strong></td>
<td>Computer display casing</td>
<td>EEE/material</td>
<td>WEEE</td>
<td>China</td>
<td>1.527/ min. ND max. 6.08</td>
<td>25%</td>
<td>yes</td>
<td>[Chen et al. 2010]</td>
<td></td>
</tr>
<tr>
<td><strong>China/Guangzhou City/Pearl Delta River</strong></td>
<td>TV casing</td>
<td>EEE/material</td>
<td>WEEE</td>
<td>China</td>
<td>7.99/ min. ND max. 45.65</td>
<td>83.3 %</td>
<td>yes</td>
<td>[Chen et al. 2010]</td>
<td></td>
</tr>
<tr>
<td><strong>China/Guangzhou City/Pearl Delta River</strong></td>
<td>Computer component</td>
<td>EEE/material</td>
<td>WEEE</td>
<td>China</td>
<td>253.9/ min. ND max. 1512</td>
<td>75%</td>
<td>yes</td>
<td>[Chen et al. 2010]</td>
<td></td>
</tr>
<tr>
<td><strong>China/Guangzhou City/Pearl Delta River</strong></td>
<td>raw materials</td>
<td>EEE/material</td>
<td>WEEE</td>
<td>China</td>
<td>108.9; ND - 677</td>
<td>81.8%</td>
<td>yes</td>
<td>[Chen et al. 2010]</td>
<td></td>
</tr>
<tr>
<td><strong>Articles in use</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Norway</td>
<td>Flashlight (headlight)</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td></td>
<td>2,800</td>
<td></td>
<td></td>
<td>[NO EA 2018]</td>
<td></td>
</tr>
<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
<td>decaBDE content (mean*/range) [mg/kg]</td>
<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
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<tr>
<td>Sweden</td>
<td>Coating/cases for electronics</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>Before 2002</td>
<td>100 - 150,000</td>
<td></td>
<td></td>
<td>[SE EPA Sub. 2018]</td>
<td></td>
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<tr>
<td>Sweden</td>
<td>Plastics from TV, Scanner and household equipment</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>Before 2012</td>
<td>30 - 70,000</td>
<td></td>
<td></td>
<td>[SE EPA Sub. 2018]</td>
<td></td>
</tr>
<tr>
<td>Europe</td>
<td>Small and large Appliances</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>0 - 200,000</td>
<td></td>
<td></td>
<td></td>
<td>[Plastic Recyc. Europe Sub. 2018]</td>
<td></td>
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<tr>
<td></td>
<td>Plastic polymers in WEEE</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>0.20-0590</td>
<td></td>
<td></td>
<td></td>
<td>[UBA Sub. 2018]</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>Printer</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>&lt;3.1 - &lt;5.2</td>
<td>17% yes</td>
<td></td>
<td></td>
<td>[IVM 2013]</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>CRT TV</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>&lt;3.2 - 72,300</td>
<td>yes</td>
<td></td>
<td></td>
<td>[IVM 2013]</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>CRT Monitor</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>&lt;3</td>
<td>yes</td>
<td></td>
<td></td>
<td>[IVM 2013]</td>
<td></td>
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<tr>
<td>Netherlands</td>
<td>Flatscreen</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>&lt;3.4</td>
<td>yes</td>
<td></td>
<td></td>
<td>[IVM 2013]</td>
<td></td>
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<tr>
<td>Netherlands</td>
<td>Scanner</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>&lt;4.8 - 320</td>
<td>yes</td>
<td></td>
<td></td>
<td>[IVM 2013]</td>
<td></td>
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<tr>
<td>Turkey</td>
<td>computer</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>81 - 98</td>
<td>83% yes</td>
<td></td>
<td></td>
<td>[Binici et al. 2013]</td>
<td></td>
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<tr>
<td>Turkey</td>
<td>hair dryer</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>75 - 191</td>
<td>80% yes</td>
<td></td>
<td></td>
<td>[Binici et al. 2013]</td>
<td></td>
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<tr>
<td>Turkey</td>
<td>toaster</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>91 - 98</td>
<td>67% yes</td>
<td></td>
<td></td>
<td>[Binici et al. 2013]</td>
<td></td>
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<tr>
<td>Turkey</td>
<td>microwave oven</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>72 - 91</td>
<td>67% yes</td>
<td></td>
<td></td>
<td>[Binici et al. 2013]</td>
<td></td>
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<tr>
<td>Turkey</td>
<td>cable</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>79 - 96</td>
<td>100% yes</td>
<td></td>
<td></td>
<td>[Binici et al. 2013]</td>
<td></td>
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<tr>
<td>Japan</td>
<td>PC boards</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>2008; Japan</td>
<td>16 - 17</td>
<td>yes</td>
<td></td>
<td>[Kajiwara et al. 2011]</td>
<td></td>
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<tr>
<td>Japan</td>
<td>Rice cooker 1</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>2008; Japan</td>
<td>11</td>
<td>yes</td>
<td></td>
<td>[Kajiwara et al. 2011]</td>
<td></td>
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<tr>
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<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
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<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
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<tr>
<td>Japan</td>
<td>Rice cooker 2</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>2008; Japan</td>
<td>0.28 - 0.3</td>
<td>yes</td>
<td></td>
<td>[Kajiwara et al. 2011]</td>
<td>different methods of extraction used</td>
</tr>
<tr>
<td>Japan</td>
<td>computer mouse</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>2008; Japan</td>
<td>0.55 - 0.83</td>
<td>yes</td>
<td></td>
<td>[Kajiwara et al. 2011]</td>
<td>different methods of extraction used</td>
</tr>
<tr>
<td>India</td>
<td>motherboard computer</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>1996; Taiwan</td>
<td>11,583</td>
<td>100%</td>
<td>yes</td>
<td>[Kumari et al. 2014]</td>
<td></td>
</tr>
<tr>
<td>India</td>
<td>electrical computer</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>1996; Taiwan</td>
<td>ND</td>
<td>0</td>
<td>no</td>
<td>[Kumari et al. 2014]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>Consumer electronics</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>&gt;1,000</td>
<td>100%</td>
<td>no</td>
<td></td>
<td>[Kant. Lab. BS 2009]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>lighting appliances</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>&gt;1,000</td>
<td>20%</td>
<td>no</td>
<td></td>
<td>[Kant. Lab. BS 2009]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>EEE</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>ND</td>
<td>0%</td>
<td>no</td>
<td></td>
<td>[Kant. Lab. BS 2009]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>cooling and freezing appliances (inside lining without drawers)</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>200</td>
<td>20%</td>
<td>yes</td>
<td></td>
<td>[Wäger et al. 2011]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>cooling and freezing appliances (all plastic, except foam)</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>400 - 500</td>
<td>33%</td>
<td>yes</td>
<td></td>
<td>[Wäger et al. 2011]</td>
<td></td>
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<tr>
<td>Switzerland</td>
<td>vacuum cleaner</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>200</td>
<td>100%</td>
<td>yes</td>
<td></td>
<td>[Wäger et al. 2011]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>M2 w/o cleaners</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>500</td>
<td>100%</td>
<td>yes</td>
<td></td>
<td>[Wäger et al. 2011]</td>
<td></td>
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<tr>
<td>Switzerland</td>
<td>small appliances for high temperature applications</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>800</td>
<td>100%</td>
<td>yes</td>
<td></td>
<td>[Wäger et al. 2011]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>CRT monitors</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>3,200; 1,000 - 7,800</td>
<td>100%</td>
<td>yes</td>
<td></td>
<td>[Wäger et al. 2011]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>flat screen monitors</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>100</td>
<td>33%</td>
<td>yes</td>
<td></td>
<td>[Wäger et al. 2011]</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>printers</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>400</td>
<td>33%</td>
<td>yes</td>
<td></td>
<td>[Wäger et al. 2011]</td>
<td></td>
</tr>
<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
<td>decaBDE content (mean*/range) [mg/kg]</td>
<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
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</tr>
<tr>
<td>Switzerland</td>
<td>CRT TVs</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td></td>
<td>4,400; 800 – 7,800</td>
<td>100%</td>
<td>yes</td>
<td>[Wäger et al. 2011]</td>
<td></td>
</tr>
<tr>
<td>Nigeria</td>
<td>PC CRT screens</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>1987 – 2006; Asia, America, Europe</td>
<td>800; 0 – 54,000</td>
<td>4.5% (10 of 224)</td>
<td>yes</td>
<td>[Sindiku et al. 2014]</td>
<td>Detected levels 2,600 – 54,000 mg/kg</td>
</tr>
<tr>
<td>Nigeria</td>
<td>TV CRT screens</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>1981 – 2004; Europe, Asia</td>
<td>8,600; 0 – 237,000</td>
<td>15% (24 of 159)</td>
<td>yes</td>
<td>[Sindiku et al. 2014]</td>
<td>Detected levels 860 – 237,000 mg/kg</td>
</tr>
<tr>
<td>Japan</td>
<td>TV set circuit board</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>1989 – 1998; Japan</td>
<td>1.0 – 38</td>
<td>100%</td>
<td>yes</td>
<td>[Takigami et al. 2008]</td>
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<td>Japan</td>
<td>TV set front cabinet</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>1989 – 1998; Japan</td>
<td>0.58 – 140,000</td>
<td>100%</td>
<td>yes</td>
<td>[Takigami et al. 2008]</td>
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<td>Japan</td>
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<td>1.3 – 120,000</td>
<td>100%</td>
<td>yes</td>
<td>[Takigami et al. 2008]</td>
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<td>U.S.</td>
<td>VCRs</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td></td>
<td>9,350</td>
<td>yes</td>
<td>[Petreas Oros 2009]</td>
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<td>telephones</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td></td>
<td>1,300</td>
<td>yes</td>
<td>[Petreas Oros 2009]</td>
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<td>U.S.</td>
<td>radios</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td></td>
<td>190</td>
<td>yes</td>
<td>[Petreas Oros 2009]</td>
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<td>U.S.</td>
<td>computers</td>
<td>EEE/articles</td>
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<td></td>
<td>640</td>
<td>yes</td>
<td>[Petreas Oros 2009]</td>
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<td>U.S.</td>
<td>microwave</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td></td>
<td>4</td>
<td>yes</td>
<td>[Petreas Oros 2009]</td>
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<td>U.S.</td>
<td>printer</td>
<td>EEE/articles</td>
<td>WEEE</td>
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<td>200</td>
<td>yes</td>
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<td>cell phone</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td></td>
<td>15</td>
<td>yes</td>
<td>[Petreas Oros 2009]</td>
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<td>Australia</td>
<td>Computer</td>
<td>EEE/articles</td>
<td>WEEE</td>
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<td>169,029</td>
<td>45%</td>
<td>[Gallen et al. 2014]</td>
<td>Br content; Max. value, 3% of measurement show conc. &gt;10,000 mg/kg; 8% &gt; 1,000 mg/kg</td>
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<td>Australia</td>
<td>Electronic consumer</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td></td>
<td>1,075</td>
<td>43%</td>
<td>[Gallen et al.</td>
<td>Br content; Max.</td>
<td></td>
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<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
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<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
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<tr>
<td>Australia</td>
<td>tool</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2014]</td>
<td>value, 0% of measurement show conc. &gt;10,000 mg/kg; 2% &gt; 1,000 mg/kg</td>
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<td>Australia</td>
<td>LHA</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>94,828</td>
<td></td>
<td>43%</td>
<td>[Gallen et al. 2014]</td>
<td>Br content; Max. value, 5% of measurement show conc. &gt;10,000 mg/kg; 7% &gt; 1,000 mg/kg</td>
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<td>Australia</td>
<td>SHA</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>172,645</td>
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<td>58%</td>
<td>[Gallen et al. 2014]</td>
<td>Br content; Max. value, 9% of measurement show conc. &gt;10,000 mg/kg; 21% &gt; 1,000 mg/kg</td>
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<td>Electronic other</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>114,422</td>
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<td>69%</td>
<td>[Gallen et al. 2014]</td>
<td>Br content; Max. value, 23% of measurement show conc. &gt;10,000 mg/kg; 35% &gt; 1,000 mg/kg</td>
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<td>Australia</td>
<td>Telephone</td>
<td>EEE/articles</td>
<td>WEEE</td>
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<td>[Gallen et al. 2014]</td>
<td>Br content; Max. value, 0% of measurement show conc. &gt;10,000 mg/kg; 29% &gt; 1,000 mg/kg</td>
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<td>Australia</td>
<td>TV</td>
<td>EEE/articles</td>
<td>WEEE</td>
<td>124,868</td>
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<td>[Gallen et al. 2014]</td>
<td>Br content; Max. value, 62% of measurement show conc. &gt;10,000 mg/kg; 68% &gt; 1,000 mg/kg</td>
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Waste
<table>
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<tr>
<th>Country/ region</th>
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<th>Product category</th>
<th>Waste category</th>
<th>Production year and country of origin</th>
<th>decaBDE content (mean*/range) [mg/kg]</th>
<th>Detection frequency</th>
<th>Evidence of other POP-BDEs</th>
<th>Reference</th>
<th>Comment/further info</th>
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<tbody>
<tr>
<td>Sweden</td>
<td>Mixed plastic from EEE</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003 – 2006</td>
<td>1,000 – 20,000</td>
<td></td>
<td></td>
<td>[SE EPA Sub. 2018]</td>
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<td>Sweden</td>
<td>Coating/case for TV</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2009</td>
<td>8</td>
<td></td>
<td></td>
<td>[SE EPA Sub. 2018]</td>
<td></td>
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<tr>
<td>Sweden</td>
<td>Coating/case for computer</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2009</td>
<td>15</td>
<td></td>
<td></td>
<td>[SE EPA Sub. 2018]</td>
<td></td>
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<tr>
<td>Austria</td>
<td>housings of TV-screens</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>[AT Ministry of Sustain. and Tourism Sub. 2018a]</td>
<td>613 – 7,500</td>
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<td>Austria</td>
<td>housings of PC-monitors</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>[AT Ministry of Sustain. and Tourism Sub. 2018a]</td>
<td>700 – 6,300</td>
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<tr>
<td>Switzerland</td>
<td>small appliances</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; 510</td>
<td>200 – 1,200</td>
<td>85%</td>
<td>yes</td>
<td>[Wäger et al. 2011]</td>
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<td>small household appliances</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; 510</td>
<td>100 – 400</td>
<td>67%</td>
<td>yes</td>
<td>[Wäger et al. 2011]</td>
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<td>Switzerland</td>
<td>ICT and consumer equipment (screens, CRTs)</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; 510</td>
<td>500 – 1,400</td>
<td>100%</td>
<td>yes</td>
<td>[Wäger et al. 2011]</td>
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<td>Switzerland</td>
<td>large household appliances</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; 510</td>
<td>300 – 1,600</td>
<td>60%</td>
<td>yes</td>
<td>[Wäger et al. 2011]</td>
<td></td>
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<td>Switzerland</td>
<td>small household appliances</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; 510</td>
<td>100 – 600</td>
<td>50%</td>
<td>yes</td>
<td>[Wäger et al. 2011]</td>
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<td>ICT equipment</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; 510</td>
<td>100 – 900</td>
<td>100%</td>
<td>yes</td>
<td>[Wäger et al. 2011]</td>
<td></td>
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<tr>
<td>Switzerland</td>
<td>consumer equipment (screens, CRTs)</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; 510</td>
<td>900</td>
<td>100%</td>
<td>yes</td>
<td>[Wäger et al. 2011]</td>
<td></td>
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<tr>
<td>Switzerland</td>
<td>sWEEE</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; 510</td>
<td>510</td>
<td></td>
<td>yes</td>
<td>[Morf et al.</td>
<td></td>
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<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
<td>decaBDE content (mean*/range) [mg/kg]</td>
<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
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<tr>
<td>Switzerland</td>
<td>Cu cable scrap</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; Switzerland</td>
<td>170</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2005]</td>
<td></td>
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<tr>
<td>Switzerland</td>
<td>printed circuit boards</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; Switzerland</td>
<td>27</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2005]</td>
<td></td>
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<td>Switzerland</td>
<td>TV housing plastic</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; Switzerland</td>
<td>4,800</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2005]</td>
<td></td>
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<td>TV housing rear cover</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; Switzerland</td>
<td>13,000</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2005]</td>
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<td>Switzerland</td>
<td>plastic parts from small e-waste</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>2003; Switzerland</td>
<td>1,800</td>
<td>yes</td>
<td></td>
<td>[Morf et al. 2005] quoted in [Li et al. 2013]</td>
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<td>China/Guang Zhou</td>
<td>cell phone shell</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
<td></td>
<td>[Chen et al. 2012]</td>
<td>Authors’ statement83</td>
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<tr>
<td>China/Guang Zhou</td>
<td>computer housing</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
<td></td>
<td>[Chen et al. 2012]</td>
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<td>China/Guang Zhou</td>
<td>TV housing</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
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<td>[Chen et al. 2012]</td>
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<td>EEE/waste</td>
<td>WEEE</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
<td></td>
<td>[Chen et al. 2012]</td>
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<td>Netherlands</td>
<td>e-waste 1</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>1860</td>
<td>yes</td>
<td></td>
<td></td>
<td>[Ballesteros-Gómez et al.]</td>
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</table>

83 It is well known that D10BDE is one of the widely used BFRs, however, many low BDEs (P5BDE, H6BDE, H7BDE, O8BDE, N9BDE) but not the D10BDE were detected in the samples used in this experiment. As for these results, we could take into consideration the possibility of preferential degradation of highly brominated diphenyl ethers, as observed by Rahman et al., Watanabe and Tatsukawa and Pijnenburg et al. Particularly, Watanabe and Tatsukawa reported that D10BDE was vulnerable to photolytic debromination, resulting in the formation of less brominated PBDEs congeners. Because the samples were collected from landfill and exposed to sunlight, the degradation might have been taken place [Chen et al. 2012].
<table>
<thead>
<tr>
<th>Country/ region</th>
<th>Product/product category</th>
<th>Product category</th>
<th>Waste category</th>
<th>Production year and country of origin</th>
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<th>Detection frequency</th>
<th>Evidence of other POP-BDEs</th>
<th>Reference</th>
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<td>e-waste 2</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>71</td>
<td>yes</td>
<td></td>
<td>[Ballesteros-Gómez et al. 2013]</td>
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<td>United Kingdom</td>
<td>TVs</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>21 – 138,000 78% yes</td>
<td>[WRc 2012a]</td>
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<td>United Kingdom</td>
<td>industrial IT equipment</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>0.162 – 32,400 100% yes</td>
<td>[WRc 2012a]</td>
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<td>printed circuit boards</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>42,500 – 129,000 100% yes</td>
<td>[WRc 2012a]</td>
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<td>United Kingdom</td>
<td>digiboxes</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>4.40 – 5,720 100% yes</td>
<td>[WRc 2012a]</td>
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<tr>
<td>United Kingdom</td>
<td>large household appliances</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>912 – 45,700 100% yes</td>
<td>[WRc 2012a]</td>
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<td>United Kingdom</td>
<td>small WEEE</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>778 100% yes</td>
<td>[WRc 2012a]</td>
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<td>United Kingdom</td>
<td>coat hangers</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>22 – 80,400 100% yes</td>
<td>[WRc 2012a]</td>
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<td>fridges</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>79 – 155 100% yes</td>
<td>[WRc 2012a]</td>
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<td>United Kingdom</td>
<td>PC monitors</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>6 – 13,900 100% yes</td>
<td>[WRc 2012a]</td>
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<td>Switzerland</td>
<td>PC monitor casing shredder mix</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>40 100% yes</td>
<td>[Zenegg et al 2014]</td>
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<td>TV casing shredder mix</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>100 100% yes</td>
<td>[Zenegg et al 2014]</td>
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<td>Norway</td>
<td>EE waste fluff</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>5 – 6 50% yes</td>
<td>[COWI 2013]</td>
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<td>Finland</td>
<td>electronic plastic</td>
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<td></td>
<td>60 100% yes</td>
<td>[Sinkkonen et al. 2004]</td>
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<td>WEEE shredded</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>6.4 – 3,300.0 92% yes</td>
<td>[IVM 2013]</td>
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<td>CRT TVs and monitors</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>500 – 3,000 100% yes</td>
<td>[Tange and Slijkhuus, 2009]</td>
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<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
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<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
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<td>Office equipment</td>
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<td>WEEE</td>
<td></td>
<td>500 – 3,000</td>
<td>yes</td>
<td>[Tange and Slijkhuis, 2009]</td>
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<td>EEE/waste</td>
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<td>ND – 1,000</td>
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<td>[Tange and Slijkhuis, 2009]</td>
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<td>ND – 1,000</td>
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<td>[Tange and Slijkhuis, 2009]</td>
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<td>Small domestic appliances</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>ND – 1,000</td>
<td>yes</td>
<td>[Tange and Slijkhuis, 2009]</td>
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<td>Switzerland</td>
<td>remainder ICT equipment</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>ND – 1,000</td>
<td>yes</td>
<td>[Tange and Slijkhuis, 2009]</td>
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<td>Switzerland</td>
<td>Small domestic appliances fraction A</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>&lt; 0.2</td>
<td>yes</td>
<td>[Tange and Slijkhuis, 2009]</td>
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<td>toner cartridge plastics</td>
<td>EEE/waste</td>
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<td></td>
<td>74.8</td>
<td>yes</td>
<td>[Tange and Slijkhuis, 2009]</td>
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<td>shredder TV housing material</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>1,615</td>
<td>yes</td>
<td>[Tange and Slijkhuis, 2009]</td>
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<td>small domestic appliances fraction B with some shredded TV housing</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>544</td>
<td>yes</td>
<td>[Tange and Slijkhuis, 2009]</td>
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<td>Switzerland</td>
<td>small domestic appliances fraction C</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>1,321</td>
<td>yes</td>
<td>[Tange and Slijkhuis, 2009]</td>
<td></td>
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<tr>
<td>Austria</td>
<td>TV waste plastic</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>Before 2006</td>
<td>613 – 7,503</td>
<td>100%</td>
<td>yes</td>
<td>[Aldrian et al. 2015]</td>
<td>analysed with a handheld XRF</td>
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<td>Austria</td>
<td>PC waste plastic</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>Before 2006</td>
<td>702 – 6,272</td>
<td>100%</td>
<td>yes</td>
<td>[Aldrian et al. 2015]</td>
<td></td>
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<td>Germany/Europe</td>
<td>single TV sets and monitor housings</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td></td>
<td>NA</td>
<td>2 of 15 (15 of 45 FR)</td>
<td>yes</td>
<td>[Schlummer et al. 2007]</td>
<td></td>
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<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
<td>decaBDE content (mean*/range) [mg/kg]</td>
<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
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<tr>
<td>Germany/Europe</td>
<td>shredded housing material from TV sets and monitors</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>ND – 21,000</td>
<td>86%</td>
<td>yes</td>
<td>[Schlummer et al. 2007]</td>
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<td>Germany/Europe</td>
<td>mixed shredder residues from WEEE</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>ND – 3,100</td>
<td>75%</td>
<td>yes</td>
<td>[Schlummer et al. 2007]</td>
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<td>China</td>
<td>e waste (printed circuit boards and plastic materials)</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>3.26/0.98 – 6.39</td>
<td>yes</td>
<td>[NO EA 2016]</td>
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<td>China</td>
<td>waste printed circuit boards</td>
<td>EEE/waste</td>
<td>WEEE</td>
<td>130</td>
<td>yes</td>
<td>[Zhou et al. 2013]</td>
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# 10.8 Occurrence and concentrations of decaBDE in material, articles and waste of the ELV waste category

## Table 81: Occurrence and concentration of decaBDE in material, articles and waste of the ELV waste category

<table>
<thead>
<tr>
<th>Country/region</th>
<th>Product/product category</th>
<th>Product category</th>
<th>Waste category</th>
<th>Production year and country of origin</th>
<th>decaBDE contentation (mean*/range) [mg/kg]</th>
<th>Detection frequency</th>
<th>Evidence of other POP-BDEs</th>
<th>Reference</th>
<th>Comment/further info</th>
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<tbody>
<tr>
<td>Netherlands</td>
<td>PUF car seat (Pontiac Transport)</td>
<td>automotive materials</td>
<td>ELV</td>
<td>1997</td>
<td>522</td>
<td>100%</td>
<td>yes</td>
<td>[IVM 2013]</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>seat cover (Pontiac Transport)</td>
<td>automotive materials</td>
<td>ELV</td>
<td>1997</td>
<td>22,500</td>
<td>100%</td>
<td>yes</td>
<td>[IVM 2013]</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>interior (Pontiac Transport)</td>
<td>automotive materials</td>
<td>ELV</td>
<td>1997</td>
<td>18</td>
<td>100%</td>
<td>no</td>
<td>[IVM 2013]</td>
<td></td>
</tr>
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<td>Netherlands</td>
<td>PUF car seat (Mazda 323)</td>
<td>automotive materials</td>
<td>ELV</td>
<td>1998</td>
<td>&lt;1.6</td>
<td>100%</td>
<td>no</td>
<td>[IVM 2013]</td>
<td></td>
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<td>seat cover (Mazda 323)</td>
<td>automotive materials</td>
<td>ELV</td>
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<td>22,700</td>
<td>100%</td>
<td>no</td>
<td>[IVM 2013]</td>
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<td>automotive materials</td>
<td>ELV</td>
<td>1998</td>
<td>18</td>
<td>100%</td>
<td>no</td>
<td>[IVM 2013]</td>
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<tr>
<td>Netherlands</td>
<td>different car components (Europa)</td>
<td>automotive materials</td>
<td>ELV</td>
<td>1992 - 2000</td>
<td>&lt;1.2 - &lt;3.6</td>
<td>0%</td>
<td>no</td>
<td>[IVM 2013]</td>
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<td>Netherlands</td>
<td>car seats</td>
<td>automotive materials</td>
<td>ELV</td>
<td>1974 - 2002</td>
<td>&lt;4.4 - 131</td>
<td>18%</td>
<td>no</td>
<td>[IVM 2013]</td>
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<tr>
<td>Netherlands</td>
<td>PUF car seats</td>
<td>automotive materials</td>
<td>ELV</td>
<td>1992 - 1999</td>
<td>0.11 - 0.97</td>
<td>100%</td>
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<td>[IVM 2013]]</td>
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<tr>
<td>Netherlands</td>
<td>seat cover (Chrysler Saratoga)</td>
<td>automotive materials</td>
<td>ELV</td>
<td>1991</td>
<td>256</td>
<td>100%</td>
<td>no</td>
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<tr>
<td>China/Guagnzh</td>
<td>car interior (plastic)</td>
<td>automotive</td>
<td>ELV</td>
<td>8; ND - 32.6</td>
<td>60%</td>
<td>yes</td>
<td></td>
<td>[Chen et al. 2010]</td>
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<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
<td>decaBDE content ration (mean*/range) [mg/kg]</td>
<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
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</tr>
<tr>
<td>ou City/Pearl Delta River</td>
<td>interior, seat PUF and textile)</td>
<td>materials</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>[Swerea 2018]</td>
</tr>
<tr>
<td>Norway</td>
<td>PP/PE</td>
<td>ELV</td>
<td>n/a</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>[Swerea 2018]</td>
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</tr>
<tr>
<td>France</td>
<td>PP/PE</td>
<td>ELV + household</td>
<td>n/a</td>
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<td>[Swerea 2018]</td>
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<tr>
<td>France</td>
<td>PP/PE</td>
<td>ELV</td>
<td>n/a</td>
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<tr>
<td>France</td>
<td>PS/ABS</td>
<td>ELV</td>
<td>140</td>
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<td>PP</td>
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<td>n/a</td>
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<td></td>
<td></td>
<td></td>
<td>[Swerea 2018]</td>
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<td>Country/region</td>
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<td>Evidence of other POP-BDEs</td>
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<td></td>
<td></td>
<td></td>
<td>[Swerea 2018]</td>
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<td>cars</td>
<td>automotive articles</td>
<td>ELV</td>
<td>625</td>
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<td>[Morf et al. 2003]</td>
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<td>rail vehicles</td>
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<td>ELV</td>
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<td>yes</td>
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<td>[Morf et al. 2003]</td>
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<td>Sweden</td>
<td>Textiles (in vehicles, furniture)</td>
<td>Automotive/furniture articles</td>
<td>ELV</td>
<td>Before 2004</td>
<td>200,000 - 300,000</td>
<td>yes</td>
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<td>[SE EPA Sub. 2018]</td>
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<td>[BSEF Sub. 2018]</td>
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<td>ELV</td>
<td>0 - 200,000</td>
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<td>[Plastic Recyc. Europe Sub. 2018]</td>
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<td>Plastic polymers ELV</td>
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<td>0.2 - 590</td>
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<td>Car accessoires</td>
<td>automotive articles</td>
<td>ELV</td>
<td>3,600</td>
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<td></td>
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<td>[Gallen et al. 2014]</td>
<td>Br content; Max. value, 4% of measurement</td>
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<td>Product category</td>
<td>Waste category</td>
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<td>Evi-dence of other POP-BDEs</td>
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<tr>
<td>Australia</td>
<td>Car dashboard/surfaces</td>
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<td>ELV</td>
<td>506</td>
<td>46%</td>
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<td>[Gallen et al. 2014]</td>
<td>show conc. &gt; 1,000 mg/kg</td>
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<td>Sweden</td>
<td>PUF from old car seats (Audi, Citroen, Dodge, Ford, Mercedes, Mitsubishi, Saab, Skoda, Subaru, Toyota, VW, Volvo)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1986 - 1999</td>
<td>ND - 2.1</td>
<td>88%</td>
<td>yes</td>
<td>[Niinipuu 2013]</td>
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<td>Norway</td>
<td>Soundproofing material sample 1 (Audi A6, Nissan Primera, Audi S442)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1993 - 1994</td>
<td>&lt;50</td>
<td>0%</td>
<td>yes</td>
<td>[MEPEX 2012]</td>
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<td>Norway</td>
<td>Soundproofing material sample 2</td>
<td>automotive waste</td>
<td>ELV</td>
<td>7,000</td>
<td>no</td>
<td></td>
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<td>[MEPEX 2012]</td>
<td></td>
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<td>Norway</td>
<td>Airbag material (Kia Ria)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>2002</td>
<td>&lt;50</td>
<td>0%</td>
<td>no</td>
<td>[MEPEX 2012]</td>
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<tr>
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<td>Seat cover material (Ford Explorer, Nissan Almera)</td>
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<td>ELV</td>
<td>1994 - 1998</td>
<td>27,000</td>
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<td>[MEPEX 2012]</td>
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<td>Country/region</td>
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<td>Product category</td>
<td>Waste category</td>
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<tr>
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<td>Interior material (door, headlining, cover) (Kia Rio, Toyota Corolla, Volvo 240)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1987 - 2002</td>
<td>17,000</td>
<td>100%</td>
<td>no</td>
<td>[MEPEX 2012]</td>
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<td>Luggage compartment material (Mazda 626)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>2000</td>
<td>&lt;50</td>
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<td>no</td>
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<td>Radiator, outer material (Nissan Primera)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>2004</td>
<td>&lt;50</td>
<td>0%</td>
<td>no</td>
<td>[MEPEX 2012]</td>
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<tr>
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<td>Printed circuit boards sample 1 (Audi A6, Audi 80, Mercedes 124 E)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1990 - 1994</td>
<td>200</td>
<td>100%</td>
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<td>[MEPEX 2012]</td>
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<td>Printed circuit boards sample 2 (Mazda 626, Mitsubishi Carisma, BMW E39, Opel Vectra)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1995 - 1999</td>
<td>&lt;10</td>
<td>0%</td>
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<td>[MEPEX 2012]</td>
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<td>Printed circuit boards sample 3 (Peugeot 406, Toyota Corolla, Volvo S40, Audi A4)</td>
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<td>ELV</td>
<td>1995 - 1999</td>
<td>&lt;10</td>
<td>0%</td>
<td>yes</td>
<td>[MEPEX 2012]</td>
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<td>Printed circuit boards</td>
<td>automotive waste</td>
<td>ELV</td>
<td>2000 - 2004</td>
<td>33</td>
<td>100%</td>
<td>yes</td>
<td>[MEPEX 2012]</td>
<td></td>
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<td>Country/region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
<td>decaBDE content ration (mean*/range) [mg/kg]</td>
<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
</tr>
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</tr>
<tr>
<td>United Kingdom</td>
<td>Seat foam materials within shredder residue fraction</td>
<td>automotive waste</td>
<td>ELV</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
<td>[BMRA 2013]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>United Kingdom</td>
<td>Seat foam from ELVs (BMW, Ford, Peugeot, Vauxhall)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1989 – 2005; Germany, America, France, UK</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
<td>[BMRA 2013]</td>
<td></td>
</tr>
<tr>
<td>United Kingdom</td>
<td>Seat fabric from ELVs (BMW, Ford, Peugeot, Vauxhall)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1989 – 2005; Germany, America, France, UK</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
<td>[BMRA 2013]</td>
<td></td>
</tr>
<tr>
<td>United Kingdom</td>
<td>cloth from car headrest</td>
<td>automotive waste</td>
<td>ELV</td>
<td>20</td>
<td>100%</td>
<td>yes</td>
<td>[WRc 2012a]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>U.S.</td>
<td>autoshredder waste</td>
<td>automotive waste</td>
<td>ELV</td>
<td>43.5</td>
<td></td>
<td>yes</td>
<td>[Petreas Oros 2009]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>car interior</td>
<td>automotive waste</td>
<td>ELV</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
<td>[Ballesteros-Gomez et al. 2013]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Norway</td>
<td>fluff from mixed feedstock</td>
<td>automotive waste</td>
<td>ELV</td>
<td>60/0 - 400</td>
<td>54%</td>
<td>yes</td>
<td>[COWI 2013]</td>
<td>[log-Werte (logarithmische Skala)]</td>
<td></td>
</tr>
<tr>
<td>Norway</td>
<td>car fluff</td>
<td>automotive waste</td>
<td>ELV</td>
<td>3 - 60</td>
<td>69%</td>
<td>yes</td>
<td>[COWI 2013]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Norway</td>
<td>complex metals fluff</td>
<td>automotive waste</td>
<td>ELV</td>
<td>7 - 12</td>
<td>40%</td>
<td>yes</td>
<td>[COWI 2013]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>United Kingdom</td>
<td>light ASR (foam, textile)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>138/130 - 151</td>
<td></td>
<td>yes</td>
<td>[WRC addendum 2012]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Country/region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
<td>decaBDE content ration (mean*/range) [mg/kg]</td>
<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
</tr>
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</tr>
<tr>
<td>United Kingdom</td>
<td>mixed light plastic</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td>118/11.5 - 229</td>
<td>yes</td>
<td>[Wrc addendum 2012]</td>
<td></td>
<td></td>
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<tr>
<td>United Kingdom</td>
<td>mixed medium density plastic</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td>2,163/644 - 3,915</td>
<td>yes</td>
<td>[Wrc addendum 2012]</td>
<td></td>
<td></td>
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<tr>
<td>United Kingdom</td>
<td>mixed high density plastic</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td>1,357</td>
<td>100%</td>
<td>yes</td>
<td>[Wrc addendum 2012]</td>
<td></td>
</tr>
<tr>
<td>Finland</td>
<td>light fraction of auto shredder residue</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td>0.01</td>
<td>100%</td>
<td>yes</td>
<td>[Sinkkonen et al. 2004]</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>plastic fraction</td>
<td>automotive waste</td>
<td>ELV</td>
<td>18 years (lifetime)</td>
<td>800</td>
<td>no</td>
<td>[ARN 2015] as cited in [NO EA 2016]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>United Kingdom</td>
<td>seat foam material from shredder fraction</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1989 - 2005; Germany, America, France, UK</td>
<td>ND</td>
<td>0%</td>
<td>yes</td>
<td>[BMRA 2013]</td>
<td></td>
</tr>
<tr>
<td>Ireland</td>
<td>ASR</td>
<td>automotive waste</td>
<td>ELV</td>
<td>1990 - 2006</td>
<td>3.50</td>
<td>100%</td>
<td>yes</td>
<td>[ELVES 2016]</td>
<td></td>
</tr>
<tr>
<td>Ireland</td>
<td>ASR fine</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td>2.55</td>
<td>100%</td>
<td>yes</td>
<td>[ELVES 2016]</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>car shredder residues</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td>0.2 - 70.0</td>
<td>100%</td>
<td>yes</td>
<td>[IVM 2013]</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>WEEE/automotive shredder residues</td>
<td>automotive/EEE waste</td>
<td>WEEE/ELV</td>
<td></td>
<td>6.0 -810</td>
<td>100%</td>
<td>yes</td>
<td>[IVM 2013]</td>
<td></td>
</tr>
<tr>
<td>Denmark</td>
<td>WEEE/automotive</td>
<td>automotive/EEE</td>
<td>WEEE/ELV</td>
<td></td>
<td>100</td>
<td>100%</td>
<td>yes</td>
<td>[Danisch EPA 2014b]</td>
<td>as</td>
</tr>
<tr>
<td>Country/region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Waste category</td>
<td>Production year and country of origin</td>
<td>decaBDE content ration (mean*/range) [mg/kg]</td>
<td>Detection frequency</td>
<td>Evidence of other POP-BDEs</td>
<td>Reference</td>
<td>Comment/further info</td>
</tr>
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</tr>
<tr>
<td>shredder residues</td>
<td>waste</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>cited in [NO EA 2016]</td>
</tr>
<tr>
<td>Vehicle equipment</td>
<td>automotive/EEE waste</td>
<td>WEEE/ELV</td>
<td>2009</td>
<td>9</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>[SE EPA Sub. 2018]</td>
</tr>
<tr>
<td>Plastic from ASR (density &lt;1.1 g/cm³)</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>[ARN Sub. 2018]</td>
</tr>
<tr>
<td>Fibres originating from automotive shredder residue</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>[ARN Sub. 2018]</td>
</tr>
<tr>
<td>ASR</td>
<td>automotive waste</td>
<td>ELV</td>
<td></td>
<td>0.01 - 590</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>[DEFRA Sub. 2018; ACEA Sub. 2018a]</td>
</tr>
<tr>
<td>Fibres from ASR (gained by separation after shredding)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>2017</td>
<td>0 - 48</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>[DEFRA Sub. 2018; ACEA Sub. 2018a]</td>
</tr>
<tr>
<td>Plastic (&lt; 1.1 g/cm³) (gained by separation after shredding)</td>
<td>automotive waste</td>
<td>ELV</td>
<td>2017</td>
<td>0 - 12</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>[DEFRA Sub. 2018; ACEA Sub. 2018a]</td>
</tr>
</tbody>
</table>

* mean values highlighted in red
### 10.9 Calculation to derive at the decaBDE concentration for plastics in ELV

We compiled literature values from studies and informal stakeholder consultations. The input for our data set has 18 data points and is given in the table below.

**Table 82: Input data from literature and stakeholder consultations**

<table>
<thead>
<tr>
<th>Nr.</th>
<th>Source</th>
<th>Country</th>
<th>decaBDE concentration [mg/kg]</th>
<th>Plastic type</th>
<th>Comment</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>[Swerea 2018]</td>
<td>France</td>
<td>140</td>
<td>PS/ABS</td>
<td>Regrind</td>
</tr>
<tr>
<td>2</td>
<td>[Swerea 2018]</td>
<td>UK</td>
<td>5.1</td>
<td>ABS</td>
<td>Regrind</td>
</tr>
<tr>
<td>3</td>
<td>[Petreas Oros 2009]</td>
<td>US</td>
<td>43.5</td>
<td>n.d.</td>
<td>Autoshredder waste</td>
</tr>
<tr>
<td>4</td>
<td>[WRc addendum 2012]</td>
<td>UK</td>
<td>138</td>
<td>n.d.</td>
<td>light ASR (foam, textile); Mean value from n=3</td>
</tr>
<tr>
<td>5</td>
<td>[WRc addendum 2012]</td>
<td>UK</td>
<td>118</td>
<td>n.d.</td>
<td>mixed light plastic; Mean value from n=5</td>
</tr>
<tr>
<td>6</td>
<td>[WRc addendum 2012]</td>
<td>UK</td>
<td>2,163</td>
<td>n.d.</td>
<td>mixed medium density plastic; Mean value from n=5</td>
</tr>
<tr>
<td>7</td>
<td>[WRc addendum 2012]</td>
<td>UK</td>
<td>1,357</td>
<td>n.d.</td>
<td>mixed high-density plastic</td>
</tr>
<tr>
<td>8</td>
<td>[Sinkkonen et al. 2004]</td>
<td>Finland</td>
<td>0.01</td>
<td>n.d.</td>
<td>light fraction of auto shredder residue</td>
</tr>
<tr>
<td>9</td>
<td>[ARN 2015] as cited in [NO EA 2016]</td>
<td>Netherlands</td>
<td>800</td>
<td>n.d.</td>
<td>plastic fraction</td>
</tr>
<tr>
<td>10</td>
<td>[MoE survey 2011]</td>
<td>Japan</td>
<td>406</td>
<td>n.d.</td>
<td>ASR from ELVs before 1996; mean value from n=70</td>
</tr>
<tr>
<td>11</td>
<td>[MoE survey 2011]</td>
<td>Japan</td>
<td>123</td>
<td>n.d.</td>
<td>ASR from 2000 or later; mean value from n=70</td>
</tr>
<tr>
<td>12</td>
<td>[ELVES]</td>
<td>Ireland</td>
<td>3.5</td>
<td>n.d.</td>
<td></td>
</tr>
<tr>
<td>13</td>
<td>[ELVES]</td>
<td>Ireland</td>
<td>2.55</td>
<td>n.d.</td>
<td></td>
</tr>
<tr>
<td>14</td>
<td>[IVM 2013]</td>
<td>Netherlands</td>
<td>70</td>
<td>n.d.</td>
<td>car shredder residues; range from 0.2 to 70</td>
</tr>
<tr>
<td>15</td>
<td>[IVM 2013]</td>
<td>Netherlands</td>
<td>810</td>
<td>n.d.</td>
<td>WEEE/automotive shredder residues; range from 6.0 to 810, n=5</td>
</tr>
<tr>
<td>16</td>
<td>[Danish EPA 2014b] as cited in [NO EA 2016]</td>
<td>Denmark</td>
<td>100</td>
<td>n.d.</td>
<td>WEEE/automotive shredder residues</td>
</tr>
<tr>
<td>Nr.</td>
<td>Source</td>
<td>Country</td>
<td>decaBDE concentration [mg/kg]</td>
<td>Plastic type</td>
<td>Comment</td>
</tr>
<tr>
<td>-----</td>
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<td>-----------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>17</td>
<td>[ACEA sub. 2018a]</td>
<td>n.d.</td>
<td>48 n.d.</td>
<td>Fibres from ASR (gained by separation after shredding), range from 0 to 48; measured by ARN in 2017</td>
<td></td>
</tr>
<tr>
<td>18</td>
<td>[ACEA sub. 2018a]</td>
<td>n.d.</td>
<td>12 n.d.</td>
<td>Plastic (&lt; 1.1 g/cm³) (gained by separation after shredding), range from 0 to 12; Measured by ARN in 2017</td>
<td></td>
</tr>
</tbody>
</table>

We calculated the median of 109 mg/kg of these 18 values and assume that this represents the decaBDE concentration in both plastic-containing fractions, SLF and SHF. As Eurostat provides tonnages of SLF and SHF, we were able to give total amounts of decaBDE within both fractions by multiplying the tonnages (712,814 t for the SLF and 188,038 t for the SHF) with the decaBDE concentration. This gives decaBDE amounts of 77.7 t (SLF) and 20.5 t (SHF).

The data for the waste treatment for both fractions is provided by Eurostat. SHF is assumed to be fully recycled while for the SLF the following shares are given: 33.1% is incinerated, 29.9% recycled and 37.0% landfilled. We assume that plastic and decaBDE in the SLF are treated in the same way as the SLF itself. Therefore, 28.7 t of decaBDE in the SLF are recycled, on top of the 20.5 t of decaBDE in the SHF. 23.2 t of decaBDE are incinerated and 25.8 t landfilled.
### 10.10 Occurrence and concentrations of HBCDD in articles from recyclate

<table>
<thead>
<tr>
<th>Country/ region</th>
<th>Product/product category</th>
<th>Product category</th>
<th>Concentration [mg/kg]</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Argentina</td>
<td>Rubik’s-like cube</td>
<td>Toy</td>
<td>1</td>
<td>[Petrlik et al. 2018]</td>
</tr>
<tr>
<td>Argentina</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0 - 1,586</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Austria</td>
<td>Toy gun</td>
<td>Toy</td>
<td>6 - 8</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Austria</td>
<td>Hair clip</td>
<td>Hair accessorie</td>
<td>0 - 2</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Austria</td>
<td>Fidget spinner</td>
<td>Toy</td>
<td>0</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Austria</td>
<td>Hair brush</td>
<td>Hair accessorie</td>
<td>0</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Bangladesh</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>1 - 5</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Belarus</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>n.a.</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Belgium</td>
<td>Hair brush</td>
<td>Hair accessorie</td>
<td>0</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Belgium</td>
<td>Hair clip</td>
<td>Hair accessorie</td>
<td>0</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Belgium</td>
<td>Telescope</td>
<td>Toy</td>
<td>1</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Brazil</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Canada</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>1 - 20</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>China</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0/n.a.</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Knife</td>
<td>Kitchen utensil</td>
<td>&lt;LOQ</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Fork</td>
<td>Kitchen utensil</td>
<td>&lt;LOQ</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Concentration [mg/kg]</td>
<td>Source</td>
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<tr>
<td>----------------</td>
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<td>-----------------------</td>
<td>-----------------------------</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Eye-glasses</td>
<td>Other</td>
<td>&lt;LOQ</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Scissors (handle, rivet)</td>
<td>Kitchen utensil</td>
<td>&lt;LOQ - 29</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Headdress</td>
<td>Hair accessorie</td>
<td>0 - 24</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Hair clip</td>
<td>Hair accessorie</td>
<td>1 - 5</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Pastry brush set</td>
<td>Kitchen utensil</td>
<td>0</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Comb</td>
<td>Hair accessorie</td>
<td>0</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Robot</td>
<td>Toy</td>
<td>0</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Finger skate-board</td>
<td>Toy</td>
<td>0</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Thermo cup</td>
<td>Toy</td>
<td>0</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
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<td>Children hockey stick</td>
<td>Toy</td>
<td>0</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Comb</td>
<td>Hair accessorie</td>
<td>0</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
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<td>Toy</td>
<td>0 - 42</td>
<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
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<td>Kitchen set</td>
<td>Kitchen utensil</td>
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<td>[Straková et al. 2018a]</td>
</tr>
<tr>
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<td>Transformer</td>
<td>Toy</td>
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<td>[Straková et al. 2018]</td>
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<tr>
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<td>Hair accessorie</td>
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<td>[Straková et al. 2018]</td>
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<td>Gun</td>
<td>Toy</td>
<td>2</td>
<td>[Straková et al. 2018a]</td>
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<td>Stapler</td>
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<td>2</td>
<td>[Straková et al. 2018a]</td>
</tr>
<tr>
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<td>Product category</td>
<td>Concentration [mg/kg]</td>
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<td>Children painting brush</td>
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<td>[DiGangi et al. 2017]</td>
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<td>0.04 – 0.09</td>
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<td>Head-dress</td>
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<td>&lt;0.01 – 5.20</td>
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<td>[Petrlik et al. 2018]</td>
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<td>Shoe toy</td>
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<td>0.30 – 0.98</td>
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<td>Toy gun</td>
<td>Toy</td>
<td>375</td>
<td>[DiGangi et al. 2017]</td>
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<td>&lt;LOQ</td>
<td>[Straková et al. 2018a]</td>
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<td>Hair accessorie</td>
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<td>[Straková et al. 2018a]</td>
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<td>Toy</td>
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<td>[Straková et al. 2018a]</td>
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<td>Hair accessorie</td>
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<td>Denmark</td>
<td>Toy gun</td>
<td>Toy</td>
<td>0 – 2</td>
<td>[Straková et al. 2018a]</td>
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<td>France</td>
<td>Diadem</td>
<td>Hair accessorie</td>
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<tr>
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<td>Musical mobile phone</td>
<td>Toy</td>
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<td>[Straková et al. 2018a]</td>
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<tr>
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<td>Toy gun</td>
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<td>[Straková et al. 2018a; Petrlik et al. 2018]</td>
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<td>Toy</td>
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<td>Hair accessorie</td>
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<td>[Straková et al. 2018a]</td>
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<tr>
<td>Germany</td>
<td>Massage roller</td>
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<td>[Straková et al. 2018a]</td>
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<td>Waste bin</td>
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<td>1 - 7</td>
<td>[Straková et al. 2018a]</td>
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<td>Germany</td>
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<td>[DiGangi et al. 2017]</td>
</tr>
<tr>
<td>Germany</td>
<td>Key fob</td>
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<td>[Petrlik et al. 2018]</td>
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<td>Toy</td>
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<tr>
<td>Germany</td>
<td>Mini hair claws</td>
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<td>[Straková et al. 2018a]</td>
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<td>Product category</td>
<td>Concentration [mg/kg]</td>
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<td>Hungar</td>
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<td>Toy</td>
<td>0/n.a.</td>
<td>[DiGangi et al. 2017]</td>
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<td>0 – 78</td>
<td>[DiGangi et al. 2017]</td>
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<td>Toy</td>
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<td>[Petrlik et al. 2018]</td>
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<td>Indonesia</td>
<td>Rubik’s cube</td>
<td>Toy</td>
<td>0 – 541</td>
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<td>[DiGangi et al. 2017]</td>
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<td>0 – 1,280</td>
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<td>[DiGangi et al. 2017]</td>
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<td>Toy</td>
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<td>[Straková et al. 2018a]</td>
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<td>Country/ region</td>
<td>Product/product category</td>
<td>Product category</td>
<td>Concentration [mg/kg]</td>
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<td>[Straková et al. 2018a]</td>
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<td>Small guitar</td>
<td>Toy</td>
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<td>Toy</td>
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<td>[DiGangi et al. 2017]</td>
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<td>Hair accessory</td>
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<td>[DiGangi et al. 2017]</td>
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<td>Hair accessory</td>
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<td>[Straková et al. 2018a]</td>
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<td>Toy</td>
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<td>[Straková et al. 2018a]</td>
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<td>Hair accessory</td>
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<td>[Straková et al. 2018a]</td>
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<td>Toy</td>
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<td>0 – 1</td>
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<td>&lt;LOQ</td>
<td>[Straková et al. 2018a]</td>
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<td>Toy</td>
<td>&lt;LOQ</td>
<td>[Straková et al. 2018a]</td>
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<td>Product category</td>
<td>Concentration [mg/kg]</td>
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<td>&lt;LOQ</td>
<td>[Straková et al. 2018a]</td>
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<td>[Straková et al. 2018a]</td>
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<td>[DiGangi et al. 2017]</td>
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<td>0 - 5</td>
<td>[DiGangi et al. 2017]</td>
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<td>UK</td>
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<td>Toy</td>
<td>0 - 5</td>
<td>[DiGangi et al. 2017]</td>
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## 10.11 Selected stakeholder reasoning for LPCLs

<table>
<thead>
<tr>
<th>Stakeholder</th>
<th>Substance</th>
<th>Proposed LPCL [mg/kg]</th>
<th>Reasoning for proposal</th>
</tr>
</thead>
<tbody>
<tr>
<td>[IPEN &amp; Arnika Sub. 2018a]</td>
<td>SCCPs</td>
<td>100</td>
<td>According to additional information submitted by IPEN, the current LPCL listed in the Annex IV of the EU POP Regulation of SCCPs (10,000 mg/kg) is considered too high in IPEN's view. IPEN suggests lowering the LPCL to 100 mg/kg. The suggestion is based on the report prepared by BiPRO for the German Environment Agency (i.e. [BiPRO 2015]), stating: &quot;In order to minimise risks, it is possible to set the LPCL for SCCPs to 1,000 mg/kg or even to 100 mg/kg. The limit value would have to lie above 350 mg/kg (for instance at 1,000 mg/kg), as otherwise a large proportion of the total rubber waste in Germany could not be directed to material recovery any longer. As an alternative, the LPCL could be set to 100 mg/kg, in case the SCCPs containing rubber waste is separated successfully from other rubber waste and is treated separately. However, it is recommended not to set the LPCL below 100 mg/kg to avoid adverse economic effects.&quot;</td>
</tr>
<tr>
<td>[VCI 2018] on behalf of European HBCD Industry Group and EUMEPS</td>
<td>HBCDD</td>
<td>1,000</td>
<td>At present no robust analytical method for HBCDD exists to provide satisfactory reproducible results at levels below 1,000 mg/kg. A 1,000 mg/kg limit would ensure alignment with the limit values deemed safe in national or international regulations, such as EU REACH which defines a level of 0.1% w/w for articles containing SVHC. Other EU legislation, such as the EU RoHS and WEEE Directives, also set an allowed threshold limit of 1,000 mg/kg for the sum of PBDEs. A low POP limit of 1,000 mg/kg would be practical, pragmatic and environmentally sound. It would ensure both the protection of human health and of the environment. It could be implemented by the stakeholders in the waste management sector easily and effectively enforced by EU MS. The European HBCD Industry Group and EUMEPS conclude that an LPCL of 1,000 mg/kg for HBCDD provides for the necessary margin of safety and can be regarded as appropriate for meeting environmental and health concerns. A low POP limit value of 1,000 mg/kg will suffice to maintain the downward trend of HBCDD in the environment, given that following the listing of HBCDD under the Stockholm Convention and the EU POPs Regulation, the use of HBCDD in products is being discontinued worldwide. A low POP level of 1,000 mg/kg represents a solid basis to ensure that all PS foam waste from demolition containing HBCDD is channelled to destruction, providing for a sound and responsible end of life management of the HBCDD-containing PS foam waste. Considering circular economy aspirations, the low POP</td>
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<tr>
<td>Stakeholder</td>
<td>Substance</td>
<td>Proposed LPCL [mg/kg]</td>
<td>Reasoning for proposal</td>
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<tr>
<td></td>
<td></td>
<td>1,000</td>
<td>content limit ought to be such that it allows for the recovery and recycling of PS foams which do not contain HBCDD. Even though investigations have shown that waste fractions of EPS packaging can contain HBCDD, levels of contamination are comparatively low. Considering the effectiveness, the cost and the logistics of analysis, a low POP limit of 1,000 mg/kg allows for a manageable recycling of PS foams which are not meant to contain HBCDD. A level lower than 1,000 mg/kg is likely to bring such recycling operations economically off balance whilst endangering their compliance obligations. Furthermore, additional costs will result from an increase of the amounts of PS foams that would have to be incinerated, including also valuable foams that do not contain any HBCDD. Therefore, lower levels than 1,000 mg/kg are likely to hinder the achievement of recycling targets and the transition to a circular economy.</td>
</tr>
<tr>
<td>[PS PlasticsSupport Sub. 2018]</td>
<td>HBCDD</td>
<td>1,000</td>
<td>The determination of a LPCL for HBCDD in waste and also the UTC of 100 mg/kg is dependent on a practicable and validated analytic method, which is also economically feasible. Currently, no standard analysis method for HBCDD in plastics exists. In 2014, efforts started by the ICE TC111 WG3 of processing an ‘IEC’ DIN EN 62321-9 (HBCDD) standardization. By the end of 2017, the first results of the global validation process were discussed in an international interlaboratory study (ILS). The ILS did not arrive at a conclusion and the investigations will continue in 2018. Certainly, HBCDD has been detected in water, sediments, fatty tissues, breast milk etc. for many years with liquid chromatography-mass spectrometry (LC-MS) and gas chromatograph-mass spectrometry (GC-MS). This practice of trace analysis of HBCDD is available, however, this procedure is not suitable for determining HBCDD in industrial plastics recycling. According to the stakeholder, at a limit value of 100 mg/kg for HBCDD the results of the analysis will be questionable, even if a LOD of 10 mg will be possible. An LOD of 1 mg/kg would be necessary which would extremely increase the laboratory efforts (equipment, resources) and hence, the costs of the analysis. The stakeholder pleads for an LPCL of 1,000 mg/kg for HBCDD, which can be ensured via available analytical measures at reasonable costs.</td>
</tr>
<tr>
<td>[Plastics Recyclers Europe 2018]</td>
<td>HBCDD</td>
<td>&gt; 1,000</td>
<td>According to the submission from the Plastics Recyclers Europe, for the sake of alignment of legislation of BFR, the threshold for HBCDD should be set higher than 1,000 mg/kg. The analysis of elemental bromine will become sufficient in most cases to make sure that HBCDD is below the limit value.</td>
</tr>
<tr>
<td>Stakeholder</td>
<td>Substance</td>
<td>Proposed LPCL [mg/kg]</td>
<td>Reasoning for proposal</td>
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<td>-----------------------------------------------</td>
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<tr>
<td>[AT Ministry of Sustain. and Tourism Sub. 2018b]</td>
<td>HBCDD</td>
<td>n.a.</td>
<td>According to information submitted by Austria regarding bromine detection in WEEE &quot;XRF is generally applied to trace sources of BFR in waste plastics (each piece of plastic is checked). This analysis technique is fast, cheap and simple and a suitable means to detect bromine. For data evaluation in the course of the Austrian study (Monitoring of WEEE plastics in regard to BFR using handheld XRF (see [Aldrian et al., 2015]), a detection limit of 30 mg/kg was assumed although the standard deviation of the XRF device for this value is already over 60%. As the detection limit strongly depends on the analysis time, the actual detection limit for bromine for this kind of application (bromine in a polymer matrix) is estimated to be at approximately 100–150 mg/kg based on the acceptable relative error of the device and the accuracy of results and data from literature. In [Aldrian et al., 2015] the applied method was validated by comparing the data of handheld XRF with results obtained by GC–MS. The results showed the expected and sufficiently accurate correlation between these two methods. It is shown that handheld XRF technique is an effective tool for fast monitoring of large volumes of WEEE plastics in regard to BFRs for on-site measurements&quot;.</td>
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<tr>
<td>[IPEN &amp; Arnika Sub. 2018a]</td>
<td>HBCDD</td>
<td>100</td>
<td>IPEN &amp; Arnika suggest establishing a more stringent LPCL for HBCDD at 100 mg/kg because high levels of HBCDD are found in children toys, food packaging and other daily use products produced from recycled plastics. Recently HBCDD was found even in polystyrene packaging for food [Rani et al. 2014; Abdallah et al. 2018]). This flow of HBCDD should be stopped. HBCDD can be removed from polystyrene before its recycling [Schlummer et al. 2017]. The establishment of a more stringent LPCL would lead to prevention of entering of this chemical into uncontrolled processes of plastic waste recycling.</td>
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10.12

PCDD/F in fly ash residues
Table 85:

PCDD/F in fly ash residues [CEWEP Sub. 2018b]
Observations in I-TEQ [ng TEQ/g]

Congeneer

Average

Max
1

2378 TCDD

12378

2

3

4

5

6

7

8

9

10

11

12

13

14

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31

32

33

34

35

0.1

0.5

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PeCDD

123478
HxCDD

123678
HxCDD

123789
HxCDD

1234678
HpCDD

OCDD

2378 TCDF

12378

387/390


| Congener   | Average | Max | 1  | 2  | 3  | 4  | 5  | 6  | 7  | 8  | 9  | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 | 20 | 21 | 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 | 30 | 31 | 32 | 33 | 34 | 35 |
|------------|---------|-----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|----|
| PeCDF      |         |     | 0.7| 0.5| 0.4| 0.5| 0.3| 0.1| 0.2| 0.1| 0.6| 0.3| 0.2| 0.9| 0.2| 0.3| 0.1| 0.4| 0.2| 0.5| 0.6| 0.8| 0.1| 8.5| 0.1| 0.3| 0.2| 0.1| 0.1| 0.1| 1.9| 0.3| 0.6| 2.5| 1.0| 0.5| 1.2| 1.2|
| HxCDF      |         |     | 0.2| 2.9| 0.1| 0.2| 0.1| 0.0| 0.0| 0.0| 0.0| 0.1| 0.0| 0.0| 0.0| 0.0| 0.0| 0.1| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0|
| OCDF       |         |     | 0.1| 0.8| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0|
| OCDF       |         |     | 0.1| 0.6| 0.0| 0.1| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0|
| OCDF       |         |     | 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0| 0.0|
| Total      | 2.5     |     | 23.9| 1.6| 1.8| 1.0| 0.2| 0.5| 0.8| 1.7| 1.0| 0.3| 3.2| 1.0| 1.2| 0.2| 1.0| 0.5| 1.7| 1.4| 4.8| 0.5| 23.9| 0.8| 0.9| 1.0| 0.6| 0.4| 0.4| 0.5| 5.9| 1.8| 2.0| 9.0| 2.7| 2.3| 4.5| 5.6|

Observations in 1-TEQ (ng TEQ/g)